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LINKING OCCURRENCE AND FITNESS TO PERSISTENCE: HABITAT-BASED APPROACH FOR ENDANGERED GREATER SAGE-GROUSE

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Abstract. Detailed empirical models predicting both species occurrence and fitness across a landscape are necessary to understand processes related to population persistence. Failure to consider both occurrence and fitness may result in incorrect assessments of habitat importance leading to inappropriate management strategies. We took a two-stage approach to identifying critical nesting and brood-rearing habitat for the endangered Greater Sage-Grouse (*Centrocercus urophasianus*) in Alberta at a landscape scale. First, we used logistic regression to develop spatial models predicting the relative probability of use (occurrence) for Sage-Grouse nests and broods. Secondly, we used Cox proportional hazards survival models to identify the most risky habitats across the landscape. We combined these two approaches to identify Sage-Grouse habitats that pose minimal risk of failure (source habitats) and attractive sink habitats that pose increased risk (ecological traps). Our models showed that Sage-Grouse select for heterogeneous patches of moderate sagebrush cover (quadratic relationship) and avoid anthropogenic edge habitat for nesting. Nests were more successful in heterogeneous habitats, but nest success was independent of anthropogenic features. Similarly, broods selected heterogeneous high-productivity habitats with sagebrush while avoiding human developments, cultivated cropland, and high densities of oil wells. Chick mortalities tended to occur in proximity to oil and gas developments and along riparian habitats. For nests and broods, respectively, approximately 10% and 5% of the study area was considered source habitat, whereas 19% and 15% of habitat was attractive sink habitat. Limited source habitats appear to be the main reason for poor nest success (39%) and low chick survival (12%). Our habitat models identify areas of protection priority and areas that require immediate management attention to enhance recruitment to secure the viability of this population. This novel approach to habitat-based population viability modeling has merit for many species of concern.

Key words: Alberta, Canada; Centrocercus urophasianus; *Cox proportional hazard; fitness; Greater Sage-Grouse; habitat; logistic regression; occurrence; persistence; population viability; sagebrush.*

Introduction

Detailed theoretical and empirical models linking resources to both animal occurrence and fitness measures are necessary to understand the underlying processes determining population persistence. Although numerous local population studies focusing on fine-scale habitat correlations with various species declines have been conducted, landscape-scale habitat models (Franklin et al. 2000, Wisdom et al. 2002*a, b,* Akçakaya et al. 2004) or range-wide analyses addressing processes and patterns of persistence have been attempted for relatively few species (see Mattson and Merrill 2002, Laliberte and Ripple 2004). Only a handful of these studies have integrated population dynamics with landscape-level resources (Wiegand et al. 1998, Akçakaya et al. 2004), with even fewer successfully decomposing models to critical life stages and addressing landscape-level drivers of fitness (see Breininger et al. 1998, Franklin et al. 2000, Larson et al. 2004). Links to fitness are a critical and necessary component for long-term conservation of many species of concern (Donovan and Thompson 2001) that allows biologists and managers to suitably assess population viability (Boyce et al. 1994, Boyce and McDonald 1999).

Ultimately, measures of habitat quality must link fitness (reproduction and survival; Van Horne 1983, Morrison 2001) to resources to accurately assess how resources affect population viability. Occurrence or abundance may not be a good indicator of fitness (Van Horne 1983, Hobbs and Hanley 1990, Morrison 2001, Tyre et al. 2001), particularly in human-dominated landscapes (Remes 2000, Bock and Jones 2004), due to the creation of ecological traps. Thus, assessments should involve the identification of (1) habitats that animals are likely to use (occurrence), in addition to (2) habitats where animals are likely to be successful (fitness). Habitat patches where animals are likely to occur and that also have high reproduction and/or survival measures are source habitats (Pulliam 1988, Breininger et al. 1998), whereas habitats with abundant

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animals but poor fitness have been referred to as attractive sinks (Delibes et al. 2001, Larson et al. 2004) or ecological traps (Donovan and Thompson 2001, Battin 2004, Bock and Jones 2004). Failure to differentiate attractive sinks from source habitats may result in incorrect assessments of habitat importance, ultimately leading to inappropriate management. However, the ability to appropriately assess habitat quality is limited by the difficulty in gathering suitable basic life-history information for many species (Donovan and Thompson 2001), particularly those that are rare or have low reproductive rates.

Sagebrush-steppe habitats have undergone extensive changes since European settlement. Today, many of these habitats are considered imperiled, facing continuing fragmentation and degradation (Knick et al. 2003, Connelly et al. 2004) due to conversion to agriculture (Connelly et al. 2004), invasion by nonnative species (Knick et al. 2003, Connelly et al. 2004), energy extraction activities and developments (Braun et al. 2002, Lyon and Anderson 2003), intense grazing pressure (Beck and Mitchell 2000, Hayes and Holl 2003, Crawford et al. 2004), and climate change (Neilson et al. 2005). As a result, species dependent on sagebrush-steppe have experienced drastic range contractions and population declines. Sage-Grouse (*Centrocercus* spp.) are a notable example. Currently, Sage-Grouse exist in about half of their historic range (Schroeder et al. 2004), with individual populations declining by 15–90% since the early 1970s (Connelly and Braun 1997, Aldridge and Brigham 2003, Connelly et al. 2004). Many populations are at risk of extirpation, reinforcing the need to appropriately assess habitat relationships for this species.

Although much research has been conducted at fine scales, addressing factors related to nest success (Aldridge and Brigham 2001, Connelly et al. 2004, Holloran et al. 2005) and some related to chick survival (Aldridge and Brigham 2001, Aldridge 2005), research assessing potential landscape features driving habitat selection and fitness is limited. Other than the recently published Greater Sage-Grouse (*Centrocercus urophasianus*) conservation assessment (Connelly et al. 2004), which summarized range-wide habitats and threats, only one study, to our knowledge, used a habitat-based landscape approach to assess Greater Sage-Grouse population persistence within the interior Columbia basin of the western United States (Wisdom et al. 2002*a*, *b*).

Within its current range, the Alberta Greater Sage-Grouse (hereafter Sage-Grouse) population has declined 66–92% since the 1970s (Aldridge and Brigham 2003, Connelly et al. 2004). This population (endangered provincially and within Canada; Aldridge and Brigham 2003) is isolated from other populations and inhabits a heavily fragmented landscape dominated by oil and gas activities (Braun et al. 2002), and has only 400–600 birds remaining (Aldridge 2005). Low productivity limits this population (Aldridge and Brigham 2001, 2002, 2003, Aldridge 2005) and the implementation of long-term habitat management initiatives may be required before increases occur (Crawford et al. 2004).

Our overall objective was to identify nesting and brood-rearing habitats critical to the persistence of Sage-Grouse in Alberta. First, we developed landscape-level occurrence models predicting where Sage-Grouse are likely to nest and raise their young. Secondly, we developed survival models to identify the most risky habitat for Sage-Grouse nests and for chicks. We validated the predictive capacity of these models using independent data sources from prior research in Alberta. We combined these two approaches to identify source habitats where Sage-Grouse are likely to occur and also be successful. Conversely, we identified ecological trap habitats that are attractive to Sage-Grouse, but are habitats where nests are likely to fail, or chicks are likely to die. We used these habitat states to identify areas that require immediate management attention. We discuss our findings within the context of potential reclamations or landscape improvements that could result in the transformation of ecological trap habitats into higher quality source habitats that are likely to sustain the Alberta Greater Sage-Grouse population.

Methods

Study area

Sage-Grouse are found within a 4000 km^2 area of the dry mixed-grass prairie of southern Alberta, Canada (Fig. 1). Our study area (49°24′ N, 110°42′ W, ~900 m elevation) encompasses the core of this range (1110 km^2; Fig. 1). Most lands are grazed by cattle, and roughly one-third of this area is influenced by oil and gas activities. Summer (July–August) temperatures average 19.1°C and annual precipitation is ~358 mm (Onefour Agriculture and Agri-food Canada Research Station [2004], *unpublished weather data*). Silver sagebrush (*Artemisia cana* Pursh) is the dominant shrub, and there are a variety of different forb species, including pasture sage (*A. frigida* Willd.), several species of clover (*Trifolium* spp. and *Melilotus* spp.), vetch (*Astragalus* spp.), and common dandelion (*Taraxacum officinale* Weber ex Wiggers). Needle-and-thread grass (*Hesperostipa comata* Trin. and Rupr.), june grass (*Koeleria macrantha* Ledeb.), blue grama (*Bouteloua gracilis* Willd. ex Kunth), and western wheatgrass (*Pascopyrum smithii* Rydb.) are the dominant grass species (Coupland 1961, Aldridge and Brigham 2003).

Field techniques

Female Sage-Grouse were captured during the breeding season from five of eight known active leks (breeding sites) in southeastern Alberta from 2001 to 2004 and were fitted with a 14-g necklace-style radiotransmitter (RI-2B transmitters, Holohil Systems, Carp, Ontario, Canada). Hens were located every second day so that nesting attempts and nest fate could be assessed. Nest initiation and hatch/failure were estimated as the midpoint between consecutive (every two days) relocations (Manolis et al. 2000) following Aldridge (2005). From 2001 to 2003, if a nest was successful (i.e., $\geq$1 egg hatched), we captured chicks by hand as soon as possible after hatch and attached 1.6-g microtransmitters (BD-2G transmitters, Holohil Systems, Carp, Ontario, Canada) to two randomly chosen chicks from each brood (see Burkepile et al. 2002, Aldridge 2005). Hens with broods (2001–2004) and chicks (2001–2003) were relocated every two days during the brood-rearing period.

GIS predictor variables

We developed a suite of variables in a GIS that may be important as predictors of Sage-Grouse nest and brood occurrence, as well as survival of nests and chicks. These variables were related to either habitat characteristics or human influences (see Table 1 for a detailed description of each variable and its data source). We used a dry mixed-grass plant community guide based primarily on soil types (Adams et al. 2005) to identify Sage-Grouse ecosite range plant communities (B. W. Adams, *personal communication*). We generated summary statistics calculating the proportion of each habitat class within a 1-km^2 moving window across the landscape. We used a July 2000 Landsat TM Satellite image to generate brightness, greenness, and NDVI (Normalized Difference Vegetation Index) values using a tasselled-cap transformation (Crist and Cicone 1984, Sellers 1985) in the program PCI Geomatica Prime 8.2 (PCI Geomatics 2001). We also estimated the mean and standard deviation (SD) of NDVI values within a 1-km^2 moving window. Higher SD values represent more heterogeneous (variable) habitat patches.

The importance of sagebrush in providing nesting habitat at local scales has been demonstrated (Sveum et al. 1998*b*, Aldridge and Brigham 2002, Holloran et al. 2005), and sagebrush may also be selected at brood-rearing sites (Aldridge and Brigham 2002, Aldridge 2005). We used a digital map of sagebrush developed from aerial photo interpretation to estimate sagebrush cover (the percentage of each landscape polygon that was covered with sagebrush plants; Jones et al. 2005) at the each pixel and 1-km^2 window scales. Sage-Grouse may select for intermediate sagebrush cover (quadratic relationship or concave selection function; Aldridge 2005), because very thick shrub cover can limit herbaceous understory and reduce a bird's ability to detect predators (Wiebe and Martin 1998). Thus, we also assessed selection for sagebrush cover metrics as quadratic functions (Table 1). Finally, we reclassified the sagebrush density distribution defined by Jones et al. (2005) into two measures of "patchy" or heterogeneous sagebrush distribution, estimated per pixel and at the 1-km^2 scale (see Table 1).

Sage-Grouse broods move to mesic habitats with greater forb (Drut et al. 1994*a*, Sveum et al. 1998*a*) and insect (Johnson and Boyce 1991, Drut et al. 1994*b*) abundance later in the summer. We used a soil-moisture index derived from a digital elevation model (DEM; see Evans 2002) called a compound topographic index (CTI), which is correlated with soil moisture and nutrients (Gessler et al. 1995). Similar to our lines of inference for NDVI, we also calculated measures of the mean CTI and the variability (SD) in CTI within a 1-km^2 moving widow (Table 1). In addition, we calculated the distance to the nearest water source (Table 1).

Anthropogenic landscape features included distance measures for roads, trails, oil well sites, crop (cultivated lands), and urban (town, farmstead, energy infrastructure) areas, as well as a density measure for each variable calculated as the linear kilometer per square kilometer for roads and trails, the number of well sites within a 1-km^2 window, and the proportion of area that was either crop or urban within a 1-km^2 window. Noise and human activity associated with road and oil wells may be avoided by (Braun et al. 2002) or may have negative consequences (Lyon and Anderson 2003) for Sage-Grouse. Thus, we also summed the number of pixels classified as either roads or well sites that were visible from any given cell within 250, 500, and 1000 m. To assess how water impoundments (e.g., dams, dugouts, canals, and so on; McNeil and Sawyer 2003) influence habitat selection by Sage-Grouse, we generated distance and density measures for water impoundments (Table 1). The final anthropogenic variables were distance and density measures (proportion of habitat within 1-km^2 window) for human habitat (roads, oil wells, urban), and nonnatural edge habitats (roads, oil wells, urban, and crop). All GIS analyses were conducted using ArcGIS 8.3 (ESRI 2002).

Model development

We conducted univariate analyses for all predictor variables (Hosmer and Lemeshow 2000), using $P < 0.25$ based on a Wald z statistic as a cutoff for inclusion in the full model. We assessed each variable for outliers and nonlinearities (Hosmer and Lemeshow 1999, 2000). If two parameters were correlated ($r > |0.6|$), we retained the variable with the smaller P value. We assessed the full model, dropping the least significant parameter (i.e., largest P value), refitting the reduced model and repeating the process until all remaining parameters were significant at $\alpha = 0.05$ (Hosmer and Lemeshow 1999, 2000). We tested for multicollinearity using variance inflation factors (VIF; Menard 1995), removing variables if VIF scores for individual parameters > 10 or mean model scores > 1 (Chatterjee et al. 2000). All analyses were conducted in STATA 8.2 (STATA 2004), and descriptive results are presented as means $\pm$ SE.

Logistic regression occurrence analyses

We define occurrence as the relative probability of Sage-Grouse resource use based on detections from radiotelemetry. We evaluated third-order habitat selection

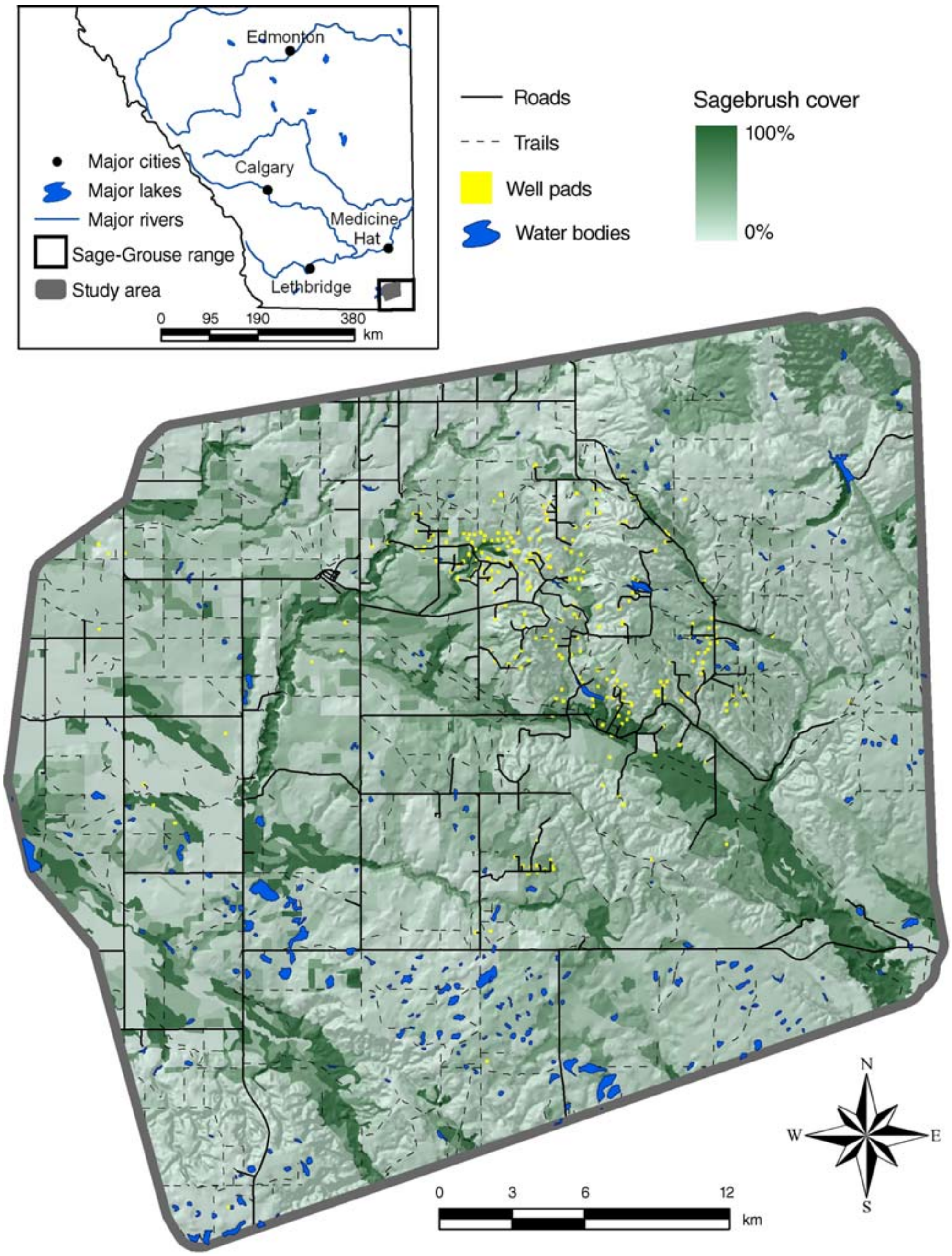

FIG. 1. Alberta Greater Sage-Grouse study area showing sagebrush density along with roads, trails, well pads, and major water bodies. The inset map shows the study area and current range of Sage-Grouse within Alberta, Canada, with major rivers, water bodies, and cities for reference.

(Johnson 1980) using resource selection functions (RSFs; Manly et al. 2002) with a design II approach, following individuals to identify a set of used resources, but assessing availability at the population level (Erickson et al. 2001). The RSF is equivalent to the logistic discriminant contrasting the distributions of used and available resource units (Keating and Cherry 2004, Johnson et al. 2006). Coefficients for RSF models are presented as unstandardized linear estimates and standard errors. We generated 5000 random locations across a 1-km buffer around a 100% minimum convex polygon surrounding all Sage-Grouse nest and brood locations combined (1110 km^2 area), resulting in a sample density of about five available resource units per square kilometer. Due to models being heavily biased toward the larger sample of available (0) resource units, we used an importance weight,

TABLE 1. Explanatory GIS variables used for Sage-Grouse nest and brood/chick occurrence and survival models in southeastern Alberta, Canada.

Variable	Data type	Description
Brit	30 m cont.	brightness generated from a Landsat 7 TM satellite image
Green	30 m cont.	greenness generated from a Landsat 7 TM satellite image
Wet	30 m cont.	wetness generated from a Landsat 7 TM satellite image
NDVI	30 m cont.	NDVI calculated from a TM satellite image
NDVI_avg	30 m cont.	mean NDVI value within a 1-km^2 moving window
NDVI_sd	30 m cont.	standard deviation of NDVI within a 1-km^2 window
CTI	30 m cont.	Compound Topographic Index (high values = increased moisture)
CTI_mean	30 m cont.	mean CTI values within a 1-km^2 moving window
CTI_sd	30 m cont.	standard deviation of CTI values within a 1-km^2 moving window
Well_dist	10 m cont.	distance to nearest standing energy well site
Well_dens	10 m cont.	count of energy well sites within a 1-km^2 moving window
vWell_1km, _500 m, _250 m	30 m cont.	no. visible 30-m pixels that are wells within radius of 1 km, 500 m, or 250 m
Rd_dst	10 m cont.	distance (km) to nearest road (any paved or gravel road)
Rd_dens	10 m cont.	linear km per km^2 of roads
vRd_1km, _500 m, _250 m	30 m cont.	no. visible 30-m pixels that are road within radius of 1 km, 500 m, or 250 m
Tr_dst	10 m cont.	distance (km) to nearest trail (non-paved or gravelled truck trail)
Tr_dens	10 m cont.	linear km per km^2 of trails
Imp_dst	10 m cont.	distance to nearest water impoundment (dam, dugout, canal, combination)
Imp_dens	10 m cont.	count of no. water impoundments within a 1-km^2 moving window
Water_dst	10 m cont.	distance to nearest natural permanent or semipermanent water body
SB	10 m cont.	sagebrush cover (%) as identified from air photo interpretation
SB^2	10 m cont.	squared term for SB
SBmean	10 m cont.	mean sagebrush cover (%) within a 1-km^2 moving window
SBmean^2	10 m cont.	squared term for SBmean
SB_pch1, SB_pch2	10 m cont.	patchy sagebrush distribution 1 (codes 7, 8, 9) or 2 (codes 7, 8, 9, 11) from Jones et al. (2005)
pSB_pch1, pSB_pch2	10 m cont.	proportion of habitat within a 1-km^2 moving window that is SB_pch1 or SB_pch2, respectively
Crop_dst	10 m cont.	distance to nearest cultivated lands
pCrop	10 m cont.	proportion of habitat within a 1-km^2 moving window that is cultivated
pUrban	10 m cont.	proportion of habitat within a 1-km^2 moving window that is urban (town, ranch, energy compressor station, and so on)
Urban_dst	10 m cont.	distance to nearest urban developments
Eco1	10 m cat.	loamy range site with well-drained soils, low sagebrush cover
Eco2	10 m cat.	saline lowlands, swales and depression, sparse low sagebrush
Eco3	10 m cat.	blowout and overflow sites, solonetzic soils; plant community varies, but higher density of sagebrush
Eco4	10 m cat.	loamy upland sites with medium texture soils, fescue and wheat grasses
Eco5	10 m cat.	thin break range sites, soils vary, characterized by greater shrub cover
Eco6	10 m cat.	badlands type habitats with juniper and needle-and-thread-blue grama
Eco7	10 m cat.	broad, wetland and shrubby (willow, rose, snowberry) riparian habitats
Eco8	10 m cat.	all altered habitats (urban, crop, wells and roads); see Hum and Edge
pEco1, pEco2, … pEco7	10 m cont.	proportion of habitat within a 1-km^2 moving window that is Eco1, Eco2, … Eco7
Hum_dst	10 m cont.	distance to any human habitat (roads, wells, urban)
pHum	10 m cont.	proportion of habitat within a 1-km^2 moving window that is human habitats
Edge_dst	10 m cont.	distance to habitat that creates nonnatural edge habitats (human above + crop)
pEdge	10 m cont.	proportion of habitat within 1-km^2 moving window that is edge habitats

Notes: All variables were first tested univariately in occurrence (logistic regression) and survival (proportional hazards) models. Candidate variables with $P < 0.25$ were removed, and correlated variables with higher P values were removed. Data type refers to continuous (cont.) or categorical (cat.) variables. All distance measures are in kilometers. NDVI is the Natural Difference Vegetation Index. Data sources are as follows: TM-derived variables were based on a 22 July 2000 Landsat 7 image (Path 39 Row 26); digital elevation models (DEM) were derived from 1:50 000 National Topographic Database Contour Lines; sagebrush, crop, urban, and water base features are from Jones et al. (2005); sagebrush cover is the percentage of the area within each polygon covered by sagebrush (Jones et al. 2005); linear features were based on a 2001 landscape from Alberta Provincial Base features (1:20 000); well locations were provided by Alberta Energy for the study area as of August 2002; water impoundments were mapped based on McNeil and Sawyer (2003); Eco1–Eco7 are dry mixed-grass rangeland ecosite plant community bins after Adams et al. (2005); a viewshed analyses tool for ArcGIS 8.3 (H. L. Beyer, ⟨http://www.spatialecology.com/htools/overview.php⟩), together with a DEM to generate these data, was used to generate density of viable wells and roads. For visibility purposes, we assumed that well sites were 9 m in height and that the average vehicle was 2 m in height.

which gave full weighting to used resource units, but available resource units received a weighting (down) proportional to the ratio of sampled use (1) points to available points (STATA 2004, Users Guide). Weighting effectively adjusts (inflates) the standard errors of the estimates, and allows for traditional inferences about standard errors and P values for coefficient estimates. Given that a shift in brood habitat to more mesic sites at about seven weeks of age (Dunn and Braun 1986, Sveum et al. 1998*a*) does not occur in Alberta (Aldridge and Brigham 2002), we combined locations throughout the brood-rearing period for all analyses.

Proportional hazards survival analyses

We used the Cox proportional hazards regression model (Cox 1972) to assess how landscape variables affect nest survival or success and chick survival. The Cox model allows for left- and right-censoring of data (Andersen and Gill 1982, Cleves et al. 2004) and estimates the hazard rate. We present coefficients for all survival models as hazard ratios (exp[β_i]) and standard errors. For chick survival models, we estimated a shared frailty Cox proportional hazards model to account for lack of independence of chicks within broods (Cleves et al. 2004, Wintrebert et al. 2005). We used the Breslow estimation of the continuous-time likelihood calculation (Cleves et al. 2004) to partition deaths with tied failure times. We assessed the proportional hazards assumption (Winterstein et al. 2001) for our models by testing for nonzero slopes of Schoenfeld residuals (Schoenfeld 1982) and by inspecting logarithm plots of the estimated cumulated hazard functions (Cleves et al. 2004).

Model assessment and validation

We used a χ^2 statistic (Hosmer and Lemeshow 2000) to asses the fit of all final models, except for the chick shared frailty model, for which we used a Wald χ^2 statistic (Hosmer and Lemeshow 1999). We estimated the cumulative daily relative risk of failure for top survival models as the sum of the predicted relative hazard for each individual nest or chick divided by exposure days. We used these predictions to assess the predictive accuracy based on receiver operator characteristic (ROC) estimates (Fielding and Bell 1997). High model accuracy results in ROC estimates above 0.9, good model accuracy between 0.7 to 0.9, and values below 0.7 indicate low model accuracy (Swets 1988, Manel et al. 2001). We used the percentage correctly classified (PCC) at the optimal cutoff (where the absolute value of the difference between sensitivity and specificity is minimized; Liu et al. 2005) to estimate of the predictive capacity of the top occurrence models. We considered PCC $\geq$ 80% as excellent model prediction and PCC $\geq$ 70% was reasonable prediction (Nielsen et al. 2004). We also validated our nest survival model by predicting it to an independent sample of 38 nests with known fate produced by 31 different females from 1998 to 2000 (Aldridge and Brigham 2002). We assessed fit and prediction as previously described for model training data. We did not have independent chick survival data for validation, and limited sample sizes (41 chicks) prevented us from folding our data for cross-validation purposes (Boyce et al. 2002). Thus, for both chick and nest survival models, we took the predicted daily hazard and tested for differences in the rate of failures or deaths (nest or chick) compared to those that survived. If the model was predictive, failed chicks or nests should have been exposed to greater daily hazards. We used a one-tailed t test with unequal variances to test for differences in daily relative hazard rates.

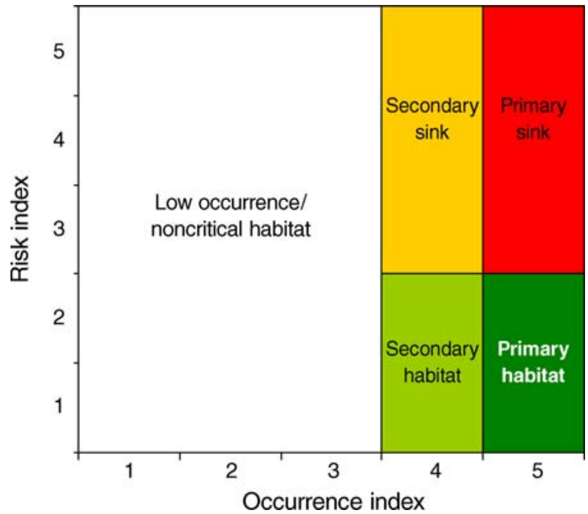

FIG. 2. A graphic representation of nesting and brood-rearing habitat states for Greater Sage-Grouse in southeastern Alberta. States include noncritical (low occurrence) habitat, primary habitat (high occurrence and low-to-moderate risk), secondary habitat (good occurrence and low-to-moderate risk), primary sink (high occurrence and moderate-to-extreme risk), and secondary sink (high occurrence and moderate-to-extreme risk). The figure is developed from the approach of Nielsen et al. (2006).

For RSF models, it is inappropriate to assess model accuracy and predictive capacity using ROCs and PCC (Boyce et al. 2002). Thus, we predicted the RSF to generate relative index-of-occurrence scores, ranking habitat pixels into five quantile bins; bin 1 was the lowest rank. For each model, we initially grouped the landscape into 10 quantile bins, each with an equal proportion of the landscape (see Boyce et al. 2002). In most cases though, some bins contained no training or validation data points, forcing us to lump bins to avoid null cells. We adjusted for availability of habitat (amount of area) within each bin as suggested by Boyce et al. (2002). We used a Spearman rank correlation to test for a correlation between frequency (area-adjusted) of use locations within increasing bin ranks (Boyce et al. 2002). Again, we validated both occurrence models using training data sets (2001 to 2004), and performed out-of-sample validation (1998 to 2000) using an independent sample of 40 nest locations produced by 33 different females, and 151 brood locations from 16 different broods (Aldridge and Brigham 2002, Aldridge 2005).

Development of habitat states

We defined the five ranked bins for nest and brood occurrence models as (1) poor, (2) low, (3) moderate, (4) good, and (5) high occurrence, with good-to-high bins indicating that Sage-Grouse were likely to occur there. Similarly, we applied survival models, ranking the predicted relative risk of failure (nest or chick) for the survival models, into five quantile risk bins: (1) minimal,

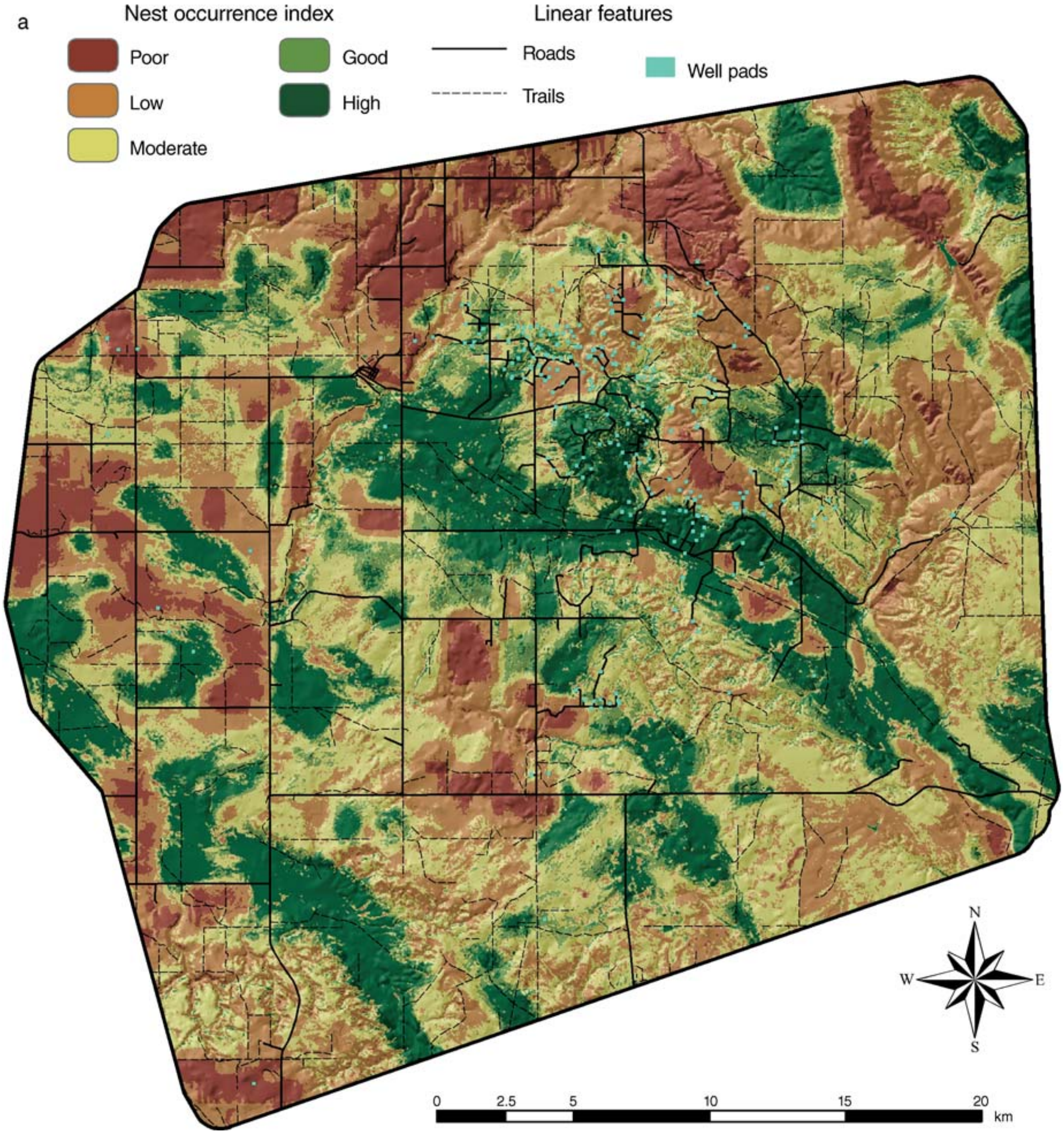

FIG. 3. Relative index of Sage-Grouse (a) nest and (b) brood occurrence in southeastern Alberta, as determined by logistic-regression occurrence models. Good and high index values indicate that Sage-Grouse are likely to use these habitats for nests or brood-rearing, respectively.

(2) low, (3) moderate, (4) high, and (5) extreme risk of failure. We used these occurrence and risk indices to identify five different habitat states, similar to the methods of Nielsen et al. (2006). Firstly, occurrence bins ranking from poor to moderate (1–3) were classified as overall low use, and it was assumed that Sage-Grouse would be unlikely to occur in those habitats, although we tested this with validation data. We refer to bin 5 as primary habitat and bin 4 as secondary habitat, based on the relative probability of use of resource units in these bins. We overlaid the respective nest or chick survival model predictions on the occurrence maps in our GIS to identify the habitat states. Primary and secondary occurrence habitats falling in areas of moderate-to-extreme risk (bins 3–5) were classified as attractive sink habitats, broken into primary and secondary sinks, respectively. Similarly, habitats with low risk (bins 1–2) but high occurrence (occurrence bins

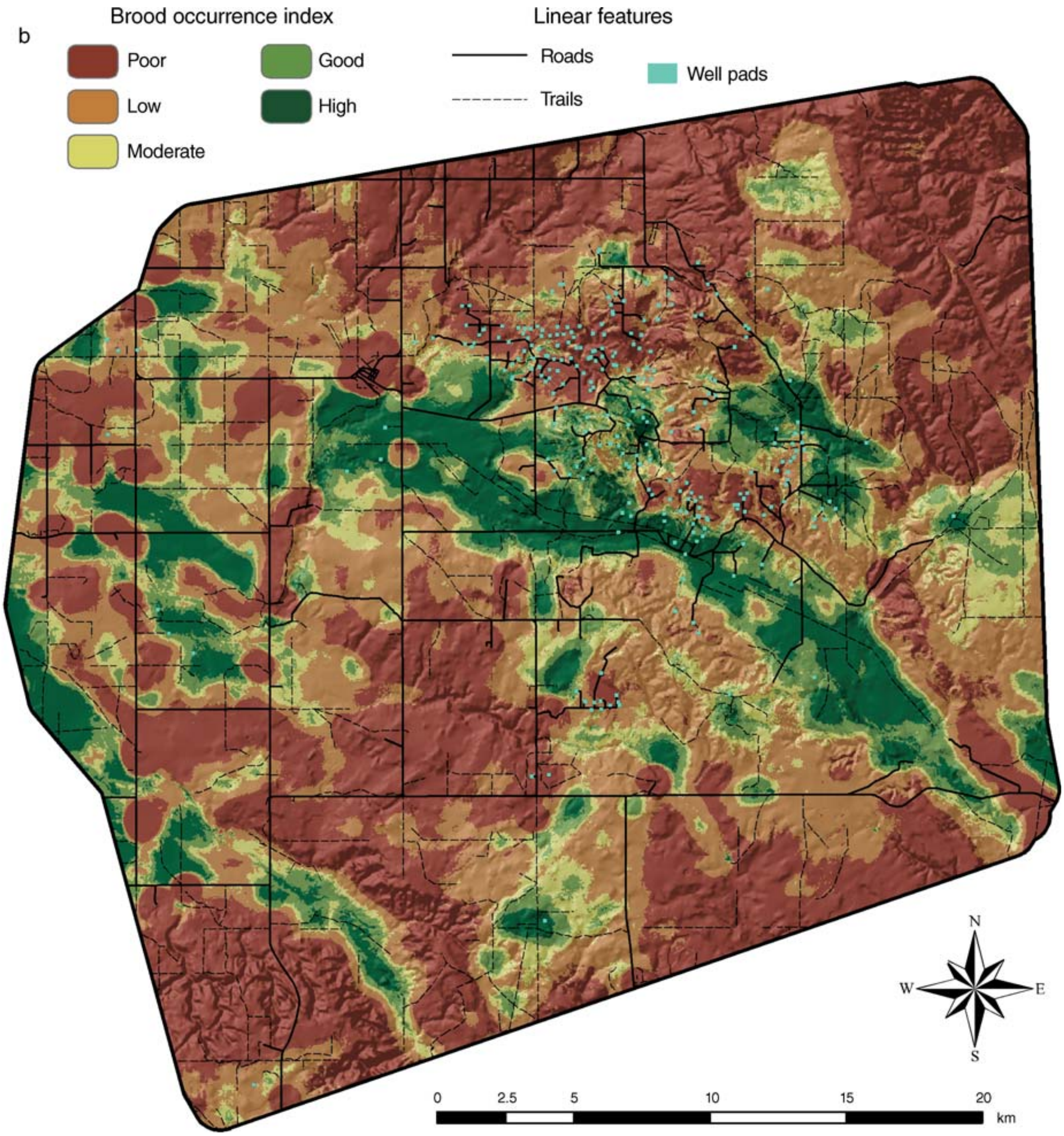

FIG. 3. Continued.

5 and 4) were considered primary or secondary source habitat. We graphically illustrate these conceptual habitat classes in Fig. 2 and develop maps depicting these habitat states for nesting and brood-rearing habitats within each habitat state.

RESULTS

From 2001 to 2004, we located 113 Sage-Grouse nests for occurrence modeling (two nests were from unmarked females). Nest survival/success over the 28 day incubation period was 39.4% ± 4.84% for 111 nests produced by 61 radio-marked females (all values reported as mean ± SE). With only five of 111 nests produced by yearlings, we were precluded from testing for age effects. There was no difference in nest survival among years of our study (log rank $\chi^2_3 = 5.50$, $P = 0.14$) and there was no difference in survival between initial (40.2% ± 5.7%, $n = 77$) and second nesting attempts (37.5% ± 9.0%, $n = 34$; log rank $\chi^2_1 = 0.07$, $P = 0.79$), allowing us to combine all nests when modeling survival.

From 2001 to 2004, we identified a total of 669 brood locations from 35 Sage-Grouse broods (19.11 ± 0.60 locations/brood), which we used to model brood occurrence. From 2001 to 2003, we radio-marked 41

TABLE 2. Estimated coefficients (β_i) and standard errors for the final nest occurrence model for 113 Sage-Grouse nests in southeastern Alberta from 2001 to 2004.

Variable	β_i	SE	P
Brit	−0.0215	0.0082	0.009
SBmean	0.1025	0.0401	0.011
SBmean^2	−0.0014	0.0007	0.047
pSB_pch2	1.5251	0.7602	0.045
pEco6	−3.0573	0.9654	0.002
pEdge	−2.8002	1.3531	0.038

Notes: To characterize habitat availability, 5000 random points were used; these points were weighted using importance weights such that the available sample was effectively 113 points. P values indicate the significance of the coefficients using a Wald z statistic.

chicks from 22 different broods. Chick survival to 56 days using the shared frailty proportional hazards model was 12.3% and there was significant correlation (at $\alpha = 0.10$) in the fate of chicks within broods ($\theta = 0.96$, $P = 0.086$).

Nest occurrence

Our stepwise modeling approach resulted in a final nest occurrence model that contained six parameters (Table 2); no interactions were significant. This model had good fit (likelihood ratio $\chi^2_6 = 53.62$, $P < 0.0001$). Sage-Grouse showed strong avoidance of badland habitats ($\beta_{\mathrm{pEco6}} = -3.0573$), areas with a high proportion of anthropogenic edge habitats ($\beta_{\mathrm{pEdge}} = -2.8002$), and areas with greater brightness values ($\beta_{\mathrm{Brit}} = -0.0212$). Conversely, Sage-Grouse selected nesting habitat that contained large patches (1 km^2) of moderate sagebrush cover (quadratic or concave relationship; $\beta_{\mathrm{SBmean}} = 0.1025 + \beta^2_{\mathrm{SBmean}} = -0.0014$), but where the distribution of sagebrush within these patches was heterogeneous ($\beta_{\mathrm{pSB\_patch2}} = 1.5251$; Table 2).

When we applied this model to the study area (Fig. 3a) and mapped the five habitat bins, only 30% of the landscape was considered to have a good-to-high likelihood of Sage-Grouse nesting there. However, the majority of nests (72% of training nests and 65% of validation nests) occurred within the good-to-high habitat bins, indicating that lower ranked habitat bins were used less frequently. Both the nests ($n = 113$ nests) that we used to build the model (years 2001–2004) and the validation sample (years 1998–2000, $n = 40$ nests) showed an increasing frequency (area-adjusted) of occurrence within the predicted nest index bin (training data: $r_S = 1.00$, $P < 0.0001$; testing data: $r_S = 1.00$, $P < 0.0001$), suggesting that the RSF for nest occurrence was approximately proportional to probability of use.

Brood occurrence

After stepwise removal of variables, the final brood occurrence model contained 15 significant variables with no interaction terms. This model had good fit (likelihood ratio $\chi^2_{15} = 583.32$, $P < 0.0001$). Similar to the nest occurrence model, hens with broods selected for large patches (1 km^2) of moderate sagebrush cover (quadratic; $\beta_{\mathrm{SBmean}} = 0.10445 + \beta^2_{\mathrm{SBmean}} = -0.0010$) that contained a patchy distribution of sagebrush ($\beta_{\mathrm{pSB\_patch2}} = 1.7924$; Table 3). Selection was strong for mesic habitats, selecting for higher wetness values ($\beta_{\mathrm{Wet}} = 0.0217$) and higher mean CTI scores ($\beta_{\mathrm{CTImean}} = 0.4835$), while avoiding high brightness values ($\beta_{\mathrm{Brit}} = -0.0076$; Table 3). Broods avoided habitats associated with a high density of urban developments ($\beta_{\mathrm{pUrban}} = -64.9741$), areas close to cultivated cropland ($\beta_{\mathrm{Crop\_dist}} = 0.1525$), and habitats composed largely of ecosite plant community types in bins 4 (loamy upland sites), 5 (thin break sites), and 6 (badland sites; Table 3). Sage-Grouse broods tended to occur in areas with a greater density of trails ($\beta_{\mathrm{Tr\_dens}} = 0.2336$) and were closer to water impoundments than random ($\beta_{\mathrm{Imp\_dist}} = -0.6305$; Table 3). Broods tended to be closer to well sites ($\beta_{\mathrm{Well\_dist}} = -0.4087$), but at the same time, they avoided areas with a greater density of visible well sites within 1 km ($\beta_{\mathrm{vWell\_1km}} = -0.2016$; Table 3).

We applied this 15-parameter brood occurrence model to the study area (Fig. 3b), binning habitats from poor to high occurrence. Only 20% of habitat fell within good-to-high habitat occurrence, but the majority of brood locations (77% of training points and 71% of testing points) fell within the good-to-high habitat, suggesting that our relative bin ranks capture brood occurrence across the landscape. The brood occurrence model was predictive, with the area-adjusted frequency of occurrence increasing with increasing bin rank; for 669 model training locations, $r_S = 1.00$, $P < 0.0001$; for 151 validation brood locations, $r_S = 1.00$, $P < 0.0001$.

Nest survival

The final nest survival model contained three variables (Table 4). Nest failure was independent of human-

TABLE 3. Estimated coefficients (β_i) and standard errors (SE) for the final brood occurrence model for 669 Sage-Grouse brood locations in southeastern Alberta from 2001 to 2004.

Variable	β_i	SE	P
Brit	−0.0076	0.0032	0.018
Wet	0.0217	0.0088	0.013
CTI_mean	0.4835	0.0872	<0.001
Well_dist	−0.4087	0.0446	<0.001
vWell_1km	−0.2016	0.0591	0.001
Tr_dens	0.2336	0.0887	0.008
Imp_dist	−0.6305	0.2134	0.003
SBmean	0.1044	0.0175	<0.001
SBmean^2	−0.0010	0.0003	<0.001
pSB_pch2	1.7924	0.3703	<0.001
Crop_dist	0.1525	0.0339	<0.001
pUrban	−64.9741	18.2819	<0.001
pEco4	−1.2791	0.3625	<0.001
pEco5	−2.1208	0.3368	<0.001
pEco6	−1.8744	0.4931	<0.001

Notes: To characterise habitat availability, 5000 random points were used; these points were weighted using importance weights such that the available sample was effectively 669 points. P values indicate the significance of the coefficients using a Wald z statistic.

TABLE 4. Estimated hazard ratios (exponentiated coefficients, exp[β_i]) and standard errors for the final proportional hazards nest survival model using 111 Sage-Grouse nest sites in southeastern Alberta from 2001 to 2004.

Variable	β_i	SE	P
NDVI_sd	10.9×10^{-8}	9.44	0.034
SB	1.0138	0.0052	0.007
pSBpch1	0.2862	0.1784	0.045

Note: P values indicate the significance of the coefficients using a Wald z statistic.

use features. Nest failure was greatly reduced in habitats that contained a heterogeneous mix of sagebrush cover ($\beta_{\mathrm{SBpch1}} = 0.2862$; Table 4). However, there was a slight increase in risk as sagebrush cover in the immediate vicinity of the nest site increased ($\beta_{\mathrm{SB}} = 1.0138$; Table 4). As the variability in NDVI increased (NDVI_sd), risk of failure decreased significantly ($\beta_{\mathrm{NDVI\_sd}} = 10.9 \times 10^{-8}$; Table 4).

Although the final nest survival model had good fit (likelihood ratio$\chi^2_3 = 12.94$, $P < 0.005$), it had moderate-to-low predictive accuracy ($\mathrm{ROC_{train}} = 0.67$; $\mathrm{ROC_{test}} = 0.59$) and low predictive capacity ($\mathrm{PCC_{train}} = 60.4\%$; $\mathrm{PCC_{test}} = 55.3\%$). Using the cumulative daily relative hazard, however, failed nests were exposed to more risky habitats for training data set ($t_{102.05} = 3.52$, $P < 0.001$), but this model had difficulty detecting failures using the independent sample of 40 nests (22 failures; $t_{24.50} = 0.82$, $P = 0.21$). When we applied this final nest survival model to the landscape, ~60% of habitat occurred within the moderate-to-extreme risk categories, in which we predict Sage-Grouse nests are likely to fail (Fig. 4a).

Chick survival

For the chick survival model, no variables were significant ($\alpha = 0.05$) after sequential removal. However, the last two variables removed were significant at $\alpha = 0.10$ ($\beta_{\mathrm{CTI}} = 1.1883$; $\beta_{\mathrm{vWell\_1km}} = 1.5219$; Table 5) and we used these in the final model, given small chick sample size (24 failures of 41 chicks). Based on these parameters, chick failure increased in habitats with a higher visible well site density within 1 km, and surprisingly, risk was also greater in habitats with higher CTI values. Model fit was moderate (Wald $\chi^2_2 = 5.74$, $P < 0.057$), predictive accuracy ($\mathrm{ROC_{train}} = 0.67$) was low, but classification accuracy ($\mathrm{PCC_{train}} = 70.7\%$) was good. Using only these two parameters, our model accurately identified chicks that failed as being exposed to more risky habitats, having higher cumulative daily relative hazard rates ($t_{38.39} = 3.03$, $P = 0.002$), but we had no independent sample for validation. When we applied this model to the landscape (Fig. 4b), areas with greater oil and gas activities fell into the extreme risk category, but the majority of the riparian areas (linear sections with high CTI values) were also identified as risky habitats. About 60% of habitat was identified as risky for Sage-Grouse chicks.

Nest habitat states

Of the 30% of the landscape that we identified as having a good-to-high likelihood of being used as nesting habitat, over half of this habitat (19% of the landscape) occurs in high-risk areas, with 11.6% of habitat classified as a primary sink and 7.4% classified as secondary sink nesting habitat (Fig. 5a). Only a small portion of the landscape is primary nesting habitat (8.4%), with just 2.6% of habitat considered secondary habitat. Primary nesting habitat averaged 5.83 ± 0.12 km (mean ± SE) from active leks in Alberta, and secondary habitat was 6.77 ± 0.22 km. The cumulative percentage of source nesting habitat increases linearly up to about 10 km, where it asymptotes and a threshold is reached, encompassing about 90% of all source habitats (Fig. 6a).

Brood habitat states

Our brood occurrence maps indicated that there is limited habitat available (20% good-to-high occurrence class) for Sage-Grouse brood-rearing. In addition, three-quarters of available habitat (15% of the landscape) is high risk and classified as habitat sinks (Fig. 5b); only 5% is source brood-rearing habitat (primary plus secondary habitat; Fig. 5b). Primary and secondary brood-rearing habitats averaged 4.52 ± 0.16 km, and 6.21 ± 0.17 km from the nearest active lek, respectively. Similar to nesting habitat, ~90% of all source brood-rearing habitats occur within ~10 km of all active lek sites (Fig. 6b).

DISCUSSION

Our landscape-scale models indicate a limited supply of habitats selected by Sage-Grouse (good-to-high occurrence bins), with about 30% of the habitat likely to be used for nesting (Fig. 3a) and 20% for brood-rearing (Fig. 3b). Over half of that 30% identified as attractive nesting habitat (19% of the landscape) is considered risky (moderate-to-extreme risk; Fig. 5a) causing an ecological trap (Delibes et al. 2001, Kristan 2003). Therefore, more than half of the nesting habitat used by Sage-Grouse will not result in successful nesting attempts (Fig. 5a), even though Sage-Grouse still occupy those habitat patches. An even greater threat to recruitment and population persistence may be the brood habitat ecological trap, with three-quarters of

TABLE 5. Estimated hazard ratios (exponentiated coefficients, exp[β_i]) and standard errors (SE) for the shared frailty final proportional hazards chick survival model using 41 Sage-Grouse chicks from 22 different broods in southeastern Alberta from 2001 to 2003.

Variable	β_i	SE	P
CTI	1.1883	0.1145	0.073
vWell_1km	1.5219	0.3437	0.063

Notes: P values indicate the significance of the coefficients using a Wald z statistic. The shared frailty variance estimate is $\theta = 0.96$, $P = 0.086$.

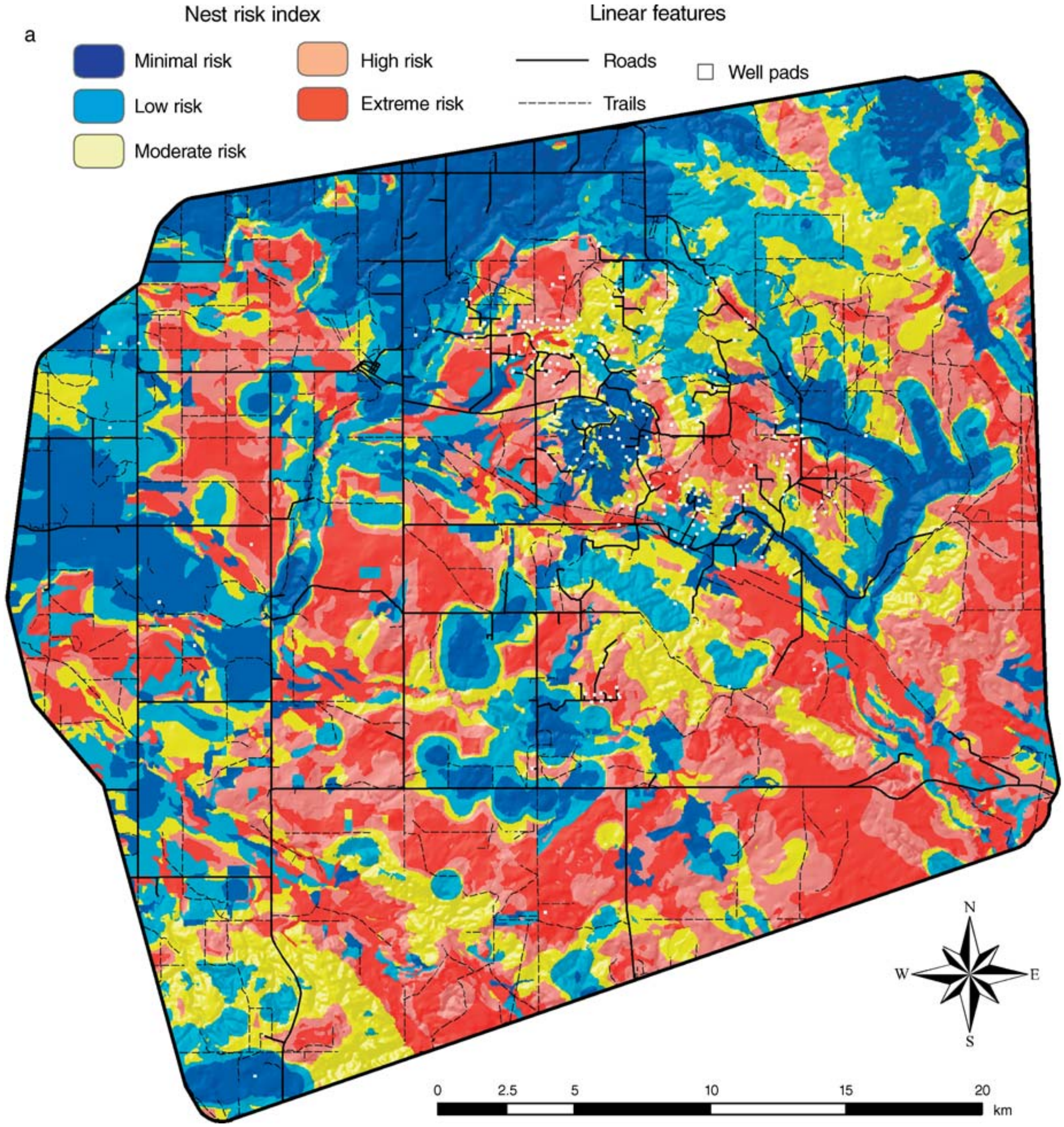

FIG. 4. Relative index of risk for Sage-Grouse (a) nest failure and (b) chick failure in southeastern Alberta, as determined by Cox proportional hazards modeling of survival. High and extreme risk values indicate that a nest is likely to fail or a chick is likely to die if it occurs in these habitats.

the attractive brood habitat (15% of the landscape out of the 20% considered attractive) likely to result in chick failure (Fig. 4b). Low nest success (39%; Sage-Grouse range 15–86%; Schroeder et al. [1999]), and poor chick survival (12%) are driven by an abundance of attractive sink habitats where Sage-Grouse have poor recruitment. Our approaches not only spatially identify habitats with poor fitness, which ultimately drive population dynamics (Van Horne 1983, Morrison 2001), but also address mechanisms driving declines.

Nesting habitat

Consistent with our predictions for nest occurrence and previous research at finer scales (Aldridge 2005), nests were more abundant in habitat patches (within a 1 km^2 area) with moderate sagebrush cover. Selection was also strong for large patches (1 km^2) that contained a heterogeneous distribution of sagebrush cover, with continuous and sparsely distributed sagebrush habitats used less than expected by chance. Sage-Grouse select locally for greater herbaceous understory cover and our

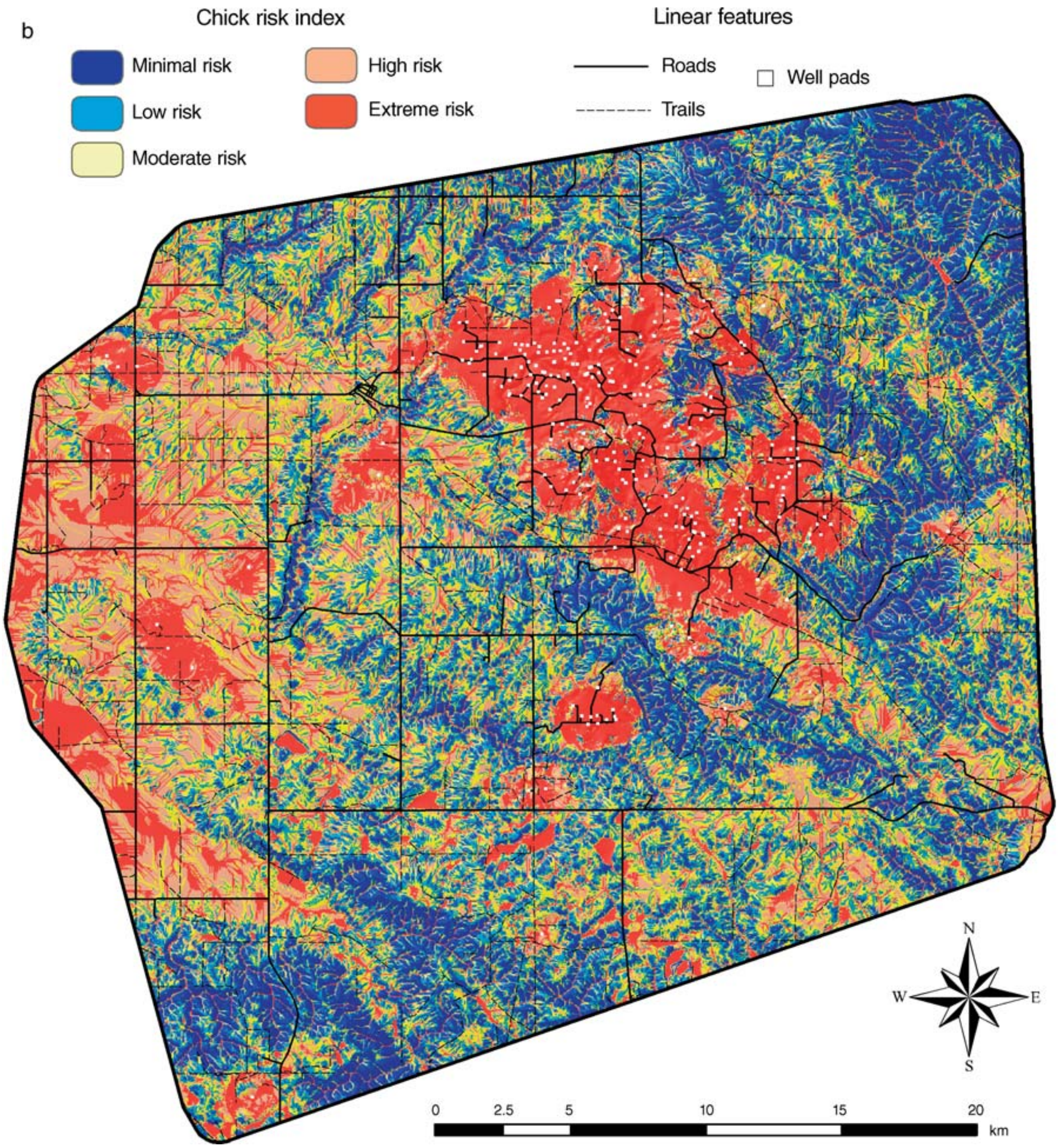

FIG. 4. Continued.

landscape models identified coarse-scale correlates for habitat that lack this understory cover. Moderate cover and patchy distributions are likely to provide suitable overstory shrub cover while allowing for the lateral herbaceous cover required to conceal nests from predators (Wallestad and Pyrah 1974, Wiebe and Martin 1998, Aldridge and Brigham 2002). Nest abundance was lower in habitats with high brightness values, suggesting that habitats with increased bare ground were avoided. This idea is reinforced by the apparent avoidance of less productive badland habitats that contain steep and dry, exposed soils (Adams et al. 2005).

As predicted, nest failure was lower in habitats that contained a heterogeneous mix of sagebrush cover ($\beta_{SBpch1} = 0.2862$), with limited or continuous dense cover resulting in nest failure (Table 4). Conceivably, this may explain the slight increase in risk with increasing sagebrush cover in the immediate vicinity of the nest (linear increase; $\beta_{SB} = 1.0138$; Table 4). Risk also was significantly reduced for increasing NDVI_sd measures. The NDVI index values were small, ranging from 0.012 to 0.099. Taking the natural logarithm of the unexponentiated β coefficient ($\beta_{NDVI\_sd} = -18.33$) times an increase in the NDVI_sd index values of 0.01 (~10%

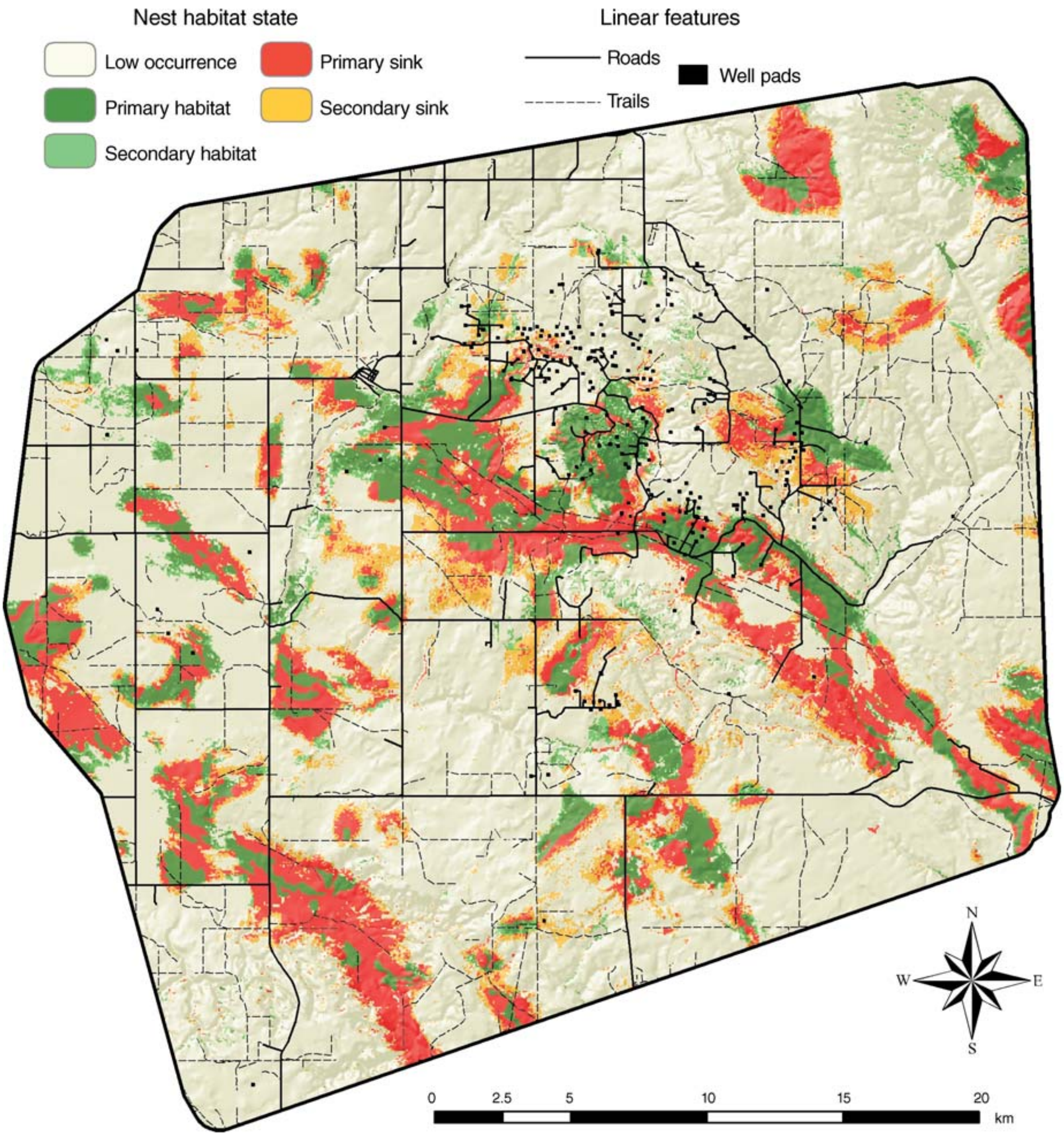

FIG. 5. Habitat states for Sage-Grouse (a) nest and (b) brood habitat in southeastern Alberta. Noncritical habitat indicates that Sage-Grouse are not likely to occur there. "Primary" and "secondary" indicate high and good likelihood of occurrence, respectively. "Habitats" are areas with minimal-to-low risk of failure, whereas "sinks" are areas with moderate-to-extreme risk. For example, primary habitat indicates areas where nests or broods are likely to occur (high occurrence values) and to be successful or survive (minimal-to-low risk values). Primary sink indicates high occurrence, where nests or broods are likely to fail or die (moderate-to-extreme risk values).

of value range) indicates that nest survival would increase by ~17% ($\exp[-18.33 \times 0.01] = 0.833$). Thus, more diverse, heterogeneous habitats reduced the risk of nest failure, as indicated by the small hazard ratio for the NDVI variability measure (Table 4).

Although the proportion of human-use features did not enter into our final nest occurrence model, when roads, well sites, urban habitats, and cropland were combined into one parameter (pEdge), Sage-Grouse strongly avoided nesting in these edge-habitat dominated landscapes. Hens may be responding to increased predator densities associated with edge-type habitats (Andrén and Angelstam 1988, Herkert et al. 2003) and agricultural landscapes (Andrén 1992, Kurki et al. 2000, Fuhlendorf et al. 2002, Manzer and Hannon 2005). However, like others (Pasitschniak-Arts and Messier

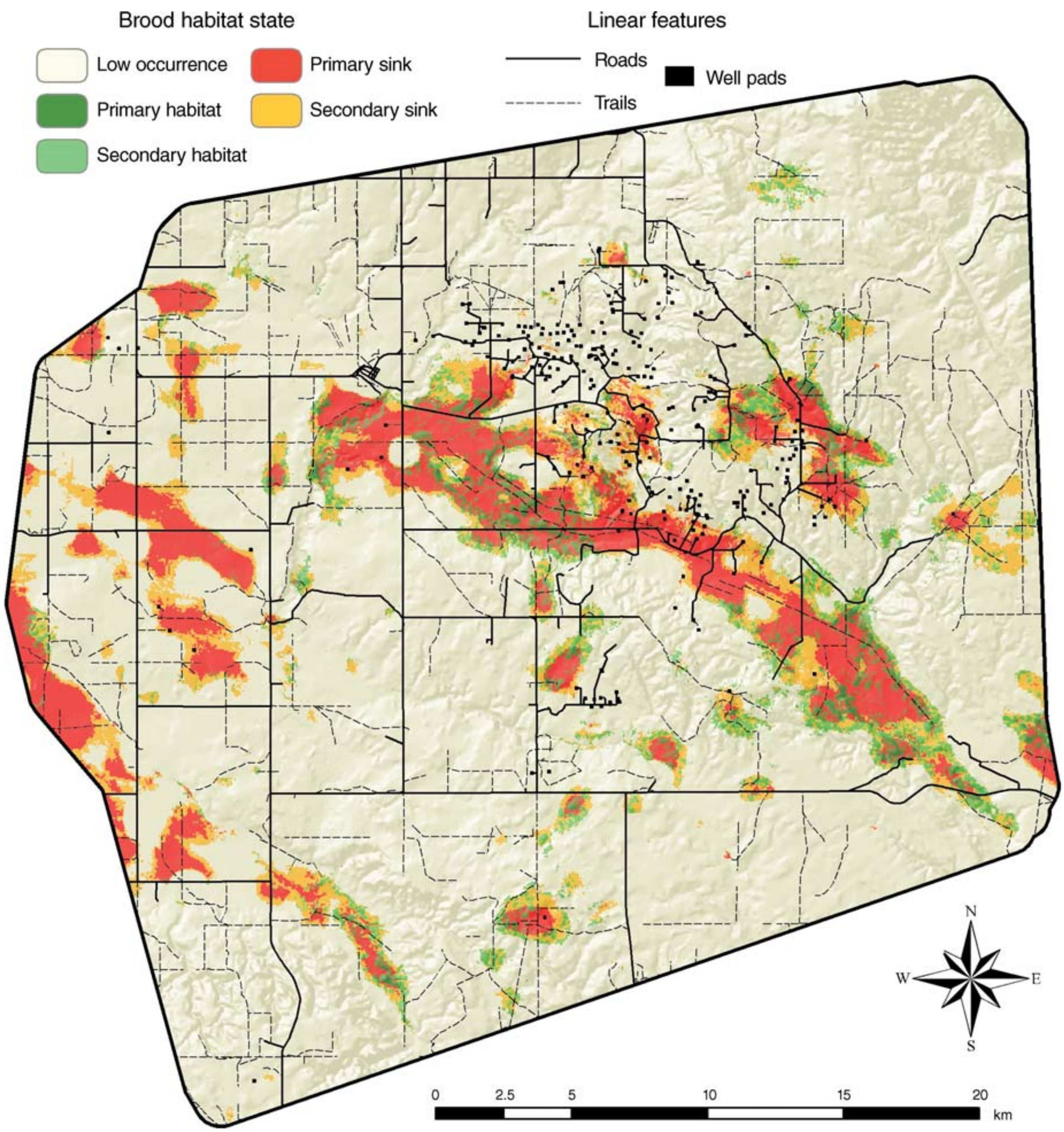

FIG. 5. Continued.

1995, Svobodova et al. 2004), we found no effect of edge habitats, or other human features, on Sage-Grouse nest success (Table 4). Nest placement for Lesser Prairie-Chickens (*Tympanuchus pallidicinctus*) in Kansas, USA was farther from paved roads than at random (Pitman 2003). In the same area, proximity to human structures greatly reduced habitat suitability, whereas roads had no obvious effect (Hagen 2003). Recent work on Sage Grouse in Wyoming, USA (Lyon and Anderson 2003, Holloran 2005) suggests that oil and gas activities within 5 km of lek sites results in sharp declines in male attendance, and avoidance by nesting females. However, Lyon and Anderson (2003) found no difference in nest success between disturbed and control leks. In our study, the mean percentage of edge habitat within a 1-km^2 window around nest sites was 2.9% ± 0.7%, compared to a mean of 10.1% ± 0.3% (mean ± SE) across the landscape. Females' strong avoidance of edge habitats ($\beta_{pEdge} = -2.80$) probably prevented us from being able to detect differences in nest success relative to these features.

Ecological traps tend to be more prevalent in human-dominated landscapes (Remes 2000, Bock and Jones 2004), where birds fail to recognize risks with which they

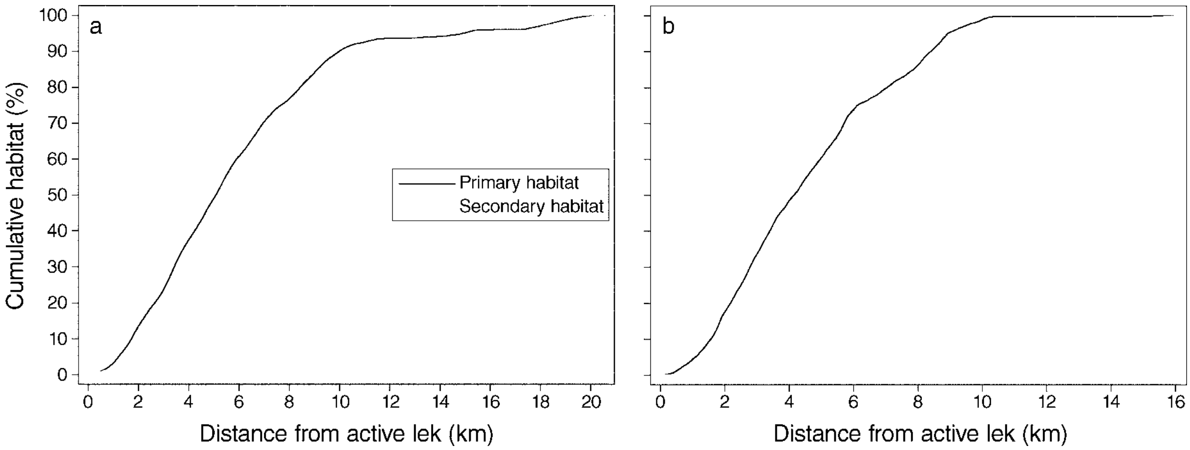

FIG. 6. Primary and secondary source (a) nest and (b) brood habitat for Sage-Grouse in southeastern Alberta, shown as a function of the distance from an active lek.

did not evolve. Sage-Grouse, however, might recognize some of these habitats as risky, avoiding potential ecological traps created in human-dominated habitat patches; at least when selecting nesting habitat. This does not mean that human features have no ill effects on nesting Sage-Grouse. Avoidance of human features removes that habitat patch from use by Sage-Grouse, and effectively removes habitat within a 1 km^2 area (functional habitat loss). This zonal-habitat influence may be greater, but we did not test the effect of edge habitat density in windows > 1 km^2. Even though Sage-Grouse might recognize and avoid these anthropogenic threats, half of all high-use nesting (good-to-high rank) habitats is considered attractive sinks (Fig. 5a), ecological traps driven by habitat features. We suggest that our habitat maps be used to identify risky nesting habitats and that managers should focus efforts at improving nest success by enhancing sagebrush cover above the currently available 5–10% cover, following recommended habitat guidelines (Connelly et al. 2000), while establishing a heterogeneous mix of sagebrush patches. Management of local range conditions (Crawford et al. 2004; see Aldridge 2005) aimed at enhancing grass and forb understory that improves visual obstruction cover in these risky nesting areas probably will be required to convert sinks into source-type habitats. Range conditions should be assessed locally and grazing could be used to adaptively manage and enhance these habitats (Aldridge et al. 2004). For instance, removing cattle or reducing grazing intensity in some areas may result in increased shrub cover and/or plant species diversity (Manier and Hobbs 2006).

Brood habitat

As predicted, Sage-Grouse also selected for moderate ranges of sagebrush cover at brood-rearing sites. Brood occurrence was greater in more heterogeneous sagebrush stands, where patchy cover reduces predator efficiency (Wiebe and Martin 1998) but still affords necessary forb resources. Sage-Grouse are more abundant in patchy habitats containing a mix of mesic, forb-rich foraging areas interspersed within suitable sagebrush escape cover (Boyce 1981).

Brooding hens appeared to avoid areas closer to cultivated cropland or with a greater proportion of urban developments. Although Sage-Grouse may forage regularly on alfalfa (Patterson 1952), or occasionally on insects found in other cereal crops, they typically do not occur in cultivated lands or landscapes heavily dominated by agriculture. Cultivation directly removes habitats and is correlated with Sage-Grouse population declines in Idaho, USA (Leonard et al. 2000).

In some cases, Sage-Grouse broods occurred close to well sites, but not often in areas with high well densities (Fig. 3b). This relationship may partially be due to the static 2002 distribution of well sites for our GIS landscape, as energy developments have increased slightly over time. However, Holloran (2005) similarly found that nest sites occurred closer to well sites in areas of lower well density. Disturbed habitats, such as trails and well pads, tend to harbor succulent invasive species such as dandelions (*Taraxacum officinale*), important forage to which Sage-Grouse are attracted. Despite this attraction, our chick survival model predicts a 1.5 times increase in risk for each additional oil well that is visible within 1 km of brood locations (see Fig. 4a). As a result, a significant portion of frequently used brood habitat is classified as attractive sink habitats (see Fig. 5b), suggesting that Sage-Grouse may only partially recognize some ecological cues related to anthropogenic features. Birds are run over by vehicles accessing these wells (C. L. Aldridge, *unpublished data*), and are killed by raptorial predators, such as Golden Eagles (*Aquila chrysaetos*) and Great Horned Owls (*Bubo virginianus*), that perch on the power lines leading to well sites. Regardless of the mechanism, chicks have a low probability of survival, which is further reduced when energy extraction activities dominate the landscape.

Sage-Grouse broods also avoided the less productive and more exposed badland range plant community habitats (pEco6), as well as thin-break range sites (pEco5) and the loamy upland sites (pEco4; Table 3). The thin-break sites are similar to badland habitats, but contain greater sagebrush cover, and the loamy upland sites are more productive range sites, but are dominated by various grasses, resulting in a lack of shrubs and forbs (Adams et al. 2005). Although these two sites might provide added cover from either sagebrush or dense grass cover, they lack the forb component required by Sage-Grouse broods.

More mesic habitats were selected by broods, with occurrence being associated with lower brightness values and higher mean CTI and wetness values (Table 3). These habitats are probably required for birds to meet dietary requirements, because forb (Drut et al. 1994*a*, Sveum et al. 1998*a*) and insect (Johnson and Boyce 1991, Drut et al. 1994*b*) abundance is higher. Hens also chose to be closer to water impoundments. The effect of altered water hydrology on the vegetation productivity, composition, and distribution within this xeric ecosystem is unknown. Removing some of these impoundments may allow water to recharge former mesic sites, rather than retain water behind a dam or within a dugout.

Although mesic habitats were selected, higher CTI values resulted in increased chick failure. Excluding the high-risk values associated with greater well-site densities (Fig. 4b), the majority of other high-CTI risky habitats occurred in riparian habitats along creeks and streams. These habitats are not frequently used by Sage-Grouse broods (see Fig. 3b), but there may be increased risk associated with these shrubby riparian corridors, which often contain a greater concentration of predators (Wilcove 1985). Aldridge (2005) showed that, at local scales, mesic, forb-rich habitats preferred by Sage-Grouse broods tend occur in more risky open habitats. Sage-Grouse may be making trade-offs between habitats that provide protective escape cover and risky open, mesic habitats that provide necessary forage resources. Recent droughts resulting in reduced cover could have made these habitats even more risky for Sage-Grouse chicks, particularly if livestock grazing intensities were not subsequently reduced. Relationships among water impoundments, drought conditions, and the availability of mesic brood habitats are poorly understood (Crawford et al. 2004) and need to be investigated within a long-term adaptive management framework (Aldridge et al. 2004).

Conclusions

For most prairie grouse species, the lek is often thought of as the focal point for year-round activities. Much research has focused on maintaining required habitats surrounding leks and attempting to identify links between habitat alterations and lek dynamics (Wakkinen et al. 1992, Niemuth 2000, Fuhlendorf et al. 2002, Niemuth and Boyce 2004). However, our approach of modeling and mapping high-quality nesting and brood-rearing habitats suggests that such a heavy focus on habitat protection around lek sites may not be suitable to ensure the viability of Sage-Grouse populations. Both nest and brood source habitats, on average, are ~6 km from active leks, but the curvilinear relationship (Fig. 6) suggests that a threshold occurs at ~10 km from leks, within which the majority (~90%) of all source habitats occur. Thus, using a fixed buffer distance around leks of <10 km to protect Sage-Grouse habitat may not suitably protect important nesting and brood-rearing habitats. Wakkinen et al. (1992) suggested that the originally recommended 3.2-km buffer around leks (Braun et al. 1977) may not be large enough to protect nesting habitats, and Connelly et al. (2000) suggested that polygons of 5 km and 18 km may be required to protect breeding habitats for nonmigratory and migratory populations, respectively. The province of Alberta uses a 1-km protection buffer around lek sites (see Alberta Provincial Government web site, a*vailable online*).[2] Complete protection of all areas within this buffer would protect <5% of the available source nesting and brood-rearing habitat identified by our models, which is unlikely to sustain this population. The buffer approach to habitat management and protection could easily result in important habitats being left unprotected and noncritical habitats being protected.

We see our empirically based modeling approach as a framework for identifying and protecting important source nesting and brood-rearing habitats for Sage-Grouse. We identify key sink habitats, which provide managers with the ideal opportunity to evaluate management alternatives aimed at increasing productivity through habitat management following an adaptive management framework (Aldridge et al. 2004), using these models as the baseline habitat accounting system for assessments and future monitoring for Sage-Grouse in Alberta. Careful attention still needs to be given to managing for other seasonal habitat requirements, such as lekking, summer, and winter habitat, and connectivity between habitats. We see great utility in applying our habitat states modeling approach to population viability assessments for many species across different ecological systems.

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[2] ⟨http://www.srd.gov.ab.ca/fw/landuse/pdf/GrasslandParkland.pdf⟩

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Literature Cited

Adams, B. W., L. Poulin-Klein, D. Moisey, and R. L. McNeil. 2005. Rangeland plant communities and range health assessment guidelines for the dry mixedgrass natural subregion of Alberta. Publication Number T/040, Rangeland Management Branch, Public Lands Division, Alberta Sustainable Resource Development, Lethbridge, Edmonton, Alberta, Canada.

Akçakaya, H. R., V. C. Radeloff, D. J. Mlandenoff, and H. S. He. 2004. Integrating landscape and metapopulation modeling approaches: viability of the sharp-tailed grouse in a dynamic landscape. Conservation Biology 18:526–537.

Aldridge, C. L. 2005. Habitats for persistence of greater sage-grouse (*Centrocercus urophasianus*) in Alberta, Canada. Dissertation. University of Alberta, Edmonton, Alberta, Canada.

Aldridge, C. L., M. S. Boyce, and R. K. Baydack. 2004. Adaptive management of prairie grouse: how do we get there? Wildlife Society Bulletin 32:92–103.

Aldridge, C. L., and R. M. Brigham. 2001. Nesting and reproductive activities of Greater Sage-Grouse in a declining northern fringe population. Condor 103:537–543.

Aldridge, C. L., and R. M. Brigham. 2002. Sage-Grouse nesting and brood habitat use in southern Canada. Journal of Wildlife Management 66:433–444.

Aldridge, C. L., and R. M. Brigham. 2003. Distribution, abundance, and status of the greater sage-grouse, *Centrocercus urophasianus*, in Canada. Canadian Field-Naturalist 117:25–34.

Andersen, P. K., and R. D. Gill. 1982. Cox regression-model for counting-processes: a large sample study. Annals of Statistics 10:1100–1120.

Andrén, H. 1992. Corvid density and nest predation in relation to forest fragmentation: a landscape perspective. Ecology 73: 794–804.

Andrén, H., and P. Angelstam. 1988. Elevated predation rates as an edge effect in habitat islands: experimental evidence. Ecology 69:544–547.

Battin, J. 2004. When good animals love bad habitats: ecological traps and the conservation of animal populations. Conservation Biology 18:1482–1491.

Beck, J. L., and D. L. Mitchell. 2000. Influences of livestock grazing on sage grouse habitat. Wildlife Society Bulletin 28: 993–1002.

Bock, C. E., and Z. F. Jones. 2004. Avian habitat evaluation: should counting birds count? Frontiers in Ecology and the Environment 2:403–410.

Boyce, M. S. 1981. Robust canonical correlation of sage grouse habitat. Pages 152–159 *in* D. E. Capen, editor. The use of multivariate statistics in studies of wildlife habitat. Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado, USA.

Boyce, M. S., and L. L. McDonald. 1999. Relating populations to habitats using resource selection functions. Trends in Ecology and Evolution 14:268–272.

Boyce, M. S., J. S. Meyer, and L. Irwin. 1994. Habitat-based PVA for the northern spotted owl. Pages 63–85 *in* D. J. Fletcher and B. F. J. Manly, editors. Statistics in ecology and environmental monitoring. University of Otago Press, Dunedin, New Zealand.

Boyce, M. S., P. R. Vernier, S. E. Nielsen, and F. K. A. Schmiegelow. 2002. Evaluating resource selection functions. Ecological Modelling 157:281–300.

Braun, C. E., T. Britt, and R. O. Wallestad. 1977. Guidelines for maintenance of sage grouse habitats. Wildlife Society Bulletin 5:99–106.

Braun, C. E., O. O. Oedekoven, and C. L. Aldridge. 2002. Oil and gas development in western North America: effects on sagebrush steppe avifauna with particular emphasis on sage grouse. Transactions of the North American Wildlife and Natural Resources Conference 67:337–349.

Breininger, D. R., V. L. Larson, B. W. Duncan, and R. B. Smith. 1998. Linking habitat suitability to demographic success in Florida scrub-jays. Wildlife Society Bulletin 26: 118–128.

Burkepile, N. A., J. W. Connelly, D. W. Stanley, and K. P. Reese. 2002. Attachment of radiotransmitters to one-day-old sage grouse chicks. Wildlife Society Bulletin 30:93–96.

Chatterjee, S., A. S. Hadi, and B. Price. 2000. Regression analysis by example. John Wiley, New York, New York, USA.

Cleves, M. A., W. W. Gould, and R. G. Gutierrez. 2004. An introduction to survival analyses using STATA. Stata Press, College Station, Texas, USA.

Connelly, J. W., and C. E. Braun. 1997. Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. Wildlife Biology 3:229–234.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming, USA.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967–985.

Coupland, R. T. 1961. A reconsideration of grassland classification in the northern Great Plains of North-America. Journal of Ecology 49:135–167.

Cox, D. R. 1972. Regression models and life-tables. Journal of the Royal Statistical Society Series B, Statistical Methodology 34:187–220.

Crawford, J. A., R. A. Olson, N. E. West, J. C. Mosley, M. A. Schroeder, T. D. Whitson, R. F. Miller, M. A. Gregg, and C. S. Boyd. 2004. Synthesis paper: ecology and management of sage-grouse and sage-grouse habitat. Journal of Range Management 57:2–19.

Crist, E. P., and R. C. Cicone. 1984. Application of the tasseled cap concept to simulated thematic mapper data. Photogrammetric Engineering and Remote Sensing 50:343–352.

Delibes, M., P. Gaona, and P. Ferreras. 2001. Effects of an attractive sink leading into maladaptive habitat selection. American Naturalist 158:277–285.

Donovan, T. M., and F. R. Thompson. 2001. Modeling the ecological trap hypothesis: a habitat and demographic analysis for migrant songbirds. Ecological Applications 11: 871–882.

Drut, M. S., J. A. Crawford, and M. A. Gregg. 1994*a*. Brood habitat use by sage grouse in Oregon. Great Basin Naturalist 54:170–176.

Drut, M. S., W. H. Pyle, and J. A. Crawford. 1994*b*. Technical note: diets and food selection of sage grouse chicks in Oregon. Journal of Range Management 47:90.

Dunn, P. O., and C. E. Braun. 1986. Summer habitat use by adult female and juvenile sage grouse. Journal of Wildlife Management 50:228–235.

Erickson, W. P., T. L. McDonald, K. G. Gerow, S. Howlin, and J. W. Kern. 2001. Statistical issues in resource selection studies with radio-marked animals. Pages 209–242 *in* J. J. Millspaugh and J. M. Marzluff, editors. Radio-tracking and animal populations. Academic Press, New York, New York.

ESRI (Environmental Systems Research Institute). 2002. ArcGIS. Release 8.3. ESRI, Redlands, California, USA.

Evans, J. 2002. Compound topographic index, AML script. ESRI Support Center Downloads. ESRI, Redlands, California, USA. ⟨http://arcscripts.esri.com/details.asp?dbid=11863⟩

Fielding, A. H., and J. F. Bell. 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. Environmental Conservation 24:38–49.

Franklin, A. B., D. R. Anderson, R. J. Gutierrez, and K. P. Burnham. 2000. Climate, habitat quality, and fitness in Northern Spotted Owl populations in northwestern California. Ecological Monographs 70:539–590.

Fuhlendorf, S. D., A. J. W. Woodward, D. M. Leslie, and J. S. Shackford. 2002. Multi-scale effects of habitat loss and fragmentation on lesser prairie-chicken populations of the US Southern Great Plains. Landscape Ecology 17:617–628.

Gessler, P. E., I. D. Moore, N. J. Mckenzie, and P. J. Ryan. 1995. Soil-landscape modeling and spatial prediction of soil attributes. International Journal of Geographical Information Systems 9:421–432.

Hagen, C. A. 2003. A demographic analysis of lesser prairie-chicken populations in southwestern Kansas: survival, population viability, and habitat use. Dissertation. Kansas State University, Manhattan, Kansas, USA.

Hayes, G. F., and K. D. Holl. 2003. Cattle grazing impacts on annual forbs and vegetation composition of mesic grasslands in California. Conservation Biology 17:1694–1702.

Herkert, J. R., D. L. Reinking, D. A. Wiedenfeld, M. Winter, J. L. Zimmerman, W. E. Jensen, E. J. Finck, R. R. Koford, D. H. Wolfe, S. K. Sherrod, M. A. Jenkins, J. Faaborg, and S. K. Robinson. 2003. Effects of prairie fragmentation on the nest success of breeding birds in the midcontinental United States. Conservation Biology 17:587–594.

Hobbs, N. T., and T. A. Hanley. 1990. Habitat evaluation: do use availability data reflect carrying-capacity? Journal of Wildlife Management 54:515–522.

Holloran, M. J. 2005. Greater Sage-Grouse (*Centrocercus Urophasianus*) population response to natural gas field development in western Wyoming. Dissertation. University of Wyoming, Laramie, Wyoming, USA.

Holloran, M. J., B. J. Heath, A. G. Lyon, S. J. Slater, J. K. Kuipers, and S. H. Anderson. 2005. Greater Sage-Grouse nesting habitat selection and success in Wyoming. Journal of Wildlife Management 69:638–649.

Hosmer, D. W., and S. Lemeshow. 1999. Applied survival analysis: regression modeling of time to event data. John Wiley, New York, New York, USA.

Hosmer, D. W., and S. Lemeshow. 2000. Applied logistic regression. John Wiley, New York, New York, USA.

Johnson, C. J., S. E. Nielsen, E. H. Merrill, T. L. McDonald, and M. S. Boyce. 2006. Resource selection functions based on use-availability data: theoretical motivation and evaluation methods. Journal of Wildlife Management 70:347–357.

Johnson, D. H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology 61:65–71.

Johnson, G. D., and M. S. Boyce. 1991. Survival, growth and reproduction of captive-reared sage grouse. Wildlife Society Bulletin 19:88–93.

Jones, P. F., R. Penniket, L. Fent, J. Nicholson, and B. Adams. 2005. Silver sagebrush community associations in southeastern Alberta, Canada. Rangeland Ecology and Management. 58:400–405.

Keating, K. A., and S. Cherry. 2004. Use and interpretation of logistic regression in habitat selection studies. Journal of Wildlife Management 68:774–789.

Knick, S. T., D. S. Dobkin, J. T. Rotenberry, M. A. Schroeder, W. M. Vander Haegen, and C. Van Riper. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. Condor 105:611–634.

Kristan, W. B. 2003. The role of habitat selection behavior in population dynamics: source–sink systems and ecological traps. Oikos 103:457–468.

Kurki, S., A. Nikula, P. Helle, and H. Lindén. 2000. Landscape fragmentation and forest composition effects on grouse breeding success in boreal forests. Ecology 81:1985–1997.

Laliberte, A. S., and W. J. Ripple. 2004. Range contractions of North American carnivores and ungulates. BioScience 54: 123–138.

Larson, M. A., F. R. Thompson, J. J. Millspaugh, W. D. Dijak, and S. R. Shifley. 2004. Linking population viability, habitat suitability, and landscape simulation models for conservation planning. Ecological Modelling 180:103–118.

Leonard, K. M., K. P. Reese, and J. W. Connelly. 2000. Distribution, movements and habitats of sage grouse *Centrocercus Urophasianus* on the upper Snake River Plain of Idaho: changes from the 1950s to the 1990s. Wildlife Biology 6:265–270.

Liu, C., P. M. Berry, T. P. Dawson, and R. G. Pearson. 2005. Selecting thresholds of occurrence in the prediction of species distributions. Ecography 28:385–393.

Lyon, A. G., and S. H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin 31:486–491.

Manel, S., H. C. Williams, and S. J. Ormerod. 2001. Evaluating presence–absence models in ecology: the need to account for prevalence. Journal of Applied Ecology 38:921–931.

Manier, D. J., and N. T. Hobbs. 2006. Large herbivores influence the composition and diversity of shrub-steppe communities in the Rocky Mountain, USA. Oecologica 146:641–651.

Manly, B. F. J., L. L. McDonald, D. L. Thomas, T. L. McDonald, and W. P. Erickson. 2002. Resource selection by animals: statistical design and analysis for field studies. Chapman and Hall, London, UK.

Manolis, J. C., D. E. Andersen, and F. J. Cuthbert. 2000. Uncertain nest fates in songbird studies and variation in Mayfield estimation. Auk 117:615–626.

Manzer, D. L., and S. J. Hannon. 2005. Relating grouse nest success and corvid density to habitat: a multi-scale approach. Journal of Wildlife Management 69:110–123.

Mattson, D. J., and T. Merrill. 2002. Extirpations of grizzly bears in the contiguous United States, 1850–2000. Conservation Biology 16:1123–1136.

McNeil, R. L., and B. J. Sawyer. 2003. Effects of water management practices and precipitation events on sagebrush habitat in southeastern Alberta. Prepared for Alberta Conservation Association and Alberta Sustainable Resource Development by Landwise Inc., Lethbridge, Alberta, Canada.

Menard, S. 1995. Applied logistic regression analysis. Sage Publications, Thousand Oaks, California, USA.

Morrison, M. L. 2001. Invited Paper: A proposed research emphasis to overcome the limits of wildlife–habitat relationship studies. Journal of Wildlife Management 65:613–623.

Neilson, R. P., J. M. Lenihan, D. Bachet, and R. J. Drapek. 2005. Climate change implications for sagebrush ecosystems. Pages 145–159 *in* Transactions of the 70th North American Wildlife and Natural Resources Conference. Wildlife Management Institute, Washington, D.C., USA.

Nielsen, S. E., R. H. M. Munro, E. L. Bainbridge, G. B. Stenhouse, and M. S. Boyce. 2004. Grizzly bears and forestry II. Distribution of grizzly bear foods in clearcuts of west-central Alberta, Canada. Forest Ecology and Management 199:67–82.

Nielsen, S. E., G. B. Stenhouse, and M. S. Boyce. 2006. A habitat-based framework for grizzly bear conservation in Alberta. Biological Conservation 130:217–229.

Niemuth, N. D. 2000. Land use and vegetation associated with greater prairie-chicken leks in an agricultural landscape. Journal of Wildlife Management 64:278–286.

Niemuth, N. D., and M. S. Boyce. 2004. Influence of landscape composition on sharp-tailed grouse lek location and attendance in Wisconsin pine barrens. Écoscience 11:209–217.

Pasitschniak-Arts, M., and F. Messier. 1995. Risk of predation on waterfowl nests in the Canadian prairies: effects of habitat edge and agricultural practices. Oikos 73:347–355.

Patterson, R. L. 1952. The sage grouse in Wyoming. Sage Books, Denver Colorado, USA.

PCI Geomatics. 2001. Geomatica Prime: Release 8.2. Richmond Hill, Ontario, Canada.

Pitman, J. C. 2003. Lesser prairie-chicken nest site selection and success, juvenile gender determination and growth, and juvenile dispersal in southwestern Kansas. Thesis. Kansas State University, Manhattan, Kansas, USA.

Pulliam, H. R. 1988. Sources, sinks, and population regulation. American Naturalist 132:652–661.

Remes, V. 2000. How can maladaptive habitat choice generate source–sink population dynamics? Oikos 91:579–582.

Schoenfeld, D. 1982. Partial residuals for the proportional hazards regression-model. Biometrika 69:239–241.

Schroeder, M. A., et al. 2004. Distribution of Sage-Grouse in North America. Condor 106:363–376.

Schroeder, M. A., J. R. Young, and C. E. Braun. 1999. Sage Grouse (*Centrocercus urophasianus*). Number 28 *in* A. Poole and F. Gill, editors. The birds of North America. The Birds of North America, Inc., Philadelphia, Pennsylvania, USA.

Sellers, P. J. 1985. Canopy reflectance, photosynthesis and transpiration. International Journal of Remote Sensing 6: 1335–1372.

STATA. 2004. STATA version 8.2. STATA Corporation, College Station, Texas, USA.

Sveum, C. M., J. A. Crawford, and W. D. Edge. 1998*a*. Use and selection of brood-rearing habitat by sage grouse in south central Washington. Great Basin Naturalist 58:344–351.

Sveum, C. M., M. D. Edge, and J. A. Crawford. 1998*b*. Nesting habitat selection by sage grouse in south-central Washington. Journal of Range Management 51:265–269.

Svobodova, J., T. Albrecht, and M. Salek. 2004. The relationship between predation risk and occurrence of black grouse (*Tetrao tetrix*) in a highly fragmented landscape: an experiment based on artificial nests. Écoscience 11:421–427.

Swets, J. A. 1988. Measuring the accuracy of diagnostic systems. Science 240:1285–1293.

Tyre, A. J., H. P. Possingham, and D. B. Lindenmayer. 2001. Inferring process from pattern: can territory occupancy provide information about life history parameters? Ecological Applications 11:1722–1737.

Van Horne, B. 1983. Density as a misleading indicator of habitat quality. Journal of Wildlife Management 47:893–901.

Wakkinen, W. L., K. P. Reese, and J. W. Connelly. 1992. Sage grouse nest locations in relation to leks. Journal of Wildlife Management 56:381–383.

Wallestad, R. O., and D. B. Pyrah. 1974. Movements and nesting of sage grouse hens in central Montana. Journal of Wildlife Management 38:630–633.

Wiebe, K. L., and K. Martin. 1998. Costs and benefits of nest cover for ptarmigan: changes within and between years. Animal Behaviour 56:1137–1144.

Wiegand, T., J. Naves, T. Stephan, and A. Fernandez. 1998. Assessing the risk of extinction for the brown bear (*Ursus arctos*) in the cordillera Cantabrica, Spain. Ecological Monographs 68:539–570.

Wilcove, D. S. 1985. Nest predation in forest tracts and the decline of migratory songbirds. Ecology 66:1211–1214.

Winterstein, S. R., K. H. Pollock, and C. M. Bunck. 2001. Analysis of survival data from radiotelemetry studies. Pages 351–380 *in* J. J. Millspaugh and J. M. Marzluff, editors. Radio-tracking and animal populations. Academic Press, New York, New York, USA.

Wintrebert, C. M. A., A. H. Zwinderman, E. Cam, R. Pradel, and J. C. Van Houwelingen. 2005. Joint modelling of breeding and survival in the kittiwake using frailty models. Ecological Modelling 181:203–213.

Wisdom, M. J., M. M. Rowland, B. C. Wales, M. A. Hemstrom, W. J. Hann, M. G. Raphael, R. S. Holthausen, R. A. Gravenmier, and T. D. Rich. 2002*a*. Modeled effects of sagebrush-steppe restoration on greater sage-grouse in the Interior Columbia Basin, USA. Conservation Biology 16: 1223–1231.

Wisdom, M. J., B. C. Wales, M. M. Rowland, M. G. Raphael, R. S. Holthausen, T. D. Rich, and V. A. Saab. 2002*b*. Performance of greater sage-grouse models for conservation assessment in the Interior Columbia Basin, USA. Conservation Biology 16:232–1242.

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January 29, 2008

MEMORANDUM

TO: Terry Cleveland and John Emmerich

FROM: Tom Christiansen and Joe Bohne

COPY TO: Jay Lawson, Bill Rudd, Reg Rothwell, Bob Oakleaf

SUBJECT: Multi-State Sage-Grouse Coordination and Research-based Recommendations

As assigned by Assistant Director Emmerich, we have been working with other state fish and wildlife agencies in WAFWA Sage-Grouse Management Zones 1 and 2 (MT, CO, UT, SD, ND, WY) in order to coordinate interpretation of recent sage-grouse research related to oil and gas development.

Attached for your review, please find the latest and final document capturing the multi-state interpretation of the recent science related to sage-grouse conservation and oil and gas development. It has been well scrutinized by staff from MT, WY, CO, ND and UT and there is consensus on the content by the participants. South Dakota was unable to attend the initial meeting in Salt Lake City on January 8-9, but they have been provided with meeting notes and the resulting document.

It is our recommendation that WGFD acknowledge this document as the correct interpretation of the recently published sage-grouse research and use this information to update and augment department documents and policies. It should be used in the forthcoming discussions with the BLM regarding their update to their sage-grouse Instruction Memorandum. In addition, we suggest that in order for this document to serve the broadest purpose for sage-grouse conservation four additional actions are needed. First, the document should be shared with Governor Freudenthal's staff. Second, we recommend that the Director's Office enter into discussions with MT FWP Director Jeff Hagener to ensure consistency in the application of these recommendations between our border states, and especially with the WY and MT BLM State Field Offices. Third, we recommend the document be submitted to WAFWA's Sage-Grouse Technical Committee as well as the WAFWA Executive Committee for their consideration and use. Finally, we recommend this document be included with other materials sent to the USFWS for consideration in their review of the status of sage-grouse and measures in place to conserve those populations.

We look forward to your direction on how to proceed.

Using the Best Available Science to Coordinate Conservation Actions that Benefit Greater Sage-Grouse Across States Affected by Oil & Gas Development in Management Zones I-II (Colorado, Montana, North Dakota, South Dakota, Utah, and Wyoming)

Background

Greater Sage-grouse are widely considered in scientific and public policy arenas to be a species of significant conservation concern. Loss, degradation and fragmentation of important sagebrush grassland habitats have negatively impacted sage-grouse populations. Much of this loss of habitat function is occurring in Sage-grouse Management Zones (MZ) 1 and 2 (Stiver et al. 2006) in Colorado, Montana, North Dakota, South Dakota, Utah, and Wyoming as a result of oil and gas development (Connelly et al. 2004). Oil and gas development is rapidly increasing within these areas. In response to those concerns, states and provinces are in various stages of completing or updating management plans in order to provide for long-term sage-grouse conservation. Special emphasis is being placed on oil and gas development as it rapidly spreads across much of the eastern range of sage-grouse.

The recent decision by B. Lynn Winmill, Chief U.S. District Judge (2007), which remands the original 2005 not warranted decision back to the USFWS for reconsideration, has highlighted the need for States to coordinate their application of best available science. Representatives from the state agencies with authority for managing fish and wildlife from the major sage-grouse and energy producing states comprising MZ 1 and 2 and sage-grouse researchers who have published new findings, met on January 8 and 9, 2008 in Salt Lake City. The objectives of the meeting were to better understand the application of most recent peer-reviewed science within the context of oil and gas development and coordinate and compare implementation of conservation actions utilizing that information.

Review Process

The participants at this meeting represented technical science and management advisors from each of the states. Researchers having the most recently peer reviewed and published articles concerning sage grouse and oil and gas development were invited to present their findings and answer questions. State agency participants agreed that the goal was not to establish state or regional policy or to determine the management actions that will be implemented in any or all states within MZ 1 or 2. Rather, the goal was to reach agreement on the conservation concepts and strategies related to oil and gas development that are supported by current published peer-reviewed and unpublished literature. If implemented, these concepts and strategies likely will not eliminate impacts to sage-grouse populations that result from energy development. However, when used in combination with other conservation measures, these actions may enhance the likelihood that sage-grouse populations will persist at levels that allow historical uses such as grazing and agriculture and maintain their current distribution and abundance, thereby avoiding the need to list sage-grouse under the federal Endangered Species Act.

Each researcher was invited to present their findings and to answer questions posed by the states. Following this, each state provided an overview of their review of the science and their resulting management actions and recommendations. The group then collectively reviewed, debated and agreed on the concepts and strategies supported by that science. The focus of the meeting was on five key issues: core areas, no-surface-occupancy zones, phased development, timing stipulations, well-pad densities, and restoration. Scientific data are available to inform many other issues related to sage-grouse management and conservation that were not reviewed (e.g., BMPs).

Core Areas

Identification and protection of core areas, sometimes also referred to as crucial areas, will help maintain or achieve target goals for populations including distribution and abundance.

Full field energy development appears to have severe negative impacts on sage-grouse populations under current lease stipulations (Lyon and Anderson 2003, Holloran 2005, Kaiser 2006, Holloran et al. 2007, Aldridge and Boyce 2007, Walker et al 2007, Doherty et al. 2008). Much of greater sage-grouse habitat in MZ 1 and 2 has already been leased for oil and gas development. These leases carry stipulations that have been shown to be inadequate for protecting breeding and wintering sage-grouse populations during full field development. (Holloran 2005, Walker et. al. 2007, Doherty et al. 2008) New leases continue to be issued utilizing these same stipulations. To ensure long-term persistence of populations and meet goals set by the states for sage-grouse, identifying and implementing greater protection within core areas from impacts of oil and gas development is a high priority.

In order to conserve core areas it is essential that they be identified and delineated. Sage-grouse populations occur over large landscapes comprising a series of leks and lek complexes with associated seasonal habitats. Therefore, core areas should capture the range required by a defined population to maintain itself. This concept is consistent with Crucial Wildlife Habitats recently endorsed by the Western Governor's Association (2007). Criteria that could be used to identify and map core areas include, but are not limited to: (1) lek densities, (2) displaying male densities, (3) sagebrush patch sizes, (4) seasonal habitats (breeding, summering, wintering areas), (5) seasonal linkages, or (6) appropriate buffers around important seasonal habitats.

Research indicates that oil or gas development exceeding approximately 1 well pad per square mile with the associated infrastructure, results in calculable impacts on breeding populations, as measured by the number of male sage-grouse attending leks (Holloran 2005, Naugle et al. 2006). Because breeding, summer, and winter habitats are essential to populations, development within these areas should be avoided. If development cannot be avoided within core areas, infrastructure should be minimized and the area should be managed in a manner that effectively conserves sagebrush habitats within that area.

No Surface Occupancy (NSO)

At the scale that NSOs are established, they alone will not conserve sage-grouse populations without being used in combination with core areas. The intent of NSOs is to maintain sage-grouse distribution and a semblance of habitat integrity as an area is developed.

Breeding Habitat - Leks

Research in Montana and Wyoming in coal-bed methane natural gas (CBNG) and deep-well fields suggests that impacts to leks from energy development are discernable out to a minimum of 4 miles, and that some leks within this radius have been extirpated as a direct result of energy development (Holloran 2005, Walker et al. 2007). Walker et al. (2007) indicates that the current 0.25-mile buffer lease stipulation is insufficient to adequately conserve breeding sage-grouse populations in areas having full CBNG development. A 0.25-mi. buffer leaves 98% of the landscape within 2 miles open to full-scale energy development. In a typical landscape in the Powder River Basin, 98% CBNG development within 2 miles of leks is projected to reduce the average probability of lek persistence from 87% to 5% (Walker et al. 2007). Only 38% of 26 leks inside of CBNG development remained active compared to 84% of 250 leks outside of development (Walker et al. 2007). Of leks that persisted, the numbers of attending males were reduced by approximately 50% when compared to those outside of CBNG development (Walker et al. 2007).

The impact analyses provided in Walker et al. (2007) are based on a 7-year dataset where probability of lek persistence is strongly related to extent of sagebrush habitat and the extent of energy development within 4 miles of the lek and the extent of agricultural tillage in the surrounding landscape. The estimated probabilities of lek persistence are only reliable for the length of the dataset, and it is not understood how other stressors (e.g., West Nile virus [Naugle et al. 2004], invasive weeds [Bergquist et al. 2007]) will cumulatively impact sage-grouse over longer time periods. While increased NSO buffers alone are unlikely to conserve sage-grouse populations, results from Walker et al. 2007 suggest they will increase the likelihood of maintaining the distribution and abundance of grouse and should increase the likelihood of successful restoration following energy development.

Additional information provided in Walker et al. (2007) allows managers and policy makers to estimate trade-offs associated with allowing development within a range of different distances from leks (Figures 1a and 1b). These probabilities will also need to be applied over larger landscapes in future analyses to better understand projected region- and state-wide population impacts under current and future development scenarios. Walker et al. (2007) studied lek persistence from 1997-2005 in relation to coal bed natural gas (CBNG) development in the Powder River Basin. These models are based on projected impacts of full-field development within (a) 2 miles and (b) 4 miles of the lek. We present results from these models (rather than models with impacts at smaller scales) because development within 2 and 4 miles of leks are known to decrease breeding populations as measured by the number of displaying males (Holloran et al. 2005, Walker et al. 2007), and 52% and 74-80% of hens are known to nest within 2 and 4 miles of leks, respectively (Holloran and Anderson 2005, Colorado Greater Sage-Grouse Conservation Plan Steering Committee 2008). Sizes of NSO buffers required to protect breeding populations may be underestimated because leks in CBNG fields have fewer males per lek and a time lag occurs (avg. 3-4 years) between development and when leks go inactive. As a result, it is expected that not only will lek persistence decline, the number of males per lek will also decline. In contrast, sizes may be overestimated where high lek densities cause buffers from adjacent leks to overlap. Additional time is required to develop models demonstrating the probabilities of lek persistence at well-pad densities less than full development.

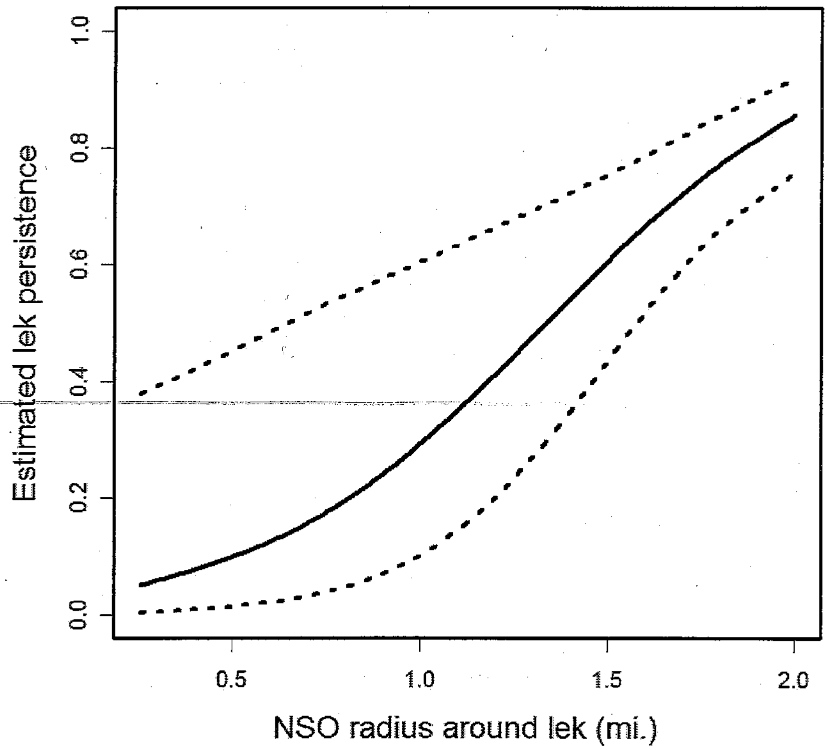

Figure 1a. Estimated probability of lek persistence (dashed lines represent 95% CIs) in fully-developed[1] coal-bed natural gas fields within an average landscape in the Powder River Basin (74% sagebrush habitat, 26% other habitats types) with different sizes of no-surface-occupancy (NSO) buffers around leks, assuming that only CBNG within 2 miles of the lek affects persistence. Buffer sizes of 0.25 mi., 0.5 mi., 0.6 mi., and 1.0 mi. result in estimated lek persistence of 5%, 11%, 14%, and 30%. Lek persistence in the absence of CBNG averages ~85%.

[1] Defined as entire area outside the NSO buffer, but within 2 miles, being within 350 meters of a well.

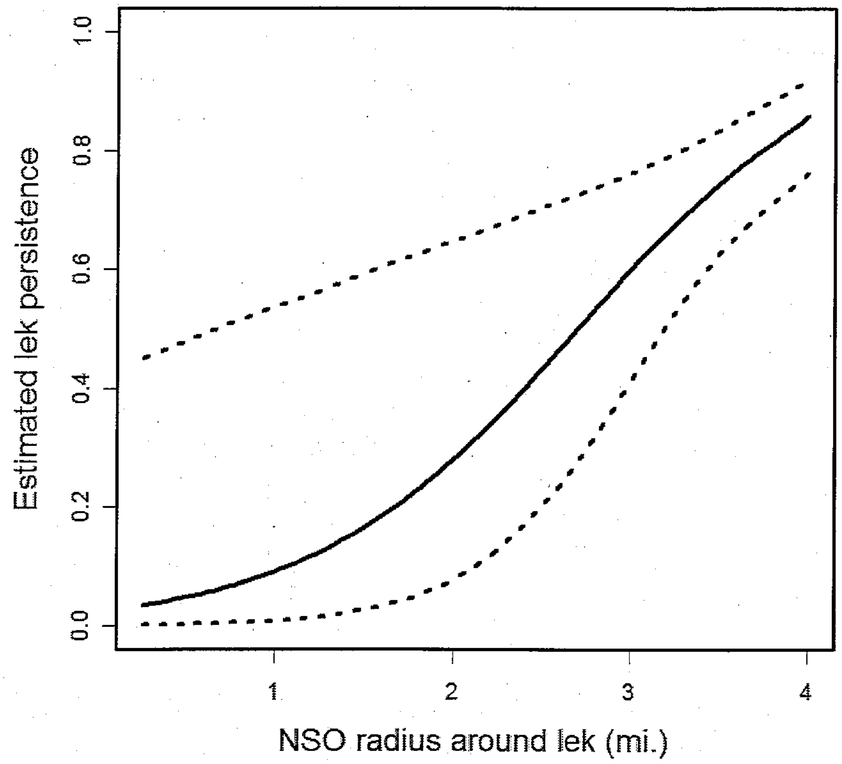

Figure 1b. Estimated probability of lek persistence (dashed lines represent 95% CIs) in fully-developed[2] coal-bed natural gas fields within an average landscape in the Powder River Basin (74% sagebrush habitat, 26% other habitats types) with different sizes of no-surface-occupancy (NSO) buffers around leks, assuming that only CBNG within 4 miles of the lek affects persistence. Buffer sizes of 0.25 mi., 0.5 mi., 0.6 mi., 1.0 mi., and 2.0 mi. result in estimated lek persistence of 4%, 5%, 6%, 10%, and 28%. Lek persistence in the absence of CBNG averages ~85%.

Figures 1a and 1b provide an illustration of the trade-offs between differing NSO buffers in relation to lek persistence in developing CBNG fields. The group does not offer a specific NSO recommendation but provides these graphs to guide decision-making.

Breeding Habitat - Nesting and Early Brood-rearing

Yearling female greater sage-grouse avoid nesting in areas within 0.6 miles of producing well pads (Holloran et al. 2007), and brood-rearing females avoid areas within 0.6 miles of producing wells (Aldridge and Boyce 2007). This suggests a 0.6-mile NSO around all suitable nesting and brood-rearing habitats is required to minimize impacts to females during these seasonal periods. In areas where nesting habitats have not been delineated, research suggests that greater sage-grouse nests are not randomly distributed. Rather, they are spatially associated with lek location within 3.1 miles in Wyoming (Holloran and Anderson 2005). However, a 4-mile buffer is needed to encompass 74-80% (Moynahan

[2] Defined as entire area outside the NSO buffer, but within 4 miles, being within 350 meters of a well.

2004, Holloran and Anderson 2005, Colorado Greater Sage-Grouse Conservation Plan Steering Committee 2008). These suggest that all areas within at least 4-miles of a lek should be considered nesting and brood-rearing habitats in the absence of mapping.

Winter Habitat

NSO or other protections may also need to be considered for crucial winter range. Survival of juvenile, yearling, and adult females are the three most important vital rates that drive population growth in greater sage-grouse (Holloran 2005, Colorado Greater Sage-Grouse Conservation Plan Steering Committee 2008). Although overwinter survival in sage-grouse is typically high, severe winter conditions can decrease hen survival (Moynahan et al 2006). Crucial wintering habitats can constitute a small part of the overall landscape (Beck 1977, Hupp and Braun 1989). Doherty et al. (2008) demonstrated that sage-grouse avoided otherwise suitable wintering habitats once they have been developed for energy production, even after timing and lek buffer stipulations had been applied (Doherty et al. 2008). For this reason, increased levels of protection may need to be considered in crucial winter habitats.

Phased Development

Population-level impacts and avoidance associated with energy development have been documented (Braun et al. 2002, Lyon and Anderson 2003, Holloran 2005, Kaiser 2006, Holloran et al. 2007, Aldridge and Boyce 2007, Walker et al 2007, Doherty et al. 2008). Phased development maximizes the amount of area within a landscape that is not being impacted by development at any one time, and can occur at multiple spatial scales (e.g., phased development of separate fields in a landscape, phased development of infrastructure within a single unit or field, or phased development within a single lease). Unitization, clustering, and geographically staggered development are all forms of phased development. As a tool to minimize impacts to sage-grouse, developing oil and gas resources by employing one of these phased methods may help maintain large, functional blocks of sage-grouse habitat.

Timing Stipulations

As with NSOs, at the scale that timing stipulations are established, they alone will not conserve sage-grouse populations without being used in combination with core areas. The intent of timing stipulations is to help maintain sage-grouse distribution and a semblance of habitat integrity as an area is developed. Timing stipulations are of lesser value at the scale of full-field development.

Breeding Habitat - Leks

Traffic during the strutting period when males are on a lek results in declines in male attendance when road-related disturbance is within 0.8 miles (Holloran 2005). The distance traveled by males from the lek during the breeding season has been reported in varying ways but generally averages 0.6 miles from a lek (Colorado Greater Sage-Grouse

Conservation Plan Steering Committee 2008 - see Appendix B). Additionally, females breeding on leks within 1.9 miles of natural gas development had lower nest initiation rates and nested farther from the lek compared to non-impacted individuals (Lyon and Anderson 2003), suggesting disturbance to leks influence females as well. Local variations may influence the application of specific dates, which are typically within a window of March 1 and May 31.

Breeding Habitat - Nesting and Early Brood-rearing

Often, timing stipulations (periods where no activity that creates disturbance are allowed) for breeding habitat have been applied using a radius around a lek. However, nesting and brood-rearing habitat is not uniformly distributed around the lek. Mapping of habitat would allow for more accurate application of this stipulation. Research on the distribution of nests relative to leks and on the timing of nesting indicates that timing stipulations to protect nesting hens and their habitat should be in place from March through June in mapped breeding habitat or (when nesting habitat has not been mapped) within 4 miles of active lek sites (Moynahan 2004, Holloran et al. 2005, Colorado Greater Sage-Grouse Conservation Plan Steering Committee 2008).

Winter Habitat

Research suggests that no surface occupancy should also be applied to important wintering habitats (Doherty et al. 2008), but if development occurs, impacts would be reduced if development activities were avoided between December 1 and March 15.

Well-Pad Densities

Leks tend to remain active when well-pad densities within 1.9 miles of leks are less than 1 pad per square mile (Holloran 2005) but leks tend to go inactive at higher pad densities (Holloran 2005, Naugle et al. 2006).

Restoration

The purpose of restoration in sage-grouse habitat should be the removal of infrastructure associated with energy development from the land surface and subsequent re-establishment of native grasses, forbs, and shrubs, including sagebrush, to promote natural ecological function. Restoration should reestablish functionality of seasonal habitats for sage-grouse. Thus a field should not be considered restored until sagebrush-grassland habitats have been reestablished.

Future Needs

Time did not allow for a detailed discussion of specific Best Management Practices for oil and gas development and restoration, seasonal habitat mapping, or future research. These topics are all recognized as needing action in the immediate future.

Literature Cited

Aldridge, C. L., and M. S. Boyce. 2007. Linking occurrence and fitness to persistence: a habitat-based approach for endangered greater sage-grouse. Ecological Applications 17:508-526.

Beck, T. D. I. 1977. Sage grouse flock characteristics and habitat selection during winter. Journal of Wildlife Management 41:18-26.

Bergquist, E., P. Evangelista, T. J. Stohlgren, and N. Alley. 2007. Invasive species and coal bed methane development in the Powder River Basin, Wyoming. Environmental Monitoring and Assessment. 128:381-394.

Braun, C. E., O. O. Oedekoven, and C. L. Aldridge. 2002. Oil and gas development in western North America: effects on sagebrush steppe avifauna with particular emphasis on sage grouse. Transactions North American Wildlife and Natural Resources Conference 67:337-349.

Colorado Greater Sage-Grouse Conservation Plan Steering Committee. 2008. The Colorado Greater Sage-Grouse Conservation Plan. Colorado Division of Wildlife. Denver, CO. Unpublished Report.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming, USA.

Doherty, K.E., D.E. Naugle, B.L. Walker, J.M. Graham. 2008. Greater sage-grouse winter habitat selection and energy development. Journal of Wildlife Management *In Press.*

Holloran, M. J. 2005. Greater sage-grouse (*Centrocercus urophasianus*) population response to natural gas field development in western Wyoming. Dissertation. University of Wyoming, Laramie, USA.

Holloran, M. J. and S. H. Anderson. 2005. Spatial distribution of greater sage-grouse nests in relatively contiguous sagebrush habitats. Condor 107:742-752.

Holloran, M. J., B. J. Heath, A. G. Lyon, S. J. Slater, J. L. Kuipers, and S. H. Anderson. 2005. Greater sage-grouse nesting habitat selection and success in Wyoming. Journal of Wildlife Management 69: 638-649.

Holloran, M. J., R. C. Kaiser, and W. A. Hubert. 2007. Population response of yearling greater sage-grouse to the infrastructure of natural gas fields in southwestern Wyoming. Completion report. Wyoming Cooperative Fish and Wildlife Research Unit, Laramie, WY, USA.

Hupp, J. W. and C. E. Braun. 1989. Topographic distribution of sage grouse foraging in winter. Journal of Wildlife Management 53: 823-829.

Kaiser, R. C. 2006. Recruitment by greater sage-grouse in association with natural gas development in western Wyoming. Thesis. University of Wyoming. Laramie, USA.

Lyon, A. G. and S. H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin 31:486-491.

Moynahan B. J. 2004. Landscape-scale factors affecting population dynamics of greater sage-grouse *(Centrocercus urophasianus)* in northcentral Montana, 2001–2004. Dissertation, The University of Montana. Missoula, USA.

Moynahan, B.J., M.S. Lindberg, and J.W. Thomas. 2006. Factors contributing to process variance in annual survival of female greater sage-grouse in north-central Montana. Ecological Applications 16:1529-1538.

Naugle, D. E., C. L. Aldridge, B. L. walker, T. E. Cornish, B. J. Moynahan, M. J. Holloran, K. Brown, G. D. Johnson, E. T. Schmidtmann, R.T. Mayer, C. Y. Kato, M. R. Matchett, T. J. Christiansen, W. E. Cook, T. Creekmore, R. D. Falise, E. T. Rinkes, and M. S. Boyce. 2004. West Nile virus: pending crisis for greater sage-grouse. Ecology Letters 7:704-713.

Naugle, D. E., B. L. Walker, and K. E. Doherty. 2006. Sage-grouse population response to coal-bed natural gas development in the Powder River Basin: interim progress report on region-wide lek-count analyses. Unpublished Report, University of Montana, Missoula, USA.

Stiver, S.J., A.D. Apa, J.R. Bohne, S.D. Bunnell, P.A. Deibert, S.C. Gardner, M.A. Hilliard, C.W. McCarthy, and M.A. Schroeder. 2006. Greater sage-grouse comprehensive conservation strategy. Western Association of Fish and Wildlife Agencies. Unpublished report. Cheyenne, Wyoming.

Walker, B.L., D. E. Naugle, and K.E. Doherty. 2007. Greater sage-grouse population response to energy development and habitat loss. Journal of Wildlife Management 71:2644-2654.

Western Governor's Association. 2007. Wildlife corridors initiative oil and gas working group report. http://www.westgov.org/wga/publicat/OilGas07.pdf. Accessed 15 January 2007.

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Impacts of Anthropogenic Noise on Wildlife: Research Priorities for the Development of Standards and Mitigation

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1. INTRODUCTION

Human development introduces anthropogenic noise sources into the environment across many elements of the modern terrestrial landscape, including roads, airports, military bases, and cities. The impacts of these introduced noise sources on wildlife are less well studied than many of the other effects human activities have on wildlife, the most well known of which are habitat fragmentation and the introduction of invasive species. A growing and substantial body of literature suggests, however, that noise impacts may be more important and widespread than previously imagined.[3] They range in effects from mild to severe. They can impact wildlife species at both the individual and population levels. The types of impacts run the gamut from damage to the auditory system, the masking of sounds important to survival and reproduction, the imposition of chronic stress and associated physiological responses, startling, interference with mating, and population declines.

Anthropogenic noise is a global phenomenon, with the potential to affect wildlife across all continents and habitat types. Despite the widespread

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[3] For a review of noise impacts on birds and other wildlife, see P. A. KASELOO & K. O. TYSON, SYNTHESIS OF NOISE EFFECTS ON WILDLIFE POPULATIONS (U.S. Department of Transportation, Federal Highway Administration, 2004); ROBERT J. DOOLING & ARTHUR N. POPPER, THE EFFECTS OF HIGHWAY NOISE ON BIRDS (California Department of Transportation, Division of Environmental Analysis, 2007).

distribution of noise, the bulk of research on the effects of noise on terrestrial wildlife has been limited to European countries and the United States. This geographic bias in research may limit the application of the results from previous studies on a global basis, since the impacts may differ among habitats and species.[4]

Since much human development involves the introduction of noise, separating out and understanding the impacts of noise pollution is a critical step in developing effective wildlife policy, particularly the setting of standards and the use of mitigation measures. The first step typically is to determine the overall impact on the population demography of a species, by measuring population declines and birth rates. Mitigation requires that the mechanisms of this effect then be understood. From an initial determination, for example, that roads decrease songbird population densities, there must next be an estimation of the extent to which noise, dust, chemical pollution, habitat fragmentation, invasive weeds, visual disturbance, or road mortality are partial and contributory causes of that impact before effective mitigation measures aimed at noise can be chosen. Quieter pavements will not help songbirds if the true cause of the problem is visual disturbance. The key challenge, then, is to measure the contribution of noise to observed impacts on animal populations while controlling for other variables.

In this article, we address three questions: what are the common sources of anthropogenic noise; what is known about the mechanisms by which noise impacts wildlife; and how can we use observational and experimental approaches to estimate the impacts of noise on whatever species are of concern?

In answering these questions we deal at length with both observational and experimental methods, the latter including both laboratory and field work. We describe observational field studies on animal abundance and reproduction in impacted areas and a method for estimating the potential of noise sources to mask animal vocalizations. We address both the feasibility and value of laboratory and field experiments and describe a case study based on an ongoing noise-playback experiment we have designed to quantify the impacts of noise from energy development on greater sage-grouse (*Centrocercus urophasianus)* in Wyoming.

[4] The geographic bias in research has lead to a focus on species that live in temperate zones, with little to no study of tropical species. Also of concern, many of the landscapes that have been the focus of research on noise and wildlife in these industrialized nations have already been profoundly influenced by human development such that the species or individuals living in these areas may be more tolerant of disturbance. Application of the results of studies from developed to less developed landscapes would potentially lead to an underestimation of the effects of noise. Anthropogenic changes to the environment are occurring at an unprecedented rate in developing nations in tropical latitudes, however, we do not yet know whether the results from existing research are applicable in these regions.

Our focus, then, is on noise impacts on animals in the terrestrial environment,[5,6] especially birds, which are the subjects of most terrestrial studies.[7] We also outline directions for future research and in a final section emphasize the importance of this research for developing flexible wildlife management strategies in landscapes that are increasingly subject to human encroachment.

2. SOURCES OF NOISE

Noise is associated with most phases in the cycle of human development activity, from early construction to the daily operation of a completed project. Transportation systems are one of the most pervasive sources of noise across all landscapes, including common sources like roads and their associated vehicular traffic, airports and airplanes, off-road vehicles, trains, and ships. Roads deserve special attention, because they are a widespread and rapidly increasing terrestrial noise source. Although the surface area covered by roads is relatively small, the ecological effects of roads, including noise, extend far beyond the road itself, impacting up to one-fifth of the land area of the United States, for example.[8] Industrial noise sources, such as military bases, factories, mining operations, and wind farms may be more localized in the landscape, but are problematic for wildlife because the noise produced can be very loud.

The characteristics of noise vary substantially among sources. Each source type exhibits variance in amplitude (i.e., loudness), frequency profile

[5] Many terrestrial noise sources produce noise that travels through the ground as well as the air. Seismic noise is likely to impact fossorial animals and animals that possess specialized receptors for seismic detection, many of which communicate by seismic signals. We do not address seismic noise in this paper, but it is an issue that warrants further discussion.

[6] For recent treatments of noise in the marine environment, its impacts on marine species, and legal and policy responses, see *Noise Pollution and the Oceans: Legal and Policy Responses Part 1*, 10 J. INT'L WILDLIFE L. & POL'Y (2007) 101–199 and *Noise Pollution and the Oceans: Legal and Policy Responses Part 2*, 10 J. INT'L WILDLIFE L. & POL'Y (2007) 219–288. See also, Committee on Characterizing Biologically Significant Marine Mammal Behavior, Marine Mammal Populations and Ocean Noise, DETERMINING WHEN NOISE CAUSES BIOLOGICALLY SIGNIFICANT EFFECTS 142 (Ocean Studies Board, Division on Earth and Life Studies, National Research Council, The National Academies, 2005).

[7] Birds have often been used in noise research because birds are generally easy to study due to their high detectability, most species use vocal communication (making them likely to be impacted by noise) and they are generally of high conservation importance.

[8] R.T.T. Forman & R.D. Deblinger, *The Ecological Road-Effect Zone of a Massachusetts (U.S.A.) Suburban Highway*, 14 CONS. BIOL. 36–46 (2000); R.T.T. Forman, *Estimate of the Area Affected Ecologically by the Road System in the United States*, 14 CONS. BIOL. 31–35 (2000); R.T.T. Forman, B. Reineking, and A.M. Hersberger, *Road Traffic and Nearby Grassland Bird Patterns in a Suburbanizing Landscape*, 29 ENVT'L. MGMT. 782–800 (2002). Due to its ubiquity, road noise is the most commonly studied type of terrestrial noise. Road noise is, in general, similar to other types of anthropogenic noise and affects a wide range of species and habitat types, so the research techniques and results can be applied to many other types of anthropogenic noise.

(i.e., pitch), and spatial and temporal patterns. The interaction of these characteristics is what determines in a narrow sense the impact of noise on wildlife, setting aside the possibly confounding influence of contextual variables.

Intuitively, loud noise is more disruptive than quiet noise[9] and noise with frequencies similar to animal vocalizations is more likely to interfere with (i.e., mask) communication than noise with different frequencies.[10] Most anthropogenic noise sources have energy concentrated in low frequencies (<250 Hz), which can travel long distances with relatively little energy loss. Such noise is also more difficult to control using traditional noise-abatement structures, such as noise reflecting or absorbing walls along highways or surrounding other fixed noise sources, such as industrial sites.[11] Spatial patterning of noise may also affect the level of disturbance. A highly localized point source, like a drilling rig, will generally impact a smaller area than a linear source, such as a highway, although the area of impact will also depend on the amplitude and frequency structure of the noise. The temporal patterning of noise can also be important, because animal behaviors are often temporally patterned. Rush hour traffic, for example, often coincides with the dawn chorus of bird song,[12] an important time for birds because this is when mates are attracted and territories defended.[13]

Environmental noise is not an entirely new problem for animals, nor is human activity the exclusive cause of it. Natural environments have numerous sources of ambient noise, such as wind, moving water, and sounds produced by other animals. There is also evidence that animals living in naturally noisy areas have made adaptations through the use of signals and signaling behaviors to overcome the masking impacts of noise.[14] However, if anthropogenic noise

[9] M.E. Weisenberger et al., *Effects of Simulated Jet Aircraft Noise on Heart Rate and Behavior of Desert Ungulates*, 60 J. WILDLIFE MGMT. 52–61 (1996).

[10] Bernard Lohr et al., *Detection and Discrimination of Natural Calls in Masking Noise by Birds: Estimating the Active Space of a Signal*, 66 ANIMAL BEHAV. 703–710 (2003).

[11] S.P. SINGAL, NOISE POLLUTION AND CONTROL STRATEGY (2005).

[12] R.A. Fuller et al., *Daytime Noise Predicts Nocturnal Singing in Urban Robins*, 3 BIOL. LETTERS 368–370 (2007).

[13] C.K. CATCHPOLE & PETER J.B. SLATER, BIRD SONG: THEMES AND VARIATIONS (1995).

[14] For example, the structural and temporal properties of many acoustic signals are adapted—by evolution or through individual plasticity—to maximize the propagation distance and/or minimize interference from natural noise sources. R. Haven Wiley & Douglas G. Richards, *Adaptations for Acoustic Communication in Birds: Sound Transmission and Signal Detection*, *in* 1 ACOUSTIC COMMUNICATION IN BIRDS 131–181 (D. Kroodsma & E.H. Miller eds., 1982); H. Brumm, *Signalling through Acoustic Windows: Nightingales Avoid interspecific Competition by Short-Term Adjustment of Song Timing*, 192 J. COMP. PHYSIOL. A 1279–1285 (2006); Henrik Brumm & Hans Slabbekoorn, *Acoustic Communication in Noise*, 35 ADVANCES STUDY BEHAV. 151–209 (2005); Hans Slabbekoorn & Thomas B. Smith, *Habitat-Dependent Song Divergence in the Little Greenbul: An Analysis of Environmental Selection Pressures on Acoustic Signals*, 56 EVOLUTION 1849–1858 (2002); G.M. Klump, *Bird Communication in the Noisy World*, *in* ECOLOGY AND EVOLUTION OF ACOUSTIC COMMUNICATION IN BIRDS 321–338 (D. Kroodsma & E.H. Miller eds., 1996); Eugene S. Morton, *Ecological Sources of Selection on Avian Sounds*, 109 AM. NATURALIST 17–34 (1975).

differs enough from natural noise in frequency, amplitude, or daily/seasonal patterns, animal adaptations to natural noise can be overwhelmed. Furthermore, the extensive introduction of anthropogenic noise into the environment on a large scale is a relatively recent phenomenon, so that animals have had only a limited opportunity to adapt to widespread and sometimes drastic changes in their acoustic environments.[15]

3. THE POTENTIAL IMPACTS OF NOISE ON WILDLIFE

Animals exhibit a variety of responses to noise pollution (also called introduced noise), depending on the characteristics of the noise and the animal's ability to tolerate or adapt to it. Noise impacts on wildlife can be observed at the individual and population levels, which we now consider in turn.

3.1 Individual-Level Impacts

Some of the most dramatic impacts of noise on individuals are acute and need to be distinguished from chronic effects. Acute impacts include physiological damage, masking of communication, disruption of behavior, and startling. The most direct physiological impact affects an animal's ability to hear, either by permanently damaging the auditory system, in which case it produces what is called a permanent threshold shift (PTS) in hearing, or by causing temporary decreases in hearing sensitivity, which are called temporary threshold shifts (TTS).[16] The noise levels required for PTS and TTS are quite loud,[17] making hearing damage unlikely in most terrestrial situations. Even extremely loud sound sources will only cause PTS and TTS over a small area, because on land sound attenuates very quickly with distance.[18] This is why most studies

[15] G. Patricelli & J. Blickley, *Avian Communication in Urban Noise: Causes and Consequences of Vocal Adjustment*, 123 THE AUK 639–649 (2006); Paige S. Warren et al., *Urban Bioacoustics: It's Not Just Noise*, 71 ANIMAL BEHAV. 491–502 (2006); Lawrence A. Rabin et al., *Anthropogenic Noise and Its Effects on Animal Communication: An Interface Between Comparative Psychology and Conservation Biology*, 16 INT'L J. COMP. PSYCHOL. 172–192 (2003); Lawrence A. Rabin & Correigh M. Greene, *Changes to Acoustic Communication Systems in Human-Altered Environments*, 116 J. COMP. PSYCHOL. 137–141 (2002); H. Slabbekorn & E.A.P. Ripmeester, *Birdsong and Anthropogenic Noise: Implications and Applications for Conservation*, 17 MOLECULAR ECOLOGY 72–83 (2008).

[16] P. Marler et al., *Effects of Continuous Noise on Avian Hearing and Vocal Development*, 70 PROC. NAT'L ACAD. SCI. 1393–1396 (1973); J. Saunders & R. Dooling, *Noise-Induced Threshold Shift in the Parakeet (*Melopsittacus undulatus), 71 PROC. NAT'L ACAD. SCI. 1962–1965 (1974); Brenda M. Ryals et al., *Avian Species Differences in Susceptibility to Noise Exposure*, 131 HEARING RES. 71–88 (1999).

[17] PTS in birds may result from sound levels of ∼125 dBA SPL for multiple impulsive sounds and ∼140 dBA SPL for a single impulsive sound. TTS can result from continuous noise levels of ∼93 dBA SPL. The term "dBA SPL" refers to the A-weighted decibel, the most common unit for noise measurements. It adjusts for human perception of sound and is scaled relative to the threshold for human hearing.

[18] Sound levels drop by approximately 6 dB (measured using dBA SPL, or any other decibel measure), which represents a halving of loudness, with every doubling in distance from a point source, and 3 dB with every doubling of distance from a linear source, such as a highway.

of impacts from highway and urban noise do not directly address PTS and TTS, although they may need to be considered in extremely noisy areas.

Other acute impacts of noise, such as masking and behavioral disruption, occur over a much larger area. Masking occurs when the perception of a sound is affected by the presence of background noise, with high levels of background noise decreasing the perception of a sound.[19] One possible consequence of masking is a decrease in the efficacy of acoustic communication. Many animals use acoustic signals to attract and retain mates, settle territorial disputes, promote social bonding, and alert other individuals to predators. Disruption of communication can, therefore, have dramatic impacts on survival and reproduction.[20] In one laboratory study, high environmental noise reduced the strength of the pair bond in monogamous zebra finches, *Taeniopygia guttata,* likely because females either had increased difficulty identifying mates or pair-bond maintenance calls were masked.[21] The broader consequence of this finding is that females in noisy areas may be more likely to copulate with extra-pair partners, and this in turn can change the social and genetic dynamics of a population.

In other research, birds have been found to change their songs and calls in response to noise in urban areas, which may reduce masking of communication.[22] However, the consequences of this vocal adjustment on reproduction in a species remain unclear. One outcome may be that populations using urban dialects have a better chance to thrive in urban areas. But by the same token they may experience a decrease in mate recognition and/or gene flow with populations in non-urban areas.[23]

Beyond interfering with communication, introduced background noise can also mask the sounds of approaching predators or prey, and increase the perception of risk from predation. Studies have yet to compare predation rates or hunting success in noisy and quiet areas while controlling for other confounding factors. The degree to which noise affects predator/prey relations

[19] Lohr et al.*,* *supra* note 5.

[20] M.A. Bee & E.M. Swanson, *Auditory Masking of Anuran Advertisement Calls by Road Traffic Noise*, 74 ANIMAL BEHAV. 1765–1776 (2007); Henrik Brumm, *The Impact of Environmental Noise on Song Amplitude in a Territorial Bird*, 73 J. ANIMAL ECOLOGY 434–440 (2004); L. Habib et al., *Chronic Industrial Noise Affects Pairing Success and Age Structure of Ovenbirds* Seiurus aurocapilla, 44 J. APPLIED ECOLOGY 176–184 (2007); Frank E. Rheindt, *The Impact of Roads on Birds: Does Song Frequency Play a Role in Determining Susceptibility to Noise Pollution?*, 144 J. ORNITHOLOGIE 295–306 (2003).

[21] J.P. Swaddle & L.C. Page, *Increased Amplitude of Environmental White Noise Erodes Pair Preferences in Zebra Finches: Implications for Noise Pollution*, 74 ANIMAL BEHAV. 363–368 (2007).

[22] Slabbekorn & Ripmeester, *supra* note 10; Brumm, *supra* note 15; Hans Slabbekoorn & Margriet Peet, *Birds Sing at a Higher Pitch in Urban Noise*, 424 NATURE 267 (2003); William E. Wood & Stephen M. Yezerinac, *Song Sparrow (*Melozpiza melodia*) Song Varies with Urban Noise*, 123 THE AUK 650–659 (2006).

[23] Patricelli & Blickley, *supra* note 10; Warren et al. *supra* note 10; Slabbekoorn & Peet, *supra* note 17.

in any species, therefore, remains largely unexplored.[24] One study found that birds nesting near noisy natural gas pads had higher nesting success, likely due to reduced presence of the most common nest predator, the western scrub jay.[25] As suggested by these authors, the higher nesting success of birds in noisy areas provides a mechanism by which noise-tolerant species could become more common in a noisy world. Noise also causes short-term disruptions in behavior, such as startling or frightening animals away from food or other resources.[26]

In addition to the acute effects of noise, animals may suffer chronic effects, including elevated stress levels and associated physiological responses. Over the short term, chronic stress can result in elevated heart rate.[27] Longer-term stress can be associated with the ability to resist disease, survive, and successfully reproduce.[28] Good measures of chronic stress come from elevated stress hormones, like corticosterone, in blood or fecal samples.[29] In noise-stressed laboratory rats, elevated corticosterone was linked with reduced food consumption and decreased weight gain,[30] raising the possibility that for some individuals there may be longer-term welfare and survival consequences from the elevated stress associated with noise introduction.

3.2 Population Level Impacts

The cumulative impacts of noise on individuals can manifest at the population level in various ways that can potentially range from population declines up to

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[24] Quinn found that chaffinchs (*Fringilla coelebs*) perceived an increased risk of predation while feeding in noisy conditions, likely due to a reduced ability to detect auditory cues from potential predators. L. Quinn et al., *Noise, Predation Risk Compensation and Vigilance in the Chaffinch* Fringilla coelebs, 37 J. AVIAN BIOL. 601–608 (2006). Research on greater sage-grouse also highlights the potential for noise to contribute to predation. One of the methods for capturing sage-grouse is to mask the sound of researcher footfalls using a noise source such as a stereo or a chain saw. With such masking, the grouse can be easily approached and netted in their night roosts for banding or blood sampling. Presumably, predators would be equally fortunate in noisy areas, though the ability of predators to use acoustic cues for hunting could be diminished by masking as well.

[25] Clinton D. Francis et al., *Noise Pollution Changes Avian Communities and Species Interactions*, 19 CURRENT BIOL. 1–5 (2009).

[26] Dooling & Popper, *supra* note 1; N. Kempf & O. Huppop, *The Effects of Aircraft Noise on Wildlife: A Review and Comment*, 137 J. ORNITHOLOGIE 101–113 (1996); D.K. Delaney et al., *Effects of Helicopter Noise on Mexican Spotted Owls*, 63 J. WILDLIFE MGMT. 60–76 (1999); L.A. Rabin, R.G. Coss, & D.H. Owings, *The Effects of Wind Turbines on Antipredator Behavior in California Ground Squirrels* (Spermophilus beecheyi), 131 BIOL. CONS. 410–420 (2006).

[27] Weisenberger et al., *supra* note 4.

[28] J.C. Wingfield & R.M. Sapolsky, *Reproduction and Resistance to Stress: When and how*, 15 J. NEUROENDOCRINOL, 711 (2003); A. Opplinger et al., *Environmental Stress Increases the Prevalence and Intensity of Blood Parasite Infection in the Common Lizard* Lacerta vivipara, 1 ECOLOGY LETTERS 129–138 (1998).

[29] Wingfield & Sapolsky, *supra* note 23; S.K. Wasser et al., *Noninvasive Physiological Measures of Disturbance in the Northern Spotted Owl*, 11 CONS. BIOL. 1019–1022 (1997); D.M. Powell et al., *Effects of Construction Noise on Behavior and Cortisol Levels in a Pair of Captive Giant Pandas (*Ailuropoda melanoleuca*)*, 25 ZOO BIOL. 391–408 (2006).

[30] P. Alario et al., *Body Weight Gain, Food Intake, and Adrenal Development in Chronic Noise Stressed Rats*, 40 PHYSIOL. BEHAV. 29–32 (1987).

regional extinction. If species already threatened or endangered due to habitat loss avoid noisy areas and abandon otherwise suitable habitat because of a particular sensitivity to noise, their status becomes even more critical. As discussed below, numerous studies have documented reduced habitat use and lower breeding success in noisy areas by a variety of animals.[31]

4. MEASURING THE IMPACTS OF NOISE ON SPECIES OF CONCERN

Species vary widely in their ability to tolerate introduced noise and can exhibit very different responses to altered acoustic environments. This variability in response to noise makes generalizations about noise impacts among species and among noise sources difficult. Generalizations relevant to a single species can also be hard to make, because the ability to tolerate noise may vary with reproductive status, prior exposure to noise, and the presence of other stressors in the environment. This is why more measurements of noise impacts and associated variables are needed for a wider range of species.

Measuring the effects of noise at the individual and population levels is, however, extremely challenging. As we noted earlier, noise is typically accompanied by other changes in the environment that may also have physiological, behavioral, and population level effects. For example, habitat fragmentation is a side effect of road development, and fragmentation alone has been shown to cause population declines and changes in communication and other behaviors.[32] So, can we measure the impacts of noise on wildlife in ways that will support biologically relevant noise standards?

[31] Affected animals include birds, mammals, reptiles, and amphibians. Forman et al., *supra* note 6; Rheindt, *supra* note 15; Rien Reijnen et al., *The Effects of Car Traffic on Breeding Bird Populations in Woodland. III. Reduction of Density in Relation to the Proximity of Main Roads*, 32 J. APPLIED ECOLOGY 187–202 (1995); Rien Reijnen et al., *The Effects of Traffic on the Density of Breeding Birds in Dutch Agricultural Grasslands*, 75 BIOL. CONS. 255–260 (1996); S.J. Peris & M. Pescador, *Effects of Traffic Noise on Passerine Populations in Mediterranean Wooded Pastures*, 65 APPLIED ACOUSTICS 357–366 (2004); R.T.T. Forman & L.E. Alexander, *Roads and Their Major Ecological Effects*, 29 ANN. REV. ECOLOGY SYSTEMATICS 207–231 (1998); E. Stone, *Separating the Noise from the Noise: A Finding in Support of the "Niche Hypothesis," That Birds Are Influenced by Human-Induced Noise in Natural Habitats*, 13 ANTHROZOOS 225–231 (2000); Ian Spellerberg, *Ecological Effects of Roads and Traffic: A Literature Review*, 7 GLOBAL ECOLOGY BIOGEOG. LETTERS 317–333 (1998); David Lesbarrères et al., *Inbreeding and Road Effect Zone in a Ranidae: The Case of Agile Frog,* Rana dalmatina *Bonaparte 1840*, 326 COMPTES RENDUS BIOLOGIES 68–72 (2003).

[32] See, *e.g*., Jeffrey A. Stratford & W. Douglas Robinson, *Gulliver Travels to the Fragmented Tropics: Geographic Variation in Mechanisms of Avian Extinction*, 3 FRONTIERS ECOLOGY & ENV'T 91–98 (2005); P. Laiolo & J. L. Tella, *Erosion of Animal Cultures in Fragmented Landscapes*, 5 FRONTIERS ECOLOGY & ENV'T 68–72 (2007).

4.1 The Observational Approach

4.1.1 Relating wildlife abundance to noise levels

Much of the evidence for noise impacts on animals comes from field observations of animal density, species diversity, and/or reproductive success in relation to noise sources. Most studies focus on the presence or absence of wildlife near roads, finding lower population densities of many birds,[33] lower overall diversity for birds, reptiles, and amphibians,[34] and road avoidance in large mammals.[35] Most of this work does not separate the impacts of noise from other road effects or measure spatial and temporal variations in noise levels along transects where animals were studied.

One influential series of studies in the Netherlands did find, however, a negative relationship between noise exposure along roadways and both bird diversity and breeding densities.[36] Noise exposure better explained decreased density and diversity than either visual or chemical disturbance. These Dutch studies have been criticized for research design and statistical analysis problems,[37] underscoring the fact that researchers in different countries have different assumptions about how to measure noise and evaluate its impacts.[38] On their own, the Dutch studies are an inadequate basis for establishing internationally standardized noise regulations, but they are among the few analyses that set measurements of noise levels beside data on species presence/absence and diversity.

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[33] Forman & Deblinger, *supra* note 3; Rheindt, *supra* note 15; Peris & Pescador, *supra* note 26; M. Kuitunen et al., *Do Highways Influence Density of Land Birds?* 22 ENVTL. MGMT. 297–302 (1998); A.N. van der Zande et al., *The Impact of Roads on the Densities of Four Bird Species in an Open Field Habitat—Evidence of a Long-Distance Effect*, 18 BIOL. CONS. 299–321 (1980).

[34] C.S. Findlay & J. Houlahan, *Anthropogenic Correlates of Species Richness in Southeastern Ontario Wetlands*, 11 CONS. BIOL. 1000–1009 (1997).

[35] Studies in large mammals typically find road avoidance, but many small mammals are found in higher densities near roads, due to increased dispersal and reduced numbers of predators. Forman & Deblinger, *supra* note 3; F. J. Singer, *Behavior of Mountain Goats in Relation to US Highway 2, Glacier National Park, Montana*, 42 J. WILDLIFE MGMT. 591–597 (1978); G.R. Rost & J.A. Bailey, *Distribution of Mule Deer and Elk in Relation to Roads*, 43 J. WILDLIFE MGMT. 634–641 (1979); L.W. Adams & A.D. Geis, *Effects of Roads on Small Mammals*, 20 J. APPLIED ECOLOGY 403–415 (1983).

[36] Reijnen et al., *supra* note 29; R. Foppen & R. Reijnen, *The Effects of Car Traffic on Breeding Bird Populations in Woodland. II. Breeding Dispersal of Male Willow Warblers (*Phylloscopus trochilus*) in Relation to the Proximity of a Highway*, 31 J. APPLIED ECOLOGY 95–101 (1994).

[37] N. Sarigul-Klign, D.C. Karnoop, & F.A. Bradley, *Environmental Effect of Transportation Noise. A Case Study: Criteria for the Protection of Endangered Passerine Birds, Final Report* (Transportation Noise Control Center (TNCC), Department of Mechanical and Aeronautical Engineering, University of California, Davis, 1977); G. Bieringer & A. Garniel, Straßenalärm und Vögel—eine kurze Übersicht über die Literatur mit einer Kritik einflussreicher Arbeiten. Bundesministerium für Verkehr, Innovation und Technologie. Schriftenreihe Straßenforschung. Unpublished manuscript, Vienna, 2010 (copy on file with the authors).

[38] Noise is commonly measured in dBA SPL, a unit that is measured differently in different countries, making extrapolation difficult. Bieringer & Garniel, *supra* note 32.

The value of observational studies of presence/absence and diversity also needs to be assessed in context. One would not want to use information about reduced occupancy of a noisy area, for example, as the only indication that noise was having population-level impacts. It is conceivable that, if noise results in increased mortality or decreased reproduction, noisy areas could become population sinks,[39] and a detriment to conservation efforts across the range of the species. But this conclusion would be premature unless the presence/absence data are assessed in the context of other measures of impact, such as breeding success, stress response, startling and other behavioral changes.

So, while observational studies can be and have been helpful in identifying noise as a conservation problem, their policy relevance and value is constrained if they are unable to separate the effects of noise from the many other confounding disturbances that can affect animal densities near roads and other human development. When Fahrig et al.[40] documented reduced densities of frogs and toads near high traffic roads compared to low traffic roads, noise was a potential causal factor. After controlling for other variables, however, their evidence suggested that differences in density more likely reflected varying levels of traffic-associated road mortality.

One way to reduce, though not eliminate, the problem of confounding variables is to compare behaviors and other response variables in the presence and absence of noise. Animals can be observed, for example, before and after noise sources are introduced, or when noise is intermittent. This approach has been used to demonstrate the impact (or lack of impact) of noise from aircraft, machinery, and vehicles on animal behavior and reproductive success.[41] Spatial variation in noise may also allow researchers to control for some confounding factors. One study examined ovenbirds (*Seiurus aurocapilla)* along the edges of clearings containing either compressor stations or gas-producing wells.[42] Both clearings had a similar level of surface disturbance and human activity, but compressors produced high-amplitude noise whereas the wells were relatively quiet. Near compressors, the analysis found reduced pairing success and evidence that the habitat was non-preferred.[43]

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[39] Sinks are areas where successful reproduction is insufficient to maintain the population without immigration. H.R. Pulliam, *Sources, Sinks, and Population Regulation*, 132 AM. NATURALIST 652–661 (1988).

[40] L. Fahrig et al., *Effect of Road Traffic on Amphibian Density*, 73 BIOL. CONS. 177–182 (1995).

[41] Delaney et al., *supra* note 24; D. Hunsaker, J. Rice, & J. Kern, *The Effects of Helicopter Noise on the Reproductive Success of the Coastal California Gnatcatcher*, 122 J. ACOUSTICAL SOC. AM. 3058 (2007); Jennifer W. C. Sun & Peter M. Narins, *Anthropogenic Sounds Differentially Affect Amphibian Call Rate*, 121 BIOL. CONS. 419–427 (2005).

[42] L. Habib, E.M. Bayne, & S. Boutin, *Chronic Industrial Noise Affects Pairing Success and Age Structure of Ovenbirds* Seiurus aurocapilla, 44 J. APPLIED ECOLOGY 176–184 (2007).

[43] Habib et al. found an increased proportion of juveniles in noisy areas, suggesting that the area is undesirable for breeding adults. *Id.*

An additional observational approach is to include noise as a factor in habitat-selection models. These spatially explicit models, typically produced in GIS (Geographic Information Systems), relate species distribution data to information about landscape characteristics in order to determine the impact of disturbance or habitat quality on habitat usage by wildlife.[44] Multiple habitat layers can be added to the model to determine what factors best predict habitat usage. While few studies have incorporated noise into these types of models, GIS layers of noise can readily be created using commercially available and freeware programs. These types of models may be the best option for measuring noise impacts on a large scale and can also be useful in predicting future areas of conflict with human activities.

Ideally, future observational studies encompassing a variety of noise sources, habitats, and species will measure noise exposure levels and then relate observed impacts to noise exposure while controlling for confounding variables. When effects cannot properly be controlled for in a single study design, a second-best choice is to use replicated studies and let statistical modeling separate out the impacts of noise. To date, only a handful of studies follow this approach.[45]

4.1.2 Estimating the masking potential of noise

There is a relatively simple technique for addressing possible noise impacts on signal detection. It involves estimating the potential of a noise source to mask communication signals and other important sounds, such as the sounds of predators or prey. Masking occurs when background noise is loud relative to the signal, such that it cannot be detected by the receiver.

The estimation of masking requires knowledge of the physiology and behavior of the organism and the nature of the noise. Masking is frequency-specific, so an acoustic signal will only be masked by the portion of the background noise that is in a similar frequency band as the signal.[46] An

[44] J.B. Dunning et al., *Spatially Explicit Population Models: Current Forms and Future Uses*, 5 ECOLOGICAL APPLICATIONS 3–11 (1995).

[45] Forman, Reineking, & Hersberger, *supra* note 6; Reijnen et al. (1995), *supra* note 29; Reijnen et al. (1996), *supra* note 29; Foppen & Reijnen, *supra* note 34; R. Reijnen & R. Foppen, *The Effects of Car Traffic on Breeding Bird Populations in Woodland. I. Evidence of Reduced Habitat Quality for Willow Warblers (*Phylloscopus trochilus*) Breeding Close to a Highway*, 31 J. APPLIED ECOLOGY 95–101 (1994).

[46] Lohr et al., *supra* note 8; E.A. Brenowitz, *The Active Space of Red-Winged Blackbird Song*, 147 J. COMP. PHYSIOLOGY 511–522 (1982); R.J. Dooling & B. Lohr, *The Role of Hearing in Avian Avoidance of Wind Turbines, in* PROC. NAT'L AVIAN-WIND PLANNING MEETING IV 115–134 (S.S. Schwartz ed., for the Avian Subcommittee, National Wind Coordinating Committee, 2001).

estimation of masking requires,[47] first, the audiogram of the focal species;[48] second, the absolute amplitude and frequency spectrum of the noise;[49] third, the absolute amplitude and frequency spectrum of the vocalization or sound of interest; and fourth, the critical ratio for the focal species.[50]

With this information, masking is estimated by determining how introduced noise changes the "active space" of the signal, which is the area around the sender where the signal can be detected by receivers.[51] Intuitively, there is less masking when signals have a different frequency profile than noise, when noise is quiet, when signals are loud and/or when animals are close together when communicating. Conversely, masking is most problematic when signal and noise have similar frequency profiles, when noise is loud, when calls are quiet, and/or when calls are used over large distances.[52]

There are, however, limitations to masking estimations. The method described addresses only the potential impacts of masking animal vocalizations or other sounds and cannot estimate other impacts of noise, such as startling or chronic stress. Further, in the absence of specific information about the auditory physiology and behaviors of the focal species, estimates of masking using this method may be either too conservative or too liberal. Estimates can be too conservative, for example, in situations in which the mere detection of a vocalization is an insufficient basis for extracting necessary information from the sound.[53] Estimates can be too liberal if as part of their communication

[47] For detailed methods on calculating masking potential, *see* R.J. Dooling & J.C. Saunders, *Hearing in the Parakeet (*Melopsittacus undulatus*): Absolute Thresholds, Critical Ratios, Frequency Difference Limens, and Vocalizations*, 88 J. COMP. PHYSIOL. 1–20 (1975).

[48] A measure of how hearing sensitivity varies with the frequency of the sound. In general, birds do not hear as well as mammals in very low or high frequencies, or use them to communicate. Dooling & Popper, *supra* note 1.

[49] A measure of how much energy is present in each frequency band of the sound.

[50] This is the difference in amplitude between signal and noise necessary for detection of the signal. For a generalized bird, the critical threshold ranges from approximately 26 to 28 dB between 2 and 3 kHz, meaning that a typical bird cannot hear a 2–3 kHz vocalization unless the vocalization exceeds the background noise in that frequency range by 26–28 dB. In general, birds have higher critical ratios than mammals, making them worse at discriminating signals in noise. If measurements for these parameters are not available for the focal species, then information from closely related species may be used as a substitute. However, this may be misleading if the species of interest has particularly strong or poor hearing capabilities relative to the substitute species. Dooling & Popper, *supra* note 1; Lohr et al., *supra* note 8; Dooling & Saunders, *supra* note 45.

[51] Lohr et al., *supra* note 5; Brenowitz, *supra* note 39.

[52] Lohr et al., *supra* note 5; Bee & Swanson, *supra* note 15; G. Ehret & H.C. Gerhardt, *Auditory Masking and Effects of Noise on Responses of the Green Treefrog (*Hyla cinerea*) to Synthetic Mating Calls*, 141 J. COMP. PHYSIOL. A 13–18 (1980); T. Aubin & P. Jouventin, *Cocktail-Party Effect in King Penguin Colonies* 265 PROC. R. SOC. B 1665–1673 (1998).

[53] This would happen when humans can detect human voices, but not discriminate the identity of the speaker or the words being said. *See* Lohr et al., *supra* note 5, for a discussion of the difference between detection and discrimination.

animals use spatial cues,[54] co-modulation of frequencies,[55] or adjust their vocalizations to reduce masking.[56]

Because so many factors affect the degree of masking, there is a critical need for additional field studies to validate estimation techniques. The available work relating the potential for masking to observed individual- and population-level impacts[57] is just not a sufficient basis for knowing whether masking potential is a reliable predictor of how noise will impact wildlife. If the predictive power of measuring masking potential can be shown, researchers will then have a low-cost tool for predicting impacts in species about which little is known. Otherwise, masking analysis is most informative when used in concert with field studies that assess actual noise impacts. If a disruption of communication or decreased rates of prey capture in noisy areas can be demonstrated, then an analysis of the masking potential of a new noise source could be used to determine the area over which individuals are likely to be affected by that new source.[58]

4.2 The Experimental Approach

Experimental manipulations of noise in the laboratory and the field are more powerful than observational studies in isolating the effects of noise and identifying the underlying causes of noise impacts because they deal more effectively with the problem of controlling for confounding variables. The following sections discuss their advantages and limitations.

4.2.1. Laboratory experiments

Laboratory studies introduce noise to captive animals and measure the impacts in a controlled environment. Studies using captive animals are the basis for much of what we know about the hearing range and sensitivity of a number of animal taxa[59] and about the ability of animals to detect and

[54] The ability to hear sounds is improved if they are separated spatially. M. Ebata, T. Sone, & T. Nimura, *Improvement of Hearing Ability by Directional Information*, 43 J. ACOUSTICAL SOC. AM. 289–297 (1968); J.J. Schwartz & H.C. Gerhardt, *Spatially Mediated Release From Auditory Masking in an Anuran Amphibian*, 166 J. COMP. PHYSIOL. A 37–41 (1989).

[55] Masking is reduced when the noise has amplitude modulation patterns that make it distinct from the signal. G.M. Klump & U. Langemann, *Co-Modulation Masking Release in a Songbird*, 87 HEARING RES. 157–164 (1995).

[56] Patricelli & Blickley, *supra* note 10; Rabin & Greene, *supra* note 10; Warren et al., *supra* note 10; Slabbekoorn & Peet, *supra* note 17.

[57] Rheindt, *supra* note 18.

[58] Lohr et al., *supra* note 8.

[59] Dooling & Saunders, *supra* note 45; K. Okanoya & Robert F. Dooling, *Hearing in the Swamp Sparrow,* Melospiza georgiana, *and the Song Sparrow,* Melospiza melodia, 36 ANIMAL BEHAV. 726–732 (1988); H.E. Heffner et al., *Audiogram of the Hooded Norway Rat*, 73 HEARING RES. 244–247 (1994); H.E. Heffner & R.S. Heffner, *Hearing Ranges of Laboratory Animals*, 46 J. AM. ASS'N LABORATORY ANIMAL SCI. 20–22 (2007).

discriminate sounds in the presence of background noise.[60] These psychoacoustic studies are critical for assessing masking potential, and provide a physiological and morphological basis for predicting which species are most likely to be impacted by introduced noise.[61] Laboratory studies also provide insight into the physiological and behavioral impacts of noise, and the potential consequences of masking for breeding individuals.[62] As noted earlier, they demonstrate impacts on pair-bonding[63] and the amplitude at which vocalizations are produced.[64] They do not address, however, the long-term consequences of these behavioral changes, which remain unclear and need further study both in the laboratory and in the field.

Traditionally, psychoacoustic studies use white noise or pure tones to measure hearing ability and noise effects.[65] Recent studies also address the effects of anthropogenic noise directly, increasing their relevance to conservation. Lohr and colleagues, for example, measured the masked thresholds of natural contact calls for budgerigars (*Melopsittacus undulates)* and zebra finches, in the lab using simulated traffic noise, allowing them to predict how traffic noise affects the distance at which vocalizations can be detected by receivers.[66]

The environmental control that gives laboratory studies their analytic power can also be a disadvantage, if there is reason to believe that the response of animals to noise in a laboratory setting will be different from that of animals in the wild, where natural variations in the environment and in animal populations can affect the impact of noise. When increased physiological stress from noise is experienced, for example, in combination with habitat loss, synergistic effects on animals will magnify the overall impact of development.

Laboratory studies also must be careful not to extrapolate findings from animals that thrive in captivity to endangered animals, particularly since the

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[60] Lohr et al., *supra* note 8; Dooling & Saunders, *supra* note 45; Klump & Langemann, *supra* note 53; L. Wollerman, *Acoustic Interference Limits Call Detection in a Neotropical frog* Hyla ebraccata, 57 ANIMAL BEHAV. 529–536 (1999).

[61] Dooling & Popper, *supra* note 1.

[62] Marler et al., *supra* note 14; Ryals et al., *supra* note 14; J. Syka & N. Rybalko, *Threshold Shifts and Enhancement of Cortical Evoked Responses After Noise Exposure in Rats*, 139 HEARING RES. 59–68 (2000); D. Robertson & B.M. Johnstone, *Acoustic Trauma in the Guinea Pig Cochlea: Early Changes in Ultrastructure and Neural Threshold*, 3 HEARING RES. 167–179 (1980).

[63] Swaddle & Page, *supra* note 19.

[64] J. Cynx, et al., *Amplitude Regulation of Vocalizations in Noise by a Songbird,* Taeniopygia guttata, 56 ANIMAL BEHAV. 107–113 (1998); Marty L. Leonard & Andrew G. Horn, *Ambient Noise and the Design of Begging Signals*, 272 PROC. R. SOC. B 651–656 (2005). This finding has been corroborated with studies of birds in the field in Brumm, *supra* note 18.

[65] Dooling & Saunders, *supra* note 45; Klump & Langemann, *supra* note 53; Wollerman, *supra* note 53; J.B. Allen & S.T. Neely, *Modeling the Relation between the Intensity Just-Noticeable Difference and Loudness for Pure Tones and Wideband Noise*, 102 J. ACOUSTICAL SOC. AM. 3628–3646 (1997).

[66] Lohr et al., *supra* note 8. For other studies that introduce anthropogenic noise, see Weisenberger et al., *supra* note 7; Bee & Swanson, *supra* note 18.

animals chosen for laboratory study are often domesticated or otherwise show tolerance for human disturbance. Endangered animals, by contrast, are often driven to rarity due to their inability to tolerate environmental change, which may include sensitivity to noise.[67] The use of surrogate species would be unnecessary if the species of concern could be tested in the lab for noise response. But small population sizes and narrow tolerances often make it impossible to bring threatened or endangered species into the lab for such tests.

The use of anthropogenic noise in laboratory studies of noise effects, particularly noise that is likely to be affecting wild animals, increases the conservation applicability of such research and should be a future priority. Laboratory experiments must also be supplemented with field studies and other methods to fully understand the impacts of noise on wildlife.

4.2.2. Noise introduction experiments in the field

Field experiments are another method for isolating and quantifying the impacts of noise on animals under natural conditions. The controlled introduction of noise can be accomplished either by creating noise in the field or by playing back the associated noise through speakers. The first approach has been used to investigate the impacts on wildlife of aircraft, machinery, and vehicles.[68] As is the case with observational studies, interpretations of this type of research are complicated by the problem of controlling for confounding variables, such as the visual and other disturbances, in addition to noise, associated with many sorts of environmental change. Compared to observational studies, however, field experiments offer greater opportunities to examine interactions among multiple associated stressors. They are also generally a more efficient use of scarce research resources and provide the ability to control for (or examine) seasonal effects, time-of-day effects, and other factors influencing responses to noise.

The second experimental approach, playing back noise that has been recorded from a source of interest or synthesized to match that source,[69] has the advantage that noise effects can be easily separated from other aspects of disturbance. Because noise introduction on a large spatial and temporal scale is logistically challenging in natural habitats, studies to date have been short-term and relatively small in scale. A short-term experiment may be appropriate

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[67] T. Caro, J. Eadie, & A. Sih, *Use of Substitute Species in Conservation Biology*, 19 CONS. BIOL. 1821–1826 (2005).

[68] Delaney, et al., *supra* note 24; P. R. Krausman, et al., *Effects of Jet Aircraft on Mountain Sheep*, 62 J. WILDLIFE MGMT. 1246–1254 (1998); A. Frid, *Dall's Sheep Responses to Overflights by Helicopter and Fixed-Wing Aircraft*, 110 BIOL. CONS. 387–399 (2003).

[69] Sun & Narins, *supra* note 39; A.L. Brown, *Measuring the Effect of Aircraft Noise on Sea Birds*, 16 ENV'T INT'L 587–592 (1990).

for studying dynamic behaviors, such as call rate, startling, or avoidance,[70] but cannot address the longer-term individual- or population-level consequences of noise.

To illustrate study design for a long-term and large-scale noise introduction experiment, we describe our ongoing experiment in Wyoming, addressing the noise impacts of energy development on greater sage-grouse.

4.2.2.1 Noise impacts on sage-grouse: A long-term field experiment

Populations of this species are declining throughout their range in the interior West of the United States,[71] enough to merit consideration for listing under the federal Endangered Species Act. Coal-bed methane (CBM) and deep natural gas extraction are increasing rapidly in sage-grouse habitats, and recent studies document dramatic declines in sage-grouse populations in areas of energy development.[72] However, incomplete knowledge of the causes of these declines is hampering the creation of effective management strategies.

Among the number of disturbances associated with energy development that impact sage-grouse, noise is particularly problematic in breeding areas downwind of development when it causes declines in male attendance, although attendance was not affected by visual disturbance from development.[73] In addition, the life history of sage-grouse makes them particularly vulnerable to disturbance from noise pollution. In the breeding season, males gather on communal breeding grounds (leks) to perform complex acoustic displays, used by females to locate leks and choose mates. The risk is that anthropogenic noise in sage-grouse habitat masks male vocalizations and interferes with reproduction. While there are rules governing the noise emitted during drilling of natural gas wells, exemptions are often granted and there has been little research demonstrating that stipulated noise levels reduce the impacts of development on sage-grouse, as well as other sensitive species.

Our multi-year, noise-introduction experiment on sage-grouse leks in an otherwise undisturbed area tries to separate the impacts of noise from other potential impacts of energy development. Two types of noise are of

[70] Weisenberger et al., *supra* note 7; Sun & Narins, *supra* note 39; Leonard & Horn, *supra* note 62; Brown, *supra* note 67.

[71] J.W. Connelly et al., Conservation Assessment of Greater Sage-Grouse and Sagebrush Habitats, Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming, 2004. Copy online at http://www.ndow.org/wild/conservation/sg/resources/greate_sg_cons_assessment.pdf

[72] M.J. Holloran, Greater Sage-Grouse (*Centrocercus urophasianus*) Population Response to Natural Gas Field Development in Western Wyoming (2005) (unpublished Ph.D. dissertation, University of Wyoming) (accessible online from http://www.sagebrushsea.org/th_energy_sage_grouse_study2.htm); Brett L. Walker et al., *Greater Sage-Grouse Population Response to Energy Development and Habitat Loss*, 71 J. WILDLIFE MGMT. (2007); Dooling & Popper, *supra* note 1.

[73] Other factors at work include habitat loss, fragmentation, dust, air pollution, and West Nile virus. Connelly et al, *supra* note 64; Holloran, *supra* note 70; D.E. Naugle et al., *West Nile Virus: Pending Crisis for Greater Sage-Grouse*, 7 ECOLOGY LETTERS 704–713 (2004).

primary interest, road noise and drilling noise. Both types are dominated by low frequencies, but drilling noise is high intensity, continuous noise, whereas road noise is intermittent with gradual increases and decreases in amplitude. Monitored leks are divided into pairs of control leks and leks with experimentally introduced noise.[74] Ideally, noise would be introduced at different levels on different leks to determine the noise threshold at which an impact can be observed. However, such a "dose-response" experiment would require a large sample of leks and that is logistically infeasible. The experiment, instead, creates a noise gradient across each lek, so that the effect of noise level on microhabitat use and behavior can be measured and noise-tolerance thresholds estimated.

This experimental approach isolates and makes it possible to assess the impacts of noise on lekking sage-grouse at both the individual and population levels. The individual effects are analyzed from audio and video recordings, to determine whether individuals change the rate, frequency structure, and amplitude of their displays in the presence of noise, as has been found in other species.[75] A non-invasive technique compares the relative stress levels of birds on experimental and control leks through analysis of stress hormones in feces.[76] Population-levels effects of noise derive from comparison of lek attendance patterns on experimental and control leks over multiple seasons. This allows detection of noise impacts while controlling for natural variations in behavior, physiology, and larger-scale fluctuations in the population.

Although introducing noise in the wild is a powerful tool for measuring noise impacts on animals, it is only appropriate in certain circumstances. Noise introduction requires access, for example, to a population of animals residing in a relatively undisturbed area. Such a population may be unavailable in some species of concern, or the species may be too sensitive or rare to risk such an experimental manipulation. In addition, animals must be at fairly high densities in order to collect sufficient data for analysis, because it is difficult to create a noise disturbance over a large area using speakers.[77] During the breeding season, noise introduction can rely on battery-powered speakers, because leks are relatively small and have a high density of birds. This same

[74] Paired leks have similar size and location and are visited by researchers for counts on the same days. Noise is introduced at 70 dBF SPL (unweighted decibels) at 16 meters using three to four battery-powered outdoor speakers. This is similar to noise levels measured at $\frac{1}{4}$-mile from drilling rigs and main haul roads in Pinedale, Wyoming. Control leks have dummy speakers and are visited for "battery changes" with the same frequency as experimental leks.

[75] Patricelli & Blickley, *supra* note 13; Warren et al., *supra* note 13; Rabin et al., *supra* note 13; Rabin & Greene, *supra* note 13; Slabbekoorn & Peet, *supra* note 20.

[76] *See*, *e.g.,* Wasser et al., *supra* note 27.

[77] Most anthropogenic noise sources are very large, and it is extremely difficult to replicate loud noise over a large area from small speakers, since amplitude (and thus propagation) is limited by source size. This challenge is even greater when speakers are powered by batteries in remote field locations.

approach is less able, however, to address noise impacts on nesting or overwintering behaviors, when sage-grouse are more dispersed.

In some situations, the use of semi-captive populations reaps some of the benefits of both field and laboratory studies, by increasing animal density in a more natural setting than is afforded by laboratory animal colonies. This approach is outside the scope of our current study. Another limitation of the experimental approach is that it underestimates (or even misses) the impacts of noise that occur in interaction with other forms of disturbance, such as the combination of noise pollution with an increase of raptor perches in energy development areas.[78] The combined effects will be larger than that attributable to either disturbance alone, but they can only be examined in observational studies and noise-source introduction experiments. This highlights, again, the need for multiple research approaches to measuring wildlife noise impacts.

There are very few experimental studies that use either noise-source introductions or noise playbacks, even though these experimental tools, used in a field setting or in naturalistic captive settings, are among the most powerful for understanding noise impacts on wild populations. Large-scale field experiments are expensive and logistically challenging. They do, however, appear to be warranted, particularly when observational studies and measurements of masking potential suggest a likely role for noise in impacting wild animals. Future field research should also focus on validating results and methods from laboratory studies, thus increasing the ability to apply lab studies and estimates of masking potential to the development of effective mitigation measures and predictions about the impacts future development is likely to have on wildlife.

5. FUTURE DIRECTIONS AND POLICY RELEVANCE

Even though the rapid spread of human development and associated anthropogenic noise have impacts on wildlife, it is not always logistically, politically, or economically feasible to eliminate or even minimize noise. The more common policy approach is to set noise standards, in the hope of limiting the levels of noise that development produces. The production of noise can then be reduced structurally[79] or operationally[80] to meet these standards. Road noise, for example, can be reduced through the use of certain types of asphalt, although these road surfaces can also have lower durability, lower traction, and higher cost than noisier varieties. Road noise can also be decreased by noise barriers, but these may cut off migration routes and exacerbate rather than

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[78] Connelly et al., *supra* note 69.

[79] Noise can be reduced structurally by using alternative materials and architecture, such as noise barriers, to reduce sound production and propagation.

[80] Noise can be reduced operationally through limitations on the timing and frequency of noisy activities, for example, by avoiding shift changes that occur at 7:00 a.m., in the peak lekking hours of sage-grouse.

reduce overall road impacts.[81] Regulations necessarily balance the economic and environmental trade-offs involved in allowing development to proceed and as a general rule the more information that can be brought to bear on this balancing process the better.

There can be no doubt that the first priority in the development of most current noise standards is the protection of human welfare. They use human criteria of disturbance, generated primarily in areas where humans are impacted.[82] These standards protect animal species with noise tolerances and distributions similar to those of humans. They are not effective, however, in reducing the impacts of noise on sensitive species of wildlife. So what should be our goal in the development of effective noise standards for the protection of wildlife? Environmental managers typically prefer a single noise standard that covers all situations. But since species differ in their ability to tolerate noise, a single noise standard is bound to be conservative for some species and insufficient for others. [83] Simply erring on the side of more conservative standards could do more harm than good in cases where it diverts money from more appropriate types of mitigation, and when noise mitigation measures introduce other environmental and economic costs, as discussed above. Rather than a single standard, a set of standards is needed, based on the measured sensitivities of indicator species and species of concern in a particular habitat type or location. Recently, a panel of experts developed a set of general and species-specific recommendations for marine mammal noise exposure criteria.[84] The development of such a set of standards for terrestrial species will require information about sensitivity to noise pollution in both abundant and rare species; the research priorities outlined here will help to achieve this goal.

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[81] Forman, Reineking, and Hersberger, *supra* note 6.

[82] Dooling & Popper, *supra* note 1; SINGAL, *supra* note 9.

[83] A single noise standard, for example, might establish a maximum acceptable noise level of 49 dBA at a one quarter mile from a noise source.

[84] B.L. Southall, A.E. Bowles, & W.T. Ellison, *Marine Mammal Noise Exposure Criteria: Initial Scientific Recommendations*, 125 J. ACOUSTICAL SOC. AM. 2517 (2009). There is no equivalent set of recommendations for terrestrial animals.

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CHAPTER 3
POTENTIAL ACOUSTIC MASKING OF GREATER SAGE-GROUSE (*CENTROCERCUS UROPHASIANUS*) DISPLAY COMPONENTS BY CHRONIC INDUSTRIAL NOISE

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Abstract.—Anthropogenic noise can limit the ability of birds to communicate by masking their acoustic signals. Masking, which reduces the distance over which the signal can be perceived by a receiver, is frequency dependent, so the different notes of a single song may be masked to different degrees. We analyzed the individual notes of mating vocalizations produced by Greater Sage-Grouse (*Centrocercus urophasianus*) and noise from natural gas infrastructure to quantify the potential for such noise to mask Greater Sage-Grouse vocalizations over both long and short distances. We found that noise produced by natural gas infrastructure was dominated by low frequencies, with substantial overlap in frequency with Greater Sage-Grouse acoustic displays. Such overlap predicted substantial masking, reducing the active space of detection and discrimination of all vocalization components, and particularly affecting low-frequency and low-amplitude notes. Such masking could increase the difficulty of mate assessment for lekking Greater Sage-Grouse. We discuss these results in relation to current stipulations that limit the proximity of natural gas infrastructure to leks of this species on some federal lands in the United States. Significant impacts to Greater Sage-Grouse populations have been measured at noise levels that predict little or no masking. Thus, masking is not likely to be the only mechanism of noise impact on this species, and masking analyses should therefore be used in combination with other methods to evaluate stipulations and predict the effects of noise exposure.

Key words: acoustic masking, *Centrocercus urophasianus*, Greater Sage-Grouse, industrial noise.

Enmascaramiento Acústico Potencial de Mayor Sage-Grouse (*Centrocercus urophasianus*) Mostrar Componentes por Ruido Industrial Crónica

Resumen.— Antropógena ruido puede limitar la capacidad de las aves para comunicarse por enmascarar sus señales acústicas. Enmascaramiento, que reduce la distancia sobre la que se puede percibir la señal por un receptor, es frecuencia dependiente, por lo que las diferentes notas de una canción pueden enmascararse en diferentes grados. Analizamos las notas individuales de apareamiento vocalizaciones producidas por mayor Sage-Grouse (Centrocercus urophasianus) y el ruido de infraestructura de gas natural para cuantificar el potencial de tal ruido a vocalizaciones de mayor Sage-urogallo de máscara en distancias cortas y largas. Hemos encontrado que ruido producido por la infraestructura de gas natural fue dominado por las frecuencias bajas, con considerable superposición en frecuencia con pantallas acústicas de mayor Sage-urogallo. Tal superposición predijo enmascaramiento sustancial, reduciendo el espacio activo de detección y discriminación de todos los componentes de vocalización y que afectan particularmente a notas de baja frecuencia y baja amplitud. Estas máscaras podrían aumentar la dificultad de evaluación de mate para lekking mayor Sage-urogallo. Analizaremos estos resultados en relación con las actuales disposiciones que limitan la proximidad de la infraestructura de gas natural a leks de esta especie en algunas tierras federales en los Estados Unidos. Impactos

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significativos a las poblaciones de mayor Sage-urogallo han sido medidos en los niveles de ruido que predicen el enmascaramiento de poca o ninguna. Así, enmascaramiento no es probable que sea el único mecanismo de impacto de ruido en esta especie, y enmascaramiento análisis debe por lo tanto, utilizarse en combinación con otros métodos para evaluar las estipulaciones y predecir los efectos de la exposición al ruido. Así, enmascaramiento no es probable que sea el único mecanismo de impacto de ruido en esta especie, y enmascaramiento análisis debe por lo tanto, utilizarse en combinación con otros métodos para evaluar las estipulaciones y predecir los efectos de la exposición al ruido.

Birds use acoustic signals to communicate with conspecifics for a host of biologically important functions, including mate attraction, territory defense, parent–offspring communication, and predator avoidance. In order for this communication to be successful, the signal must travel from the signaler to the receiver through the local environment. The local physical and acoustic environment, therefore, plays an important role in determining the active space of a signal, the area in which a receiver can successfully perceive it (Brenowitz 1982, Dooling et al. 2009). Background noise, a conspicuous feature of most natural environments, can result in acoustic masking if this noise is loud in relation to the signal of interest. Animals have numerous acoustic and behavioral adaptations to maximize the active space of their signals in the presence of natural background noise. For example, the structural and temporal properties of many acoustic signals appear to be adapted to maximize the propagation distance and minimize masking from abiotic and biotic noise sources in the environment (Marten and Marler 1977, Wiley and Richards 1982, Ryan and Brenowitz 1985, Brumm 2006). However, the spread of humans into natural landscapes has resulted in the proliferation of anthropogenic noise sources, with the potential to affect many of the animal species that live and communicate in these environments (Barber et al. 2010). Acoustic signals that are adapted to deal with natural noise sources may still be susceptible to masking from anthropogenic noise sources if the anthropogenic noise differs enough from natural noise sources in frequency, duration, or daily or seasonal pattern.

Effective communication requires that a receiver be able to detect a given signal, discriminate that signal from other possible signals, and recognize features that may convey information about the specific signaler. The active space of a signal may be different for each of these receiver tasks (Lohr et al. 2003). Detection provides the receiver with the lowest level of information—simply that a signal is present—and requires the lowest contrast between the signal and background noise. For a signal to be successfully detected in a noisy environment requires that the ratio of the signal to the background noise (i.e., signal-to-noise ratio [SNR], the difference between signal and noise amplitudes measured in decibels) within a frequency band exceed a critical detection threshold (Klump 1996). The critical detection threshold for a "typical bird" ranges from 18 dB to 37 dB across frequency bands. Discrimination of the signal from other signals, as would be required to identify the species of the sender or the functional category of the signal, requires a higher SNR than detection. In a laboratory study of two bird species, Lohr et al. (2003) found that discrimination of conspecific song required an SNR approximately 3 dB higher than the levels required for detection. An even more challenging task for a receiver is signal recognition, discerning variation among signals within a category, such as information about individual identity or reproductive quality. For example, receivers may use the acoustic features of the signal such as frequency structure, relative amplitude of notes, and note duration to recognize the identity of the signaling individual. Signal recognition may require an even higher SNR (Dooling and Popper 2007); however, we do not yet know how much higher the signal must be for recognition to occur.

The fitness consequences of being able to detect a signal versus discriminate or recognize a signal is likely to be signal specific. For example, a predator alert call, which functions to alert a conspecific to danger, may be effective so long as it exceeds the critical ratio for detection. However, a mate-attraction call that is used by females to assess the quality of a potential mate may need to exceed the critical recognition threshold in order to be effective. For example, the ability to recognize individual signals is critical to mate choice in the Swamp Sparrow (*Melospiza georgiana*): females use song features such as trill rate and frequency bandwidth to assess the quality of potential mates (Ballentine et al. 2004). Introduced noise has been demonstrated to weaken pair bonds in captive Zebra Finches (*Taeniopygia guttata*; Swaddle and Page 2007), which suggests that reduced recognition can have fitness consequences.

Active space can vary within a given signal as well as among signals. Many bird vocalizations are highly complex and are composed of multiple acoustic components (bouts, phrases, syllables, or notes). Some multicomponent signals may encode either distinct ("multiple messages hypothesis") or redundant ("redundancy hypothesis") information about the signaler (Møller and Pomiankowski 1993, Hebets and Papaj 2005). For example, the trill note and note complex of White-crowned Sparrow (*Zonotrichia leucophrys*) song each convey distinct information about dialect and individual identity, respectively (Nelson and Poesel 2007). Each component can vary in frequency structure, duration, and relative amplitude; these factors interact with the local physical and acoustic environment to determine the active space of the signal component (Patricelli et al. 2008). The result of this variation is that each component of a complex vocalization may have a different active space and be uniquely susceptible to masking by a given noise source.

Anthropogenic noise is typically dominated by low frequencies, so low-frequency signal components and features are most susceptible to masking (Brumm and Slabbekoorn 2005, Slabbekoorn and Ripmeester 2008). Even if a signal is not completely masked, low-frequency background noise could distort a signal, resulting in a higher-frequency note being perceived as having higher relative amplitude than a masked lower-frequency note. Such distortion could result in increased difficulty in assessment or identification.

Our focal species, the Greater Sage-Grouse (*Centrocercus urophasianus*; hereafter "sage-gouse"), is a medium-bodied gallinaceous bird that has long been used as a model system for studies of sexual selection and communication (Wiley 1973; Gibson 1989, 1996). During the breeding season, males gather on strutting grounds (leks) where they establish small display territories that are visited by females for courtship. Males produce a complex visual and acoustic display. Sound is critical to the breeding system on both large and small spatial scales because females use the acoustic component of the display to locate strutting males and, once on a lek, to select a male (Gibson 1989, 1996; Patricelli and Krakauer 2010).

The sage-grouse vocal display is composed of three major note types: a series of low-frequency "coo" notes, two broadband "pops," and a frequency-modulated "whistle" (Fig. 1). The rate of display (strut rate) is positively correlated with male success in mating (Gibson and Bradbury 1985, Gibson 1996, Patricelli and Krakauer 2010). In addition, the time interval between the two pop notes during which the whistle note occurs, the inter-pop interval (IPI), is positively correlated with mating success (Gibson et al. 1991, Gibson 1996). This suggests that assessment of the two pop notes might be particularly critical in female mating decisions. Whistles may also be important in female choice. Gibson and Bradbury (1985) found that the time interval from the first pop to the whistle peak as well as the maximum frequency of the whistle at the apex are related to male mating success. Female sage-grouse also may assess amplitude of the whistle; unpublished results suggest that whistle amplitude may be positively correlated with mating success (J. W. Bradbury pers. comm.), and males orient during courtship so that the highly directional whistle is beamed toward females (Dantzker et al. 1999). This female preference for male-display quantity

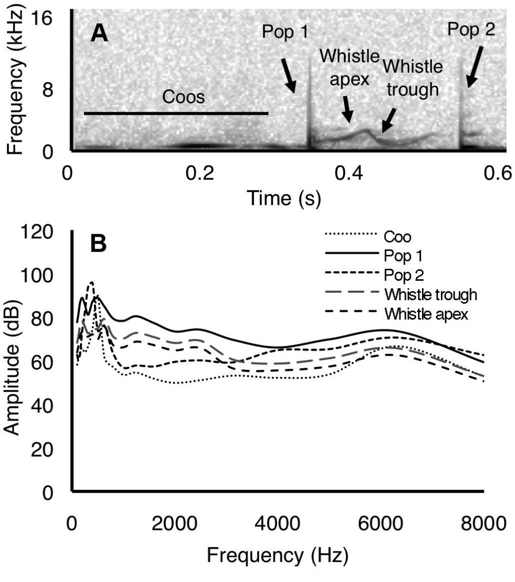

FIG. 1. Spectrogram and (B) power spectra of a male Greater Sage-Grouse strut display with distinct display components labeled. Low-frequency coos are followed by a broadband pop (pop 1), a frequency-modulated whistle with an apex of ~2,500 Hz (whistle apex) and a minimum of ~630 Hz (whistle trough), and another broadband pop (pop 2).

and quality suggests that masking of one or all of these notes by background noise may negatively affect a female's ability to assess males on the lek.

Sage-grouse populations are declining across their range (Connelly et al. 2004, Garton et al. 2011), leading sage-grouse to be listed as endangered under Canada's Species at Risk Act and designated as a candidate species for listing in the United States under the federal Endangered Species Act. Natural gas development has expanded rapidly over the past decade and has been implicated in contributing to population declines (Holloran 2005, Walker et al. 2007, Copeland et al. 2009, Holloran et al. 2010). In particular, noise associated with energy development has been demonstrated to result in reduced attendance on leks (Blickley et al. 2012) and is associated with increased stress hormones in males on noisy leks (J. L. Blickley and G. L. Patricelli unpubl. data). Masked communication has been suggested as a mechanism of this impact, so understanding the potential for introduced noise sources to mask signals used in mating could lead to improved management of vulnerable sage-grouse populations.

The present study addresses the potential for noise pollution from natural gas development to mask or distort acoustic signals that are used in breeding by sage-grouse. We analyzed the individual acoustic components of sage-grouse vocalizations (Fig. 1) and noise from natural gas infrastructure (a compressor station, generator, and drilling rig; Fig. 2) to quantify the potential for such noise to mask sage-grouse vocalizations over both long and short distances. We compared the effect of such noise on the level of both detection and discrimination and discuss the utility of this approach for predicting the impacts of noise on this and other species. For the masking analysis, we focused primarily on noise measurements at 75 m and 400 m (~1/4 mile), which represent a typical distance to the edge of surface disturbance (the pad) from a compressor station or drilling rig and the distance stipulated as the minimum surface-disturbance buffer around leks in our study region, respectively (Bureau of Land Management 2008).

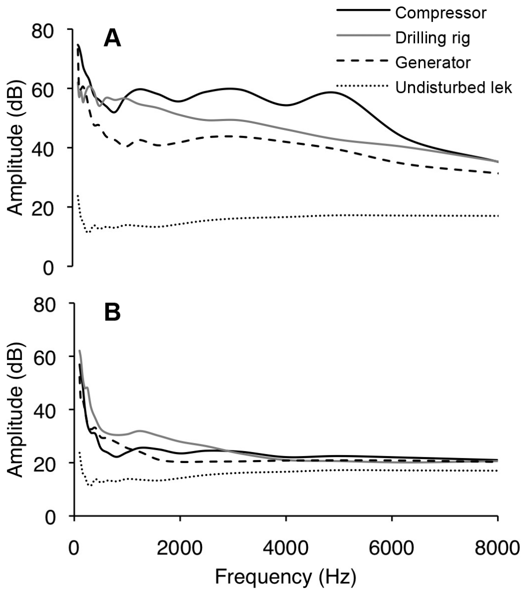

FIG. 2. Power spectra of ambient noise levels at (A) 75 m and (B) 400 m from a natural gas compressor station, natural gas drilling rig, and generator in Sublette County, Wyoming, and on an undisturbed lek (quiet) in Fremont County, Wyoming. Values were interpolated if a measurement for that distance was not available. Noise was dominated by low frequencies at both short and medium distances from the source.

Methods

Field recordings and measurements.—Between 1 and 5 May 2010, we collected field recordings and vocal amplitude measurements from adult male sage-grouse on Preacher Reservoir lek (42°53.597′N, 108°28.417′W) in Fremont County, Wyoming. Recordings and amplitude measurements were collected simultaneously from a blind on the lek using a handheld Larson Davis 824 sound level meter (software version 3.12) using the logging function with a time-history resolution of 1/32 s and an amplitude resolution of 0.1 dB. A Marantz PMD670 portable solid-state recorder continuously recorded the audio stream from the SPL meter (through the AC/DC output) at 16-bit linear PCM format at 44.1 kHz. Each sound level measurement started prior to the initiation of a display by an individual male. The SPL meter measured and logged the average and peak amplitude in unweighted decibels (dB) at each time interval (0.03 s). Immediately after the vocalization was recorded, the distance between the vocalizing bird and the microphone was measured with a range finder (Leupold RX750). Sage-grouse strut displays are highly directional (Dantzker et al. 1999), so the orientation of the bird and distance to the microphone were also noted for each display measured. We used only high-quality and comparable measurements in the analysis, including only vocalizations that we recorded from individuals in a small range of orientations and at similar distances in relation to the microphone. All vocalizations included in the analysis were from individuals with side-facing orientations ranging from 30 to 90 degrees (if zero degrees reflects an orientation with the bird directly facing the observer). We did not use recordings if there was temporal overlap with other strutting males or background noises, such as songbirds. Because of the difficulty of obtaining such recordings, a total of only 6 vocalizations, collected from 2 individuals (2 from one male, 4 from the other), were used in the final analysis.

Ambient noise levels were measured on Chugwater Reservoir lek (42°47.192′N, 108°26.292′W), a lek with little human disturbance in Fremont County, Wyoming. Noise was quantified as a 2-min L_{eq} (equivalent sound pressure level); this is a type of average, defined as the equivalent steady sound level that would produce the energetic equivalent of the actual fluctuating sound levels over the defined 2-min period. The sound level meter calculated an overall L_{eq} for the noise level as well as the 2-min L_{eq} for each 1/3-octave band frequency, which was used for SNR analysis (see below). Ambient measurements were made after lekking in the morning. Ambient noise levels tend to be slightly higher during this time than during the lekking hours (J. L. Blickley and G. L. Patricelli unpubl. data), so this measure is a slight overestimate of ambient levels on an undisturbed lek, leading to a slight underestimate of masking on disturbed leks.

Sound level measurements were made on a large compressor station (Falcon Compressor, which consisted of two Ariel JGC-4 compressors driven by 3,500-HP engines; 42°31.319′N, 109°40.271′W) and a deep natural-gas drilling rig (Questar Drilling Rig no. 232; 42°43.501′N, 109°50.876′W) on the Pinedale Anticline Project Area in Sublette County, Wyoming, and at a generator (East Litton Generator, a 300-kW MQ Power diesel generator powered by a Volvo engine; 43°31.501′N, 105°25.573′W) in the Powder River Basin, Campbell County, Wyoming. These noise sources are all commonly found in areas of natural gas development and typically operate 24 h day–1, year round. Noise was measured along one transect extending from each noise source. Noise measurements were taken at points 75, 200, 300, and 400 m from the Falcon Compressor; at points 8, 16, 32, 64, 128, 256, and 512 m from the East Litton Generator; and at points 75 and 400 m from the Questar Drilling Rig. At each point, distance from the source was measured with a laser range finder (Bushnell Yardage Pro). Noise levels were measured using a Larson Davis 824 sound level meter. During measurements, the sound level meter was held 25 cm from the ground, similar to the height of a grouse. The sound level meter calculated an overall L_{eq} for the noise level as well as the 2-min L_{eq} for each 1/3-octave band frequency. Noise levels are reported in unweighted decibels (reported as dB) re 20 µPa because an unweighted measure of amplitude is required for the estimation of masking potential; A-weighted values (dB[A]) are also presented for comparison. All noise measurements were made in the early morning, before the wind rose to detectable levels. Because of the similarity of noise from each of these sources (see Fig. 2), only noise measurements from the Falcon Compressor were used in the masking analysis; results from other noise sources should be very similar. Noise levels were estimated at distances >400 m from Falcon Compressor using NMSIM software (Wyle Laboratories, Arlington, Virginia). NMSIM generates spatially explicit estimates of noise propagation utilizing input topography, ground impedance, and source spectra. We developed a custom source spectrum for Falcon Compressor using noise measurements from transect data and modeled propagation from the source across flat and open ground using a topographic layer from a location at similar elevation to our study site at 200 rayls ground impedance and –1.1°C air temperature. We used NMSim to estimate the noise spectra at receiver points placed along a transect extending from the source.

Sound analysis.—Individual vocalizations were identified from a spectrogram of the field recording using RAVEN, version 1.3 beta (Cornell Lab of Ornithology, Ithaca, New York; Hann window function, FFT = 512 with 50% overlap). Audio recordings were synchronized with SPL measurements by identifying distinctive high-frequency device noise produced by the SPL meter with the initiation of the measurement; this allowed us to identify the 1/32-s sample(s) in the SPL-meter output that corresponds to each note on the spectrogram and measure the overall amplitude of that note. Each vocalization was then extracted and low-pass filtered at 8.0 KHz to exclude this device noise. For each vocalization, the amplitude of the 1/3-octave band frequencies was measured at intervals of 0.004 s using SPECTRAPLUS (Pioneer Hill Software, Poulsbo, Washington). Call components were identified in the audio recordings in RAVEN and matched with the corresponding overall amplitude measurement from synchronized SPL measurement data. The absolute amplitude of each component was calibrated using the equation

$$\text{Peak dB} = \sum 10^{(aX/10)}$$

where a represents a scaling factor and *X* represents the average amplitude for each 1/3-octave band frequency. By adjusting the value of the scaling factor, we could adjust the overall average amplitude (dB) of the vocalization while maintaining the same relative power at each frequency band. The scaling factor was adjusted to yield different overall average amplitudes (dB) for each vocalization for analysis of masking potential at different source levels. Frequency-specific amplitudes for each call component were averaged across vocalizations.

In order to determine the masking potential of the noise sources at different distances from the vocalizing bird and the noise source, SNRs were calculated for each vocalization by subtracting the average amplitude (dB) for 1/3-octave band frequencies of noise sources (taken from 2-min L_{eq} measurements; see above) from the average amplitude (dB) for 1/3-octave band frequencies of vocalizations as measured in SPECTRAPLUS. Each note of the sage-grouse vocalizations was calibrated to absolute amplitude measures made using the SPL meter (see above). We calculated the expected amplitude of the vocalization at distances 2, 4, 8, 16, 32, 64, and 128 m from the vocalizing bird, based on a 6-dB decrease in amplitude for every doubling of distance due to spherical spreading and frequency-specific rate of excess attenuation. Excess attenuation is attenuation caused by propagation of sound through the environment and is determined by habitat characteristics (e.g., groundcover, temperature) and distance of the vocalizing bird from the ground. To model propagation of vocalizations, we estimated frequency-specific rates of excess attenuation by comparing the overall rate of sound attenuation measured along noise transects with predicted amplitude loss due to spherical spreading alone. These estimated amplitudes were used to scale the vocalizations (see scaling equation above), in order to calculate the SNR for the maximum SNR frequency at different distances from the bird and from the noise source. Vocalizations were defined as "masked" if the SNR of the peak SNR frequency did not exceed the minimum threshold (critical ratio) for detection or discrimination (Dooling 2002, Lohr et al. 2003). Minimum masked distance was used to estimate the maximum detection or discrimination distance (active space). Estimates of sage-grouse critical ratios for detection were drawn from the average critical ratios for detection of 15 bird species, the only ones that have been measured to date (Dooling 2002), and ranged from 22 dB at 400–630 Hz to 27 dB at 2,500 Hz. The critical ratios for discrimination at each frequency band were estimated to be 3 dB higher than the critical ratio for detection in that band (Lohr et al. 2003). The critical ratios for detection and discrimination have not been measured specifically for sage-grouse, but there is relatively little variation in hearing abilities among bird species tested thus far, so estimates of the critical ratio are likely to be accurate to within 5 dB (Dooling 2002). All results are presented ± SE unless otherwise noted.

Results

Noise measurements.—Noise produced by Falcon Compressor was 48.9 dB louder than ambient levels at an undisturbed lek at a distance of 75 m from the source and 34.2 dB louder than ambient at a distance of 400 m (Table 1). Noise produced by the Questar Drilling Rig was 43.5 dB louder than ambient levels at a distance of 75 m from the source and 31.8 dB louder than ambient at a distance of 400 m. Noise produced by East Litton Generator was 24.9 dB louder than ambient levels at a distance of 75 m from the source and 18.4 dB louder than ambient at a distance of 400 m (Table 1). The noise produced by all noise sources was dominated by low frequencies (Fig. 2).

Vocalization measurements.—Individual components of the sage-grouse vocal display varied in amplitude and peak frequency (the frequency at which amplitude was the highest; Table 2). The pop 1 and pop 2 components had the highest peak amplitudes, with measures of 96 ± 2.1 and 98 ± 1.6 dB at 1 m, respectively. The coo components had an overall peak amplitude of 94 ± 1.3 dB at 1 m. The whistle component, by far the quietest component, had a peak amplitude of 84 ± 0.9 dB for the whistle trough (lowest frequency of the whistle component) and 82 ± 1.5 dB for the

TABLE 1. Overall noise levels (2-min L_{eq} measurements) measured along a transect extending from Falcon Compressor in Sublette County, Wyoming. For comparison, values from an undisturbed lek of Greater Sage-Grouse after the birds departed in late morning are also included (Chugwater Reservoir lek in Fremont County, Wyoming).

Distance	Amplitude (dB[F])	Amplitude (dB[A])
75 m	89.4	70.4
200 m	82.8	58.1
300 m	77.9	52.9
400 m	74.7	47.7
Undisturbed lek (quiet)	40.5	30.5

whistle apex (highest frequency of the whistle component) at 1 m. All vocal components had peak frequencies (400–630 Hz) overlapping with noise produced by natural gas infrastructure, except the apex of the frequency-modulated whistle, which had a peak frequency (2,500 Hz) above most of the noise.

Masking analysis.—We estimated the masking potential of compressor noise for five components of the sage-grouse vocalization: the coos, pop 1, pop 2, whistle trough, and whistle apex. Across all conditions modeled, the maximum detection and discrimination distance (i.e., the active space) for the highest-amplitude frequency band was greatest for the pop 2 component, the loudest note of the display. Overall amplitude of the note was not necessarily an indicator of greater active space—the coo component had a greater maximum detection distance than the pop 1 component (Fig. 3) despite lower overall amplitude, due to the higher amplitude of the maximum frequency. Active space of detection and discrimination for all components was substantially reduced at the noise levels found within 400 m of the compressor station in relation to the ambient conditions on an undisturbed lek (Fig. 3). At 75 m from the noise source, the maximum detection distance and maximum discrimination distance were reduced by 97% and 98%, respectively, for the coo; by 98% and 98% for pop 1; by 97% and 97% for pop 2; by 98% and 98% for the whistle trough; and by 100% and 100% for the whistle apex, in relation to the maximum distances on an undisturbed lek. At 400 m from the noise source, the maximum detection distance and maximum discrimination distance were reduced by 59% and 65%, respectively, for the coo; by 48% and 47% for pop 1; by 59% and 63% for pop 2; by 54% and 57% for the whistle trough; and by 64% and 58% for the whistle apex, in relation to the maximum distances on an undisturbed lek.

The distance from the source at which the active space for detection and discrimination were equal to that in ambient conditions (i.e., the maximum active space) varied for each component. The whistle apex reached maximum active space at 600 m from the noise source. The whistle trough reached maximum active space at 700 m from the source, whereas the coo and pop 1 required a minimum of 700 m from the source before they reached maximum active space. Pop 2 did not reach maximum active space until a minimum of 1,000 m from the noise source.

The SNR varied across frequencies for each component. Peak frequencies for coos, pops, and the whistle trough were relatively low (<1,000 Hz), leading to high overlap with the low-frequency noise produced by the Falcon Compressor and other natural gas infrastructure (Figs. 2 and 4). The SNR was substantially reduced at low frequencies at both short and medium distances to the compressor in relation to quiet lek conditions for all components (Fig. 4). For the whistle, coo, and pop 2 components, the frequency with the peak SNR remained the same under all noise conditions, indicating that no signal distortion would be expected. For the pop 1 component, the frequency with the peak SNR differed under different noise conditions, shifting from 400 Hz under quiet

TABLE 2. Amplitude and frequency characteristics of Greater Sage-Grouse vocalizations recorded in Fremont County, Wyoming. Measurements are normalized to 1 m from the source.

Note	Peak amplitude (dB)	Peak amplitude range (dB)	Frequency range (Hz)	Peak frequency (Hz, ⅓-octave band)
Coo	94 ± 1.3	89–98	100–800	500
Pop 1	96 ± 2.1	87–99	100–10,500	500
Pop 2	98 ± 1.6	90–100	100–11,500	400
Whistle apex	82 ± 1.3	76–87	2,200–2,600	2,500
Whistle trough	84 ± 0.9	81–87	450–800	630

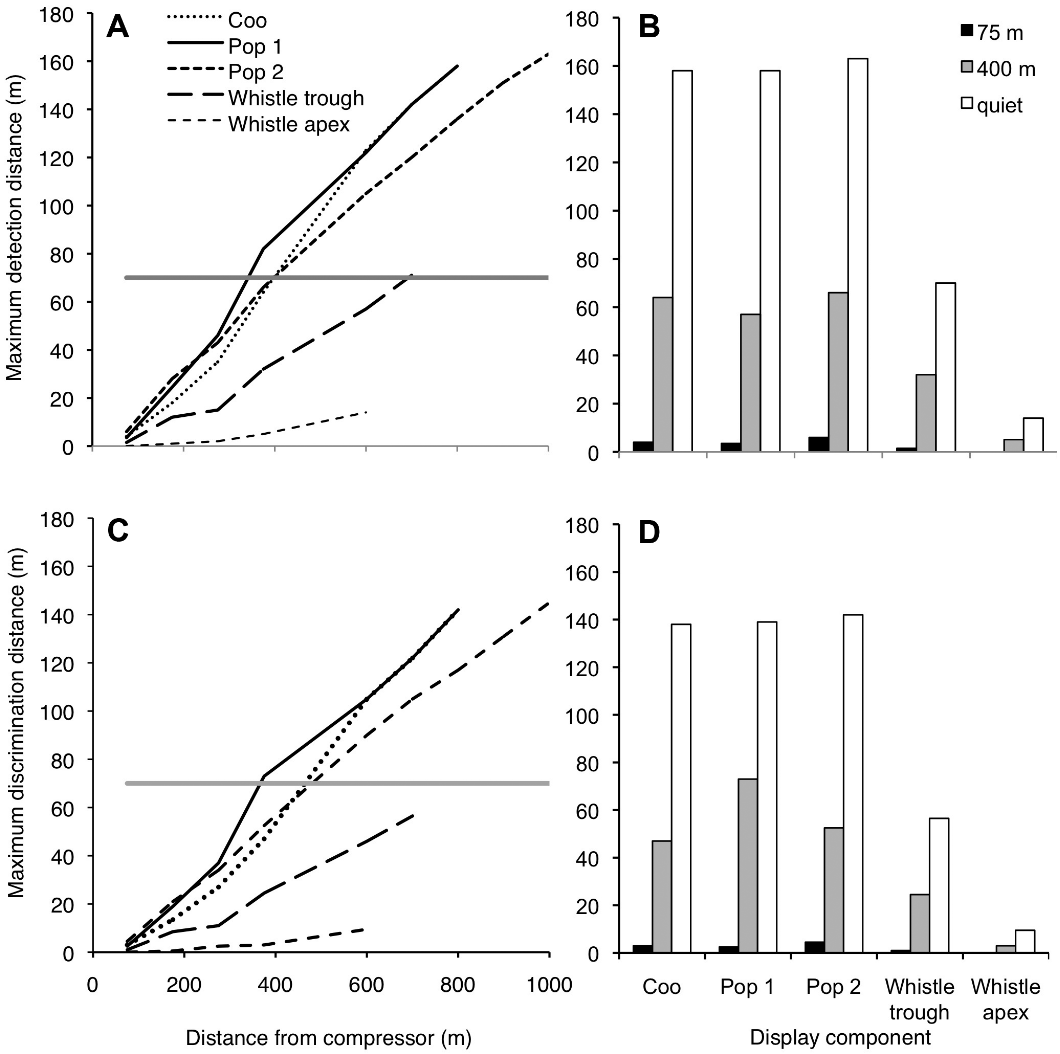

Fig. 3. Maximum (A) detection and (C) discrimination distance of Greater Sage-Grouse strut display components at varying distances from a natural gas compressor station. Gray solid line represents half the length of a typical lek in Fremont County, Wyoming. Lines end at the point where the active space is equal to that under quiet ambient conditions. Maximum (B) detection and (D) discrimination distance of vocalization components at points 75 and 400 m from a natural gas compressor station and under quiet ambient conditions.

conditions to 500 Hz in noisy conditions (Fig. 4B), potentially causing distortion of the signal.

Discussion

We assessed the potential impact of anthropogenic noise on the transmission of sage-grouse vocalizations used for mate attraction (Wiley 1973; Gibson 1989, 1996; Patricelli and Krakauer 2010). Our results indicate that there are marked differences in the active space of individual notes of the sage-grouse acoustic display, both in noisy and quiet conditions. These differences in active space are primarily determined by the frequency structure and amplitude of the different notes of the sage-grouse vocalization, and by differences in the amplitude of the background noise. These factors and their effects on the active space for detection and discrimination are discussed below.

Frequency structure.—The active space of a vocalization is determined, in part, by the frequency structure—including peak frequency and

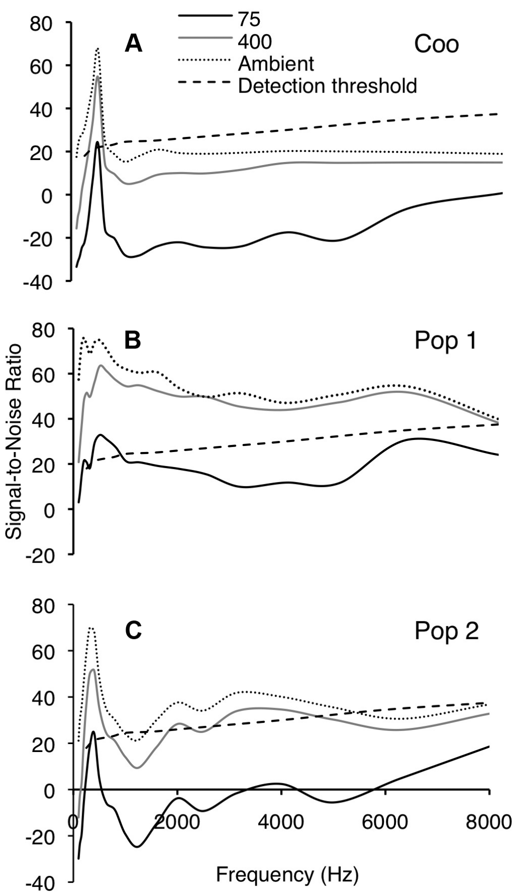

Fig. 4. Signal-to-noise ratio (SNR) of Greater Sage-Grouse acoustic display components (A) coo, (B) pop 1, and (C) pop 2 at a distance of 5 m from the vocalizing male (average close courtship distance) in ambient noise conditions measured 75 and 400 m from a natural gas compressor and on an undisturbed lek. Frequencies with an SNR that exceed the critical ratio for detection (dashed line) can be detected by a receiver. For pop 1, the frequency with the highest SNR is different in noisy and quiet environments, potentially leading to distortion of the vocalization.

frequency range—of both the acoustic signal and the background noise (Lohr et al. 2003). Both of these measures of frequency structure differed among the notes of the sage-grouse display vocalization. Notes with low peak frequencies (the coos, pops, and whistle trough) had high overlap with the noise produced by the Falcon Compressor and other natural gas infrastructure (Figs. 2 and 4), leading to predictions of a substantial reduction in active space of detection and discrimination for these notes in noisy conditions (Fig. 2). The whistle apex had a peak frequency above most of the compressor noise energy, but was still masked because of its lower source amplitude, as discussed below.

The frequency range of a note is also important in determining the degree of overlap with background noise. The coo note of the sage-grouse display is tonal and has a very small frequency range, so the entire note is likely to be masked by low-frequency noise (Fig. 4A). For notes with a broad frequency range, like the broadband pops and the frequency-modulated whistle, some of the higher-frequency energy of the signal is likely to be detectable above background noise that is predominantly low frequency. However, higher frequencies suffer greater attenuation over distance than lower frequencies (Marten and Marler 1977), which reduces the advantage of high-frequency signals in maximizing active space. Because most anthropogenic noise is dominated by low frequencies, species that have low-frequency vocalizations, such as the sage-grouse, will disproportionately experience masking. Indeed, several studies have found that anthropogenic noise more severely affects species with lower-frequency vocalizations (Rheindt 2003; Francis et al. 2009, 2011; Goodwin and Shriver 2011).

Amplitude.—The amplitude of each note is also important in determining the active space, such that quieter notes suffer increased masking at a given distance from the noise source and vocalizing individual. Pops and coos could be detected at greater distances than the whistle apex and whistle trough, despite greater overlap with the background noise, because of greater source amplitudes. The whistle apex, which had the lowest source amplitude, had the smallest active space in noise despite the low overlap with the noise frequencies.

The acoustic directionality of a vocalization may also affect the degree to which masking reduces the overall active space. Many vocalizations radiate from the signaler in a directional pattern, such that the amplitude varies with the orientation of the vocalizing individual. Because of our small sample size, we did not include the effects of directionality on active space in our analysis, but instead assessed the impact of noise on the average active space of the signal across multiple orientations. The whistle is highly directional, with differences of up to 22 dB depending on the relative orientation of the individual (Dantzker et al. 1999). We used values from the loudest orientations of those that we measured;

therefore, masking in the quieter orientations may be much greater than described here. Given that the loudest orientation can vary for different strut components (Dantzker et al. 1999), it is possible that using this small range and averaging across vocalizations may have underestimated the maximum active space for some components. Males that adjust their orientation to beam a highly directional vocalization toward a female may gain an advantage over other males, even under quiet conditions (Brumm 2002, Brumm and Todt 2003, Patricelli and Krakauer 2010); this advantage may be even more pronounced in a noisy environment.

Potential consequences of masking.—Reductions in the active space of detection and discrimination, as predicted by our analysis, could have significant effects on the fitness of individuals in noisy landscapes. Female sage-grouse use acoustic signals to locate lekking males (Bradbury et al. 1989); thus, their ability to find leks could be compromised in noisy environments because of the reduced active space of detection. Once on the lek, females can detect males visually, making detection using acoustic signals less critical. Discrimination and recognition are likely to be more critical on this smaller spatial scale. Female sage-grouse use the acoustic components of the display to select a mate (Gibson et al. 1991, Gibson 1996). In particular, acoustic features such as the IPI, and possibly the whistle, are thought to play a role in attracting females from across the lek (Gibson 1996). Thus, noise that reduces the maximum distance of discrimination to less than half the length of leks in our study population (half average lek length = ~70 m; J. L. Blickley unpubl. data) could negatively affect a male's ability to attract females. Further, background noise could make active comparison of males difficult for females if the maximum discrimination distance is reduced to less than the average distance between males (Forrest and Raspet 1994).

If the interfering noise only overlaps partially with a vocalization, the frequency with the maximum active space may be different under noisy conditions than under normal ambient conditions, leading to the reception of a signal that is distorted. For example, in the pop 1 component of the sage-grouse display, we found that the frequency with the maximum active space was different in noisy compared with quiet conditions. Therefore, a receiver hearing pop 1 under noisy conditions would hear a call dominated by frequencies in the 500 Hz 1/3-octave band; but under quiet conditions, the receiver would hear a call dominated by frequencies in the 200 Hz 1/3-octave band. Depending on which characteristics of the vocalization are assessed by females or competing males, this distortion may lead to difficulty in discrimination or recognition. Previous studies have suggested that female sage-grouse do not assess natural variation among males in peak frequency during mate choice (Gibson et al. 1991), but further behavioral studies would be needed to determine what, if any, effect such distortion might have on female response to male sage-grouse vocalizations. Distortion may have more significant effects on species in which mate choice is based on the frequency of the signal. For example, in species in which females prefer males with low-frequency song (Halfwerk et al. 2011) or assess the fundamental frequency of song as an indicator of male body size, (Ryan and Brenowitz 1985), distortion may lead to increased difficulty in comparing potential mates.

Ultimately, increased difficulty in finding leks or assessing males on the leks may lead to lower female attendance on noisy leks compared with quieter locations. Males may also avoid leks with high levels of noise if they perceive that their vocalizations are masked. Blickley et al. (2012) found lower male and female attendance on leks with experimentally introduced noise from roads and drilling rigs, both of which produce primarily low-frequency sounds similar to the compressor station modeled here. These declines may be due in part to masking, which would be predicted given the substantial overlap in the frequency range of the introduced noise and the sage-grouse strut display. However, the average level of introduced noise across leks in this experiment was relatively low, especially on leks with intermittent road noise, so masking is not likely the only cause of the observed declines. As discussed below, masking is only one possible effect of noise, and other effects may have a larger impact.

Masking in the context of noise regulations.—Are current noise regulations predicted to limit the impact of masking on sage-grouse? Outside of the breeding season, energy development activities are limited within 400 m (1/4 mile) of active sage-grouse leks on federal lands at our study site (Bureau of Land Management 2008). Our analysis indicates that a compressor station, or a similar noise source such as a drilling rig, placed at or inside this stipulated minimum surface-disturbance buffer would have a substantial effect on the ability of sage-grouse to detect a nearby lek and, potentially, to discriminate among individuals on the lek.

Regulations also institute a 2-mile (3.2-km) buffer around leks for permanent infrastructure and lekking-season drilling activities on federal lands in this region (Bureau of Land Management 2008). Our results suggest that the masking footprint of a single compressor station or drilling rig is unlikely to exceed this buffer. Within the range of the peak frequencies for sage-grouse vocalizations (400–2,500 Hz), the noise produced by the compressor station was estimated to drop to ambient levels ≤1,000 m. Even if noise travels farther during temperature inversions common in the early morning, when sage-grouse are actively lekking (Sutherland and Daigle 1998), masking on the lek is likely to be negligible for sources outside the 2-mile (3.2-km) buffer. However, off-lek communication, such as parent–offspring communication, occurs well beyond the boundaries of a lek (Lyon and Anderson 2003) and may still be susceptible to masking. Further, our analysis considered the masking impact of only a single, stationary noise source, but many developed areas contain a network of such sources connected by roads; this will lead to a much greater area of total impact.

Mechanisms to reduce masking.—Features of sound perception and flexibility in signal production may improve the ability of animals to detect signals in noise beyond the active-space predictions calculated by this method. Animals may use directional cues to separate a sound from background noise if the two sound sources are spatially separated (Schwartz and Gerhardt 1989, Dent et al. 1997). Amplitude fluctuations across the spectrum of a sound, or comodulation, may also increase the detectability of the sound against background noise, especially if the noise is relatively constant (Klump and Langemann 1995) like the noise sources investigated here. Animals in noisy areas may adjust their vocalizations to compensate for the increased background noise (Patricelli and Blickley 2006), increasing the amplitude (Brumm 2004) or redundancy (Brumm and Slater 2006) or shifting the peak or minimum frequencies to reduce overlap with background noise frequencies (e.g., Slabbekoorn and Peet 2003, Wood and Yezerinac 2006, Potvin et al. 2011). The potential for these forms of compensation is species specific; the degree to which hearing ability and vocal adjustment affect the active space of sage-grouse vocalizations is unknown.

Noise impacts beyond masking.—Masking is one potential effect of noise on wildlife, but it is certainly not the only one (Barber et al. 2010, Blickley and Patricelli 2010, Kight and Swaddle 2011). Blickley et al. (2012) found strong evidence that sage-grouse leks with experimentally introduced intermittent road noise experienced much greater declines in male attendance than those with more continuous drilling noise, despite the lower masking potential of road noise. Even light vehicular traffic (1–12 vehicles day^{-1}) has been found to substantially reduce nest initiation rates and increase the distance of nests from lek sites in sage-grouse (Lyon and Anderson 2003), despite minimal opportunity for masking. Together, these studies suggest that masking is not the only potential effect of noise or noisy infrastructure on sage-grouse. So, although a masking analysis can be powerful in making predictions about the effects of noise on lek communication in sage-grouse, this type of analysis may not provide sufficient predictive power for estimating the overall impact of the noise on this species.

Noise pollution has been found to induce stress, disrupt physiological processes and behaviors, cause physical trauma to the auditory system, or mask other natural sounds important to survival and reproduction, such as the sound of predator approach, in a variety of species (Marler et al. 1973, Bowles 1995, Kight and Swaddle 2011). For sage-grouse, these effects may extend beyond the area in which masking of the strut display is an issue, particularly for time spent off lek. Wildlife managers that seek to reduce the overall impact of anthropogenic noise on sage-grouse and other species affected by human encroachment must address all the potential effects of noise, including masking potential.

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Literature Cited

Ballentine, B., J. Hyman, and S. Nowicki. 2004. Vocal performance influences female response to male bird song: An experimental test. Behavioral Ecology 15:163–168.

Barber, J. R., K. R. Crooks, and K. M. Fristrup. 2010. The costs of chronic noise exposure for terrestrial organisms. Trends in Ecology & Evolution 25:180–189.

Blickley, J. L., D. Blackwood, and G. L. Patricelli. 2012. Experimental evidence for the effects of chronic anthropogenic noise on abundance of Greater Sage-Grouse at leks. Conservation Biology 26: in press.

Blickley, J. L., and G. L. Patricelli. 2010. Impacts of anthropogenic noise on wildlife: Research priorities for the development of standards and mitigation. Journal of International Wildlife Law and Policy 13:274–292.

Bowles, A. E. 1995. Responses of wildlife to noise. Pages 109–156 *in* Wildlife and Recreationists: Coexistence through Management and Research (R. L. Knight and K. J. Gutzwiller, Eds.). Island Press, Washington, D.C.

Bradbury, J. W., R. M. Gibson, C. E. McCarthy, and S. L. Vehrencamp. 1989. Dispersion of displaying male sage grouse. II. The role of female dispersion. Behavioral Ecology and Sociobiology 24:15–24.

Brenowitz, E. A. 1982. The active space of Red-winged Blackbird song. Journal of Comparative Physiology 147:511–522.

Brumm, H. 2002. Sound radiation patterns in nightingale (*Luscinia megarhynchos*) songs. Journal für Ornithologie 143:468–471.

Brumm, H. 2004. The impact of environmental noise on song amplitude in a territorial bird. Journal of Animal Ecology 73:434–440.

Brumm, H. 2006. Signalling through acoustic windows: Nightingales avoid interspecific competition by short-term adjustment of song timing. Journal of Comparative Physiology A 192:1279–1285.

Brumm, H., and H. Slabbekoorn. 2005. Acoustic communication in noise. Advances in the Study of Behavior 35:151–209.

Brumm, H., and P. J. B. Slater. 2006. Ambient noise, motor fatigue, and serial redundancy in chaffinch song. Behavioral Ecology and Sociobiology 60:475–481.

Brumm, H., and D. Todt. 2003. Facing the rival: Directional singing behaviour in nightingales. Behaviour 140:43–53.

Bureau of Land Management. 2008. Pinedale Anticline Project Area Supplemental Environmental Impact Statement Record of Decision. U.S. Department of Interior, Bureau of Land Management, Pinedale Field Office, Wyoming.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of Greater Sage-Grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming. [Online.] Available at www.ndow.org/wild/conservation/sg/resources/greate_sg_cons_assessment.pdf.

Copeland, H. E., K. E. Doherty, D. E. Naugle, A. Pocewicz, and J. M. Kiesecker. 2009. Mapping oil and gas development potential in the US intermountain west and estimating impacts to species. PLoS One 4:e7400.

Dantzker, M. S., G. B. Deane, and J. W. Bradbury. 1999. Directional acoustic radiation in the strut display of male sage grouse *Centrocercus urophasianus*. Journal of Experimental Biology 202:2893–2909.

Dent, M. L., O. N. Larsen, and R. J. Dooling. 1997. Free-field binaural unmasking in Budgerigars (*Melopsittacus undulatus*). Behavioral Neuroscience 111:590–598.

Dooling, R. [J.] 2002. Avian hearing and the avoidance of wind turbines. Technical Report NREL/TP-500-30844. National Renewable Energy Lab, Golden, Colorado.

Dooling, R. J., M. Leek, and E. West. 2009. Predicting the effects of masking noise on communication distance in birds. Journal of the Acoustical Society of America 125:2517.

Dooling, R. J., and A. N. Popper. 2007. The Effects of Highway Noise on Birds. [Report prepared for the California Department of Transportation, Sacramento; Jones and Stokes Associates.] [Online.] Available at www.dot.ca.gov/hq/env/bio/files/caltrans_birds_10-7-2007b.pdf.

Forrest, T. G., and R. Raspet. 1994. Models of female choice in acoustic communication. Behavioral Ecology 5:293–303.

Francis, C. D., C. P. Ortega, and A. Cruz. 2009. Noise pollution changes avian communities and species interactions. Current Biology 19:1415–1419.

Francis, C. D., C. P. Ortega, and A. Cruz. 2011. Noise pollution filters bird communities based on vocal frequency. PLoS One 6:e27052.

Garton, E. O., J. W. Connelly, J. S. Horne, C. A. Hagen, A. Moser, and M. A. Schroeder. 2011. Greater Sage-Grouse population dynamics and probablity of persistence. Pages 293–382 *in* Greater Sage-Grouse: Ecology and Conservation of a Landscape Species and Its Habitats (S. T. Knick and J. W. Connelly, Eds.). Studies in Avian Biology, no. 38.

Gibson, R. M. 1989. Field playback of male display attracts females in lek breeding sage grouse. Behavioral Ecology and Sociobiology 24:439–443.

Gibson, R. M. 1996. Female choice in sage grouse: The roles of attraction and active comparison. Behavioral Ecology and Sociobiology 39:55–59.

Gibson, R. M., and J. W. Bradbury. 1985. Sexual selection in lekking grouse: Phenotypic correlates of male strutting success. Behavioral Ecology and Sociobiology 18:117–123.

Gibson, R. M., J. W. Bradbury, and S. L. Vehrencamp. 1991. Mate choice in lekking sage grouse revisited: The roles of vocal diplay, female site fidelity, and copying. Behavioral Ecology 2:165–180.

Goodwin, S. E., and W. G. Shriver. 2011. Effects of traffic noise on occupancy patterns of forest birds. Conservation Biology 25:406–411.

Halfwerk, W., S. Bot, J. Buikx, M. van der Velde, J. Komdeur, C. ten Cate, and H. Slabbekoorn. 2011. Low-frequency songs lose their potency in noisy urban conditions. Proceedings of the National Academy of Sciences USA 108:14549–14554.

Hebets, E. A., and D. R. Papaj. 2005. Complex signal function: Developing a framework of testable hypotheses. Behavioral Ecology and Sociobiology 57:197–214.

Holloran, M. J. 2005. Greater Sage-Grouse (*Centrocercus urophasianus*) population response to natural gas field development in western Wyoming. Ph.D. dissertation, University of Wyoming, Laramie.

Holloran, M. J., R. C. Kaiser, and W. A. Hubert. 2010. Yearling Greater Sage-Grouse response to energy development in Wyoming. Journal of Wildlife Management 74:65–72.

Kight, C. R., and J. P. Swaddle. 2011. How and why environmental noise impacts animals: An integrative, mechanistic review. Ecology Letters 14:1052–1061.

Klump, G. M. 1996. Bird communication in the noisy world. Pages 321–338 *in* Ecology and Evolution of Acoustic Communication in Birds (D. E. Kroodsma and E. H. Miller, Eds.). Cornell University Press, Ithaca, New York.

Klump, G. M., and U. Langemann. 1995. Comodulation masking release in a songbird. Hearing Research 87:157–164.

Lohr, B., T. F. Wright, and R. J. Dooling. 2003. Detection and discrimination of natural calls in masking noise by birds: Estimating the active space of a signal. Animal Behaviour 65:763–777.

Lyon, A. G., and S. H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin 31:486–491.

Marler, P., M. Konishi, A. Lutjen, and M. S. Waser. 1973. Effects of continuous noise on avian hearing and vocal development. Proceedings of the National Academy of Sciences USA 70:1393–1396.

Marten, K., and P. Marler. 1977. Sound transmission and its significance for animal vocalization. I. Temperate habitats. Behavioral Ecology and Sociobiology 2:271–290.

Møller, A. P., and A. Pomiankowski. 1993. Why have birds got multiple sexual ornaments? Behavioral Ecology and Sociobiology 32:167–176.

Nelson, D. A., and A. Poesel. 2007. Segregation of information in a complex acoustic signal: Individual and dialect identity in White-crowned Sparrow song. Animal Behaviour 74:1073–1084.

Patricelli, G. L., and J. L. Blickley. 2006. Avian communication in urban noise: Causes and consequences of vocal adjustment. Auk 123:639–649.

Patricelli, G. L., M. S. Dantzker, and J. W. Bradbury. 2008. Acoustic directionality of Red-winged Blackbird (*Agelaius phoeniceus*) song relates to amplitude and singing behaviours. Animal Behaviour 76:1389–1401.

Patricelli, G. L., and A. H. Krakauer. 2010. Tactical allocation of effort among multiple signals in sage grouse: An experiment with a robotic female. Behavioral Ecology 21:97–106.

Potvin, D. A., K. M. Parris, and R. A. Mulder. 2011. Geographically pervasive effects of urban noise on frequency and syllable rate of songs and calls in Silvereyes (*Zosterops lateralis*). Proceedings of the Royal Society of London, Series B 278:2464–2469.

Rheindt, F. E. 2003. The impact of roads on birds: Does song frequency play a role in determining susceptibility to noise pollution? Journal für Ornithologie 144:295–306.

Ryan, M. J., and E. A. Brenowitz. 1985. The role of body size, phylogeny, and ambient noise in the evolution of bird song. American Naturalist:87–100.

Schwartz, J. J., and H. C. Gerhardt. 1989. Spatially mediated release from auditory masking in an anuran amphibian. Journal of Comparative Physiology A 166:37–41.

Slabbekoorn, H., and M. Peet. 2003. Birds sing at a higher pitch in urban noise—Great Tits hit the high notes to ensure that their mating calls are heard above the city's din. Nature 424:267.

Slabbekoorn, H., and E. A. Ripmeester. 2008. Birdsong and anthropogenic noise: Implications and applications for conservation. Molecular Ecology 17:72–83.

Sutherland, L. C., and G. A. Daigle. 1998. Atmospheric Sound Propagation. Wiley, New York.

Swaddle, J. P., and L. C. Page. 2007. High levels of environmental noise erode pair preferences in Zebra Finches: Implications for noise pollution. Animal Behaviour 74:363–368.

Walker, B. L., D. E. Naugle, and K. E. Doherty. 2007. Greater Sage-Grouse population response to energy development and habitat loss. Journal of Wildlife Management 71:2644–2654.

Wiley, R. H. 1973. Territoriality and non-random mating in sage grouse (*Centrocercus urophasianus*). Animal Behaviour Monographs 6:85–169.

Wiley, R. H., and D. G. Richards. 1982. Adaptations for acoustic communication in birds: Sound transmission and signal detection. Pages 131–181 *in* Acoustic Communication in Birds (D. E. Kroodsma, E. H. Miller, and H. Ouellet, Eds.). Academic Press, New York.

Wood, W. E., and S. M. Yezerinac. 2006. Song Sparrow (*Melospiza melodia*) song varies with urban noise. Auk 123:650–659.

Conservation Biology

Contributed Paper

Experimental Evidence for the Effects of Chronic Anthropogenic Noise on Abundance of Greater Sage-Grouse at Leks

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Abstract: *Increasing evidence suggests that chronic noise from human activities negatively affects wild animals, but most studies have failed to separate the effects of chronic noise from confounding factors, such as habitat fragmentation. We played back recorded continuous and intermittent anthropogenic sounds associated with natural gas drilling and roads at leks of Greater Sage-Grouse (*Centrocercus urophasianus*). For 3 breeding seasons, we monitored sage grouse abundance at leks with and without noise. Peak male attendance (i.e., abundance) at leks experimentally treated with noise from natural gas drilling and roads decreased 29% and 73%, respectively, relative to paired controls. Decreases in abundance at leks treated with noise occurred in the first year of the study and continued throughout the experiment. Noise playback did not have a cumulative effect over time on peak male attendance. There was limited evidence for an effect of noise playback on peak female attendance at leks or male attendance the year after the experiment ended. Our results suggest that sage-grouse avoid leks with anthropogenic noise and that intermittent noise has a greater effect on attendance than continuous noise. Our results highlight the threat of anthropogenic noise to population viability for this and other sensitive species.*

Keywords: chronic noise, energy development, *Centrocercus urophasianus*, roads

Evidencia Experimental de los Efectos de Ruido Antropogénico Crónico sobre la Abundancia de *Centrocercus urophasianus* en Leks

Resumen: *El incremento de evidencias sugiere que el ruido crónico de actividades humanas afecta negativamente a los animales silvestres, pero la mayoría de los estudios no separan los efectos del ruido crónico de los factores de confusión, como la fragmentación del hábitat. Reprodujimos sonidos antropogénicos intermitentes y continuos asociados con la perforación de pozos de gas natural y caminos en leks de* Centrocercus urophasianus. *Durante 3 épocas reproductivas, monitoreamos la abundancia de* C. urophasianus e *leks con y sin ruido. La abundancia máxima de machos (i.e., abundancia) en leks tratados con ruido de la perforación de pozos de gas natural y caminos decreció 29% y 73% respectivamente en relación con los controles pareados. La disminución en abundancia en leks tratados con ruido ocurrió en el primer año del estudio y continuó a lo largo del experimento. La reproducción de ruido no tuvo efecto acumulativo en el tiempo sobre la abundancia máxima de machos. Hubo evidencia limitada para un efecto de la reproducción de ruido sobre la abundancia máxima de hembras en los leks o sobre la asistencia de machos el año después de que concluyó el experimento. Nuestros resultados sugieren que* C. urophasianus *evita leks con ruido anropogénico y que el ruido intermitente tiene un mayor efecto sobre la asistencia que el ruido continuo. Nuestros*

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resultados resaltan amenaza del ruido antropogénico para la viabilidad poblacional de esta y otras especies sensibles.

Palabras Clave: *Centrocercus urophasianus*, desarrollo energético, ruido crónico, caminos

Introduction

Noise associated with human activity is widespread and expanding rapidly in aquatic and terrestrial environments, even across areas that are otherwise relatively unaffected by humans, but there is still much to learn about its effects on animals (Barber et al. 2009). Effects of noise on behavior of some marine organisms are well-documented (Richardson 1995). In terrestrial systems, the effects of noise have been studied less, but include behavioral change, physiological stress, and the masking of communication signals and predator sounds (Slabbekoorn & Ripmeester 2008; Barber et al. 2009). These effects of noise on individual animals may lead to population decreases if survival and reproduction of individuals in noisy habitats are lower than survival and reproduction of individuals in similar but quiet habitats (Patricelli & Blickley 2006; Warren et al. 2006; Slabbekoorn & Ripmeester 2008). Population declines may also result if animals avoid noisy areas, which may cause a decrease in the area available for foraging and reproduction.

There is evidence of variation among species in their sensitivity to noise. Noise sensitivity may also differ with the type of noise, which varies in amplitude, frequency, temporal pattern, and duration (Barber et al. 2009). Duration may be particularly critical; most anthropogenic noise is chronic and the effects of chronic noise may differ substantially from those of short-term noise in both severity and response type. For example, brief noise exposure may cause elevated heart rate and a startle response, whereas chronic noise may induce physiological stress and alter social interactions. Therefore, when assessing habitat quality for a given species, it is critical to understand the potential effects of the full spectrum of anthropogenic noise present in the species' range.

The effects of noise on wild animals are difficult to study because noise is typically accompanied by other environmental changes. Infrastructure that produces noise may be associated with fragmentation of land cover, visual disturbance, discharge of chemicals, or increased human activity. Each of these factors may affect the physiology, behavior, and spatial distribution of animals, which increases the difficulty of isolating the effects of the noise.

Controlled studies of noise effects on wild animals in terrestrial systems thus far have focused largely on birds. Recent studies have compared avian species richness, occupancy, and nesting success near natural gas wells operating with and without noise-producing compressors. In these studies, spatial variation in noise was used to control for confounding visual changes due to infrastructure (Habib et al. 2007; Bayne et al. 2008; Francis et al. 2009). Results of these studies show that continuous noise affects density and occupancy of a range of bird species and leads to decreases or increases in abundance of some species and has no effect on other species (Bayne et al. 2008; Francis et al. 2009; Francis et al. 2011). Results of these studies also show that noise affects demographic processes, such as reproduction, by reducing the pairing or nesting success of individuals (Habib et al. 2007; Francis et al. 2009).

Although these studies in areas near natural gas wells controlled for the effects of most types of disturbance besides noise, they could not address the effect of noise on naïve individuals in areas without natural gas wells and compressors. Furthermore, there have been no controlled experiments that address the effects of chronic but intermittent noise, such as traffic, which may be more difficult for species to habituate. Road noise may have large negative effects because it is widespread (affecting an estimated 20% of the United States) (Forman 2000) and observational studies indicate that noise may contribute to decreases in abundance of many species near roads (e.g., Forman & Deblinger 2000).

Noise playback experiments offer a way to isolate noise effects on populations from effects of other disturbances and to compare directly the effects of noise from different sources. Playback experiments have been used to study short-term behavioral responses to noise, such as effects of noise on calling rate of amphibians (Sun & Narins 2005; Lengagne 2008), heart rate of ungulates (Weisenberger et al. 1996), diving and foraging behavior of cetaceans (Tyack et al. 2011), and song structure of birds (Leonard & Horn 2008), but have not been used to study effects of chronic noise on wild animals because producing long-term noise over extensive areas is challenging. We conducted a playback experiment intended to isolate and quantify the effects of chronic noise on wild animals. We focused on the effects of noise from natural gas drilling on Greater Sage-Grouse (*Centrocercus urophasianus*).

Greater Sage-Grouse occur in the western United States and Canada and have long been a focus of sexual selection studies (Wiley 1973; Gibson 1989; Gibson 1996). Greater Sage-Grouse populations are decreasing in density and number across the species' range, largely due to extensive habitat loss (Connelly et al. 2004; Garton et al. 2010). The species is listed as endangered under Canada's

Species at Risk Act and is a candidate species for listing under the U.S. Endangered Species Act. Deep natural gas and coal-bed methane development have been expanded rapidly across the species' range since 2000 and substantial evidence suggests that these processes may contribute to observed decreases in the number of Greater Sage-Grouse (Holloran 2005; Walker et al. 2007; Holloran et al. 2010). Many factors associated with deep natural gas and coal-bed methane development are thought to lead to these decreases, including habitat loss, increased occurrence of West Nile Virus, and altered fire regimes due to the expansion of nonnative invasive species (Naugle et al. 2004; Walker et al. 2007; Copeland et al. 2009).

The noise created by energy development may also affect sage grouse by disrupting behavior, causing physiological stress, or masking biologically important sounds. During the breeding season (February–May), male sage grouse gather on communal breeding grounds called leks. Male attendance (number of male birds on the lek) at sage grouse leks downwind of deep natural gas development decreases up to 50% per year compared with attendance at other leks, which suggests noise or aerial spread of chemical pollution as factors contributing to these decreases (Holloran 2005).

We sought to test the hypothesis that lek attendance by male and female sage grouse is negatively affected by both chronic intermittent and continuous noise from energy development. To do so, we conducted a noise playback experiment in a population that is relatively unaffected by human activity. Over 3 breeding seasons (late February to early May), we played noise recorded from natural gas drilling rigs and traffic on gas-field access roads at sage grouse leks and compared attendance patterns on these leks to those on nearby control leks.

We conducted our experiment at leks because lekking sage grouse are highly concentrated in a predictable area, which makes them good subjects for a playback experiment. More importantly, sage grouse may be particularly responsive to noise during the breeding season, when energetic demands and predation risk are high (Vehrencamp et al. 1989; Boyko et al. 2004). Additionally, noise may mask sexual communication on the lek. Lekking males produce a complex visual and acoustic display (Supporting Information) and females use the acoustic component of the display to find lekking males and select a mate (Gibson 1989; Gibson 1996; Patricelli & Krakauer 2010). Furthermore, lek attendance is commonly used as a metric of relative abundance of sage grouse at the local and population level (Connelly et al. 2003; Holloran 2005; Walker et al. 2007). We used counts of lek attendance (lek counts) to assess local abundance relative to noise versus control treatments.

Methods

Study Site and Lek Monitoring

Our study area included 16 leks (Table 1 & Supporting Information) on public land in Fremont County, Wyoming, U.S.A. (42° 50′, 108° 29′). Dominant vegetation in this region is big sagebrush (*Artemisia tridentata wyomingensis*) with a grass and forb understory. The primary land use is cattle ranching, and there are low levels of recreation and natural gas development.

We paired leks on the basis of similarity in previous male attendance and geographic location (Table 2 & Supporting Information). Within a pair, one lek was

Table 1. Pairing, treatment type, location, and baseline attendance for leks used in noise playback experiment.

Lek	*Pair*	*Pair noise type*	*Noise or control*	*Years of playback*	*Baseline attendance**
Gustin	A	drilling	control	3	26
Preacher Reservoir	A	drilling	noise	3	49
North Sand Gulch	B	road	control	3	32
Lander Valley	B	road	noise	3	67
East Twin Creek	C	drilling	control	3	44
Coal Mine Gulch	C	drilling	noise	3	83
East Carr Springs	D	road	control	3	67
Carr Springs	D	road	noise	3	92
Powerline	E	drilling	control	2	49
Conant Creek North	E	drilling	noise	2	44
Monument	F	road	control	2	53
Government Slide Draw	F	road	noise	2	55
Nebo	G	drilling	control	2	18
Arrowhead West	G	drilling	noise	2	24
Onion Flats 1	H	road	control	2	41
Ballenger Draw	H	road	noise	2	38

**Baseline attendance is the average peak male attendance value (annual maximum number of males observed averaged across years) for that lek from 2002 to 2005.*

Table 2. Mixed-effect candidate models used to assess change in peak attendance of male Greater Sage-Grouse at leks from pre-experiment baseline attendance during the natural gas drilling noise playback (2006–2008) and after the experiment (2009).

Model (year)[a]	K[b]	ΔAIC_c[c]	w_i[d]
Male experiment (2006–2008)			
treatment×type+season[e]	9	0	0.64
treatment×type[e]	7	1.8	0.26
treatment+experiment year	6	6.1	0.03
treatment+season	7	6.8	0.02
treatment	5	7.3	0.02
treatment×experiment year	7	8.0	0.01
treatment×type+treatment×season+experiment year	12	8.6	<0.01
treatment×type+treatment×season	11	9.9	<0.01
treatment×type+treatment×season+treatment×experiment year	13	10.0	<0.01
treatment+type	6	10.4	<0.01
treatment×season	9	16.2	<0.01
null- random effects only	4	57.0	<0.01
Male after experiment (2009)			
null, random effects only[e]	3	0.0	0.84
treatment	4	3.3	0.16

[a]*All models contain pair as a random effect, and experiment (2006–2008) models also include year as a random effect. Covariates: treatment, lek treatment (noise or control) assigned to individual leks within a pair; type, pair noise treatment type (road or drilling assigned to pair); season, time of year (early [late February to 1 week prior to peak female attendance for that lek; female peak ranged from 15 March to 6 April], mid [1 week before and after female peak], and late [starting 1 week after female peak]); experiment year, years of experimental noise exposure.*
[b]*Number of parameters in the model.*
[c]*Difference in AIC_c (Akaike's information criterion for small sample size) values from the model with lowest AIC_c.*
[d]*Akaike weight.*
[e]*Model with substantial support ($\Delta AIC_c < 2$).*

randomly assigned to receive experimental noise treatment and the other lek was designated a control. We randomly assigned the experimental leks to receive playback of either drilling or road noise. In 2006, we counted attendance at 8 leks (2 treated with drilling noise, 2 treated with road noise, and 4 control). In both 2007 and 2008, we included an additional 8 leks for a total of 16 leks (4 treated with drilling noise, 4 treated with road noise, and 8 controls).

Throughout the breeding season, we counted males and females on leks with a spotting scope from a nearby point selected to maximize our visibility of the lek. We visited paired leks sequentially on the same days between 05:00 and 09:00, alternating the order in which each member of the pair was visited. We visited lek pairs every day during the breeding season in 2006 and, after expanding our sample size in 2007, every 2–4 days in 2007 and 2008. Peak estimates of male attendance from >4 visits are a highly repeatable measure of abundance at individual leks (Garton et al. 2010), so the lower frequency of visits in 2007 and 2008 was unlikely to have a substantial effect on estimates of peak male attendance. At a minimum, we conducted 2 counts per visit at 10- to 15-min intervals. The annual peak attendance was the highest daily attendance value at each lek for the season for males or females. For males we also calculated the peak attendance in 3 nonoverlapping date ranges: early (late February to 1 week prior to peak female attendance for that lek; female peak ranged from 15 March to 6 April), mid (1 week before and after female peak), and late (starting 1 week after female peak).

Noise Introduction

We recorded noise used for playback near natural gas drilling sites and gas-field access roads in a region of extensive deep natural gas development in Sublette County, Wyoming (Pinedale Anticline Gas Field and Jonah Gas Field). We recorded drilling noise in 2006 within 50 m of the source on a digital recorder (model PMD670, 44.1 kHz/16 bit; Marantz, Mahwah, New Jersey) with a shotgun microphone (model K6 with an ME60 capsule; Sennheiser, Old Lyme, Connecticut). We recorded road noise in 2005 with a handheld computer (iPAQ h5550 Pocket PC, 44.1 KHz/16 bit; Hewlett Packard, Palo Alto, California) and omnidirectional microphone (model K6 with an ME62 capsule; Sennheiser). Drilling noise is relatively continuous and road noise is intermittent (Supporting Information). Both types of noise are predominantly low frequency (<2 kHz).

We played noise on experimental leks from 2 to 4 rock-shaped outdoor speakers (300 W Outdoor Rock Speakers; TIC Corporation, City of Industry, California) hooked to a car amplifier (Xtant1.1; Xtant Technologies, Phoenix, Arizona) and an MP3 player (Sansa m240; SanDisk, Milpitas, California). The playback system was powered with 12 V batteries that we changed every 1–3 days when no birds were present. We placed the speakers

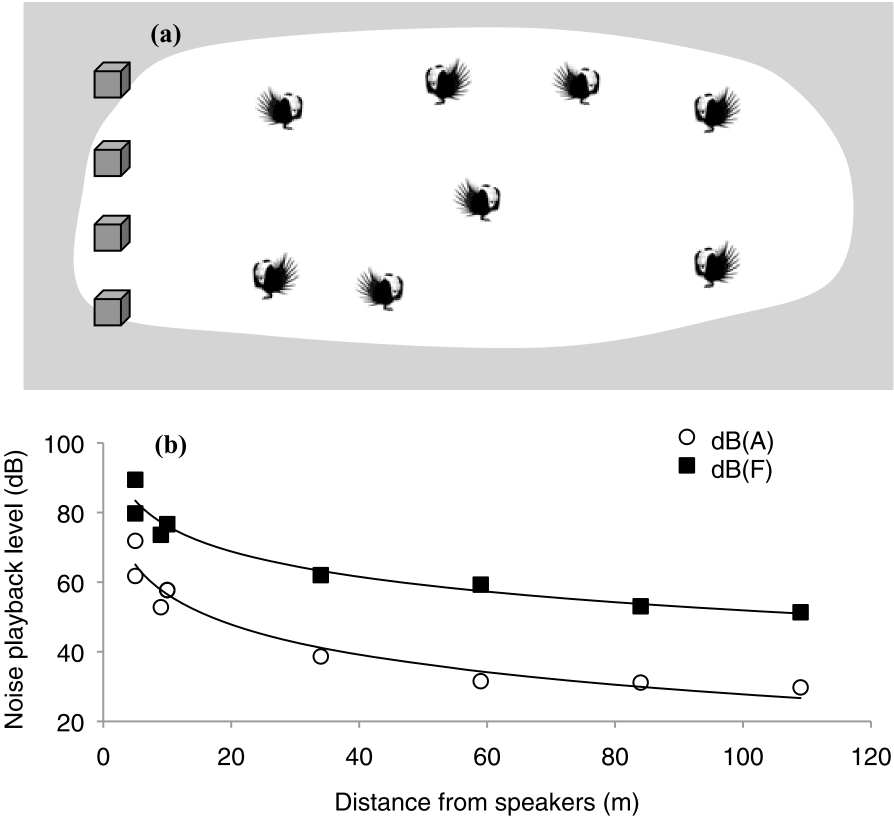

Figure 1. (a) Placement of speakers (on noise-treated leks) or dummy speakers (on control leks) (boxes) at Greater Sage-Grouse leks. (b) Mean maximum noise level (unweighted decibels, dB[F], and A-weighted decibels, dB[A], measured in Lmax [highest root-mean-square sound pressure level within the measurement period]) at Greater Sage-Grouse leks measured on transects at 25-m intervals from the line of speakers on a typical lek treated with road noise. Playback levels of natural gas drilling noise (measured in Leq) followed the same pattern. Ambient levels of noise at control leks ranged from 30 to 35 dB(A).

in a straight line across one end of the lek (Fig. 1a). In 2006 we placed 3 speakers at leks treated with drilling noise and 2 speakers at leks treated with road noise. In 2007 and 2008, we increased the number of speakers, placing 4 at each noise-treated lek to increase the area in which noise was present on the lek. At control leks, we placed dummy speakers of similar size and color to playback speakers (68-L plastic tubs). Within each lek pair, dummy and real speakers were placed in similar configurations. To control for playback-related disturbance, the leks in each pair were visited an equal number of times during the morning for counts of birds and in the afternoon for battery changes.

We played drilling noise and road noise on leks at 70 dB(F) sound pressure level (unweighted decibels) measured 16 m directly in front of the speakers (Fig. 1 & Supporting Information). This is similar to noise levels measured approximately 400 m from drilling rigs and main access roads in Pinedale (J. L. Blickley and G. L. Patricelli, unpublished data). Four hundred meters (0.25 miles) is the minimum surface disturbance buffer around leks at this location (BLM 2008). We calibrated and measured noise playback levels with a hand-held meter that provides sound-pressure levels (System 824; Larson-Davis, Depew, New York) when wind was <9.65 k/h. On drilling-noise-treated leks, where noise was continuous, we calibrated the noise playback level by measuring the average sound level (Leq [equivalent continuous sound level]) over 30 s. On leks treated with road noise, where the amplitude of the noise varied during playback to simulate the passing of vehicles, we calibrated the playback level by measuring the maximum sound level (Lmax [highest root-mean-square sound pressure level within the measurement period]).

For leks treated with drilling noise, recordings from 3 drilling sites were spliced into a 13-min mp3 file that played on continuous repeat. On leks treated with road noise, we randomly interspersed mp3 recordings of 56 semitrailers and 61 light trucks with 170 thirty-second silent files to simulate average levels of traffic on an access road (Holloran 2005). Noise playback on experimental leks continued throughout April in 2006, from mid February or early March through late April in 2007, and from late February through late April in 2008. We played back noise on leks 24 hours/day because noise from deep natural gas drilling and vehicular traffic is present at all times. This experimental protocol was reviewed and approved by the Animal Care and Use Committee at University of California, Davis (protocol 16435).

To measure noise levels across experimental leks, we measured the average amplitude (15 s Leq) of white-noise played at 1–5 points along transects that extended across the lek at 25-m intervals roughly parallel to the line of speakers. We calibrated white-noise measurements by measuring the noise level of both the white noise and either a representative clip of drilling noise or a semitrailer

10 m directly in front of each speaker. To minimize disturbance, we took propagation measurements during the day. Daytime ambient noise levels are typically 5–10 dBA higher than those in the early morning (J. L. Blickley and G. L. Patricelli, unpublished data) and are likely higher than those heard by birds at a lek.

After the experiment, we counted individuals on all leks 2–6 times from 1 March through 30 April 2009. In 2009 we continued to play noise on 2 experimental leks as part of a related experiment, so we did not include these lek pairs in our analysis of postexperiment male attendance at a lek.

Response Variables and Baseline Attendance Levels

Sage grouse leks are highly variable in size and, even within pairs, our leks varied up to 50% in size. To facilitate comparison of changes in attendance on leks of different sizes, we calculated the attendance relative to attendance levels before treatment (i.e., baseline attendance levels). We obtained male baseline abundance from the Wyoming Game and Fish Department. We used the standard lek-count protocol (Connelly et al. 2003) to count birds at leks approximately 3 times/breeding season. Due to the small number of counts in pre-experiment years, we calculated male baseline attendance by averaging the annual peak male attendance at each individual lek over 4 years (2002–2005). We assessed changes in early-, mid-, and late-season peak male attendance from this 4-year baseline attendance. Female attendance was highly variable throughout the season with a short (1–3 day) peak in attendance at each lek. Due to the limited number of annual counts, female counts from 2002 to 2005 were not reliable estimates of peak female attendance and could not be used as baseline attendance levels. Because we introduced noise to experimental leks after the peak in female attendance in 2006, we used maximum female counts from 2006 as a baseline for each of the 8 leks monitored that year. We assessed changes in annual peak female attendance from this 1-year baseline attendance. The 8 leks added to the experiment in 2007 were not included in statistical analyses of female attendance due to the lack of a baseline.

Statistical Analyses

We used an information-theoretic approach to evaluate the support for alternative candidate models (Table 2). All candidate models were linear mixed-effect models that assessed the relation between covariates and the proportional difference in annual and within-season peak attendance and baseline attendance (both males and female) (Tables 2 & 3). We ranked models on the basis of differences in Akaike's information criterion for small sample sizes (ΔAIC_c) (Burnham & Anderson 2002). Akaike weights (w_i) were computed for each model on the basis of ΔAIC_c scores. We calculated model-averaged variable coefficients, unconditional 95% CI, and variable importance (weight across models) for variables contained in models that were strongly supported ($\Delta AIC_c < 2$). All statistical analyses were performed in R (version 2.12.1) (R Development Team 2010).

The detection probability for males and females is likely to vary across a season and among leks (Walsh et al. 2004). We sought to minimize sources of error and maximize detection by conducting frequent counts from locations with a clear view of the lek and by implementing a paired treatment design (each noise lek is compared with a similar control lek, monitored by the same observer on the same days). To ensure that detection probability did not differ among noise and control leks, we corrected our data for detection probability. First, we used detection error rates, estimated as difference between the maximum count and the count immediately before or after the maximum count within a day (for both males and females), and then we applied the bounded-count method (for males only; Walsh et al. 2004). With the multiple-count estimator, estimates of detection between noise and control leks did not differ (males: $t = 1.02$, df $= 6$, $p = 0.35$; females: $t = 0.21$, df $= 3$, $p = 0.84$). We analyzed both corrected and uncorrected counts and found that neither correction qualitatively changed our results; therefore, results are presented for uncorrected counts.

Table 3. Mixed-effect candidate models used to assess change in peak annual attendance of female Greater Sage-Grouse at leks from pre-experiment baseline attendance in 2006 during noise playback.

Model[a]	K[b]	ΔAIC_c[c]	w_i[d]
Null, random effects only[e]	4	0	0.71
Treatment[e]	5	1.9	0.27
Treatment+experiment year	6	8	0.01
Treatment×experiment year	7	14	<0.001

[a] *All models contained pair and year as random effects. Due to the small sample size (4 pairs), pair type variable (road versus drilling) was not included in the model set. Covariates: treatment, lek treatment (noise or control assigned to individual leks within a pair); experiment year, years of experimental noise exposure.*
[b] *Number of parameters in the model.*
[c] *Difference in AIC_c (Akaike's information criterion for small sample size) values from the most strongly supported (lowest AICc) model.*
[d] *Akaike weight.*
[e] *Model with substantial support (ΔAIC_c < 2).*

Results

Male Attendance

Peak male attendance at both types of noise leks decreased more than attendance at paired control leks, but the decreases varied by noise type. In the most strongly supported models of the candidate set ($w_i = 0.90$, all

Table 4. Model-averaged parameter direction and effect sizes and variable importance for all variables present in strongly supported models ($\Delta AIC_c < 2$ in Table 2) of changes in peak attendance of male greater sage-grouse at leks from baseline attendance during experimental noise playback.

Variable	*Percent effect size (SE)*	*Variable importance**
Intercept	31 (22)	1.0
Treatment, noise	−29 (7)	0.91
Type, road	33 (22)	0.91
Treatment, noise∗type, road	−40 (10)	0.91
Season, mid	18 (6)	0.66
Season, late	23 (6)	

**Variable importance is the summed weight of all models containing that variable.*

other models $\Delta AIC_c > 6.1$) (Table 2), there was an interaction of the effects of experimental treatment (control versus noise) and noise type (drilling versus road) on annual peak male attendance. At leks treated with road noise, decreases in annual peak male attendance were greater (73%), relative to paired controls, than at drilling noise leks (29%). As indicated by the effect size for the main effect of pair type, attendance at control leks paired with road noise leks was 33% greater relative to the baseline than control leks paired with drilling noise leks (Table 4). However, changes in attendance were compared within a pair to control for such differences. Male attendance increased over the course of a season, with 18% and 23% increases in peak male attendance in mid and late season from the early-season peaks, but seasonal increases were similar across noise and control leks (Table 4 & Fig. 2b).

There was no evidence that the effect of noise on attendance changed as years of exposure to noise increased. The models with substantial support did not contain a main effect of years of exposure or an interaction of years of exposure and treatment type (control versus noise) (Table 2). In spite of decreases in attendance throughout the experiment, peak male attendance exceeded baseline attendance on all leks in 2006, 13 leks in 2007, and 11 leks in 2008 (Table 4 & Fig. 2c). There was an increase in sage grouse abundance regionally in 2006 (Fig. 3).

After the experiment (2009), attendance at leks we experimentally exposed to drilling and road noise was lower relative to paired controls (Table 2). The model that included the treatment variable showed an effect size of −30% (across road and drilling noise leks) but had only moderate support ($\Delta AIC_c = 3.3$) relative to the null model.

Female Attendance

Peak female attendance at leks treated with noise in 2007 and 2008 decreased from the 2006 baseline, relative to control leks (Table 3). The most strongly supported model in the set was the null model; however, the model that included noise treatment was highly supported ($\Delta AIC_c < 2$). The effect size of noise treatment on female attendance was −48% (10% SE), which is similar to the effect of noise on male attendance averaged across both noise types (51%).

Discussion

Results of previous studies show abundance of Greater Sage-Grouse decreases when natural gas and coal-bed methane fields are developed (Holloran 2005; Walker et al. 2007; Doherty et al. 2008). Our results suggest that chronic noise may contribute to these decreases. Peak male attendance relative to the baseline was lower on noise leks than paired control leks, and the decrease was larger at road noise leks (73% decrease in abundance compared with paired controls) than drilling noise leks (29%; Fig. 3). These decreases were immediate and sustained. The effects of noise occurred in the first year of the study and were observed throughout the experiment, although patterns of male attendance within a season were similar at noise and control leks. Differences in male attendance between noise and control leks in the year after the experiment were not supported in the top models, which suggests attendance rebounded after noise ceased. However, the sample size for this analysis was small, and the effect size (30% average decreases in male attendance for both noise types) suggests a residual effect of noise.

There are 2 mechanisms by which noise may reduce male attendance. First, males on noise leks may have had higher mortality than males on control leks. Noise playback was not loud enough to cause direct injury to individuals, but mortality could be increased indirectly by noise playback if the sounds of predators (coyotes [*Canis latrans*] or Golden Eagles [*Aquila chrysaetos*]) were masked by noise. However, on-lek predation events were rare. We observed ≤1 predation event per lek per season during the experiment (observations of sage-grouse carcasses or feathers at a lek [J. L. Blickley, personal observation]). The cumulative effect of rare predation events would lead to a gradual decrease in attendance, rather than the rapid and sustained decrease we observed. Furthermore, experimental noise was likely too localized to substantially affect off-lek predation because noise levels decreased exponentially as distance to the speakers increased (Fig. 1b). To date, increased predation risk of adults due to anthropogenic noise has not been demonstrated in any species, but some species increase vigilance when exposed to noise, leaving less time for feeding, displaying, and other important behaviors (Quinn et al. 2006; Rabin et al. 2006). Noise may also affect off-lek mortality indirectly. For example, noise-stressed males may be more susceptible to disease due to a suppressed

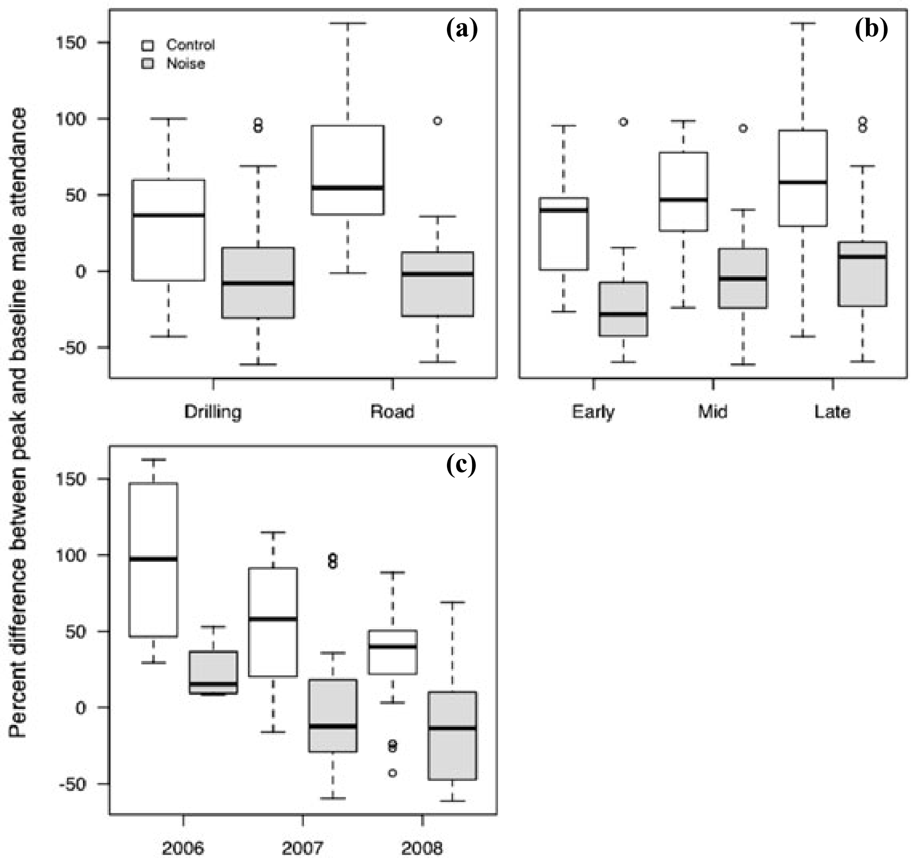

Figure 2. Percent difference between baseline attendance (i.e., abundance before experiments) of male Greater Sage-Grouse and (a) peak male attendance on control leks and leks treated with noise from natural gas drilling and road noise, (b) peak male attendance in the early (late February to 1 week prior to peak female attendance for that lek), mid (1 week before and after female peak [female peak ranged from 15 March to 6 April]), and late (starting 1 week after female peak) breeding season; on control leks and leks treated with noise, and (c) peak male attendance at control leks and leks treated with noise in experimental years 2006, 2007, and 2008 in Fremont County, Wyoming (U.S.A.) (horizontal lines, median value; box ends, upper and lower quartiles, whiskers, maximum and minimum values). Data are observed values, not model output.

immune response (Jankowski et al. 2010). Although long-term stress from noise is unlikely to be the primary cause of the rapid decreases in attendance we observed here, it may have been a contributing factor over the course of the experiment. Furthermore, in areas of dense industrial development, where noise is widespread, noise effects on mortality may be more likely.

Alternatively, noise may lower male attendance through displacement, which would occur if adult or juvenile males avoid leks with anthropogenic noise. Such behavioral shifts are consistent with the rapid decreases in attendance we observed. Adult male sage grouse typically exhibit high lek fidelity (Schroeder & Robb 2003) and visit leks regularly throughout the season, whereas juvenile males visit multiple leks and their attendance peaks late in the season (Kaiser 2006). If juveniles or adults avoid noise by visiting noisy leks less frequently or moving to quieter leks, overall attendance on noisy leks could be reduced. We could not reliably differentiate between juveniles and adults, so we do not know the relative proportion of adults and juveniles observed. Consistent with displacement due to noise avoidance, radio-collared juvenile males avoid leks near deep natural gas developments in Pinedale, Wyoming, which has resulted in decreases in attendance at leks in close proximity to development and increased attendance at nearby leks with less human activity (Kaiser 2006; Holloran et al. 2010). Reduced recruitment of juvenile males is unlikely to be the only driver of the patterns we observed because we did not observe larger decreases in lek attendance on noise-treated leks later in the season, when juvenile attendance peaks. Rather, we found immediate decreases in attendance early in the season when playback began (Fig. 2b), at which time there are few juveniles on the lek. This

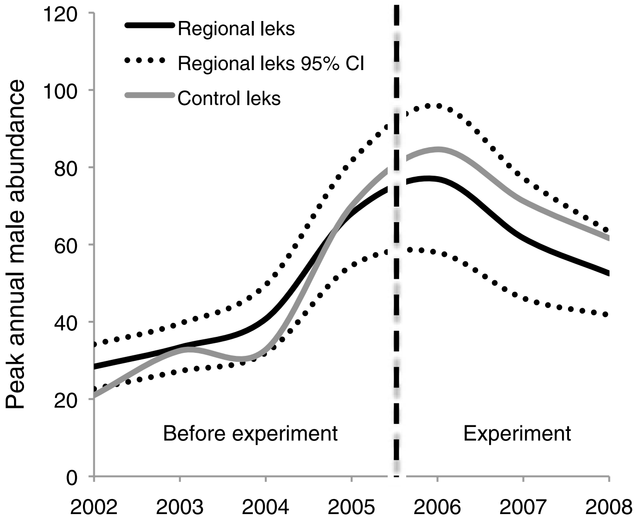

*Figure 3. Maximum abundance of male Greater Sage-Grouse from 2002 to 2008 at control leks (*n = 8*) (no anthropogenic sound played) and other leks in the region that were not part of the experiment (regional leks) (*n = 38*).*

is consistent with both adult and juvenile noise avoidance. We did not find evidence for a cumulative negative effect of noise on lek attendance, although cumulative effects may have been masked by regional population declines after 2006, a year of unusually high abundance (Fig. 3).

Female attendance at leks treated with noise was lower than that on control leks; however, the null model and the model that included noise treatment were both highly supported, providing only moderate support for the effects on noise on attendance. For this model, the overall estimated effect of noise on female attendance (−48%) was similar to that of the effect of noise on male attendance. Due to the high variability of female daily maximum attendance throughout the season and small sample size for this analysis (female attendance data available for only 4 of the 8 lek pairs), our statistical power to detect differences in female attendance was limited and effect sizes may not be representative of actual noise effects.

Our results suggest that males and possibly females avoid leks exposed to anthropogenic noise. A potential cause of avoidance is the masking of communication. Masked communication is hypothesized to cause decreases in abundance of some animal species in urban and other noisy areas. For example, bird species with low-frequency vocalizations are more likely to have low abundance or be absent from natural gas developments, roads, and urban areas than species with high-frequency vocalizations, which suggests that masking is the mechanism associated with differences in abundance (Rheindt 2003; Francis et al. 2009; Hu & Cardoso 2010). Sage-grouse may be particularly vulnerable to masked communication because their low-frequency vocalizations are likely to be masked by most sources of anthropogenic noise, including the noises we played in our experiment (Supporting Information). This may be particularly important for females if they cannot use acoustic cues to find leks or assess displaying males in noisy areas.

Alternatively, individuals may avoid noisy sites if noise is annoying or stressful, particularly if this noise is associated with danger (Wright et al. 2007). Intermittent road noise was associated with lower relative lek attendance than continuous drilling noise, in spite of the overall higher mean noise levels and greater masking potential at leks treated with drilling noise (Supporting Information). Due to the presence of roads in our study area, sage grouse may have associated road noise with potentially dangerous vehicular traffic and thus avoided traffic-noise leks more than drilling-noise leks. Alternatively, the pattern of decrease may indicate that an irregular noise is more disturbing to sage grouse than a relatively continuous noise. Regardless, our results suggest that average noise level alone is not a good predictor of the effects of noise (Slabbekoorn & Ripmeester 2008) and that species can respond differently to different types of noise.

Our results cannot be used to estimate the quantitative contribution of noise alone to observed decreases in Greater Sage-Grouse abundance at energy development sites because our experimental design may have led us to underestimate or overestimate the magnitude of these effects. Decreases in abundance due to noise could be overestimated in our study if adults and juveniles are displaced from noise leks and move to nearby control leks, which would have increased the difference in abundance between paired leks. Similar displacement occurs in areas of energy development, but over a much larger extent than is likely to have occurred in response to localized playbacks in our experiment (Holloran et al. 2010).

In contrast, we could have underestimated noise effects if there were synergistic effects of noise and other disturbances associated with energy development. For example, birds with increased stress levels due to poor forage quality may have lower tolerance for noise-induced stress, or vice versa. Noise in our experiment was localized to the immediate lek area and only played during the breeding season, so we cannot quantify the effects of noise on wintering, nesting, or foraging birds. Noise at energy development sites is less seasonal and more widespread than noise introduced in this study and may thus affect birds at all life stages and have a potentially greater effect on lek attendance. Leks do not represent discrete populations; therefore, local decreases in lek attendance do not necessarily reflect population-level decreases in abundance. However, at large energy development sites, similar displacement of Greater Sage-Grouse away from the ubiquitous noise may result in population-level declines due to spatially extensive changes in land use or increases in dispersal-related and density-dependent sources of mortality (Aldridge & Boyce 2007). Enforcement and refinement of existing seasonal restrictions on human activity could potentially reduce these effects.

We focused on the effect of noise associated with deep natural gas and coal-bed methane development on sage grouse, but our results may increase broader understanding of the effects of noise on animals. Both intermittent and constant noise from energy development affected sage grouse. Other noise sources with similar frequency range and temporal pattern, such as wind turbines, oil-drilling rigs, and mines, may have comparable effects. Similar effects may also be associated with highways, off-road vehicles, and urbanization so that the potential for noise to have an effect is large.

We believe that noise should be investigated as one potential cause of population declines in other lekking North American grouse species that are exposed to similar anthropogenic development. Populations of many bird (van der Zande et al. 1980; Rheindt 2003; Ingelfinger & Anderson 2004) and mammal (Forman & Deblinger 2000; Sawyer et al. 2009) species have been shown to decrease in abundance in response to road, urban, and energy development, and noise produced by these activities may contribute to these decreases. Our results also demonstrate that wild animals may respond differently to chronic intermittent and continuous noise, a comparison that should be expanded to other species. Additionally, we think these results highlight that experimental noise playbacks may be useful in assessing the response of wild animals to chronic noise (Blickley & Patricelli 2010).

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Supporting Information

Spectrograms and power spectrums of drilling noise, road noise and male sage-grouse vocal display (Appendix S1), map of experimental and control leks (Appendix S2), and noise playback levels on experimental leks (Appendix S3) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

Literature Cited

Aldridge, C., and M. Boyce. 2007. Linking occurrence and fitness to persistence: habitat-based approach for endangered greater sage-grouse. Ecological Applications **17:**508–526.

Barber, J. R., K. R. Crooks, and K. M. Fristrup. 2009. The costs of chronic noise exposure for terrestrial organisms. Trends in Ecology & Evolution **25:**180–189.

Bayne, E., L. Habib, and S. Boutin. 2008. Impacts of chronic anthropogenic noise from energy-sector activity on abundance of songbirds in the boreal forest. Conservation Biology **22:**1186–1193.

Blickley, J. L., and G. L. Patricelli. 2010. Impacts of anthropogenic noise on wildlife: research priorities for the development of standards and mitigation. Journal of International Wildlife Law & Policy **13:**274–292.

BLM (Bureau of Land Management). 2008. Pinedale Anticline Project area. Supplemental environmental impact statement record of decision. Appendix A. BLM, Pinedale, Wyoming.

Boyko, A. R., R. M. Gibson, and J. R. Lucas. 2004. How predation risk affects the temporal dynamics of avian leks: greater sage grouse versus golden eagles. The American Naturalist **163:**154–165.

Burnham, K. P., and D. R. Anderson 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer-Verlag, New York.

Connelly, J. W., K. P. Reese, and M. A. Schroeder. 2003. Monitoring of greater sage-grouse habitats and populations. College of Natural Resources Experiment Station, University of Idaho, Moscow.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming.

Copeland, H. E., K. E. Doherty, D. E. Naugle, A. Pocewicz, and J. M. Kiesecker. 2009. Mapping oil and gas development potential in the US intermountain west and estimating impacts to species. Public Library of Science One **4:**e7400. DOI: 10.1371.

Doherty, K., D. Naugle, B. Walker, and J. Graham. 2008. Greater sage-grouse winter habitat selection and energy development. Journal of Wildlife Management **72:**187–195.

Forman, R. T. T. 2000. Estimate of the area affected ecologically by the road system in the United States. Conservation Biology **14:** 31–35.

Forman, R. T. T., and R. D. Deblinger. 2000. The ecological road-effect zone of a Massachusetts(U. S. A.) suburban highway. Conservation Biology **14:**36–46.

Francis, C. D., C. P. Ortega, and A. Cruz. 2009. Noise pollution changes avian communities and species interactions. Current Biology **19:**415–419.

Francis, C., C. Ortega, and A. Cruz. 2011. Vocal frequency change reflects different responses to anthropogenic noise in two suboscine tyrant flycatchers. Proceedings of the Royal Society B: Biological Sciences **278:**2025–2031.

Garton, E. O., J. W. Connelly, J. S. Horne, C. A. Hagen, A. Moser, and M. A. Schroeder. 2010. Greater sage-grouse population dynamics and probablity of persistence. Pages 293–382. Studies in avian biology. University of California Press, Berkeley.

Gibson, R. M. 1989. Field playback of male display attracts females in lek breeding sage grouse. Behavioral Ecology and Sociobiology **24:**439–443.

Gibson, R. M. 1996. Female choice in sage grouse: the roles of attraction and active comparison. Behavioral Ecology and Sociobiology **39:**55–59.

Habib, L., E. M. Bayne, and S. Boutin. 2007. Chronic industrial noise affects pairing success and age structure of ovenbirds Seiurus aurocapilla. Journal of Applied Ecology **44:**176–184.

Holloran, M. J. 2005. Greater sage-grouse (*Centrocercus urophasianus*) population response to natural gas field development in western

Wyoming. Department of Zoology and Physiology, University of Wyoming, Laramie.

Holloran, M., R. Kaiser, and W. Hubert. 2010. Yearling greater sage-grouse response to energy development in Wyoming. Journal of Wildlife Management **74:**65–72.

Hu, Y., and G. C. Cardoso. 2010. Which birds adjust the frequency of vocalizations in urban noise? Animal Behaviour **79:**863–867.

Ingelfinger, F., and S. Anderson. 2004. Passerine response to roads associated with natural gas extraction in a sagebrush steppe habitat. Western North American Naturalist **64:**385–395.

Jankowski, M. D., J. C. Franson, E. Möstl, W. P. Porter, and E. K. Hofmeister. 2010. Testing independent and interactive effects of corticosterone and synergized resmethrin on the immune response to West Nile virus in chickens. Toxicology **269:**81–88.

Kaiser, R. 2006. Recruitment by greater sage-grouse in association with natural gas development in western Wyoming. Department of Zoology and Physiology, University of Wyoming, Laramie.

Lengagne, T. 2008. Traffic noise affects communication behaviour in a breeding anuran, *Hyla arborea*. Biological Conservation **141:**2023–2031.

Leonard, M., and A. Horn. 2008. Does ambient noise affect growth and begging call structure in nestling birds? Behavioral Ecology **19:**502–507.

Naugle, D. E., et al. 2004. West Nile virus: pending crisis for greater sage-grouse. Ecology Letters **7:**704–713.

Patricelli, G., and J. Blickley. 2006. Avian communication in urban noise: causes and consequences of vocal adjustment. The Auk **123:**639–649.

Patricelli, G., and A. Krakauer. 2010. Tactical allocation of effort among multiple signals in sage grouse: an experiment with a robotic female. Behavioral Ecology **21:**97–106.

Quinn, L., J. Whittingham, J. Butler, and W. Cresswell. 2006. Noise, predation risk compensation and vigilance in the chaffinch Fringilla coelebs. Journal of Avian Biology **37:**601–608.

Rabin, L. A., R. G. Coss, and D. H. Owings. 2006. The effects of wind turbines on antipredator behavior in California ground squirrels (*Spermophilus beecheyi*). Biological Conservation **131:**410–420.

Rheindt, F. E. 2003. The impact of roads on birds: does song frequency play a role in determining susceptibility to noise pollution? Journal of Ornithology **144:**295–306.

Richardson, W. J. 1995. Marine mammals and noise. Academic Press, San Diego, California.

Sawyer, H., M. Kauffman, and R. Nielson. 2009. Influence of well pad activity on winter habitat selection patterns of mule deer. Journal of Wildlife Management **73:**1052–1061.

Schroeder, M., and L. Robb. 2003. Fidelity of greater sage-grouse *Centrocercus urophasianus* to breeding areas in a fragmented landscape. Wildlife Biology **9:**291–299.

Slabbekoorn, H., and E. A. P. Ripmeester. 2008. Birdsong and anthropogenic noise: implications and applications for conservation. Molecular Ecology **17:**72–83.

Sun, J. W. C., and P. M. Narins. 2005. Anthropogenic sounds differentially affect amphibian call rate. Biological Conservation **121:**419–427.

Tyack, P. L., et al. 2011. Beaked whales respond to simulated and actual navy sonar. Public Library of Science One **6:**e17009. DOI: 10.1371.

van der Zande, A. N., W. J. ter Keurs, and W. J. van der Weijden. 1980. The impact of roads on the densities of four bird species in an open field habitat—evidence of a long-distance effect. Biological Conservation **18:**299–321.

Vehrencamp, S., J. Bradbury, and R. Gibson. 1989. The energetic cost of display in male sage grouse. Animal Behaviour **38:**885–896.

Walker, B. L., D. E. Naugle, and K. E. Doherty. 2007. Greater sage-grouse population response to energy development and habitat loss. Journal of Wildlife Management **71:**2644–2654.

Walsh, D. P., G. C. White, T. E. Remington, and D. C. Bowden. 2004. Evaluation of the lek-count index for greater sage-grouse. Wildlife Society Bulletin **32:**56–68.

Warren, P. S., M. Katti, M. Ermann, and A. Brazel. 2006. Urban bioacoustics: it's not just noise. Animal Behaviour **71:**491–502.

Weisenberger, M. E., P. R. Krausman, M. C. Wallace, D. W. D. Young, and O. E. Maughan. 1996. Effects of simulated jet aircraft noise on heart rate and behavior of desert ungulates. The Journal of Wildlife Management **60:**52–61.

Wiley, R. H. 1973. Territoriality and non-random mating in sage grouse (*Centrocercus urophasianus*). Animal Behaviour Monographs **6:**85–169.

Wright, A., et al. 2007. Anthropogenic noise as a stressor in animals: a multidisciplinary perspective. International Journal of Comparative Psychology **20:**250–273.

Experimental Chronic Noise Is Related to Elevated Fecal Corticosteroid Metabolites in Lekking Male Greater Sage-Grouse (*Centrocercus urophasianus*)

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Abstract

There is increasing evidence that individuals in many species avoid areas exposed to chronic anthropogenic noise, but the impact of noise on those who remain in these habitats is unclear. One potential impact is chronic physiological stress, which can affect disease resistance, survival and reproductive success. Previous studies have found evidence of elevated stress-related hormones (glucocorticoids) in wildlife exposed to human activities, but the impacts of noise alone are difficult to separate from confounding factors. Here we used an experimental playback study to isolate the impacts of noise from industrial activity (natural gas drilling and road noise) on glucocorticoid levels in greater sage-grouse (*Centrocercus urophasianus*), a species of conservation concern. We non-invasively measured immunoreactive corticosterone metabolites from fecal samples (FCMs) of males on both noise-treated and control leks (display grounds) in two breeding seasons. We found strong support for an impact of noise playback on stress levels, with 16.7% higher mean FCM levels in samples from noise leks compared with samples from paired control leks. Taken together with results from a previous study finding declines in male lek attendance in response to noise playbacks, these results suggest that chronic noise pollution can cause greater sage-grouse to avoid otherwise suitable habitat, and can cause elevated stress levels in the birds who remain in noisy areas.

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Introduction

Anthropogenic noise is becoming ubiquitous as natural landscapes are increasingly dominated by humans, but we still have much to learn about the impacts of chronic noise exposure on wildlife [1–3]. Recent studies have shown that some species avoid developed areas with high noise levels, reducing available habitat and potentially leading to reduced populations [4–6]. However, there is variation among species and individuals in the tendency to avoid noise [4,5,7], which raises the question of whether animals that remain suffer detrimental effects, or if these individuals are better able to habituate to noise or are less susceptible to its effects. It has been suggested that animals remaining in (or unable to leave) noisy areas may have lower survival and reproductive success [8–10]; indeed, recent studies have demonstrated complex effects of noise on community structure and on breeding and pairing success [4–6,11]. Given the ubiquity of noise in the environment, it is critical that we understand noise impacts on animals whether they remain in or avoid disturbed areas.

One possible impact of introduced noise on animals is the induction of stress, which may be defined broadly as nonspecific adverse effects in vertebrates but is most often characterized by its influence on neuroendocrine physiology. The duration of noise exposure affects the stress response of animals exposed to it [12]. Exposure to a brief but loud noise event, such as a single sonic boom, will result in an acute stress response. An acute stress response is characterized by a rapid release of epinephrine and norepinephrine (the "fight or flight" response) followed by a hypothalamic-pituitary-adrenal (HPA) cascade. The HPA cascade results in increased secretion of glucocorticoid hormones, cortisol or corticosterone, in the blood. Long-term exposure to a chronic noise stressor, such as a high-traffic freeway, can lead to chronic stress, defined as long-term overstimulation of coping mechanisms. This in turn can lead to less predictable changes in the HPA axis. Acclimation or exhaustion may result in reduced glucocorticoid release to the same or novel stressors; facilitation, conversely, can lead to elevated glucocorticoid release in response to novel stressors, and even in cases of reduced peak glucocorticoid response, deficits in negative feedback may develop that result in greater overall exposure to glucocorticoids due to prolonged elevation [12,13].

Glucocorticoid hormones and their metabolites are commonly used to measure a stress response [14–16]. Glucocorticoid hormones can be measured from blood samples or their metabolites may be measured non-invasively from fecal samples as an index of the relative physiological stress of animals [17–19]. Glucocorticoid hormones play a major role in allocating energy, and prolonged exposure due to chronic stress can affect fitness by inhibiting resource allocation to reproductive or immune activities, a condition known as allostatic overload [12,20–24].

Studies in captive animals have found that noise can increase HPA activity and glucocorticoid levels [25,26]; indeed studies of stress physiology often use noise exposure as a method to induce a stress response [27,28]. Previous observational and experimental studies on the impacts of anthropogenic noise on glucocorticoid levels in wild animals have yielded mixed results. Snowmobile and wheeled-vehicle traffic was associated with elevated fecal glucocorticoid metabolites in wolves and elk [14]. Noise is one potential mechanism of this impact, but visual and other types of disturbance may also contribute to these responses; indeed, the quieter activity of Nordic skiing also correlates with FCMs in capercaillie (*Tetrao urogallus*) [29]. Delaney et al. [30] found behavioral responses in spotted owls to loud noise from visually hidden chainsaws and helicopters, but subsequent studies found no evidence of change in FCMs with exposure to quieter chainsaw noise (below behavioral response threshold) or road proximity to nesting sites [31]. Results from chronic noise studies on humans have also been mixed [32]. Studies of children in areas with high road noise have found increased overnight glucocorticoid levels in urine, as well as impaired circadian rhythms, sleep, memory and concentration, [33] and increased heart-rate responsiveness to acute stressors [34]. However, a study in children living in communities near airports found increases in some measures of stress (blood pressure, epinephrine and norepinephrine) but no similar elevation in overnight urinary cortisol [35]. These results indicate that noise may have a significant effect on glucocorticoids and other stress-related variables in many species, but that further study is needed to determine the degree and extent of these effects and how the effects may vary with different types of noise.

In this study, we test the hypothesis that chronic noise causes an increase in stress levels of lekking greater sage-grouse. We used fecal levels of immunoreactive corticosteroid metabolites (FCMs) as an index of physiological stress and compared FCMs for breeding males on display grounds (leks) with and without experimentally introduced noise. The greater sage-grouse, an iconic species once widespread in western North America, is now declining throughout its range, leading to its listing as an endangered species in Canada and its recent designation as "warranted but precluded" for listing under the Endangered Species Act in the USA [36,37]. Over the last decade, natural gas development has expanded rapidly across much of the sage-grouse range and has been implicated in reduced lek attendance and abandonment of long-occupied (often for decades) lek sites by males [e.g. 38,39–41]. Males typically gather on lekking grounds for several hours in the early morning when conditions are quiet and still, a time when they may be particularly vulnerable to disturbance from noise pollution from natural gas development and other sources [42]. To investigate whether noise exposure may have contributed to declines in lek attendance, Blickley et al. [43] experimentally introduced noise from natural gas development activities (drilling and road noise) on leks over three breeding seasons (2006–2008). This noise playback caused immediate and sustained declines in sage-grouse lek attendance. Further, different types of noise had different degrees of impact, with drilling noise and road noise causing an average 29% and 73% decline in lek attendance, respectively, compared to their paired controls. That study provides evidence that anthropogenic noise from energy development causes some males to avoid attending leks with introduced noise, but we do not yet know whether noise also has a negative impact on the individuals that remain on noisy leks. The lekking season is a time of high metabolic demand [44] and stress [45] for males, so exposure to noise during this period may have a greater fitness cost.

Here we compare the FCM levels of male sage-grouse on control leks and leks with experimentally introduced noise in the second and third seasons of experimental noise playback (2007 and 2008) [43]. We predict that if noise exposure leads to chronic stress, male sage-grouse on experimental leks will have higher FCMs than males on control leks. Such differences in observed FCM levels may also be observed if males with low glucocorticoid levels are more likely to disperse from noise-treated leks, so we compared the variance in FCM levels on noise and control leks. We also investigated whether elevated FCM levels were associated with declines in peak male attendance on leks to determine the value of this metric as a tool for predicting lek declines.

Materials and Methods

Study Area & Experimental Design

Study sites were located on federal land relatively undisturbed by human development in Fremont County, Wyoming (42° 50′, 108° 29′30). We monitored a total of 16 leks that were divided into 8 pairs, with the leks of a pair matched according to size and location (6 pairs near the town of Hudson and 2 pairs near the town of Riverton) (Figure 1). Of the 8 lek pairs, 4 pairs were randomly assigned to each noise type, such that there were 4 "drilling pairs", each including one lek exposed to drilling noise and a similar lek as its control, and 4 "road pairs," each with one road noise and a matched control. For 3 of the pairs, one lek within a pair was randomly assigned to the treatment (noise) group and the other assigned as control. For the fourth pair, the treatment and control leks were deliberately assigned due to another study that was in progress. During sample collection periods, both leks in a pair were normally visited on the same day.

Noise and playback methods have been previously described [43] and are summarized here. Noise was played beginning in mid-February to early March and continuing through the end of April of each year. Noise was recorded from drilling and main road sites at the Pinedale Anticline natural gas fields and played back using a commercial car amplifier and 3–4 rock-shaped outdoor speakers placed along one edge of the lek. On leks with road-noise playback, recordings of semi-trailer trucks and pickup trucks were combined with 30- and 60-second files of silence at a ratio reflecting the average number of each truck type found on a main energy field access road; these files were then played using the "random shuffle" feature on an MP3 player. Most shift changes occur at 8 am, so our playback may underestimate actual traffic levels during the lekking time. On leks with drilling noise, a 14-minute recording of a drilling rig was played on continuous loop. Natural gas development activities occur 24 hours a day, so noise was broadcast continuously day and night at playback levels that approximate the noise level at 0.25 mile (402 m) from a typical drilling site (JLB and GLP unpublished data). Drilling-noise recordings were broadcast on experimental leks at an equivalent sound level (L_{eq}) of 71.4±1.7 dBF (unweighted decibels) SPL re 20 μPa (56.1±0.5 dBA [A-weighted decibels]) as measured at 16 meters; on road-noise leks, where the amplitude of the noise varied with the simulated passing of vehicles, noise was broadcast at an L_{max} (maximum RMS amplitude) of 67.6±2.0 dBF SPL (51.7±0.8 dBA) (see Blickley, et al. [43], for detailed noise-exposure measurements). Noise from playback was localized to each lek due to the small size of our speakers. To control for visual disturbance of the speaker system and researcher presence, control

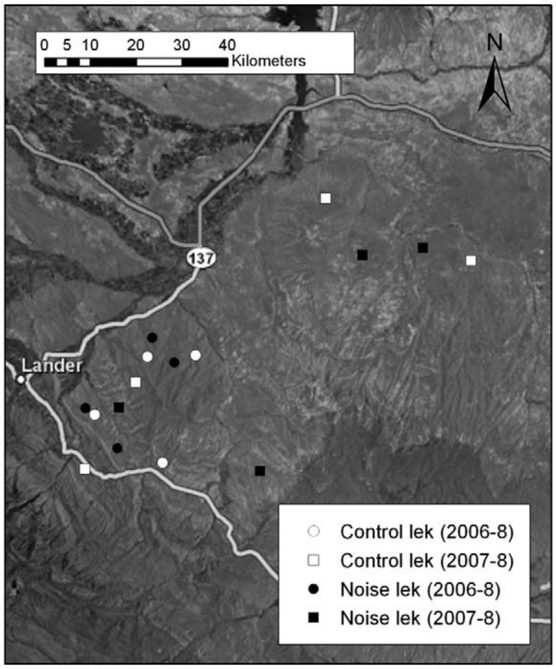

Figure 1. Noise playback study area in Fremont County, Wyoming, USA, 2006–2009. Experimental and control leks were paired on the basis of size and geographic location (the four leks in the upper right are part of the Riverton region, whereas the rest of the leks are in the Lander region).
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leks had dummy speakers placed in the same arrangement and were also visited to simulate the periodic battery changes on noise leks. This experimental protocol was reviewed and approved by the Animal Care and Use Committee at UC Davis (Protocol # 16435) and the Wyoming Game and Fish Department (Permit # 33–405).

In the first year of the experiment (2006), we played noise on only 4 of the 8 lek pairs (2 experimental leks with introduced drilling noise, 2 with introduced road noise). Therefore, some leks had been exposed to noise the breeding season prior to the first year of FCM measurement; however, we detected no significant impact of duration of noise exposure on lek attendance [43], so years of noise exposure was not included as a potential explanatory variable in candidate model sets.

Collection of Fecal Samples

Fecal samples were collected from leks soon after all sage-grouse had left the lek for the morning. Samples were collected twice per year from each lek (once during the mid season [April 4–6 in 2007, April 6–8 in 2008] and once during the late season [April 23–26 in 2007, April 22–24 in 2008]) and were collected from paired leks on the same day. Samples were collected using a sweep-search method in which the entire lek was systematically searched and fresh fecal samples were collected individually in Whirl-Pak bags and labeled with a location on the lek relative to the speakers (or dummy speakers). To minimize the chance of collecting multiple fecal samples from the same individual, we collected samples that were a minimum of 5 meters apart, roughly the minimum territory size of a male sage-grouse. Jankowski [45] found lower FCM levels in female sage-grouse than in breeding male sage-grouse. Therefore to avoid collecting samples from females, we collected samples on dates when female visitation is rare; if there were more than 1–2 females on the lek on a potential collection day, sampling for that lek pair was postponed until the next day. Time to collect samples varied among leks from 20–80 minutes. Samples were frozen at -20°C within a few hours of collection until processing. Jankowski et al. [45] found no difference in FCM levels for greater sage-grouse samples held for variable times up to 16 hours prior to freezing.

Extraction & Radioimmunoassay of Cort

We used extraction and assay procedures, with minor modifications, that were previously validated for application to greater sage-grouse by Jankowski et al. [46]. Individual fecal pellets were kept on ice while uric acid (often present in a discrete cap on the pellet) was removed and discarded. Samples were then lyophilized and returned to storage at -20°C. On the day of extraction, individual fecal pellets were weighed to the nearest 0.0001 g, then manually homogenized, vortexed, and shaken in 5 mL of 80% methanol for at least 30 minutes. Longer incubation in methanol often occurred due to the large number of tubes in each assay, but experimentation with overnight extraction produced no substantial change in detected metabolites. Samples were centrifuged at 5000 rpm for 30 minutes, then 1.5mL of supernatant was drawn off, placed in a separate tube, dried under streaming air in a 70°C water bath and reconstituted in 1.0 mL of steroid diluent provided in the RIA kit (see below). For some very large samples, it was not possible to remove 1.5 mL; in these cases, 500 μL of supernatant was drawn off and reconstitution volume was adjusted accordingly after drying. Extracts were covered with Parafilm and stored at 4°C until assayed.

A pooled sample was made by homogenizing a collection of multiple samples from one control lek (Monument lek) in a blender prior to lyophilization. From this pooled sample, 0.5 g was assayed initially to determine parallelism with the RIA standard curve, and one or more pooled samples were included in each extraction and assay.

Radioimmunoassays were conducted according to the manufacturer's instructions (catalog # 07-120103, MP Biomedicals, Costa Mesa, CA) using 1:16 dilution of reconstituted extract. This RIA kit utilizes a rabbit-produced BSA IgG polyclonal antibody against corticosterone-3-carboxymethyloxime. This antibody has been widely used for fecal assays due to its ability to bind a broad spectrum of corticosteroid metabolites [47]. Samples were randomly distributed among assays with respect to year and treatment to minimize any impacts of inter-assay variation.

FCM measures were adjusted for the mass of the fecal sample (ng ICM/g sample) to account for differences among leks in fecal pellet mass. In dividing ICM by sample mass, we effectively assume that the relationship between sample mass and fecal transit time (during which corticosteroid metabolites are secreted into the lumen of the gut) is positive and linear. To guard against faults in this assumption, we ran the same statistical analyses using "per sample" FCM data and found no difference in the main effects as reported.

Statistical Analysis

Fecal glucocorticoid metabolites levels were natural log-transformed to meet assumptions of normality and homoscedasticity prior to analysis. We used an information theoretic approach to evaluate the support for alternative candidate models using Akaike's Information Criterion for small sample sizes (AIC_c) [48]. Candidate models for the overall effect of noise (Noise effect models) were linear mixed-effect models that assessed the relationship between explanatory variables and the concentration of FCMs collected from experimental and control leks. Potential explanatory variables included pair type (NoiseType, drilling or road noise), control status (Treatment, noise or control), pellet/collection distance from speakers (SpeakerDist), maximum lek size for that year (MaxSize), location (Hudson or Riverton), season (early or late April), and relevant interactions (see Table 1 for full set of candidate models). All models contained lek pair ID, and year (2007 or 2008) as random effects.

We also evaluated a set of candidate models that assessed the relationship between the concentration of FCMs on experimental leks and the declines in peak male attendance from the previous year (attendance models). Models contained lek ID and year (2007 or 2008) as random effects. Models were ranked on the basis of differences in AICc scores (ΔAICc) and were assigned Akaike weights (w_i) corresponding to the degree of support. We calculated model-averaged coefficients and variable importance (sum of variable weights for all models in which the variable was included) for variables contained in all models that received strong support (ΔAICc <2). We also compared the variance in FCM concentrations measured on noise and control leks using a Levene's test. All statistical analyses were performed in R (version 2.12.1, R Development Team 2010).

Results

We measured baseline fecal immunoreactive corticosterone metabolites of 103.2 and 119.9 ng/g for control and treatment groups, respectively (Table 2). These values are lower than baseline measures of approximately 149 ng/g obtained previously for breeding male greater sage-grouse in Nevada, from which fecal samples were collected after capture [45].

Males on leks exposed to noise had higher (16.7% on average) FCM levels compared with controls ($w_i = 0.96$, Table 1, 2; Figure 2). While models that included the effect of Treatment (noise versus control) were highly supported by the data, there was little support for an interaction of Treatment with NoiseType variable ($w_i = 0.01$, Table 1), indicating that while noise exposure was associated with increased cort, there was little difference in FCM levels between leks with drilling versus road-noise playback. Candidate models containing other possible explanatory variables, including distance from the nearest speaker (SpeakerDist), maximum size of the lek (MaxSize), the regional location of the lek in the Hudson area or Riverton area (Location) and time of the season (Season), received little support relative to the null model (Table 1, Figure 2B), indicating that none of these factors had a strong influence on FCM levels.

To determine whether noise-playback leks with a higher stress response were associated with larger declines in lek attendance, we compared candidate models for the relationship between FCM level and change in lek attendance from the previous year. Only the null model received support (Table 3), indicating that fecal FCM level was not associated with the magnitude of changes in lek attendance on noise leks.

Finally, we examined whether there was a difference in variance among samples on noise leks and control leks. We found no significant differences in variance between treatment types in 2007 (variance on noise leks = 7729.94, control leks = 6168.28, Levene's

Table 1. Mixed-effect candidate models for the effect of noise playback on mass-dependent FCM concentrations (natural log-transformed).

Model[a,b]	K[c]	ΔAIC_c [d]	w_i [e]
Treatment[f]	5	0	0.66
Treatment + Location	6	2.4	0.20
Treatment + Location + Treatment:Location	7	4.7	0.06
Null- random effects only	4	5.5	0.04
Treatment + Season	6	6.5	0.03
Treatment + Season + Treatment:Season	7	10.0	<0.01
Treatment + NoiseType + Treatment:NoiseType	7	10.8	<0.01
Treatment + Location + NoiseType + Treatment:Location + Treatment:NoiseType	9	11.2	<0.01
Treatment + NoiseType + Season + Treatment:Season + Treatment:NoiseType	9	20.7	<0.01
Treatment + MaxSize + Treatment:MaxSize	7	25.3	<0.01
Treatment + NoiseType + Season + Treatment:NoiseType + Treatment:Season + Treatment:NoiseType:Season	11	27.3	<0.01
Treatment + SpeakerDistance + Treatment:SpeakerDistance	7	27.5	<0.01
Treatment + NoiseType + MaxSize + Treatment:NoiseType + Treatment:MaxSize	10	35.4	<0.01
Treatment + NoiseType + SpeakerDistance + Treatment:NoiseType + Treatment:SpeakerDistance	9	38.2	<0.01
Treatment + NoiseType + MaxSize + Treatment:NoiseType + Treatment:MaxSize + Treatment:NoiseType:MaxSize	12	45.1	<0.01
Treatment + NoiseType + SpeakerDistance + Treatment:NoiseType + Treatment:SpeakerDistance + Treatment:NoiseType:SpeakerDistance	11	60.4	<0.01

[a]Abbreviations of predictor variables in methods.
[b]All models contain lek pairing and year as a random effect.
[c]Number of parameters in the model.
[d]Difference in AIC_c (Akaike's Information criteria for small sample size) values from the top ranking model.
[e]Akaike weight (Probability that the model is the best fit model giving the data and model candidate set).
[f]Model with substantial support (ΔAIC_c <2).
doi:10.1371/journal.pone.0050462.t001

Table 2. Parameter estimates (± SE) and relative variable importance for variables in highly supported models ($\Delta AIC_c < 3$).

Variable	Parameter estimates[a]	Parameter estimates (back-transformed)[b]	Relative variable importance[c]
Intercept	4.63 (.06)	103.2[d]	-
Treatment:Noise	.15 (.04)	16.7[d]	0.96
Location: Hudson	0.02(.01)	2.9[d]	0.26

[a]Parameter estimates are natural-log transformed.
[b]SE not included due to back-transformation.
[c]Relative variable importance is the summed total of the model weights for models containing that variable.
[d]Intercept value was added to parameter estimates prior to back-transformation and then subtracted.
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W = 0.6327, p = 0.427). Variance on noise leks was significantly higher than on control leks in 2008 (variance on noise leks = 4462.28, control leks = 2758.69, Levene's W = 6.6064, p = 0.01).

Discussion

We found higher (16.7%) FCM levels on noise-treated leks compared to controls, supporting the hypothesis that chronic noise pollution increases stress levels in male greater sage-grouse. Combined with results from monitoring of lek attendance in the same experiment [43], these results suggest that noise from natural gas development activities can dramatically decrease male attendance on leks and cause physiological impacts on males that remain on noisy leks. The mean level of FCMs in remaining birds was not a good predictor of the degree of decline in peak male attendance on a lek compared with the previous year, indicating that the FCM level measured on a lek is not diagnostic of an effect of noise on peak male attendance (Table 3). Further, we did not find support for an effect of distance from the speakers on FCM levels. Male sage-grouse typically maintain a fixed territory on a lek throughout the season. Within a noise-treated lek, each individual's exposure to noise varied, depending on the location of their territory relative to the speakers. Since noise levels decline exponentially with distance from the speakers, the lack of a distance effect suggests that stress is not exclusively dependent on the noise exposure of individuals. Instead, noise impacted FCM levels on a lek-wide basis.

Blickley et al. [43] found a decline in lek attendance on road-noise leks more than twofold larger than the decline in lek attendance on drilling-noise leks, yet we found no difference in FCM levels between noise-playback types (Table 1, Figure 1). Both noise sources have most of their sound energy ≤2 kHz, but road noise is less predictable than drilling noise and more intermittent,

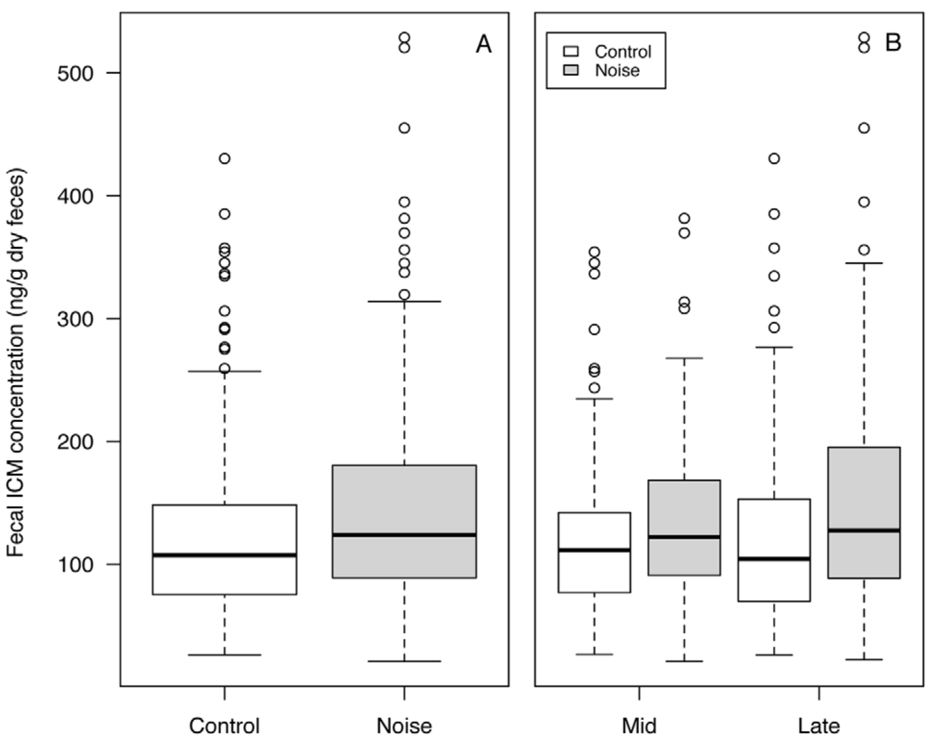

Figure 2. FCM concentrations from control and noise-treated groups. Data shown (A) pooled by season and (B) for mid and late season samples. Horizontal line represents the median value, box ends represent upper and lower quartiles, whiskers represent maximum and minimum values and open circles represent outliers. Plots present measured FCM values, not model output, which is presented in Table 2.
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Table 3. Mixed-effect candidate models assessing the relationship of FCM concentrations and changes in lek attendance from the previous year on noise-playback leks.

Model[a,b]	K[c]	ΔAIC_c [d]	w_i [e]
Null- random effects only[f]	5	0	0.90
Fecal cort	6	4.6	0.10

[a]Abbreviations of predictor variables in methods.
[b]All models contain lek pairing and year as a random effect.
[c]Number of parameters in the model.
[d]Difference in AIC_c (Akaike's Information criteria for small sample size) values from the top ranking model.
[e]Akaike weight.
[f]Model with substantial support (ΔAIC_c <3).
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leading to a lower average noise exposure across road-noise leks (43.2±0.89 dBA L_{eq}) than drilling-noise leks (56.1±0.45 dBA L_{eq}) [43]. Studies on physiological stress in rodents indicate that stressors administered at unpredictable intervals result in greater elevations in plasma corticosterone [49]. Since cort levels may also be implicated in decisions to escape from deleterious conditions [50], we cannot say with certainty that noise type has no differential impact on FCM levels, only that there was no difference observed among males that chose to remain. If road noise did result in a greater cort response in some birds, but the most susceptible birds were also the most likely to disperse, differences would not necessarily be expected among remaining birds. In this scenario, it is likely that variance would be reduced in leks with high losses, reflecting disappearance of individuals with higher FCM levels. Levene's tests did not identify any such difference in variance (indeed, there was a significant difference in one year of the study, but in the opposite direction to predictions). However, the possibility that dispersal is linked to FCM levels cannot be ruled out. Regardless of whether the stress levels of birds on noise leks increased, or whether only high-stress-level individuals remained on noisy leks, these results indicate that chronic noise at leks creates less desirable habitat for greater sage-grouse.

The unknown status of dispersed grouse – and their unknown destinations – leaves several other possible scenarios that should be considered. It is possible that the individuals most likely to disperse could have had different cort profiles at the outset compared with those more prone to remain. If noise playback caused individuals with lower integrated cort to disperse away from noisy leks, that coupled with the possible addition of those birds to control leks could cause trends similar to those observed here. Two possible sources of variation in pre-experiment cort levels among individuals are age and social status [51–53]. Reduced juvenile recruitment may have contributed to the observed declines in lek attendance on noise leks, potentially leading to a difference in age structure on noise and control leks [43]; however, this is unlikely to explain the results of this study. Studies of altricial and semi-altricial birds have found lower stress responsiveness shortly after hatching, but responses resemble those of adults by the age of fledging or first molt [54–57]. Since young male sage-grouse attending leks are likely to be at least 10 months old and after their first molt, it is unlikely that they would have lower stress response than adults. Social status can also be related to corticosteroid levels [58], therefore social upheaval caused by dispersal between noise and control leks may have contributed to observed FCM levels. Further studies are needed determine whether age-class- and social-status-dependent dispersal in response to noise contributed to the observed results.

Unlike noise sources in most energy development sites, our noise introduction in this study was localized to the immediate lek area, so birds were exposed to noise for only a few hours a day, and only during the breeding season. Therefore, we cannot quantify the effects of noise on FCMs for wintering, nesting or foraging males. Noise at energy development sites is less seasonal and more widespread and may thus affect birds at all life stages, with a potentially greater impact on stress levels. In addition, we looked only at male stress levels in this study, but males and females may respond differently to stress. For example, Jankowski et al. [45] measured FCM levels in sage-grouse in habitats with and without cattle grazing; they found no difference in male FCM levels in response to grazing regime, however, breeding females showed elevated stress response in grazed areas. This suggests that females may be more vulnerable to some types of disturbance; further studies are needed to assess whether female stress levels are influenced by noise.

Why might noise be stressful?

Increased adrenocortical activity occurs in response to circumstances perceived as threatening by an animal. Although we cannot determine from this study the extent to which noise itself is a threat to sage-grouse, noise may affect social dynamics and increase the perception of threat. Noise may have social impacts on sage-grouse by masking acoustic communication on the lekking grounds [42]. Masking occurs when the perception of a sound is decreased by the presence of background noise, which may reduce the efficacy of acoustic communication. Acoustic signals play an important role in many social interactions, including mate attraction and assessment, territorial interactions, recognition of conspecifics and alarm calling in response to environmental threats [9,10,59]. Masking of these acoustic signals may alter or interfere with social interactions and mate choice behaviors [60,61].

For prey species such as sage-grouse, noise may also increase stress levels by masking the sounds of approaching predators and increasing the perception of risk from predation [62,63]. The degree to which noise directly affects mortality through changes in predation is largely unknown, as few studies have compared predation rates or hunting success in noisy and quiet areas while controlling for other confounding factors. Francis et al. [4] did so and found that nest predation rates in some songbirds decline in noise-impacted areas, as the dominant nest predator avoided noise. This suggests that noise may cause complicated changes in predator-prey dynamics. Noise may also cause stress due to short-term disruptions in behavior, such as startling or frightening animals away from food or other resources [2,64]. Further, if individuals associate a particular type of noise, such as road noise, with a danger, such as vehicular traffic, this may provoke a stress response [43].

The impacts of chronic stress

Glucocorticoid release under challenging conditions is an adaptation to life in an unpredictable and threatening world [20]; individuals benefit from curtailing reproduction, altering behavioral patterns, and redirecting metabolic substrates to maximize glucose availability for action in response to genuine threats. Glucocorticoid levels alone are not directly or inversely correlated with fitness measures under all conditions [65], however, chronic adrenal activation has many known trade-offs that result in vulnerability to disease and death [22]. Unlike threats from predators, food shortages and inclement weather, noise typically does not directly threaten the survival of an individual or its offspring (though there may be exceptions, as discussed below). Therefore, the cost of chronic adrenal activation in response to noise pollution is unlikely to be outweighed by the benefits in most cases, and thus the net result may be adverse.

One important trade-off is the effect of corticosterone on immune response. Chickens infected with West Nile Virus (WNV) and administered corticosterone had increased oral shedding and lengthened duration of viremia compared to those without elevated cort [66]. For sage-grouse, which are highly susceptible to WNV [67,68], reduced immune response due to elevated glucocorticoid levels could have a significant effect on survival in areas where they are exposed to WNV. Therefore, despite the adaptive nature of the stress response under natural conditions, elevated glucocorticoid levels due to human disturbance may have detrimental long-term impacts on welfare and survival of sage-grouse and other wildlife.

Stress as an indicator of human impacts on sage-grouse

Measurement of FCMs may provide a non-invasive monitoring tool to assess the impact of human development (e.g. oil and gas drilling, wind farms, highways, off-road vehicle traffic) on stress levels of greater sage-grouse and other species. However comparisons between disturbed and undisturbed areas would need to account for differences in age, sex, and breeding condition of individuals sampled as well as for differences in the environmental conditions between sites in order to isolate stress as the likely cause of change [15,18,69]. We controlled for such differences by using an experimental presentation of noise that minimized effect on other habitat variables, limiting our collection to lekking birds, collecting only on days with limited female attendance and collecting samples from all leks within a short 2–3 day window. We did not find support for differences in FCM levels from samples collected in early versus late April within each season (~20 days apart in a 2–3 month breeding season), and only limited evidence for an effect of location (Hudson vs. Riverton, ~32 kilometers apart), suggesting that these temporal and spatial differences did not affect FCM levels in our study. However with a larger sample of leks or in another region or time period, it is possible that such differences might emerge.

Conclusions

Taken together, results from Blickley et al. [43] and this study suggest that noise alone can cause greater sage-grouse to avoid otherwise suitable habitat and increase the stress responses of birds that remain in noisy areas. Thus, noise mitigation may be a fruitful conservation measure for this species of concern. In this study, we focused on the effects of noise from roads and drilling rigs in natural gas development areas; other natural gas development infrastructure, including compressor stations and generators, produces noise similar to drilling rigs, with the potential for similar effects on FCM levels. Likewise, other types of energy development produce noise similar in frequency, timing, and amplitude to the noise sources used here, including shale gas, coal-bed methane, oil, and geothermal development. The noise sources used in this study also share some characteristics with other anthropogenic noise sources that are increasing across the landscape, like wind turbines, off-road vehicles, highways and urban development; this suggests that the impacts on greater sage-grouse observed here may be widespread. More generally, populations of many species of birds [4,70–74] and mammals [75–78] decline with proximity to noisy human activities, such as roads, urban and industrial developments. While further study is needed to determine whether chronic noise exposure contributes to the impacts of these human activities by activating the chronic stress response, this study adds to a growing body of evidence that such noise pollution is a threat to wildlife [1,2], significantly increasing our estimates of the footprint of human development beyond the boundaries of visible disturbance.

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Author Contributions

Conceived and designed the experiments: JLB GLP AHK SNS. Performed the experiments: JLB AHK GLP SNS KRW JLP JCW. Analyzed the data: JLB KRW GLP AHK JLP CCT JCW. Contributed reagents/materials/analysis tools: GLP JCW. Wrote the paper: JLB KRW GLP.

References

1. Kight CR, Swaddle JP (2011) How and why environmental noise impacts animals: an integrative, mechanistic review. Ecology Letters 14: 1052–1061.
2. Barber JR, Crooks KR, Fristrup KM (2009) The costs of chronic noise exposure for terrestrial organisms. Trends in Ecology & Evolution 25: 180–189.
3. Blickley JL, Patricelli GL (2010) Impacts of Anthropogenic Noise on Wildlife: Research Priorities for the Development of Standards and Mitigation. Journal of International Wildlife Law and Policy 13: 274–292.
4. Francis CD, Ortega CP, Cruz A (2009) Noise Pollution Changes Avian Communities and Species Interactions. Current Biology 19: 1415–1419.
5. Habib L, Bayne EM, Boutin S (2007) Chronic industrial noise affects pairing success and age structure of ovenbirds Seiurus aurocapilla. Journal of Applied Ecology 44: 176–184.
6. Bayne E, Habib L, Boutin S (2008) Impacts of Chronic Anthropogenic Noise from Energy-Sector Activity on Abundance of Songbirds in the Boreal Forest. Conservation Biology 22: 1186–1193.
7. Hu Y, Cardoso GC (2009) Are bird species that vocalize at higher frequencies preadapted to inhabit noisy urban areas? Behavioral Ecology 20: 1268–1273.
8. Slabbekoorn H, Ripmeester EAP (2008) Birdsong and anthropogenic noise: implications and applications for conservation. Molecular Ecology 17: 72–83.
9. Warren PS, Katti M, Ermann M, Brazel A (2006) Urban bioacoustics: it's not just noise. Animal Behaviour 71: 491–502.
10. Patricelli GL, Blickley JL (2006) Overview: Avian communication in urban noise: the causes and consequences of vocal adjustment. The Auk 123: 639–649.
11. Francis CD, Kleist NJ, Ortega CP, Cruz A (2012) Noise pollution alters ecological services: enhanced pollination and disrupted seed dispersal. Proceedings of the Royal Society B: Biological Sciences.
12. Wikelski M, Cooke SJ (2006) Conservation physiology. Trends in Ecology & Evolution 21: 38–46.
13. Romero LM (2004) Physiological stress in ecology: lessons from biomedical research. Trends in Ecology & Evolution 19: 249–255.
14. Creel S, Fox JE, Hardy A, Sands J, Garrott B, et al. (2002) Snowmobile Activity and Glucocorticoid Stress Responses in Wolves and Elk. Conservation Biology 16: 809–814.
15. Wasser SK, Bevis K, King G, Hanson E (1997) Non-Invasive Physiological Measures of Disturbance in the Northern Spotted Owl. Conservation Biology 11: 1019–1022.
16. Thiel D, Jenni-Eiermann S, Palme R (2005) Measuring corticosterone metabolites in droppings of capercaillies (*Tetrao urogallus*). Bird Hormones And Bird Migrations: Analyzing Hormones In Droppings And Egg Yolks And Assessing Adaptations In Long-Distance Migration. pp. 96–108.
17. Wasser SK, Bevis K, King G, Hanson E (1997) Noninvasive Physiological Measures of Disturbance in the Northern Spotted Owl. Conservation Biology 11: 1019–1022.
18. Millspaugh JJ, Washburn BE (2004) Use of fecal glucocorticoid metabolite measures in conservation biology research: considerations for application and interpretation. General and Comparative Endocrinology 138: 189–199.

19. Lucas JR, Freeberg TM, Egbert J, Schwabl H (2006) Fecal corticosterone, body mass, and caching rates of Carolina chickadees (*Poecile carolinensis*) from disturbed and undisturbed sites. Hormones and Behavior 49: 634–643.
20. Wingfield JC (2005) The concept of allostasis: coping with a capricious environment. Journal of Mammalogy 86: 248–254.
21. McEwen BS, Wingfield JC (2003) The concept of allostasis in biology and biomedicine. Hormones and Behavior 43: 2–15.
22. Sapolsky RM, Romero LM, Munck AU (2000) How do glucocorticoids influence stress responses? Integrating permissive, suppressive, stimulatory and preparative actions. Endocrine Reviews 21: 55–89.
23. Wingfield JC, Sapolsky RM (2003) Reproduction and resistance to stress: when and how. J Neuroendocrinol 15: 711–724.
24. Opplinger A, Clobert J, Lecomte J, Lorenzon P, Boudjemadi K, et al. (1998) Environmental stress increases the prevalence and intensity of blood parasite infection in the common lizard Lacerta vivipara. Ecology Letters 1: 129–138.
25. Alario P, Gamallo A, Beato MJ, Trancho G (1987) Body weight gain, food intake and adrenal development in chronic noise stressed rats. Physiology & Behavior 40: 29–32.
26. Barrett AM, Stockham MA (1963) The effect of housing conditions and simple experimental procedures upon the corticosterone level in the plasma of rats. Journal of Endocrinology 26: 97–105.
27. Davis M, Walker DL, Lee Y (1997) Amygdala and bed nucleus of the stria terminalis: differential roles in fear and anxiety measured with the acoustic startle reflex. Philosophical Transactions of the Royal Society of London, B 352: 1675–1687.
28. Atkinson H, Wood S, Kershaw Y, Bate E, Lightman S (2006) Diurnal Variation in the Responsiveness of the Hypothalamic-Pituitary-Adrenal Axis of the Male Rat to Noise Stress. Journal of neuroendocrinology 18: 526–533.
29. Thiel D, Jenni-Eiermann S, Palme R, Jenni L (2011) Winter tourism increases stress hormone levels in the Capercaillie *Tetrao urogallus*. Ibis 153: 122–133.
30. Delaney DK, Grubb TG, Beier P, Pater LL, Reiser MH (1999) Effects of helicopter noise on Mexican spotted owls. Journal of Wildlife Mangement 63: 60–76.
31. Tempel DJ, Gutiérrez RJ (2003) Fecal Corticosterone Levels in California Spotted Owls Exposed to Low-Intensity Chainsaw Sound. Wildlife Society Bulletin 31: 698–702.
32. Babisch W (2003) Stress hormones in the research on cardiovascular effects of noise. Noise Health 5: 1–11.
33. Ising H, Ising M (2002) Chronic cortisol increases in the first half of the night caused by road traffic noise. Noise Health 4: 13–21.
34. Evans GW, Lercher P, Meis M, Ising H, Kofler WW (2001) Community noise exposure and stress in children. Journal Of The Acoustical Society Of America 109: 1023–1027.
35. Evans GW, Bullinger M, Hygge S (1998) Chronic Noise Exposure and Physiological Response: A Prospective Study of Children Living under Environmental Stress. Psychological Science 9: 75–77.
36. Department of the Interior (2010) Endangered and Threatened Wildlife and Plants; 12-Month Findings for Petitions to List the Greater Sage-Grouse (*Centrocercus urophasianus*) as Threatened or Endangered. Federal Register 75: 13910–14014.
37. Connelly JW, Knick ST, Schroeder MA, Stiver SJ (2004) Conservation Assessment of Greater Sage-grouse and Sagebrush Habitats. Cheyenne, WY: West. Assn. Fish and Wildlife Agencies. 610 p.
38. Holloran MJ (2005) Greater Sage-Grouse (*Centrocercuc urophasianus*) Population Response to Natural Gas Field Development in Western Wyoming [Dissertation]. Laramie: University of Wyoming. 114 p.
39. Walker BL, Naugle DE, Doherty KE (2007) Greater Sage-Grouse Population Response to Energy Development and Habitat Loss. Journal of Wildlife Management 71: 2644–2654.
40. Holloran M, Kaiser R, Hubert W (2010) Yearling greater sage-grouse response to energy development in Wyoming. Journal of Wildlife Management 74: 65–72.
41. Naugle DE, Doherty KE, Walker BE, Holloran MJ, Copeland HJ (2011) Energy development and Greater Sage-Grouse. In: Knick ST, Connelly JW, editors. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Berkeley, CA.: University of California Press.
42. Blickley JL, Patricelli GL (2012) Potential acoustic masking of greater sage-grouse display components by chronic industrial noise. Ornithological Monographs 74: 23–35.
43. Blickley JL, Blackwood D, Patricelli GL (2012) Experimental Evidence for the Effects of Chronic Anthropogenic Noise on Abundance of Greater Sage-Grouse at Leks. Conservation Biology 26: 461–471.
44. Vehrencamp S, Bradbury J, Gibson R (1989) The energetic cost of display in male sage grouse. Animal Behaviour 38: 885–896.
45. Jankowski MD (2007) The influence of habitat disturbance and synergized resmethrin on avian immunocompetence: The University of Wisconsin, Madison.
46. Jankowski MD, Wittwer DJ, Heisey DM, Franson JC, Hofmeister EK (2009) The Adrenocortical Response of Greater Sage Grouse (*Centrocercus urophasianus*) to Capture, ACTH Injection, and Confinement, as Measured in Fecal Samples. Physiological and Biochemical Zoology 82: 190–201.
47. Wasser SK, Hunt KE, Brown JL, Cooper K, Crockett CM, et al. (2000) A Generalized Fecal Glucocorticoid Assay for Use in a Diverse Array of Nondomestic Mammalian and Avian Species. General and Comparative Endocrinology 120: 260–275.
48. Burnham KP, Anderson DR (2002) Model selection and multimodel inference: a practical information-theoretic approach: Springer Verlag.
49. Weiss JM (1970) Somatic Effects of Predictable and Unpredictable Shock. Psychosomatic Medicine 32: 397–408.
50. Wingfield JC, Maney DL, Breuner CW, Jacobs JD, Sharon L, et al. (1998) Ecological Bases of Hormone-Behavior Interactions: The "Emergency Life History Stage". American Zoologist 38: 191–206.
51. Goymann W, Wingfield JC (2004) Allostatic load, social status and stress hormones: the costs of social status matter. Animal Behaviour 67: 591–602.
52. Angelier F, Shaffer SA, Weimerskirch H, Chastel O (2006) Effect of age, breeding experience and senescence on corticosterone and prolactin levels in a long-lived seabird: The wandering albatross. General and Comparative Endocrinology 149: 1–9.
53. Angelier F, Moe B, Weimerskirch H, Chastel O (2007) Age-specific reproductive success in a long-lived bird: do older parents resist stress better? Journal of Animal Ecology 76: 1181–1191.
54. Schwabl H (1999) Developmental Changes and Among-Sibling Variation of Corticosterone Levels in an Altricial Avian Species. General and Comparative Endocrinology 116: 403–408.
55. Wada H, Hahn TP, Breuner CW (2007) Development of stress reactivity in white-crowned sparrow nestlings: Total corticosterone response increases with age, while free corticosterone response remains low. General and Comparative Endocrinology 150: 405–413.
56. Freeman BM (1982) Stress non-responsiveness in the newly-hatched fowl. Comparative Biochemistry and Physiology Part A: Physiology 72: 251–253.
57. Dickens MJ, Romero LM (2010) Stress Responsiveness Decreases With Age in Precocial, Juvenile Chukar. The Wilson Journal of Ornithology 122: 762–766.
58. Rubenstein DR (2007) Stress hormones and sociality: integrating social and environmental stressors. Proceedings of the Royal Society B: Biological Sciences 274: 967–975.
59. Brumm H, Slabbekoorn H (2005) Acoustic Communication in Noise. Advances in the Study of Behavior 35: 151–209.
60. Mockford EJ, Marshall RC (2009) Effects of urban noise on song and response behaviour in great tits. Proceedings of the Royal Society B: Biological Sciences 276: 2979–2985.
61. Swaddle JP, Page LC (2007) High levels of environmental noise erode pair preferences in zebra finches: implications for noise pollution. Animal Behaviour 74: 363–368.
62. Rabin LA, Coss RG, Owings DH (2006) The effects of wind turbines on antipredator behavior in California ground squirrels (Spermophilus beecheyi). Biological Conservation 131: 410–420.
63. Quinn L, Whittingham J, Butler J, Cresswell W (2006) Noise, predation risk compensation and vigilance in the chaffinch Fringilla coelebs. Journal of Avian Biology 37: 601–608.
64. Dooling RJ, Popper AN (2007) The Effects of Highway Noise on Birds. Sacramento, CA: The California Department of Transportation Division of Environmental Analysis. 74 p.
65. Bonier F, Martin PR, Moore IT, Wingfield JC (2009) Do baseline glucocorticoids predict fitness? Trends in Ecology & Evolution 24: 634–642.
66. Jankowski MD, Franson JC, Möstl E, Porter WP, Hofmeister EK (2010) Testing independent and interactive effects of corticosterone and synergized resmethrin on the immune response to West Nile virus in chickens. Toxicology 269: 81–88.
67. Naugle DE, Aldridge CL, Walker BL, Cornish TE, Moynahan BJ, et al. (2004) West Nile virus: Pending crisis for Greater Sage-Grouse. Ecology Letters 7: 704–713.
68. Walker BE, Naugle DE (2011) West Nile Virus ecology in sagebrush habitat and impacts on Greater Sage-Grouse populations. In: Knick ST, Connelly JW, editors. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Berkeley, CA.: University of California Press.
69. Sheriff MJ, Wheeler H, Donker SA, Krebs CJ, Palme R, et al. (2012) Mountain-top and valley-bottom experiences: the stress axis as an integrator of environmental variability in arctic ground squirrel populations. Journal of Zoology 287: 65–75.
70. Kuitunen M, Rossi E, Stenroos A (1998) Do Highways Influence Density of Land Birds? Environmental Management 22: 297–302.
71. van der Zande AN, ter Keurs WJ, van der Weijden WJ (1980) The impact of roads on the densities of four bird species in an open field habitat–evidence of a long-distance effect. Biological Conservation 18: 299–321.
72. Rheindt FE (2003) The impact of roads on birds: Does song frequency play a role in determining susceptibility to noise pollution? J für Ornithologie 144: 295–306.
73. Ingelfinger F, Anderson S (2004) Passerine response to roads associated with natural gas extraction in a sagebrush steppe habitat. Western North American Naturalist 64: 385–395.
74. Peris SJ, Pescador M (2004) Effects of traffic noise on paserine populations in Mediterranean wooded pastures. Applied Acoustics 65: 357–366.
75. Singer FJ (1978) Behavior of Mountain Goats in Relation to US Highway 2, Glacier National Park, Montana. The Journal of Wildlife Management 42: 591–597.
76. Rost GR, Bailey JA (1979) Distribution of Mule Deer and Elk in Relation to Roads. The Journal of Wildlife Management 43: 634–641.
77. Forman RTT, Deblinger RD (2000) The Ecological Road-Effect Zone of a Massachusetts (U. S. A.) Suburban Highway. Conservation Biology 14: 36–46.
78. Sawyer H, Kauffman M, Nielson R (2009) Influence of well pad activity on winter habitat selection patterns of mule deer. Journal of Wildlife Management 73: 1052–1061.

A Blueprint for Sage-grouse Conservation and Recovery

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TABLE OF CONTENTS

Abstract: The distribution of greater sage-grouse (*Centrocercus urophasianus*) has declined by at least 44% while overall abundance has decreased by up to 93% from presumed historic levels. These decreases are the result of habitat loss, fragmentation, and degradation. Federal and state public land management agencies currently are responsible for about 70% of the remaining sagebrush (*Artemisia* spp.) steppe, with the Bureau of Land Management and U.S. Forest Service managing most of these lands for multiple uses. The goals of strategies outlined here are to improve sagebrush habitats to increase greater sage-grouse abundance by at least 33% by 2015, and overall distribution of greater sage-grouse by at least 20% by 2030. The abundance goal is achievable following recommendations presented in this document while the distribution goal will be more difficult to obtain. Federal land management agencies are key to achieving both goals, as they are responsible for managing public lands, which support most of the remaining populations of greater sage-grouse. Improved vegetation management to restore degraded habitat (from domestic livestock grazing and development, such as from mining and gas/oil extraction) followed by reduction of habitat fragmentation has the greatest potential for maintaining and enhancing viable populations of greater sage-grouse. While the habitat management strategies and recommendations in this report focus on greater sage-grouse, they are also applicable to Gunnison sage-grouse (*Centrocercus minimus*).

Introduction

Sage-grouse (*Centrocercus urophasianus, C. minimus)* are dependent upon sagebrush (*Artemisia* spp.) and were historically widespread and at least locally abundant (Patterson 1952, Schroeder et al. 2004). Concern about the decrease in the abundance of sage-grouse is not only recent (Connelly and Braun 1997, Braun 1998, Connelly et al. 2004) but also long-term (Hornaday 1916, Patterson 1952). Sagebrush was also historically widely distributed in western North America (Küchler 1964, Vale 1975, Miller and Eddleman 2001, Schroeder et al. 2004). In the United States, about 70% of the remaining sagebrush steppe and distribution of sage-grouse is on public land, with most (~50% of all publicly owned sagebrush steppe) managed by the U. S. Department of Interior, Bureau of Land Management (BLM) (Connelly et al. 2004). Thus, the BLM and the U.S. Forest Service (USFS) (U.S. Department of Agriculture) have the greatest potential to positively impact sage-grouse abundance and distribution provided effective policies and conservation actions are implemented that will benefit sagebrush steppe habitats. Overall, the "responsibility for maintaining sagebrush habitats and [sage-grouse] populations rests squarely on public land management agencies because most [of the] species' [home] range [is] owned publicly and managed by state or federal agencies" (Knick et al. 2003:627, Connelly et al. 2004).

Statement of Problem

The abundance and distribution of greater sage-grouse (*Centrocercus urophasianus*) have declined. Sage-grouse historically occupied at least 1,247,004 km² in western North America of which at least 1,200,483 km² were occupied by greater sage-grouse (Schroeder et al. 2004). Greater sage-grouse now occupy about 668,412 km² of their estimated historical distribution and have been extirpated from 1 state (Nebraska) and 1 Canadian province (British Columbia) (Braun 1998). There are no data on historical numbers (pre-European settlement) but estimates range from at least 2 to 10 million birds (C. E. Braun, illustrated presentation to the Western Association of Fish and Wildlife Agencies, Jackson Hole, Wyoming, July 1998). Braun (1998) further presented estimated breeding population levels by state and province based on counts of male sage-grouse in spring 1998 as reported by state and provincial biologists. The total was presented as ~142,000 sage grouse (Braun 1998:141). This suggests a decrease of ~93% in overall abundance if the minimum historical estimate of 2 million sage grouse is used. Braun (1998) generally classified reasons for the apparent decrease in sage-grouse abundance as the result of habitat loss, habitat fragmentation, and habitat degradation. More recently, Connelly et al. (2004:13-4) indicated that of 41 populations defined for their analysis, 5 populations have been extirpated or have numbers too small to monitor, and 14 additional populations face a high risk of extinction. The vast majority of remaining sage-grouse are in only 8 populations. Additionally, Connelly et al. (2004: 6-67) reported that an examination of all trend data from the 1940s to 2003 "suggest a substantial decline in the overall sage-grouse population in North America." Sage-grouse populations declined at an overall rate of 2.0% per year from 1965 to 2003 (Connelly et al. 2004). These authors (2004:6-71) concluded, "Continued loss and degradation of habitat and other factors…do not provide causes for optimism."

Goals

With respect to conservation of sage-grouse and the species' habitats as well as other sagebrush obligate species, the overall goal of management of public lands should be to (1) maintain the present abundance and distribution of greater sage-grouse and (2) enhance the population viability of the species through habitat management that results in increased abundance and distribution. While it is necessary to understand past changes in abundance and distribution of greater sage-grouse, it is also important to understand the present status of the species and to work towards a goal of no net loss of sagebrush steppe presently or potentially useful to sage-grouse, no further loss of populations or subpopulations, and enhancement of sage-grouse numbers by one-third (33%) and overall distribution by one-fifth (20%) (from ~668,412 km^2 to 835,000 km^2). The abundance goal can likely be achieved by 2015 while the enhanced distribution goal is longer term (2030). Both desired increases (33% in abundance, 20% in distribution) were selected (by C. E. Braun) because they should be achievable, detectable, and measurable using current technology. A 20% increase in distribution was selected, as it should be detectable. Smaller increases in distribution are not likely to be detectable or measurable.

Habitat Needs Overview

The habitat needs of greater sage-grouse are reasonably well understood based on knowledge of what has been described as "used" by sage-grouse (extensive literature summarized in Braun et al. 1977, Connelly et al. 2000*b*, Braun et al. 2005). The basic seasonal periods relating to sage-grouse habitat needs have been described as winter (early to mid-December to early to mid-March), spring (early to mid-March to early to mid-June), summer (early to mid-June to late September), and fall (late September to early to mid-December) depending upon elevation and weather conditions (Braun et al. 2005). A summary (Braun et al. 2005) of the existing literature is attached as an appendix.

Management of Development

Development of sagebrush steppe could include agricultural uses (usually permanent loss), which includes converting sagebrush habitats to cropland, placement of ranch/farm buildings, or the replacement of native sagebrush habitats with seeded pasture lands. Development may also refer to permanent conversion of sagebrush habitats to urban, suburban, and exurban uses (housing), and related infrastructure. “Development” as used in this section refers primarily to energy development, which includes mining (coal, gold, trona, and other mineral deposits) and extraction of natural gas (including coal bed methane) and oil. The following are minimum recommendations for development in sage-grouse habitats as it has been documented that some populations of greater sage-grouse require larger areas for breeding, brood-rearing, winter-use, and security depending upon whether they are migratory or non-migratory (Connelly et al. 2000*b*).

Noise

Sage-grouse are known to select display sites (leks) that are highly visible and which have good acoustic properties (Patterson 1952, Connelly et al. 2000*b*, Lyon 2000, Braun et al. 2002). Sage-grouse numbers on leks within 1.6 km (1 mile) of coal bed methane (CBM) compressor stations in Campbell County, Wyoming, were consistently lower than on leks not affected by this disturbance (Braun et al. 2002). Holloran and Anderson (2005) reported that lek activity by sage-grouse decreased downwind of drilling activities, suggesting that noise had measurable negative impacts on sage-grouse. Roads also generate noise and Connelly et al. (2004) indicated there were no active sage-grouse leks within 2 km of Interstate 80 (I-80) across southern Wyoming and only 9 leks were known to occur between 2 and 4 km of I-80. Lyon and Anderson (2003) reported that oil and gas development influenced the rate of nest initiation of sage-grouse in excess of 3 km of construction activities. Clearly, the amount and (likely) frequency of noise associated with development has major negative effects on greater sage-grouse.

Consequently, all drilling activities for gas and oil development should be prohibited within 5.5 km (3.3 miles) of active leks and their associated nesting areas (Holloran 2005). Further, all existing and new compressor stations should add noise abatement devices (mufflers) to reduce audible noise within 5.5 km of active leks. The actual level of noise (measured in decibels) that would not negatively affect greater sage-grouse breeding and nesting activities is presently unknown.

Physical Disturbance

Greater sage-grouse are known to be negatively impacted by activities associated with mining, and oil and gas development (Remington and Braun 1991, Aldridge 1998, Lyon and Anderson 2003, Holloran and Anderson 2005). Besides the actual physical disturbance to the landscape caused by mining and oil and gas development activities, the impacts of roads are also negative for sage-grouse (Connelly et al. 2004). There are numerous examples of active leks being abandoned once road use associated with mining and gas/oil development increased in close proximity (< 1 km) to leks and nesting habitat (Braun 1986).

All surface activity should be prohibited within 5.5 km (Holloran and Anderson 2004, 2005) of active sage-grouse leks. No surface occupancy is preferred to simply limiting use of areas to specific periods, as the latter does not appear to benefit sage-grouse. Roads should not be placed within 5.5 km (3.3 miles) of active leks. If roads are present, they should be seasonally closed during the sage-grouse breeding season from 1 March to 20 June.

Management of Fire

Prescribed Fire

Fire has been demonstrated to be negative for greater sage-grouse (Hulet 1983; Connelly et al. 2000*a*, *b*; Nelle et al. 2000) as it destroys winter and nesting habitats. Use of fire has been promoted by public land management agencies (both BLM and USFS) to reduce sagebrush cover and increase forbs. However, the only presumed value of this practice is to improve brood-use areas or remove encroaching conifers. The problems with use of prescribed fire relate to control of the fire (escapement is frequent), what is actually burned versus what was desired to be burned, and size of the planned burn. Too often, what is burned is nesting or winter-use areas and burned areas are too large (> 20 ha).

Prescribed fire should not be used in areas where invasion of cheatgrass (*Bromus tectorum*) or other exotic species is likely. Burned areas should be smaller than 20 ha in size and no more than 20% of the landscape (128 ac per section [640 ac]) should be burned over a 30-year interval in taller sagebrush types. Burning should not be permitted in low sagebrush habitat types (i.e., *Artemisia arbuscula, A. longiloba, A. nova*). Burning that benefits sage-grouse will most likely be that which affects brood habitat. There should be a demonstrated need for additional brood habitat before use of prescribed fire is considered. The goal is to not exceed 20% fire coverage (128 ac per section [640 ac]) over a 30-year period regardless of the total area planned to be burned. Reseeding should not be necessary for prescribed burns, as areas should be sufficiently small so that surrounding sagebrush habitat can reseed the areas naturally.

Wild Fire

All wild fires should be vigorously suppressed except in areas where juniper (*Juniperus* spp.) or pinyon pine (*Pinus edulis*) has invaded (>20 trees/ha). Most wild fires are negative for sage-grouse in the short-term. If wild fires occur, grazing by domestic livestock should be immediately suspended and should not be reinstated for a minimum of 3 years. The present 2-year rest period from grazing that is often prescribed on public lands following wild fires is not based on data. Replicated studies are needed across the gradient of moisture regimes and habitat types to learn if 3 years or more are adequate for ecosystem renewal following wild fire. Most areas burned by wild fire do not require reseeding, as disking and other forms of site preparation can be harmful to site restoration. These are practices that promote livestock grazing, not habitat restoration. If reseeding must be done to reduce soil erosion, it should occur in linear strips perpendicular to the prevailing wind except on steeper (>30%) slopes. Strips should be planted with dryland alfalfa, biennial sweet clover, native bunch grasses, and sagebrush seed in a ratio of 1 strip (10 m width) per 50 m. Areas closest to a potential fire source (roads or railroads) should be planted with a 20-m wide strip of fire resistant vegetation.

Management of Grazing

Sound grazing management in sagebrush steppe should promote light use of herbaceous forage while having a neutral or positive impact on plant vigor. Further, proper livestock grazing should maintain or enhance desirable plant communities, improve vegetation palatability, increase native plant diversity, and promote residual vegetative cover. Extreme caution should be exercised in grazing sagebrush steppe until scientific evidence is obtained through replicated studies that demonstrate grazing improves, restores, or maintains the ecosystem. It is questionable if grazing of sagebrush-dominated rangelands that produce less than 448 kg per ha (400 lbs/ac) per year of herbaceous forage should be permitted. Domestic livestock grazing should not be permitted of any sagebrush steppe habitats that produce less than 224 kg per ha (200 lbs/ac) of herbaceous vegetation per year if successful sage-grouse nesting and brood rearing is an objective. Unfortunately, there are no replicated long-term studies of the effects of stocking rates for cattle in sagebrush grasslands (Holechek et al. 1999:12).

Livestock

Grazing by domestic cattle can negatively impact nesting success of ground-nesting birds (Walsberg 2005). Several studies have demonstrated that greater sage-grouse nest success is higher where grass height and density is greater than at random sites (Wakkinen 1990, Gregg 1991). Thus, livestock grazing that reduces herbaceous cover in sagebrush steppe may negatively affect nest success of sage-grouse. Sites used by sage-grouse broods are characterized by higher plant species richness (Dunn and Braun 1986, Klott and Lindzey 1990, and others) with strong grass and forb components (Sveum et al. 1998). Excessive livestock use may damage these important areas.

Livestock stocking rates are most important in affecting forage use and residual herbaceous cover followed by timing of grazing and length of the grazing season. The most common prescription used by public land management agencies on public lands is that of ‘moderate use’. Holechek et al. (1999:12) equated ‘moderate use’ to removal of an average of 43% (their Table 2) of the primary forage species. These authors found that moderate use resulted in rangeland deterioration in semi-arid grasslands. Holechek et al. (1999:15) recommended that no more than 30-35% use of annual herbaceous production would be necessary for improvement in rangeland vegetation versus the common recommendation of 50% use by the Natural Resources Conservation Service.

My recommendation, if livestock grazing is permitted on public rangelands, is to not exceed 25-30% utilization of herbaceous forage each year. Grazing should not be allowed until after 20 June and all livestock should be removed by 1 August with a goal of leaving at least 70% of the herbaceous production each year to form residual cover to benefit sage-grouse nesting the following spring. Twice-over grazing systems, where livestock pass through an area twice in a grazing season, should be avoided, and full rotation of each subdivision of an allotment or at least on a pasture basis should occur once every 4 years. Winter grazing is generally less negative for herbaceous vegetation and sage-grouse than grazing during the growing season. Care should be used in calculating stocking rates to ensure that no more than 25-30% forage utilization is achieved. Winter grazing should not be initiated until plant growth has ceased for the year and should generally occur in the 15 November to 1 March interval. Larger pastures with fewer fences are better than smaller pastures. Water and salt should be placed near fences or fence corners, as these areas (fences and fence corners) tend to 'naturally' attract livestock. The goal should be to reduce livestock impacts in the centers of pastures or allotments. Because fences are generally negative for sage-grouse (Connelly et al. 2004), placement of water and salt near fences can be used to concentrate livestock impacts in areas removed from the more valuable habitats for sage-grouse.

Wildlife

Native wildlife, primarily elk (*Cervus elaphus*), but also deer (*Odocoileus* spp.), pronghorn (*Antilocapra americana*), and hares (*Lepus* spp.), graze sagebrush steppe. Except in limited situations, such as within fenced pastures (to benefit domestic sheep which may prevent pronghorn movement), severe winter conditions, or unique situations (especially with hares), grazing by native wildlife species of particular sites is non-repetitive (unlike with domestic livestock). Hunting regulations by state and provincial agencies should keep populations of game animals within herd objectives. Management of elk can be difficult in achieving adequate harvests. State and provincial wildlife agencies should rigorously seek to manage elk within stated herd objectives or to reduce their numbers when sage-grouse habitat objectives are at risk. In areas where herd objectives cannot be met through legal hunting, reintroduction of native large predators should be considered.

'Wild' horses and burros also occupy some public lands and can cause habitat deterioration in areas important to sage-grouse. Efforts should be made to reduce or eliminate undocumented or permitted horses and burros on public lands important to sage-grouse where habitat deterioration is occurring.

Management of Habitat Fragmentation

Fragmentation of habitats useful for greater sage-grouse is not of recent origin, but only recently has it been accorded proper recognition (Braun 1998, Connelly et al. 2004). There are many factors that can fragment habitats from conversion of habitat type (agriculture adjacent to sagebrush steppe), to fences, power lines, roads, reservoirs, wild fire, and prescribed burns. Essentially, any land use, development, or treatment that subdivides blocks of intact sagebrush causes fragmentation. Management of sagebrush steppe should focus on maintaining large (>1 cadastral section [2.59 km^2 or 1 mi^2]) blocks of sagebrush steppe and preferably in excess of 20 cadastral sections [51.8 km^2 or 20 mi^2] in size. These blocks should conserve habitat at the landscape scale with at least 1 large block per Township (36 cadastral sections [93.2 km^2 or 36 mi^2]) throughout the sagebrush steppe. This recommendation is based on personal observations as well on published literature (Toepfer et al. 1990).

Continuity among habitat patches is desirable. Dispersal corridors should be preserved between and among blocks of habitats useful to greater sage-grouse. These corridors should be at least 1.6 km (1 mi) in width to reduce predator concentrations. Corridors should not contain roads, power lines, oil and gas developments, fences, or buildings.

Management of Invasive Plant Species

Invasive plant species are becoming more widespread throughout public lands as a result of disturbance from livestock grazing, livestock feeding operations, roads, development, and other land uses. While there are numerous invasive species that may occur across the sagebrush steppe, those most important over large areas include cheatgrass, juniper and pinyon pine (both native species), as well as other exotic species. Control or elimination of exotic species should have the highest priority.

Cheatgrass

Livestock management practices, fire, plowing/chaining, various types of development, and other practices have facilitated the spread of cheatgrass. Cheatgrass is palatable to livestock for only a short period during early growth in spring. It is a highly proficient seed producer and cannot be easily controlled by disking, plowing, grazing, or herbicides during the growing period or when mature. However, several pre-emergent herbicides have been demonstrated to reduce germination of cheatgrass (Connelly et al. 2000*b*). Reseeding cheatgrass-dominated areas with dryland alfalfa and native bunch grasses in strips (20 m width with every other strip being alfalfa/bunch grasses/biennial sweet clover/sagebrush) would appear to be effective in reducing cheatgrass abundance and may be more economical than use of herbicides.

Pinyon/Juniper

Management of pinyon pine or juniper invasion can be achieved through cutting and burning (either or both) individual trees as well as use of prescribed fire over larger landscapes. Treatment of individual trees is most effective (but more expensive), as the live sagebrush and grass/forb understory is not burned (Commons et al. 1999).

Management of Rangeland Seedings

Hundreds of thousands of hectares of former sagebrush steppe have been seeded with non-native forage species following plowing (to benefit livestock) or wild fire. Much of this area was reseeded with crested wheatgrass (*Agropyron cristatum*). Unfortunately, crested wheatgrass is of little use to sage-grouse as it provides poor cover and no food value. Sage-grouse seasonally consume forbs, insects, and sagebrush and do not eat grass seeds or leaves. Further, crested wheatgrass is a prolific seed producer with the ability to remain dominant on the landscape for periods exceeding 40 years. Crested wheatgrass is preferred forage for livestock and wild ungulates, especially during the growing period. It is capable of withstanding substantial grazing pressure and, once established, crested wheatgrass is difficult to replace with native bunchgrasses and sagebrush (due to competition and lack of seed sources).

Benign neglect has allowed portions (primarily the edges) of many seedings on public lands to revert in part to sage-grouse habitat. This is the result of sagebrush regeneration from seeds of live sagebrush in adjacent areas. Sage-grouse use these areas as density of sagebrush seedlings and canopy cover increases. Unfortunately, forb abundance in most crested wheatgrass seedings is very low (<3-5% cover) and sage-grouse use is mostly confined to foraging on young sagebrush plants. Crested wheatgrass seedings with less than 5% sagebrush canopy cover should be disked and reseeded in strips perpendicular to the prevailing wind to aid restoration of native habitats. Strips should be no more than 20 m in width in a ratio of 1 strip every 100 m. Strips should be planted with a mixture of dryland alfalfa, biennial sweet clover, native bunch grasses, and taller sagebrush (either mountain big sagebrush [*Artemisia tridentata vaseyana*] or Wyoming big sagebrush [*A. t. wyomingensis*] depending upon the site).

Biological control of crested wheatgrass seedings through manipulation of grazing intensity is possible but is negative to overall rangeland health as it results in severe overgrazing of all areas including adjacent native sagebrush steppe. This practice should not be promoted, as it will fail to control or eliminate crested wheatgrass. Chemical control of crested wheatgrass seedings also has little chance of success because of the abundant but dormant seed in the upper levels of the soil profile that are not affected by herbicides. Mechanical control through plowing or disking of the entire seeding followed by reseeding with desirable plant species also has little merit as it is expensive and exposes large expanses to wind erosion and exotic weeds. Plowing or disking (with or without reseeding) also has little chance of success because of the abundant amount of crested wheatgrass seed in the upper soil profile. Thus, the best scenario is to disk strips into crested wheatgrass seedings horizontal to the prevailing wind and replant desired vegetation (in strips) while protecting all larger sagebrush plants that may be present to serve as seed sources. Additional strips should be disked and reseeded at 3-5 year intervals depending upon site and results from the initial strips (adaptive management).

Management of Roads

Roads are known to reduce the value of potential breeding habitats for greater sage-grouse (Connelly et al. 2004), cause lek abandonment (Braun 1986), and lead to death (from collisions). Road densities are increasing within occupied sage-grouse habitats. A recent study in the Upper Green River Valley, Wyoming found that all remaining greater sage-grouse leks were within 5 km (3.1 miles) of a road and that 95% of the Jonah gas field had road densities greater than 3.2 km per 2.59 km^2 (2 miles/mile^2) (Thomson et al. 2005). Distinction should be made among primary roads (usually paved), secondary roads (mostly gravel), and trails (usually dirt, commonly expressed as 2-tracks). Primary roads are most negative for greater sage-grouse because of vehicle frequency, speed, and noise. Secondary roads can also be very negative depending again upon vehicle frequency, speed, and noise. Generally, trails are used seasonally and receive light vehicle use. Consequently, they are least problematic for sage-grouse.

Public land management agencies should have transportation plans for each forest, district, and resource area. Both permanent and seasonal road/trail closures are appropriate to reduce disturbance to sage-grouse during breeding activities and winter.

Most trails within occupied sage-grouse habitat should be closed during the breeding period and winter. Some secondary roads within 5 km of active leks should be closed during the 1 March-20 June period as well as during winter (December-February). All secondary roads and trails that traverse important sage-grouse areas should be reviewed and considered for permanent closure and revegetation.

Off-road vehicles (ORVs) should be prohibited except on designated trails and roads where sage-grouse use does not occur.

Management of Structures

Greater sage-grouse did not evolve with structures. Sage-grouse commonly collide with fences, and power lines have been demonstrated to be negative as they may result in collisions resulting in injury to or death of birds (Connelly et al. 2004). Structures can also provide perch locations for raptors, especially golden eagles (*Aquila chrysaetos*), which prey upon sage-grouse during all seasons of the year, and corvids that prey on nests. Prior to the advent of human-made structures, raptors and corvids in sagebrush steppe used elevated natural sites from which to hunt. The addition of power line poles, fences, hay equipment and stacks, and abandoned buildings have greatly expanded the number of suitable perches for raptors in a landscape that is mostly devoid of trees (Connelly et al. 2004). Historically, there were large expanses of suitable habitat for sage-grouse with few elevated perch sites.

Utility companies should be required to fit all potential perch sites (poles, towers) for golden eagles with devices to deter perching (including power poles associated with oil and gas development). All unused power poles (and towers) should be removed and consideration should be given to elimination (and removal) of unnecessary power lines that traverse sage-grouse habitats. Existing power lines should be placed in corridors that follow road systems, especially those that are paved, to minimize impacts on the landscape. First priority for fitting power poles with raptor guards and or for removal of power lines should be given to areas within 5.5 km (3.3 miles) of active leks (at least line of sight). Second priority should be given to known sage-grouse winter-use areas, especially along windswept ridges and near large expanses of sagebrush that are not typically covered by snow in winter. Raptor predation during summer and early fall is usually a local problem and more a product of habitat quality (i.e., sage-grouse are limited to few areas of suitable habitat) than at other times of the year.

Metal fence posts are preferable to wooden posts for fencing as the former better discourage raptors from using them as perches. Fencing within 2 km of active leks should be discouraged as sage-grouse are more likely to collide with them as they fly to and from leks, frequently at low levels and in low light. Fences designed to prevent domestic sheep from escaping pastures should be eliminated as walking sage-grouse frequently will follow and not readily fly over them. Fences in sage-grouse areas should be of no more than 3-strands of wire with both the top and bottom wires being barbless. All unnecessary fences should be removed (wire and posts). If fences known to result in sage-grouse mortality cannot be removed, the top wire should be marked with permanent visual flagging.

Management of Vegetation

Native sagebrush steppe vegetation should be given highest priority for management. Management should revolve around proper livestock grazing practices and not use of chemical or mechanical treatments. Grazing should be managed to ensure that sagebrush-dominated rangelands have the opportunity to recover from past management practices. The goal is to have healthy, self-sustaining native vegetation in which sagebrush comprises 10 to 25% of the vegetative canopy cover, grasses comprise 30-40%, and forbs comprise 15 to 20% of the ground cover. Holechek et al. (1999:15) indicate that livestock grazing, if the intent is to improve rangeland vegetative condition, should remove no more than 30-35% of the annual herbaceous growth. Some areas may require complete removal of livestock grazing for 3-5 years before grazing at lower stocking rates can resume. Improved management of grazing is the least expensive practice to restore degraded sagebrush steppe and should have the highest priority.

Chemicals such as 2,4-D and tebuthiuron have been widely used in attempts to eliminate or reduce sagebrush to increase livestock forage on public rangelands (Braun 1987, 1998). Use of 2,4-D has mostly been phased out for a variety of human health and environmental reasons (Braun 1998). Tebuthiuron is now favored for controlling sagebrush, especially to 'thin' sagebrush stands. Unfortunately, the effectiveness of this chemical is site dependent and is greatly affected by soil characteristics (Braun 1998) and continued livestock grazing. Application rates are critical and use of high rates or any chemical use on inappropriate soils can lead to total kill of sagebrush and forbs. For this reason, use of chemicals to 'thin' or control sagebrush is usually inappropriate for winter and breeding habitat.

Mechanical methods to manage sagebrush date to the 1930's and have involved brush beating, disking, chaining, and railing (Pechanic et al. 1954). These methods are relatively expensive and have mostly been used on small scales. They have the advantage of being able to be tailored to specific sites and will not 'escape' or 'drift' when compared to fire or use of chemicals. Of the available mechanical methods, use of brush beating is most appropriate as the desired results in terms of vegetation can reasonably be predicted. Brush beating or any other type of mechanical method to manage sagebrush should only be considered for 'better' range sites where vegetation response can be expected. These are normally areas where sagebrush canopy cover is >30%. Brush beating should be done in strips (usually 10-20 m in width) not to exceed one-quarter (25%) of the width of untreated strips. Strips should conform to the terrain and should not be straight lines but should be perpendicular to the prevailing wind. The design should result in a mosaic of sagebrush types with no more than 20-30% of the area being treated every 10-15 years (depending upon site). The goal is to set back sagebrush height (causing resprouting) and not death of all sagebrush plants. This can be accomplished by adjustment of the height of the mower blades. More recent advances such as the 'Dixie Harrow' and 'Lawson Aerator' may have merit but more scientific analysis of the results of using these devices is needed. Management of livestock grazing (reduction in or elimination of use for at least 2 years) is normally needed following brush beating or any mechanical treatment.

Use of fire to manage sagebrush steppe vegetation is usually inappropriate as it is difficult to control and frequently burns primarily winter and nesting habitats (Connelly et al. 2000*a*). Fire should generally be avoided or, at the least, restricted to small (<20 ha) sites where a lack of brood habitat has been documented to limit increases in sage-grouse populations.

Management of Water

Greater sage-grouse have been documented to use open water, especially during dry seasons. They readily eat snow in winter and forage during summer and fall on succulent vegetation in mesic sites. This vegetation may be adjacent to agricultural areas, riparian habitats, or where water is allowed to flow over land at springs and ponds. The need for so-called wildlife "guzzlers" is questionable, as studies have failed to demonstrate increases in sage-grouse density in areas with guzzlers (Connelly and Doughty 1989). Surface water flow in summer is important as it promotes growth of succulent forbs, which are attractive to greater sage-grouse. Pipes and tanks (for livestock) have no value for sage-grouse unless water is available at ground level or is allowed to spill onto the ground. There should be no emphasis placed on improving water distribution for livestock as this negatively affects sage-grouse habitats in most cases outside of ponds. All seeps and springs, and associated mesic sites should be fenced to exclude large grazing animals including domestic sheep, cattle, horses, and burros.

Livestock grazing has also impacted water tables by increasing sagebrush density and increasing soil erosion by reducing surface litter that slows runoff. Techniques useful to increasing water table levels include reduction of livestock grazing, sagebrush mowing, filling eroded drainages with (certified weed-free) straw bales, and creating check dams. These techniques are also useful in creating brood habitat for sage-grouse.

Where Should Management Focus Be Placed?

Areas with existing sage-grouse populations should have the highest priority for conservation. The best scenario for improved sage-grouse abundance and distribution is to conserve habitats with existing populations and then work outward from those core areas to improve habitats in more peripheral areas. GIS (Geographic Information Systems) derived maps of present vegetation and soil potential should be used with overlays of past and planned treatments to prevent too much area from being treated in a 10-15+ year period. The goal should be to increase sage-grouse abundance and distribution. Increases in abundance will be easier to achieve.

Areas contiguous to existing populations which do not presently have sage-grouse or which have very small populations (100-300 birds) should have second priority for management. Review of GIS maps of vegetation and soil potential will frequently identify factors that are depressing sage-grouse populations when compared to similar maps where sage-grouse still persist in some number. Treatments to improve abundance and distribution of populations will vary from area to area. Grazing practices and development are the most obvious factors depressing sage-grouse populations followed by fragmentation caused by vegetation treatments, including fire.

How Should Success Be Measured?

Changes in abundance of greater sage-grouse are best measured by monitoring the number of active leks in a discrete area (leks/10 km²) over a 3-5 year period. Total number of males counted in a given area over a 3-5 year period can also be used. Changes in estimated nest success and percent young based on wing surveys of hunter-harvested birds (where appropriate) may also provide useful data (Autenrieth et al. 1982, Connelly et al. 2003). Changes in the proportion of young to adult (and yearling) hens in the harvest can also be used to detect improvement in sage-grouse production.

Changes in distribution of greater sage-grouse can be derived from intensive searches for active leks in areas (based on GIS derived maps of potential habitat) where sage-grouse were not present in the previous 3-5 years. Random transects to assess seasonal changes in distribution of sage-grouse fecal pellets can also be used to assess changes in distribution. Even presence or absence line transect counts of either sage-grouse or their sign (pellets) can be useful. These surveys should be made at 3-5 year intervals.

Changes in vegetation such as % bare ground, % forb coverage, % grass coverage, % sagebrush cover, as well as height of residual herbaceous material can be used to assess changes in vegetative composition and quality of habitats. However, vegetation surveys are labor intensive, costly, and may be affected by weather conditions, rodents, insects, and grazing animals. It is highly unlikely that short-term changes can be detected without standardized plots, which are marked and uniformly evaluated. This is not likely to be done on a consistent basis over large areas of western North America. It will be difficult to measure success in vegetation improvement except over time in very localized sites.

Conclusions

Habitat conservation strategies to improve the abundance and distribution of greater sage-grouse have not been scientifically tested because of the reluctance of public land management agencies to invest in replicated management experiments over sufficiently large areas to be able to detect responses. However, sufficient information is available to make management recommendations given that negative responses of sage-grouse (decreases in abundance and distribution) are measurable. Habitat loss is certainly measurable as are fragmentation and degradation of habitats. The most notable changes in the sagebrush steppe since European settlement are associated with repetitive grazing by domestic livestock and developments (no matter how ‘development’ is defined). It is logical to expect improvement in sage-grouse abundance, at the least, with changes in policies, regulations, and practices involving grazing of domestic livestock and development. Both of these factors are managed by the key public land management agencies (BLM and USFS) that together control in excess of 60% of the remaining sagebrush steppe occupied by greater sage-grouse. Improvement in distribution will be more difficult as restoration of useful sagebrush habitats in areas that have been burned or plowed and seeded to exotic grasses will be exceedingly slow.

Management practices that significantly reduce wild fire, reduce grazing intensity and forage utilization, and reduce or eliminate the spread of introduced annuals have the best chance to positively impact abundance of greater sage-grouse. They will be the least expensive to implement. Development practices such as gas and oil exploration and production including surface infrastructure, which are obviously negatively affecting sage-grouse abundance and distribution, will be more expensive to change, but collectively changes in these practices could equal the gains expected to result from changes in livestock grazing practices.

Sufficient knowledge is available to begin implementing recommended practices that will positively affect greater sage-grouse. The key is to develop public support and the resolve within federal agencies to make the necessary changes.

Recommendations

-First priority for habitat management should be areas where larger sage-grouse populations are still present. Management practices chosen should maintain the present abundance and distribution of sage-grouse.

-The second priority for habitat management is for areas where sage-grouse populations are small (<300 birds or 100 males counted on a 3-year moving average). Management practices should enhance sage-grouse abundance and distribution.

-A third priority should be to improve habitats in areas adjacent to existing populations.

-Sagebrush steppe management should focus on maintaining large (>1 cadastral section and preferably >20 cadastral sections in size) blocks of sagebrush habitat per Township (36 cadastral sections).

-No surface occupancy should be allowed within 5.5 km of all active sage-grouse leks.

-No roads should be constructed within 5.5 km of active sage-grouse leks.

-Existing roads within 5.5 km of active sage-grouse leks should have seasonal closures (1 March-20 June).

-Prescribed fires should be no larger than 20 ha with no more that 40% of each cadastral section being burned over a 15-year period.

-Wild fires in sagebrush steppe should be vigorously suppressed except in areas with >20 invasive conifer trees per ha.

-Livestock grazing should be deferred for 3 years following fires for recovery of herbaceous native vegetation.

-Livestock grazing should not remove more than 25-30% of the annual growth of herbaceous vegetation with grazing delayed until after 20 June. True rest rotation systems should be used and winter grazing is preferred.

-Where wildlife (deer and elk) herd objectives cannot be achieved through legal hunting, reintroduction and expansion of populations of large predators should be encouraged.

-Rangeland seedings of exotic grasses should be converted using reseeded strips of native bunchgrasses, adapted subspecies or species of sagebrush, and dryland alfalfa.

-Power lines should be placed only into existing road/utility corridors.
-Power poles and other existing human structures should either be removed, if not used, or fitted with raptor-deterrence devices.
-Fences in sage-grouse use areas should be no more than 3 strands with the top and bottom wires being barbless. Unused fences should be removed.
-Use of chemicals to 'manage' sagebrush should not be permitted. If sagebrush is to be managed to reduce density or to enhance vigor, mechanical methods are preferred.
-Sage-grouse have not been shown to need open water. However, water should be allowed to flow (seep) over the ground to encourage growth of succulent forbs.
-Active leks per unit of area and total number of male sage-grouse counted at proscribed (4 counts per breeding period spaced at 7-10 day intervals) should be used as the measure of success of management treatments followed by changes in % bare ground, % forb coverage, % grass cover, % sagebrush canopy cover, and height of residual herbaceous vegetation.
-Sage-grouse pellet transects should be used to measure expansion of birds into vacant or former habitat.

Literature Cited

Aldridge, C. L. 1998. Status of the sage grouse (*Centrocercus urophasianus urophasianus*) in Alberta. Wildlife Status Report 13. Wildlife Management Division, Alberta Environmental Protection and Alberta Conservation Association, Edmonton, Canada.

Autenrieth, R. E., W. Molini, and C. E. Braun 1982. Sage grouse management practices. Technical Bulletin 1. Western States Sage Grouse Committee, Twin Falls, Idaho, USA.

Braun. C. E. 1986. Changes in sage grouse lek counts with advent of surface coal mining. Thorne Ecological Institute. Proceedings, Issues and Technology in The Management of Impacted Western Wildlife 2:227-231.

Braun, C. E. 1987. Current issues in sage grouse management. Proceedings of the Western Association of Fish and Wildlife Agencies 67:134-144.

Braun, C. E. 1998. Sage grouse declines in western North America: what are the problems. Proceedings of the Western Association of Fish and Wildlife Agencies 78:139-156.

Braun, C. E., T. Britt, and R. O. Wallestad. 1977. Guidelines for maintenance of sage grouse habitats. Wildlife Society Bulletin 5:99-106.

Braun, C. E., O. O. Oedekoven, and C. L. Aldridge, 2002. Oil and gas development in western North America: effects on sagebrush steppe avifauna with particular emphasis on sage grouse. Transactions of the North American Wildlife and Natural Resources Conference 67:337-349.

Braun, C. E., J. W. Connelly, and M. A. Schroeder. 2005. Seasonal habitat requirements for sage-grouse: spring, summer, fall, and winter. Pages 38-42 *in* N. L. Shaw, M. Pellant, and S. B. Monsen, compilers. Sage-grouse habitat restoration symposium proceedings, 4-7 June 2001, Boise, Idaho, USA. U. S. Department of Agriculture, Forest Service, RMRS-P-38.

Commons, M.L., R. K. Baydack, and C. E. Braun. 1999. Sage grouse response to pinyon-juniper management. Pages 238-239 *in* S. B. Monsen and R. Stevens, compilers. Proceedings: ecology and management of pinyon-juniper communities within the Interior West. U. S. Department of Agriculture, Forest Service, RMRS-P-9.

Connelly, J. W., and C. E. Braun. 1997. Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. Wildlife Biology 3:229-234.

Connelly, J. W., and L. A. Doughty. 1989. Sage grouse use of wildlife water developments in southeastern Idaho. Pages 167-173 *in* S. Stiver and G. Tsukomoto, editors. Symposium on wildlife water developments. Nevada Department of Fish and Game, Reno, USA.

Connelly, J. W., K. P. Reese, and M. A. Schroeder. 2003. Monitoring of greater sage-grouse habitats and populations. College of Natural Resources Experiment Station Bulletin 80. University of Idaho, Moscow, USA.

Connelly, J. W., K. P. Reese, R. A. Fischer, and W. L. Wakkinen. 2000*a*. Response of a sage grouse breeding population to fire in southeastern Idaho. Wildlife Society Bulletin 28:90-96.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000*b*. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Unpublished Report. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming, USA.

Dunn, P. O., and C. E. Braun. 1986. Summer habitat use by adult female and juvenile sage grouse. Journal of Wildlife Management 50:228-235.

Gregg, M. A. 1991. Use and selection of nesting habitat by sage grouse in Oregon. Thesis. Oregon State University, Corvallis, USA.

Holechek, J. L., H. Gomez, F. Molinar, and D. Galt. 1999. Grazing studies: what we've learned. Rangelands 21(2): 12-16.

Holloran, M. J. 2005. Greater sage-grouse (*Centrocercus urophasianus*) population response to natural gas field development in western Wyoming. Dissertation. University of Wyoming, Laramie, USA.

Holloran, M. J., and S. H. Anderson. 2004. Sage-grouse response to natural gas field development in northwestern Wyoming. Proceedings of the Western Agencies Sage and Columbian sharp-tailed grouse Technical Committee 24:16.

Holloran, M. J., and S. H. Anderson. 2005. Greater sage-grouse population response to natural gas field development in western Wyoming: are regional populations affected by relatively localized disturbance? Transactions of the North American Wildlife and Natural Resources Conference 70:In Press.

Hornaday, W. T. 1916. Save the sage grouse from extinction, a demand from civilization to the western states. New York Zoological Park Bulletin 5:179-219.

Hulet, B. V. 1983. Selected responses of sage grouse to prescribed fire, predation, and grazing by domestic sheep in southeastern Idaho. Thesis. Brigham Young University, Provo, Utah, USA.

Klott, J. H., and F. G. Lindzey. 1990. Brood habitats of sympatric sage grouse and Columbian sharp-tailed grouse in Wyoming. Journal of Wildlife Management 54:84-88.

Knick, S. T., D. S. Dobkin, J. T. Rotenberry, M. A. Schroeder, W. M. Vander Haegen, and C. van Riper III. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. Condor 105:611-634.

Küchler, A. W. 1964. Potential natural vegetation of the conterminous United States (map and manual). Special Publication 36. American Geographical Society, New York, USA.

Lyon, A. G. 2000. The potential effects of natural gas development on sage grouse (*Centrocercus urophasianus*) near Pinedale, Wyoming. Thesis. University of Wyoming, Laramie, USA.

Lyon, A. G., and S. H. Anderson. 2003. Potential gas development impacts on sage-grouse nest initiation and movement. Wildlife Society Bulletin 31:486-491.

Miller, R. F., and L. E. Eddleman. 2001. Spatial and temporal changes of sage grouse habitat in the sagebrush biome. Agricultural Experiment Station Technical Bulletin 151. Oregon State University, Corvallis, USA.

Nelle, P. J., K. P. Reese, and J. W. Connelly. 2000. The long-term effect of fire on sage grouse nesting and brood-rearing habitats on the Upper Snake River Plain. Journal of Range Management 53:586-591.

Patterson, R. L. 1952. The sage grouse in Wyoming. Sage Books, Denver, Colorado, USA.

Pechanic, J. F., G. Stewart, A. P. Plummer, J. H. Roberson, and A. C. Hull. 1954. Controlling sagebrush on rangelands. Farmer's Bulletin 2072. U. S. Department of Agriculture, Washington, D.C., USA.

Remington, T. E., and C. E. Braun. 1991. How surface coal mining affects sage grouse, North Park, Colorado. Thorne Ecological Institute. Proceedings, Issues and Technology in the Management of Impacted Western Wildlife 5: 128-132.

Schroeder, M. A., C. L. Aldridge, A. D. Apa, J. R. Bohne, C. E. Braun, S. D. Bunnell, J. W. Connelly, P. A. Deibert, S. C. Gardner, M. A. Hilliard, G. D. Kobriger, S. M. McAdam, C. W. McCarthy, J. J. McCarthy, D. L. Mitchell, E. V. Rickerson, and S. J. Stiver. 2004. Distribution of sage-grouse in North America. Condor 106:363-376.

Sveum, C. M., J. A. Crawford, and W. D. Edge. 1998. Use and selection of brood-rearing habitat by sage grouse in south-central Washington. Great Basin Naturalist 58:344-351.

Thomson, J. L., T. S. Schaub, N. W. Culver, and P. C. Aengst. 2005. Wildlife at a crossroads: energy development in western Wyoming. Effects of roads on habitat in the Upper Green River Valley. The Wilderness Society, Washington, D.C., USA.

Toepfer, J. E., R. L. Eng, and R. K. Anderson. 1990. Translocating prairie grouse: what have we learned? Transactions of the North American Wildlife and Natural Resources Conference 55:569-579.

Vale, T. R. 1975. Presettlement vegetation in the sagebrush grass area of the Intermountain West. Journal of Range Management 28: 32-36.

Wakkinen, W. L. 1990. Nest site characteristics and spring-summer movements of migratory sage grouse in southeastern Idaho. Thesis. University of Idaho, Moscow, USA.

Walsberg, G. E. 2005. Cattle grazing in a national forest greatly reduces nesting success in a ground-nesting sparrow. Condor 107:714-716.

About The Author

Clait E. Braun has worked with sage-grouse as a researcher (1973-99) and consultant (2000-06), and has been a leader in publishing research and management articles on sage-grouse. Dr. Braun is a Certified Wildlife Biologist and has either worked in or extensively visited all states and provinces with current populations of sage-grouse. He retired from the Colorado Division of Wildlife where he was responsible for sage-grouse research from 1973 into 1999 and now operates Grouse Inc. providing professional guidance and reviews on sage-grouse and their habitats. This ‘Blueprint’ represents his professional experience and selected literature based on 30+ years of work with sage-grouse.

Appendix

Seasonal Habitat Requirements for Sage-grouse:

Spring, Summer, Fall, and Winter[1]

(Citation: Braun, C. E., J. W. Connelly, and M. A. Schroeder. 2005. Pages 38-42 *in* N. L. Shaw, M. Pellant, and S. B. Monsen, compilers. Sage-grouse habitat restoration symposium proceedings, 4-7 June 2001, Boise, Idaho, USA. U. S. Department of Agriculture, Forest Service, RMRS-P-38.)

[1]The contents of this 'Blueprint' document have not been reviewed or approved by either of the 2 coauthors of the published paper referenced in the Appendix.

Seasonal Habitat Requirements for Sage-Grouse: Spring, Summer, Fall, and Winter

Clait E. Braun
John W. Connelly
Michael A. Schroeder

Abstract—Sage-grouse (*Centrocercus minimus, C. urophasianus*) are dependent upon live sagebrush (*Artemisia* spp.) for all life processes across their entire range. This paper describes habitats used by sage-grouse as documented in the scientific literature. The leaves of sagebrush are eaten by sage-grouse throughout the entire year and comprise 99 percent of their winter diets. Spring (late March through May) habitats are those with intermixed areas of taller (40 to 80 cm) sagebrush with canopy cover of 15 to 25 percent and taller (>18 cm) grass/forb cover of at least 15 percent. Sites used for display have shorter vegetation, frequently few or only short sagebrush plants, but with taller, more robust sagebrush within 100 to 200 m that is used for escape cover. Nesting cover mimics that used overall during spring but with clumps of tall (>50 cm), dense (about 25 percent) live sagebrush and abundant forbs (>10 to 12 percent cover). Early brood rearing areas are those within 200 m (initial 3 to 7 days posthatch) to 1 km (up to 3 to 4 weeks posthatch) of nest sites. Forbs and taller (>18 cm) grasses are important for broods; forbs provide succulent foods, grasses provide hiding cover, and the grass/forb mixture supports insects used by chicks. Summer use areas are those with abundant succulent forbs with live, taller (>40 cm), and robust (10 to 25 percent canopy cover) sagebrush useful for cover. These areas continue to be used into fall when sage-grouse move to higher benches/ridges where they forage on remaining succulent forbs such as buckwheat (*Eriogonum* spp.) and switch to more use of sagebrush leaves. Winter (early December to mid-March) use areas are often on windswept ridges, and south to southwest aspect slopes as well as draws with tall, robust live sagebrush. Height (25 to 35 cm) of sagebrush above the surface of the snow in areas used in winter is important, as is canopy cover (10 to 30 percent). Management of habitats used by sage-grouse should initially focus on maintaining all present use areas. Practices to enhance sagebrush habitats to benefit sage-grouse are reviewed, as is the need to annually monitor sage-grouse numbers along with systematic monitoring of the health of sagebrush ecosystems.

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Introduction

Sage-grouse (*Centrocercus minimus, C. urophasianus*) historically occurred in at least 16 States and three Canadian Provinces (Aldrich 1963; American Ornithologists' Union 1957; Johnsgard 1973). They have been extirpated in five States and one Canadian Province (Braun 1998; Connelly and Braun 1997) and their overall distribution has become discontinuous (fig. 1). The changes in sage-grouse distribution have been attributed to loss, fragmentation, and degradation of habitats (Braun 1995, 1998; Connelly and Braun 1997), and it is probable that at least one-half of the original occupied area can no longer support sage-grouse (Braun 1998). Because of the reduced amount of available habitat, sage-grouse abundance has also markedly decreased with reported declines of 10 to 51 percent (Connelly and Braun 1997) and as much as 45 to 82 percent since 1980 (Braun 1998). The known decreases in distribution and abundance have led to concern about stability of sage-grouse populations and the health of sagebrush ecosystems upon which they depend. Petitions to list sage-grouse under the Federal Endangered Species Act have been filed for northern sage-grouse (*C. urophasianus*) and for Gunnison sage-grouse (*C. minimus*).

Sage-grouse are dependent upon ecosystems with vast and relatively continuous expanses of live, robust, taller sagebrushes (*Artemisia* spp.) with a strong grass and forb component. This dependency upon sagebrush, especially the subspecies of big sagebrush (*A. tridentata vaseyana, A. t. wyomingensis, A. t. tridentata*), low sagebrush (*A. arbuscula*), black sagebrush (A. *nova*), silver sagebrush (*A. cana*), and three-tip sagebrush (*A. tripartita*), as well as a variety of less apparent and abundant species, has been well documented (Patterson 1952; reviews by Braun and others 1977 and Connelly and others 2000a). Since the early 1960s, the sage-grouse/sagebrush relationship has focused attention by Western States and Provinces on the need to maintain healthy sagebrush-steppe communities over large expanses. Guidelines for maintenance of sage-grouse habitats were developed from the scientific literature (Braun and others 1977, completely revised by Connelly and others 2000a) and promoted by the Western States Sage-Grouse Technical Committee. The purpose of this paper is to present an overview of the habitat needs of sage-grouse based on the scientific literature, identify the issues that affect maintainance of useful habitats for sage-grouse, and discuss management strategies to maintain, enhance, and restore habitats

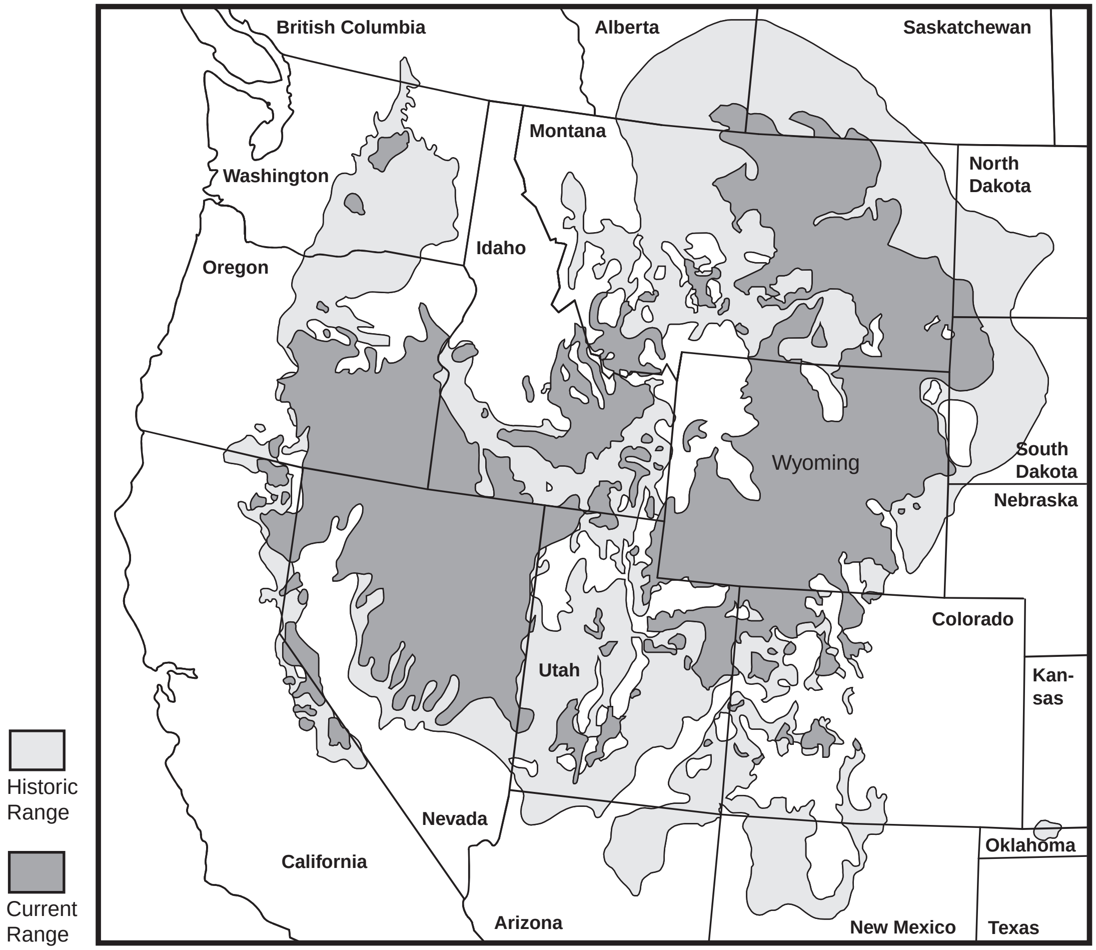

Figure 1—Historic and current distribution of sage-grouse (map prepared by M. A. Schroeder).

for sage-grouse. This paper draws extensively on the published *Guidelines to Manage Sage Grouse Populations and Their Habitats* (Connelly and others 2000a).

Habitat Overview

Spring

Timing of spring breeding activities of sage-grouse is dependent on elevation and amount of persistent snow cover. Attendance at leks may start in early to mid-March or, at higher elevations, in early April. Males may attend and display at leks until late May but most display and mating activities are greatly reduced by mid-May. Amount and depth of snow cover greatly influence sage-grouse breeding activities; thus, snow-free areas are important components of spring habitat. Habitats used by sage-grouse during the breeding period are those associated with foraging, leks, escape, and nesting. Depending upon moisture regimes, height of sagebrush in used habitats varies from 30 to 80 cm with canopy cover from 15 to 25 percent (Connelly and others 2000a). Lek sites typically have low amounts of sagebrush and appear relatively bare, but they may have extensive cover of low grasses and forbs. Taller, robust live sagebrush used as escape cover is normally within 100 to 200 m of active leks. The average distance from a nest to the nearest lek varies from 1.1 to 6.2 km, and the actual size of the breeding habitat appears largely dependent on the migratory characteristics of the sage-grouse population as well as distribution of sagebrush cover with respect to lek location (Connelly and others 2000a). Habitats selected for nesting are those with abundant (15 to 30 percent canopy cover) live, taller (30 to 80 cm) sagebrush plants within a community with >15 percent ground cover of taller (40 to 80 cm) grasses and forbs (Connelly and others 2000a). Early brood-rearing habitats (fig. 2) are normally those within 100 m to 1 km of nesting sites, especially areas with high plant species richness, moisture, and taller grasses and forbs (Connelly and others 2000a). Adult sage-grouse, while still foraging extensively on leaves of live sagebrush, eat leaves and flower parts of forbs during spring, as do chicks (Apa 1998; Drut and others 1994; Dunn and Braun 1986; Klott and Lindzey 1990).

Summer

Habitats used by sage-grouse in summer (early to mid-June to mid to late September) are those that provide

Figure 2—Sage-grouse brood hen in good quality Wyoming big sagebrush habitat, North Park, Colorado (photograph by C. E. Braun).

adequate forage, especially succulent forbs, and cover useful for escape. These habitats may include those used for agriculture, especially for native and cultivated hay production, edges of bean and potato fields, as well as more typical sagebrush uplands and moist drainages. Taller (>40 cm) and robust (10 to 25 percent canopy cover) sagebrush is needed for loafing and escape cover as well as a source of food. Grass and forb ground cover can exceed 60 percent (hayfields). Provided moisture is available through water catchments or from succulent foliage, sage-grouse may be widely dispersed over a variety of habitats during this period (Connelly and others 2000a). As late summer approaches, there is movement from lower sites to benches and ridges (fig. 3) where sage-grouse forage extensively on leaves of sagebrush.

Fall

Fall (late September into early December) is a time of change for sage-grouse from being in groups of hens with chicks or males and unsuccessful brood hens to separation into larger flocks frequently segregated by gender. Some birds may continue to use lower riparian or hayfield habitats, but there is movement onto higher, frequently north-aspect slopes where succulent native forbs, such as buckwheats, provide green forage. Use of sagebrush leaves for food becomes more common as does use of extensive stands (>20 percent canopy cover) of taller (>25 cm), live sagebrush (Connelly and others 2000a). Movements can be slow but there is a general shift toward traditional winter use areas (Connelly and others 1988).

Winter

Flocks of sage-grouse are somewhat nomadic in early winter but may remain within chosen areas for periods of several weeks or more depending upon extent of snow cover and depth (Beck 1977; Hupp and Braun 1989b). Sagebrush height (>20 cm, but usually >30 cm, above the surface of the snow) is important as is the robust (>10 to 30 percent canopy cover) structure of live sagebrush (Connelly and others 2000a). Sage-grouse use a variety of sites in winter including windswept ridges with open (10 to 20 percent canopy cover) (fig. 4) stands of sagebrush to draws with dense (>25 percent canopy cover) stands. Quality of the snow can be important because sage-grouse are known to use snow roosts and burrows (Back and others 1987). Aspect is also important with south and southwest slopes most used in hilly terrain (Hupp and Braun 1989b). Leaves of live, vigorous sagebrush plants provide >99 percent of the foods eaten during the winter period (early December until early to mid-March) (Patterson 1952; Remington and Braun 1985; Wallestad and others 1975). Generally, winter is a time of body mass gain (Beck and Braun 1978), although severe winter conditions over prolonged intervals can reduce the amount of area available for foraging and cover (Beck 1977) and thus affect body condition (Hupp and Braun 1989a). Overall movement during winter may be extensive and home ranges can be large (Connelly and others 2000a). As winter wanes, flocks of sage-grouse move toward breeding areas that may be immediately adjacent to or far distant from winter use areas (Connelly and others 2000a).

Figure 3—Radio-tracking sage-grouse in high-elevation summer range with a stand of mountain big sagebrush in the background (photograph by J. W. Connelly).

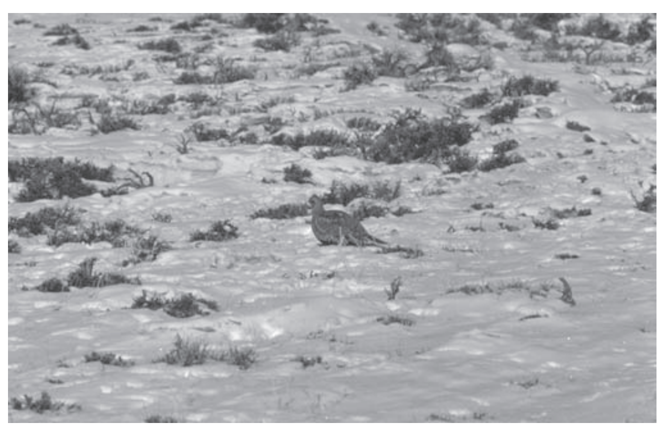

Figure 4—Sage-grouse winter range in Wyoming big sagebrush habitat in North Park, Colorado (photograph by C. E. Braun).

Issues

Decreases in distribution and abundance of sage-grouse have been ascribed to a complexity of factors (Braun 1987, 1998; Connelly and Braun 1997). The three major causes, (1) habitat loss (mostly permanent), (2) fragmentation (frequently permanent but reversible at times), and (3) degradation (usually can be corrected), are generally accepted but the latter two are poorly recognized and understood. Examples of permanent habitat loss include conversion of sagebrush rangelands to agricultural crops, town and subdivision developments, placement of power plants or surface mines, and reservoir construction. Fragmentation of habitats occurs with power lines, paved and other high-speed road development (including maintenance and improvement of farm roads), habitat-type conversion projects, fire, or any permanent development that reduces the size of existing habitat patches. Less understood are the impacts of fences, seasonal use trails, oil and gas wells with surface pipelines, noise, and so on. Some of these impacts can be resolved and sage-grouse will reoccupy some formerly disturbed areas (Braun 1987).

Distribution of habitat types useful to sage-grouse is also important, as these species are habitat specialists using a variety of areas within a larger landscape mosaic. Thus, not only is the quantity of sagebrush habitats important, but also the juxtaposition and quality of those habitats. All sagebrush habitats are not equal in their acceptability to sage-grouse, and location of areas used may affect sage-grouse distribution. Size of habitat patches is important and larger (>30 km^2) is better than smaller, although the spatial relationships of habitats for sage-grouse are not well understood. Sage-grouse use a mosaic of habitats that is normally present in sagebrush-steppe because of differences in soils, moisture, topography, aspect, insect defoliation, wildfires, and other factors. Sagebrush naturally regenerates as overmature plants die and seedlings become established. Use of the term “decadent” for sagebrush is generally inappropriate because it implies that sagebrush communities are not dynamic with a variety of age classes from seedlings to overmature. Since most sagebrush communities are resilient and represent a continuum of age classes within a mosaic of habitats, creation of “edge” to benefit sage-grouse is rarely needed. Because of human activities, the presence of too much edge (especially in straight lines) is more common than too little edge and results in degradation of sage-grouse habitats.

Sagebrush ecosystems have been managed through a variety of treatments from domestic livestock grazing, mechanical and chemical clearing or thinning, to use of prescribed fire (Braun 1998). Fire was a natural event in more mesic sagebrush communities but was infrequent as demonstrated by the lack of resprouting of big sagebrush, black sagebrush, and low sagebrush. Fire was more common in areas with three-tip sagebrush and silver sagebrush because both species resprout. Recent research suggests there is little gain in forage production of grasses and forbs after fire, because it can take longer than 30 years to return to preburn conditions (Wambolt and others 2001).

Treatments of sagebrush communities have primarily been conducted to benefit another treatment (livestock grazing). Use of some treatments has led to plantings of exotic grasses, invasion of areas by exotic plants, conifer invasion of sagebrush habitats, and increased fire frequency. Many, if not most, of these treatments have been applied to improve rangelands for domestic livestock but have had negative impacts on sagebrush communities and animals dependent on them (Braun and others 1976). Further, successive treatments have been applied to landscapes with little understanding of the cumulative effects that may impact both sagebrush-dependent animals, such as sage-grouse, and the overall health of the plant community. The impacts of natural events such as periodic drought are further exacerbated by human treatments of sagebrush communities. All of these issues emphasize the need for active protection of habitats presently used by sage-grouse as well as restoration of habitats that formerly supported sage-grouse populations.

Sage-Grouse Habitat Management Strategies

The objectives of habitat management to benefit sage-grouse, in order of importance, should be (1) to protect and maintain existing occupied habitats, (2) enhance existing occupied habitats, (3) restore degraded habitats that still receive some sage-grouse use, and (4) rehabilitate significantly altered habitats that no longer support sage-grouse. Strategies to accomplish these objectives should include:

- Vigorous suppression of wildfire.
- Reconsideration of any use of prescribed fire.
- Proper livestock management (including reconsideration of time of grazing, stocking rates, season of use, and frequency of use).
- Use of nitrogen fertilizer, except in areas infested by annual weeds.
- Mechanical chopping of sagebrush.
- Fence type and placement.
- Water management.
- Rehabilitation and restoration techniques discussed in these proceedings.

At times, manipulation of some occupied sage-grouse habitat may be necessary to enhance the overall quality of a seasonal range. An example would be removing or reducing some sagebrush canopy cover in known breeding habitat to enhance a depleted understory. Removal of 57 percent of sagebrush cover resulted in a significant decline in a sage-grouse breeding population (Connelly and others 2000b) and degradation of early brood-rearing habitat (Fischer and others 1996). More recently, a wildfire that removed about 30 percent of the sagebrush cover in a breeding habitat resulted in a 60 percent decline in sage-grouse nest success (Connelly, unpublished data, 1998). Because of this information and the fact that wildfires, drought, and insect infestations cannot be predicted, any sagebrush removal efforts should affect a relatively small portion of the occupied habitat. Connelly and others (2000a) suggested that >80 percent of breeding and winter habitat with vegetative characteristics necessary for productive sage-grouse habitat should remain intact to adequately provide for the needs of sage-grouse. However, an even greater percentage should be protected if sage-grouse populations are declining or the population status is unknown. All proposed habitat manipulations should carefully consider the current condition of habitat, status of the sage-grouse population, and likely outcome of the vegetation treatment, including recovery time necessary for the area to again provide adequate habitat for sage-grouse nesting and early brood rearing.

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References

Aldrich, J. W. 1963. Geographic orientation of American Tetraonidae. Journal of Wildlife Management. 27: 529–545.

American Ornithologists' Union. 1957. Check-list of North American birds. 5th ed. Baltimore, MD: Lord Baltimore Press. 691 p.

Apa, A. D. 1998. Habitat use and movements of sympatric sage and Columbian sharp-tailed grouse in southeastern Idaho. Moscow, ID: University of Idaho. 199 p. Dissertation.

Back, G. N.; Barrington, M. R.; McAdoo, J. K. 1987. Sage grouse use of snowburrows in northeastern Nevada. Wilson Bulletin. 99: 488–490.

Beck, T. D. I. 1977. Sage grouse flock characteristics and habitat selection in winter. Journal of Wildlife Management. 41: 18–26.

Beck, T. D. I.; Braun, C. E. 1978. Weights of Colorado sage grouse. Condor. 80: 241–243.

Braun, C. E. 1987. Current issues in sage grouse management. Proceedings of the Western Association of State Fish and Wildlife Agencies. 67: 134–144.

Braun, C. E. 1995. Distribution and status of sage grouse in Colorado. Prairie Naturalist. 27: 1–9.

Braun, C. E. 1998. Sage grouse declines in western North America: what are the problems? Proceedings of the Western Association of State Fish and Wildlife Agencies. 78: 139–156.

Braun, C. E.; Baker, M. F.; Eng, R. L.; Gashwiler, J. S.; Schroeder, M. H. 1976. Conservation Committee report on the effects of alteration of sagebrush communities on the associated avifauna. Wilson Bulletin. 88: 165–171.

Braun, C. E.; Britt, T.; Wallestad, R. O. 1977. Guidelines for maintenance of sage grouse habitats. Wildlife Society Bulletin. 5: 99–106.

Connelly, J. W. 1998. Unpublished data on file at: Idaho Department of Fish and Game, Pocatello, ID.

Connelly, J. W.; Braun, C. E. 1997. Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. Wildlife Biology. 3: 229–234.

Connelly, J. W.; Browers, H. W.; Gates, R. J. 1988. Seasonal movements of sage grouse in southeastern Idaho. Journal of Wildlife Management. 52: 116–122.

Connelly, J. W.; Reese, K. P.; Fischer, R. A.; Wakkinen, W. L. 2000b. Response of a sage grouse breeding population to fire in southeastern Idaho. Wildlife Society Bulletin. 28: 90–96.

Connelly, J. W.; Schroeder, M. A.; Sands, A. R.; Braun C. E. 2000a. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin. 28: 967–985.

Drut, M. S.; Crawford, J. A.; Gregg, M. A. 1994. Brood habitat use by sage grouse in Oregon. Great Basin Naturalist. 54: 170–176.

Dunn, P. O.; Braun, C. E. 1986. Summer habitat use by adult female and juvenile sage grouse. Journal of Wildlife Management. 50: 228–235.

Fischer, R. A.; Reese, K. P.; Connelly, J. W. 1996. An investigation on fire effects within xeric sage grouse brood habitat. Journal of Range Management. 49: 194–198.

Hupp, J. W.; Braun, C. E. 1989a. Endogenous reserves of adult male sage grouse during courtship. Condor. 91: 266–271.

Hupp, J. W.; Braun, C. E. 1989b. Topographic distribution of sage grouse foraging in winter. Journal of Wildlife Management. 53: 823–829.

Johnsgard, P. A. 1973. Grouse and quails of North America. Lincoln, NE: University of Nebraska Press. 553 p.

Klott, J. H.; Lindzey, F. G. 1990. Brood habitats of sympatric sage and sharp-tailed grouse in Wyoming. Journal of Wildlife Management. 54: 84–88.

Patterson, R. L. 1952. The sage grouse in Wyoming. Denver, CO: Sage Books, Inc. 341 p.

Remington, T. E.; Braun, C. E. 1985. Sage grouse food selection in winter, North Park, Colorado. Journal of Wildlife Management. 49: 1055–1061.

Wallestad, R. O.; Peterson, J. G.; Eng, R. L. 1975. Foods of adult sage grouse in central Montana. Journal of Wildlife Management. 39: 628–630.

Wambolt, C. L.; Walhof, K. S.; Frisina, M. R. 2001. Recovery of big sagebrush communities after burning in south-western Montana. Journal of Environmental Management. 61: 243–252.

Fernleaf biscuitroot

Rangeland Ecol Manage 64:431–438 | September 2011 | DOI: 10.2111/REM-D-09-00165.1

Fire and Invasive Plants Special Feature

Resistance to Invasion and Resilience to Fire in Desert Shrublands of North America

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Abstract

Settlement by Anglo-Americans in the desert shrublands of North America resulted in the introduction and subsequent invasion of multiple nonnative grass species. These invasions have altered presettlement fire regimes, resulted in conversion of native perennial shrublands to nonnative annual grasslands, and placed many native desert species at risk. Effective management of these ecosystems requires an understanding of their ecological resistance to invasion and resilience to fire. Resistance and resilience differ among the cold and hot desert shrublands of the Great Basin, Mojave, Sonoran, and Chihuahuan deserts in North America. These differences are largely determined by spatial and temporal patterns of productivity but also are affected by ecological memory, severity and frequency of disturbance, and feedbacks among invasive species and disturbance regimes. Strategies for preventing or managing invasive plant/fire regimes cycles in desert shrublands include: 1) conducting periodic resource assessments to evaluate the probability of establishment of an altered fire regime; 2) developing an understanding of ecological thresholds associate within invasion resistance and fire resilience that characterize transitions from desirable to undesirable fire regimes; and 3) prioritizing management activities based on resistance of areas to invasion and resilience to fire.

Resumen

Los asentamientos de Anglo-Americanos en los desiertos de matorrales de Norteamérica resultaron en la introducción y subsecuente invasión de varias especies de pastos no nativos. Estas invasiones, han alterado el régimen de fuego preestablecido, convirtiendo los matorrales de especies nativas en pastizales de gramíneas anuales inducidas y poniendo en riesgo varias especies desérticas nativas. El manejo efectivo de estos ecosistemas requiere de un entendimiento de la resistencia ecológica a la invasión y la resiliencia al fuego. La resistencia y resiliencia difieren entre los desiertos de matorral fríos y cálidos de Norteamérica tales como Great Basin, Mojave, Sonorense, y Chihuahuense. Estas diferencias son determinadas en gran medida por patrones espaciales y temporales de productividad pero también es afectado por la memoria ecológica, la severidad y frecuencia del disturbio y la retroalimentación entre las especies invasoras y el régimen de disturbio. Las estrategias para prevenir o manejar plantas invasoras/ciclos de régimen de fuego en los desiertos de matorral incluyen: 1) realizar evaluaciones periódicas de los recursos para evaluar la probabilidad de que se establezca un régimen de fuego alterado; 2) desarrollar un entendimiento de los umbrales ecológicos asociados entre la resistencia a la invasión y la resiliencia al fuego que caracteriza la transición entre regímenes de fuego deseables e indeseables; y 3) priorizar las actividades de manejo basadas en la resistencia de las áreas a la invasión y la resiliencia al fuego.

Key Words: Chihuahuan Desert, ecological resilience, ecological resistance, Great Basin Desert, Mojave Desert, Sonoran Desert

INTRODUCTION

Plant invasions and their interactions with fire regimes are recognized as threats to biodiversity and other natural resources worldwide (Brooks et al. 2004). In the desert regions of North America, invasive plants have altered fire regimes, which, in many cases, have resulted in large-scale conversions of native plant communities to invasive plant dominance (D'Antonio and Vitousek 1992; Brooks et al. 2004). These changes are affecting ecological processes including water cycles (Wilcox and Thurow 2006), nutrient dynamics (Evans et al. 2001), carbon budgets (Bradley et al. 2006), and regional albedos (Millennium Ecosystem Assessment 2005). Many of the native species associated with these desert ecosystems are at risk, and several are either listed or are being considered for listing under the Endangered Species Act (1973). Examples include the desert tortoise (*Gopherus agassizii*) and the sage grouse (*Centrocercus* spp.).

The concepts of ecological resistance and resilience are used increasingly to develop approaches for sustainable ecosystem management (Walker et al. 2004; Briske et al. 2008) and can provide useful insights into the factors influencing plant invasions and fire both within and among North American desert ecosystems. These concepts allow comparisons over a variety of spatial scales, and can be used to develop management approaches that are appropriate at scales ranging from landscapes (Walker et al. 2004) to ecological sites (Briske et al. 2008). In this paper, we discuss the concepts of resistance

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and resilience in relation to plant invasions and fire in the deserts of North America with a specific focus on resistance to invasions and resilience to fire. We provide examples of how plant invasions have altered fire regimes from both cold and hot desert shrublands and present management strategies designed to prevent or mitigate these changes.

RESISTANCE TO PLANT INVASIONS AND RESILIENCE TO FIRE

We define ecological resistance to plant invasion as a function of the biotic and abiotic factors and ecological processes in an ecosystem that limit the establishment and population growth of an invading species (D'Antonio and Thomsen 2004). We define ecological resilience to fire as the amount of disturbance that an ecosystem can withstand before changes in processes and structures occur that are of sufficient magnitude to result in new alternative states (Holling 1973; Gunderson 2000). Thresholds define the limits of natural variability within ecosystems and are crossed when they do not return to the original state via natural processes after disturbance and instead transition to new alternative states that are adjusted to the altered processes (Laycock 1991; Whisenant 1999). When thresholds to invasion or fire are crossed, active restoration involving invasive species control, native plant revegetation, and in some cases direct fire management are required to return ecosystems to their original states.

The structure and function of desert systems and, consequently, resistance to invasion and resilience to fire differ based on variations in underlying abiotic characteristics, especially the amount and timing of precipitation. The deserts of North America contain four major regions that vary in both the annual amount and seasonal distribution of precipitation (Fig. 1). Regions that receive lower amounts of precipitation have relatively lower net primary productivity and biomass. Those that receive a higher percentage of their annual precipitation during winter are dominated by woody perennials, whereas those that receive most of their precipitation in summer are dominated by perennial grasses. The resilience of desert ecosystems to disturbances like fire typically increases along gradients of increasing available resources (water and nutrients) and annual net primary productivity (Chambers et al. 2007; Wisdom and Chambers 2009). Greater resources and a higher level of productivity by functionally diverse native plant communities increase the capacity of the native community to regenerate following disturbance and to effectively compete with invaders. Thus, the most productive desert ecosystems in the Great Basin and Chihuahuan deserts (Fig. 1) tend to be most resilient to fire and resistant to plant invasions.

The structure and function of desert systems are determined not only by the amount and seasonality of precipitation but also by their inherent variability (Noy-Meir 1973). Variability in precipitation tends to increase as total precipitation decreases and is highest for desert ecosystems that receive the least precipitation (Ehleringer 1985). When biomass of extant vegetation is low (such as in deserts), it has limited capacity for utilizing soil resource increases during episodic periods when precipitation is high (Davis and Pelsor 2001). The "fluctuating resource hypothesis" predicts that resistance to invasion decreases when

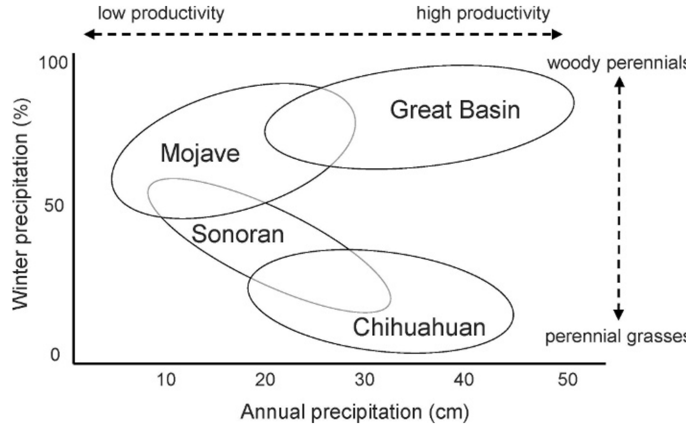

Figure 1. General precipitation patterns for the deserts of North America as they relate to average productivity and vegetation lifeform (adapted from MacMahon and Wagner 1985).

resource availability is higher than resource uptake, leaving resources for invading plants to utilize (Davis et al. 2000). Thus, ecosystems subject to pronounced fluctuations in resource supply may be more susceptible to invasion than systems with a more stable resource supply (Rejmanek 1989). The relationship between low precipitation and increased variability in precipitation is observed over elevation gradients within mountain ranges in the Great Basin cold desert and has been related to increased invasion potential at lower elevations (Chambers et al. 2007). An important caveat is that lower elevation hot desert areas with the most extreme environmental conditions may be relatively resistant to invasion because few nonnatives can establish and persist in these exceedingly harsh environments (Brooks 2009).

In desert areas with relatively low resistance to invasion, nonnative plants can severely compromise ecological resilience because of their effects on fuel characteristics, ignitability of landscapes, fire behavior, and, consequently, fire regimes (Brooks 2008). A fire regime is characterized by type (e.g., surface vs. crown fire), frequency (return interval), intensity (heat release), severity (effects on soils and/or vegetation), size, spatial complexity, and seasonality of fire within a given geographic area or vegetation type (Sugihara et al. 2006). "Presettlement" values are typically used as the baseline to determine if current fire regimes have been altered. Plant invasions that cause new fuel conditions and altered fire regimes can result in a self-perpetuating invasive plant/fire regime cycle (Fig. 2; Brooks et al. 2004). In addition, the invader may have direct negative effects on native vegetation through competition or other mechanisms that further promote the new fire regime. One of the most widely recognized examples of these types of changes is the grass/fire cycle in which invasive grasses invade native shrublands, increase fine fuels, and result in more frequent and larger fires than occurred prior to invasion (D'Antonio and Vitousek 1992). In the process, the landscape is converted from a native shrubland with a moderately long time between fires to nonnative grassland with very short periods of time between fires. It is also important to note that not all plant invasions increase the size, frequency, or intensity of fire. In some cases, invasive plants may expand into a landscape that has evolved with frequent fire, change fuel characteristics in ways that suppress burning, and alter historic fire regimes. Such is the case for native creosote bush that is expanding into hot desert

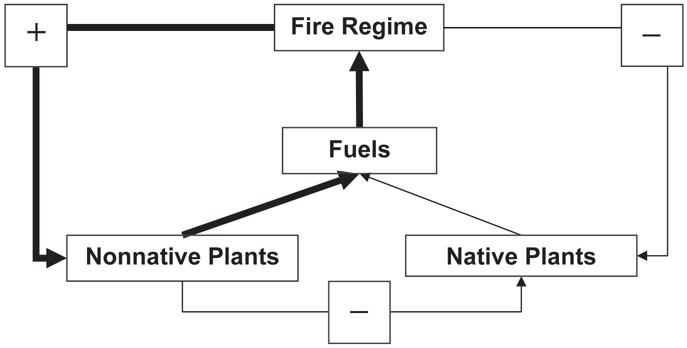

Figure 2. The invasive plant/fire regime cycle by which nonnative plants alter fire regimes through changes in fuel characteristics (reprinted with permission from Brooks 2008).

grasslands, changing the fuelbed to a more heterogenous distribution, impeding the spread of fire, and reducing fire frequency in the Chihuahuan Desert (Archer 1994; Archer et al. 1995).

Several interacting factors influence resistance to invasion and resilience to fire in desert ecosystems including ecological memory, severity and frequency of disturbance, and feedbacks among invasive species and disturbance regimes. Ecological memory consists of the legacies of information, materials, processes, and relationships that contribute to the continued functioning of an intact system and the recovery of that system following disturbance (Franklin and MacMahon 2000; Gunderson 2000; Peterson 2002; Bengtsson et al. 2003). A basic element of ecological memory is the capacity to support a given ecological site type as indicated by the climatic regime and soil characteristics. Factors that contribute to ecological memory include ecological condition as indicated by soil characteristics, the composition and abundance of residual native plants and animals, seed banks and seed sources, and the composition and abundance of invasive species. The severity and frequency of disturbance can alter the ecological memory of a site and, consequently, its capacity to support desirable alternative states and, in the worst case scenario, the historical ecological site type (Whisenant 1999; Briske et al. 2008). In the deserts of North America, inappropriate livestock grazing has significantly influenced ecological memory by reducing a major structural and functional component, specifically native perennial herbaceous species, and by serving as a dispersal agent for nonnative invaders (Milchunas et al. 1988; Van de Koppel et al. 2002). Loss of perennial herbaceous species decreases the resistance of desert ecosystems to invasion (Chambers et al. 2007) and resilience to disturbances like drought and wildfires (D'Antonio et al. 2009). Once established, invasive species can promote shorter fire return intervals and larger fire sizes than many deserts experienced historically. These changes can result in positive feedbacks for the invader and negative effects on native species, especially woody perennials (Fig. 2).

PRESETTLEMENT AND CURRENT FIRE REGIMES

The productivity and dominant life forms of the North American deserts affect fuels and fire behavior and, thus, the characteristics of both presettlement and current fire regimes. Desert ecosystems with relatively high productivity, like many middle to high elevation ecological types in cold desert shrublands, have relatively high fuel abundance and continuity, exhibited more frequent presettlement fires (Miller et al. 2011), and typically have many fire-tolerant species (Wright and Bailey 1982). Higher productivity coupled with the presence of fire-tolerant species result in greater resilience to fire. In contrast, in desert ecosystems with low productivity, including almost all hot desert shrubland ecological types and most lower elevation cold desert ecological types, fuel production and continuity are limited, presettlement fires were both infrequent and small (Humphrey 1974), and many of the species that characterize these ecological types evolved in the near-absence of fire and are fire intolerant (Wright and Bailey 1982; Brooks and Minnich 2006). Consequently, resilience to fire is typically low.

In the sections that follow, we describe presettlement and current fire regimes within cold and hot desert shrublands. We explain how invasive plants and land-use activities have altered fire regimes from presettlement conditions and discuss the implications of ecological resistance and resilience.

Cold Desert Shrublands

Cold desert shrublands dominated largely by woody plant communities typify the Great Basin desert of North America. Relative to other desert ecosystems, they are characterized by moderate to high productivity and relatively high precipitation that arrives primarily during the winter months (Fig. 1). However, cold desert shrubland types occur over elevation gradients that exhibit distinct differences in available resources and, consequently, in site productivity, vegetation composition, and fuel characteristics. Salt-desert shrublands are dominated by species in the Chenopodiaceae, occur typically on halomorphic soils, and are characterized by the lowest effective precipitation, productivity, and fuel loads (West 1983a). Wyoming big sagebrush (*Artemisia tridentata wyomingensis*), mountain big sagebrush (*Artemisia tridentata vaseyana*), and mountain brush types occur at progressively higher elevations and are associated with increasing amounts of precipitation, productivity, and fuels (West and Young 2000). During presettlement times, salt-desert shrublands rarely if ever burned due to inherently low productivity and fuels (Brooks and Pyke 2001). Sagebrush dominated shrublands had highly variable fire return intervals ranging from decades to centuries (Frost 1998; Brown and Smith 2000; Baker 2006; Miller et al. 2011). At coarse regional scales, fire return intervals in sagebrush were determined by climate and its effects on fuel abundance and continuity. Consequently, fire frequency was higher both in sagebrush types with greater productivity and during periods of increased precipitation (West 1983b; Mensing et al. 2006). At fine scales within sagebrush types, fire return intervals in sagebrush shrublands likely were determined by topographic and soil effects on productivity and fuels and also were highly variable (Miller and Heyerdahl 2008).

Anglo-American settlement of cold desert shrublands beginning in the mid 1800s initiated a series of changes in vegetation composition and structure that interacted with other global change processes to alter fire regimes across the cold desert region. The first major change occurred when overgrazing by livestock led to a decrease in native perennial grasses and forbs and effectively reduced the abundance of fine fuels in shrublands (Knapp 1996; Miller and Eddleman 2001). Decreased competition from perennial herbaceous species in combination with ongoing climate change and favorable conditions for woody species establishment at the turn of the century resulted in increased abundance of shrubs (primarily *Artemisia* species) and trees including juniper (*Juniperus occidentalis*, *Juniperus osteosperma*) and pinyon pine (*Pinus monophylla*) (Miller et al. 2011). The initial effect of these changes in fuel structure was a reduction in fire frequency and size (Miller and Tausch 2001). The second major change occurred when annual grasses (*Bromus tectorum, Bromus madritensis ssp. rubens, Taeniatherum caput-medusa*) were introduced from Eurasia in the late 1800s and spread rapidly into low to mid-elevation shrublands with depleted understories (Knapp 1996). The annual grasses increased fine fuels, and the rate of fire spread in many shrubland communities and initiated grass/fire cycles characterized by shortened fire return intervals and larger, more contiguous fires. In recent decades, salt-desert shrublands began to burn for the first time in known history, and Wyoming sagebrush types began burning as frequently as every few years (Whisenant 1990; Brooks and Pyke 2001). The final change occurred as a result of expansion of juniper and pinyon pine trees into mid- to high elevation shrublands. Progressive infilling of the trees is increasing woody fuels and causing fires of greater size and intensity (Miller and Tausch 2001). The highly competitive trees also are resulting in depletion of species associated with sagebrush shrublands and reduced resilience to fire.

Resilience of cold desert shrublands to fire increases along gradients of increasing available resources and annual net primary productivity (Chambers et al. 2007; Wisdom and Chambers 2009). Resistance to annual grasses is associated with their ecological amplitude and is lowest for lower-elevation salt-desert shrub and Wyoming sagebrush types and highest for mountain big sagebrush and mountain brush types (Wisdom and Chambers 2009). In contrast, resistance to woodland expansion is lowest for mountain big sagebrush and mountain brush types (Miller and Eddleman 2001). Factors that result in depletion of native perennial herbaceous species like overgrazing by livestock and infilling of pinyon and juniper trees decrease resistance to invasion by annual grasses and resilience following fire. In sagebrush shrublands, the removal of perennial herbaceous species can increase cheatgrass biomass and seed production two- to threefold, whereas fire alone can result in a two- to sixfold increase in these variables (Chambers et al. 2007). However, in these same shrublands, the removal of herbaceous perennials coupled with fire can cause 10- to 30-fold increases in biomass and seed production of cheatgrass (Chambers et al. 2007).

Hot Desert Shrublands

Hot desert shrublands characterize most of the Mojave and Sonoran deserts of North America. Precipitation in these deserts is relatively low and occurs largely during the winter months (Fig. 1). Native vegetation types exhibit generally low productivity and fuel levels. However, similar to cold desert shrublands, elevation gradients and local edaphic conditions influence productivity and, thus, fuel loads and continuity (Brooks and Matchett 2006; Brooks and McPherson 2008). Low elevations are dominated by creosotebush (*Larrea tridentata*) scrub, while middle elevations generally are characterized by blackbrush (*Coleogyne ramosissima*) scrub (Brooks and Minnich 2006). Higher elevations are characterized by chaparral ecological types that are dominated by woody evergreen shrubs with dense crowns like buckbrush (*Ceonothus* spp.) and manzanita (*Arctostaphylos* spp.) or by cold desert types that are dominated by big sagebrush, juniper, and pinyon pine (Brooks and Minnich 2006).

In low elevation shrublands of the hot deserts, presettlement fires were infrequent. Fine fuels were derived primarily from winter annuals and were sparse except after very wet winters (Brown and Minnich 1986; Brooks and Esque 2002; Esque and Schwalbe 2002; Salo 2005; Brooks and Minnich 2006). In both the Mojave and western Sonoran deserts, invasion of nonnative annual grasses (*B. madritensis* subsp. *rubens* and *Schismus barbatus*) significantly increased fine fuel loads in creosotebush scrub (Rogers and Vint 1987; Brooks and Minnich 2006) and created conditions conducive to fire spread (Brooks 1999). Between 1955 and 1983, fire frequency increased in the Sonoran Desert (Schmid and Rogers 1988), and during the 1980s and early 1990s, fire frequency increased in the Mojave Desert (Brooks and Esque 2002; Brooks and Matchett 2006). High rainfall years result in significant increases in nonnative annual grass biomass (fine fuels) and can result in large fires (Rogers and Vint 1987; Schmid and Rogers 1988; Brooks and Matchett 2006; Brooks and Minnich 2006). In creosotebush scrub of the eastern Sonoran Desert, invasion of nonnative perennial grasses such as Lehmann lovegrass (*Eragrostis lehmanniana*), buffelgrass (*Pennisetum ciliare*), and purple fountaingrass (*Pennisetum setaceum*) have resulted in similar increases in fire frequency and size (Brooks and McPherson 2008).

In middle elevation shrublands characterized by blackbrush, presettlement fire return intervals appear to have been on the order of centuries (Webb et al. 1987). Low amounts of fine fuels in interspaces likely limited fire spread except during extreme fire weather conditions (high winds, low relative humidity, and low fuel moisture) when stand-replacing crown fires could occur. After settlement, extensive burning to remove blackbrush for range improvement coupled with livestock grazing contributed to invasion of nonnative brome grasses (*B. tectorum* and *B. madritensis* subsp. *rubens*) and red-stemmed filaree (*Erodium cicutarium*; Brooks and Matchett 2003; Brooks and McPherson 2008). The nonnative annual grasses and forbs increased fine fuels, and during high precipitation years greater production of these fine fuels is correlated with larger fires (Brooks and Matchett 2006). An increase in nonnative grass abundance after fire has the potential to promote recurrent fire and decrease resilience in blackbrush types.

In high elevation shrublands, woody fuels and fuel continuity are typically higher than in the middle elevation zone. Greater fuel loads and continuity coupled with more frequent lightning and steeper slopes that promote fire spread resulted in historical fire return intervals of 50 to 100+ yr, although local fire return intervals probably varied widely (Cable 1975; Brooks and Minnich 2006). In the Sonoran Desert, many lower elevation chaparral sites have been managed for livestock grazing since the 1880s (Pase and Brown 1994), and the use of fire to maintain grass dominance likely limited chaparral encroachment into lower elevation grasslands (Brooks and McPherson 2008). However, in areas where fire was not used as a management tool, removal of fine fuels by livestock grazing and fire suppression likely decreased fire frequency and resulted in chaparral expansion into these areas (Brooks and McPherson 2008). Higher elevation sites likely did not receive as much grazing pressure, but fire suppression, especially at the interface with ponderosa pine forests, may have resulted in tree expansion into shrublands. Nonnative annual grasses occur in high elevation shrublands and often increase in abundance immediately after fire. However, in most cases native woody vegetation quickly recovers, overtops, and suppresses annual grasses within the first decade unless some other disturbance factor is present such as recurrent fire or significant ungulate grazing (M. Brooks, personal observation, 2006).

Similar to cold desert shrublands, resilience of hot desert shrublands to fire tends to increase with elevation and productivity gradients. More mesic conditions at higher elevations result in greater vegetation production and, historically, these areas had more frequent fires. Many of the species that occur in higher elevation ecosystems evolved with more frequent fire and have higher tolerance to fire than those that occur at more arid lower elevations (Brooks and Minnich 2006). Higher elevation ecosystems also appear to be more resistant to dominance by annual grasses. This is largely a function of greater resilience to fire and a higher probability of recovery of these ecosystems to native species dominance following disturbance. Consequently, annual grass invasions have their greatest influence on fire regimes in low- to mid-elevation shrublands where they increase fuel continuity and repeated fires decrease the recovery potential of native species with low fire tolerances.

MANAGEMENT TOOLS TO PREVENT THE INVASIVE PLANT/FIRE REGIME CYCLE

A core objective for managing invasive plants and fire regimes in desert ecosystems is maintaining or increasing ecological resistance to plant invasions and resilience to fire prior to threshold crossings and the initiation of an invasive plant/fire regime cycle (D'Antonio and Chambers 2006; D'Antonio et al. 2009). Once a threshold has been crossed it is often ecologically and economically difficult, if not impossible, to return the system to its original state.

Managing for resistance to invasives and resilience to fire requires obtaining the necessary information for prioritizing restoration and other management activities and long-term monitoring data for adaptive management. In the sections below we describe three guiding principals that should be incorporated into management plans in order to prevent or minimize the invasive plant/fire regime cycle.

Conduct Resource Assessments and Periodic Monitoring

The first step is to assess the vegetation types and current ecological conditions, ideally the ecological types and their states and phases at landscape scales. This information should be obtained using consistent methods, and geographic information systems databases should be developed that are widely accessible (Chambers et al. 2009). This type of information provides the basis for determining priority management areas and appropriate management activities at scales that allow the preservation of ecosystems and conservation of species (Wisdom and Chambers 2009). It also provides the basis for monitoring the rate and magnitude of invasion, changes in fire return intervals, and effects of wildfire management activities on ecosystems and species.

Develop an Understanding of Ecological Resistance and Resilience

Managing for resistance to invasions and resilience to fire requires both developing an understanding of the abiotic and biotic factors that determine ecological resistance to invasives and resilience to fire and defining the ecological thresholds that exist in desert ecosystems. Both monitoring data and research and management experiments can be used to determine the abiotic and biotic conditions that influence resistance to invasion and resilience to fire and that result in threshold crossings. For example, the Joint Fire Science Program, Sagebrush Treatment Evaluation Project (www.sagestep.org) is using a collaborative research and management approach that spans the Great Basin Desert to define the ecological conditions (soils, vegetation composition, and structure) that influence resistance and resilience and that result in threshold crossings in sagebrush steppe ecosystems exhibiting annual grass invasion and pinyon and juniper tree expansion. Management treatments are applied over a gradient of ecological conditions (e.g., increasing tree cover, increasing annual grass cover, and decreasing herbaceous perennial cover) and the responses are quantified to determine the point at which ecological resilience is lost and a threshold is crossed to an alternative state. Once an appropriate set of metrics has been defined for evaluating resistance to invasives and resilience to disturbance, they can be incorporated into existing state and transition models (e.g., Briske et al. 2008). Providing for unanticipated states or phases will be necessary to accommodate changes due to climate change and alterations in land use.

Prioritize Management Activities Based on Resistance and Resilience

Determining priority management areas and appropriate management activities using an understanding of resistance to invasions and resilience to fire allows a strategic approach that can be used to address multiple objectives over larger scales. Using this approach, areas with a high priority for protection are those with inherently low invasion resistance and fire resilience, like many hot desert shrub communities and lower elevation cold desert shrub communities. Areas of high conservation value for threatened and endangered species, like the Snake River Birds of Prey Area in Idaho, also receive priority status for protection. Protection focuses on eliminating or reducing stressors such as repeated fire and inappropriate livestock grazing, controlling surface disturbances and invasion corridors like roads and trails, and increasing efforts to detect and eradicate invasive species.

Maintaining or increasing resistance to invasion and resilience to fire in areas that have declining ecological conditions or that are in the initial stages of invasion but that have not crossed ecological thresholds also receives high priority. Eliminating or reducing stressors and factors that increase invasion is still a primary focus. In addition, preventative vegetation management is used in areas that receive greater amounts of effective moisture, are characterized by inherently higher invasion resistance and fire resilience, and have a high probability of improved ecological conditions following treatment. Preventative management can be a viable approach in desert shrublands with reduced native herbaceous perennials and increased shrubs or trees (D'Antonio et al. 2009) and hot desert grasslands with encroaching shrub species (Drewa et al. 2001). Management objectives typically include increasing native perennial grass and forb dominance through competitive release from shrubs and trees, and reducing woody fuel loads to minimize the risk of high severity fires. Treatments are specific to the ecosystem and ecological conditions, but typically involve prescribed fire, mechanical shrub and tree thinnings, or herbicides. After wildfires, seeding with native species through the US Department of the Interior Bureau of Land Management, Emergency Stabilization and Rehabilitation program, and the US Department of Agriculture Forest Service Burn Area Emergency Response program may increase ecological resistance to invasives and resilience to fire in areas at risk of crossing an ecological threshold. Unfortunately, many areas where seedings successfully establish are on the high end of the regional productivity gradient, are naturally more resilient to fire, and do not need active management. In addition, seeding with introduced grasses in fire resilient areas that are capable of recovering on their own to native species can result in alternative stable states with altered ecological processes and reduced species diversity (Lesica and DeLuca 1996; Richards et al. 1998). Thus, decisions to apply postfire seeding treatments should be based on careful evaluation of a site's inherent resistance to invasion and resilience to fire in order to prevent unnecessary treatments and avoid undesirable effects.

Restoring or rehabilitating areas that have already crossed ecological thresholds to states that are dominated by invasive species is ecologically challenging and expensive and is of lower priority except in special situations. Lower priority status is assigned to these areas not because they are not valuable to society but because the magnitude of the problem relative to available human and financial resources indicates that greater benefit will be obtained by maintaining or increasing the invasive resistance and fire resilience of areas that have not yet crossed ecological thresholds. Areas that may be assigned priority status for restoration or rehabilitation include those that are located adjacent to intact vegetation communities that can serve as buffers or fire breaks, occur at the wildland–urban interface, represent endangered ecosystems, or provide critical habitat for threatened and endangered species. Restoration or rehabilitation of these areas typically involves integrated management strategies in which pretreatments are used to reduce the propagule pool or adult population of the invader followed by revegetation to establish the desired plant community (Brooks et al. 2004; D'Antonio and Chambers 2006; D'Antonio et al. 2009). The choice of seeded species depends on the management objective. Restoration of critical habitat or endangered ecosystems by definition requires diverse native species mixtures. In contrast, management objectives for wildland–urban interface areas and buffers or fire breaks may include high resistance to the invader and fuel characteristics that minimize the likelihood of fire (Brooks et al. 2004; Brooks 2009). In this case, it is appropriate to rehabilitate the area with native or introduced species that are highly competitive with invaders and have low flammability but that are not likely to become significant land management problems. Regardless of the objective, it is necessary to monitor the success of restoration and rehabilitation efforts and plan for the possibility of reseeding and repeated removal of the invader.

MANAGEMENT IMPLICATIONS

Land managers often have limited financial and human resources and are faced with managing a wide range of natural, recreational, and economic resources that can be negatively affected by multiple threats. The effectiveness of land management can be improved by using ecological concepts that transcend individual resources and threats, distill interacting factors into a subset of manageable parts, and can be applied at a variety of scales. The concepts of ecological resistance to invasion and resilience to fire exhibit these properties and can be used to manage the interrelated threats of plant invasions and altered fire regimes in the deserts of North America.

In this paper, we explain how resistance to invasion and resilience to fire differ both within and among the desert shrublands of North America. An understanding of the abiotic and biotic factors and the processes that determine invasion resistance and fire resilience in these desert shrublands provides critical information for management. Specifically, when and where plant invasions are most likely to occur, the ecological and environmental conditions that confer resistance to invasions and/or resilience to fire, and, conversely, the conditions that result in threshold crossings. This information can be used to:

- Prioritize land management activities at landscape scales in order to restore and maintain ecosystems and to meet conservation objectives (Wisdom and Chambers 2009).
- Develop ecological site descriptions based on ecological resilience that incorporate process-based indicators and describe triggers, feedback mechanisms, and restoration pathways (Briske et al. 2008).
- Develop invasive species management plans that are specific to the ecosystems of interest and that are based on abiotic and biotic factors and ecological processes that influence ecological resistance to plant invasions.
- Develop fire management plans that are specific to the ecosystems of interest and that are based on abiotic and biotic factors and ecological processes that influence ecological resilience to fire.

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LITERATURE CITED

Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns and proximate causes. *In:* M. Vavra, W. Laycock, and R. Pieper [eds.]. Ecological implications of livestock herbivory in the West. Denver, CO, USA: Society for Range Management. p. 13–68.

Archer, S., D. S. Schimel, and E. A. Holland. 1995. Mechanisms of shrubland expansion: land use, climate or CO_2? *Climatic Change* 29:91–99.

Baker, W. L. 2006. Fire and restoration of sagebrush ecosystems. *Wildlife Society Bulletin* 34:177–185.

Bengtsson, J., P. Angelstam, T. Elmqvist, U. Emanuelsson, C. Folke, M. Ihse, F. Moberg, and M. Nystrom. 2003. Reserves, resilience and dynamic landscapes. *Ambio* 32:389–396.

Bradley, B. A., R. A. Houghton, and J. F. Mustard. 2006. Invasive grass reduces aboveground carbon stocks in shrublands of the Western US. *Global Climate Change* 12:1815–1822.

Briske, D. D., B. T. Bestelmeyer, T. K. Stringham, and P. L. Shaver. 2008. Recommendations for development of resilience based state and transition models. *Rangeland Ecology & Management* 61:359–367.

Brooks, M. L. 1999. Alien annual grasses and fire in the Mojave Desert. *Madroño* 46:13–19.

Brooks, M. L. 2008. Plant invasions and fire regimes. *In:* K. Zouhar, J. Kapler Smith, S. Sutherland, and M. L. Brooks [eds.]. Wildland fire in ecosystems: fire and nonnative invasive plants. RMRS-GTR-42-volume 6. Ogden, UT, USA: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. p. 33–46.

Brooks, M. L. 2009. Spatial and temporal distribution of non-native plants in upland areas of the Mojave Desert. *In:* R. H. Webb, L. F. Fenstermaker, J. S. Heaton, D. L. Hughson, E. V. McDonald, and D. M. Miller [eds.]. The Mojave Desert: ecosystem processes and sustainability. Reno, NV, USA: University of Nevada Press. p. 101–124.

Brooks, M. L., C. M. D'Antonio, D. M. Richardson, J. B. Grace, J. E. Keeley, J. M. DiTomaso, R. J. Hobbs, M. Pellant, and D. Pyke. 2004. Effects of invasive alien plants on fire regimes. *BioScience* 54:677–688.

Brooks, M. L., and T. C. Esque. 2002. Alien annual plants and wildfire in desert tortoise habitat: status, ecological effects, and management. *Chelonian Conservation and Biology* 4:330–340.

Brooks, M. L., and J. R. Matchett. 2003. Plant community patterns in unburned and burned blackbrush (*Coleogyne ramosissima*) shrublands in the Mojave Desert. *Western North American Naturalist* 63:283–298.

Brooks, M. L., and J. R. Matchett. 2006. Spatial and temporal patterns of wildfires in the Mojave Desert, 1980–2004. *Journal of Arid Environments* 67:148–164.

Brooks, M. L., and G. McPherson. 2008. Ecological role of fire and causes and ecological effects of altered fire regimes in the southwest. *In:* [anonymous]. Proceedings from the Southwest Region Threatened, Endangered, and At-Risk Species Workshop; 22–25 October 2007; Tucson, AZ, USA. Reston, VA, USA: HydroGeoLogic. 8–1 through 8–3.

Brooks, M. L., and R. A. Minnich. 2006. Southeastern deserts bioregion. *In:* N. G. Sugihara, J. W. van Wagtendonk, K. E. Shaffer, J. Fites-Kaufman, and A. E. Thode [eds.]. Fire in California's ecosystems. Berkeley, CA, USA: The University of California Press. p. 391–414.

Brooks, M. L., and D. A. Pyke. 2001. Invasive plants and fire in the deserts of North America. *In:* K. E. M. Galley and T. P. Wilson [eds.]. Proceedings of the Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species. Fire Conference 2000: the First National Congress on Fire Ecology, Prevention, and Management; 27 November–1 December 2000; San Diego, CA, USA. Miscellaneous Publication No. 11. Tallahassee, FL, USA: Tall Timbers Research Station. p. 1–14.

Brown, D. E., and R. A. Minnich. 1986. Fire and creosote bush scrub of the western Sonoran Desert, California. *American Midland Naturalist* 116:400–422.

Brown, J. K., and J. K. Smith. 2000. Wildland fire in ecosystems: effects of fire on flora. Ogden, UT, USA: USDA Forest Service, Rocky Mountain Research Station. General Technical Report RMRS, GTR-42, volume 2. 257 p.

Cable, D. R. 1975. Range management in the chaparral type and its ecological basis: the status of our knowledge. Fort Collins, CO, USA: US Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. Research Paper RM-155. 30 p.

Chambers, J., E. Leger, and E. Goergen. 2009. Cold desert fire and invasive species management: resources, strategies, tactics and responses. *Rangelands* 31:14–20.

Chambers, J. C., B. A. Roundy, R. R. Blank, S. E. Meyer, and A. Whittaker. 2007. What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? *Ecological Monographs* 77:117–145.

D'Antonio, C., and J. C. Chambers. 2006. Using ecological theory to manage or restore ecosystems affected by invasive plant species. *In:* D. Falk, M. Palmer, and J. Zedler [eds.]. Foundations of restoration ecology. Covelo, CA, USA: Island Press. p. 260–279.

D'Antonio, C. M., J. C. Chambers, R. Loh, and J. T. Tunison. 2009. Applying ecological concepts to the management of widespread grass invasions. *In:* R. L. Inderjit [ed.]. Ecological invasions and restoration. Dordrecht, The Netherlands: Springer. p. 123–149.

D'Antonio, C. M., and M. Thomsen. 2004. Ecological resistance in theory and practice. *Weed Technology* 18:1572–1577.

D'Antonio, C. M., and P. M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics* 23:63–87.

Davis, M. A., J. P. Grime, and K. Thompson. 2000. Fluctuating resources in plant communities: a general theory of invasibility. *Journal of Ecology* 88:528–534.

Davis, M. A., and M. Pelsor. 2001. Experimental support for a mechanistic resource-based model of invasibility. *Ecology Letters* 4:421–428.

Drewa, P. B., P. C. Peters, and K. M. Havstad. 2001. Fire, grazing, and honey mesquite invasion in black grama-dominated grasslands of the Chihuahuan Desert: a synthesis. *In:* K. E. M. Galley and T. P. Wilson [eds.]. Proceedings of the Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species. Fire Conference 2000: the First National Congress on Fire Ecology, Prevention, and Management; 27 November–1 December 2000; San Diego, CA, USA. Miscellaneous Publication No. 11. Tallahassee, FL, USA: Tall Timbers Research Station. p. 31–39.

Ehleringer, J. 1985. Annuals and perennial of warm deserts. *In:* B. F. Chabot and H. A. Mooney [eds.]. Physiological ecology of North American plant communities. New York, NY, USA: Chapman and Hall. p. 162–180.

Esque, T. C., and C. R. Schwalbe. 2002. Non-native annual plants and their relationships to fire and biotic change in Sonoran Desertscrub. *In:* B. Tellman [ed.]. Invasive exotic species in the Sonoran region. Tucson, AZ, USA: Arizona-Sonora Desert Museum and The University of Arizona Press. p. 165–194.

Evans, R. D., R. Rimer, L. Sperry, and J. Belnap. 2001. Exotic plant invasion alters nitrogen dynamics in an arid grassland. *Ecological Applications* 11:1301–1310.

Franklin, J. F., and J. A. MacMahon. 2000. Messages from a mountain. *Science* 288:1183–1184.

Frost, C. C. 1998. Presettlement fire frequency regimes of the United States: a first approximation. *In:* T. T. Pruden and L. A. Brennan [eds.]. Fire in ecosystem management: shifting the paradigm from suppression to prescription. Proceedings 20th Tall Timbers Fire Ecology Conference. Tallahassee, FL, USA: Tall Timbers Research Station. p. 70–82.

Gunderson, L. H. 2000. Ecological resilience in theory and practice. *Annual Review of Ecology and Systematics* 31:425–439.

Holling, C. S. 1973. Resilience and stability in ecological systems. *Annual Review of Ecology and Systematics* 4:1–23.

Humphrey, R. R. 1974. Fire in deserts and desert grassland of North America. *In:* T. T. Kozlowski and C. E. Ahlgren [eds.]. Fire and ecosystems. New York, NY, USA: Academic Press. p. 365–401.

Knapp, P. A. 1996. Cheatgrass (*Bromus tectorum*) dominance in the Great Basin Desert. *Global Environmental Change* 6:37–52.

Laycock, W. A. 1991. Stable states and thresholds of range condition on North American rangelands: a viewpoint. *Journal of Range Management* 44:427–433.

Lesica, P., and T. DeLuca. 1996. Long-term harmful effects of crested wheatgrass on Great Plains grassland ecosystems. *Journal of Soil and Water Conservation* 51:408–409.

MacMahon, J. A., and F. H. Wagner. 1985. The Mojave, Sonoran, and Chihuahuan deserts of North America. *In:* M. Evanari, I. Noy-Meir, and D. W. Goodall. [eds.] Hot deserts and arid shrublands. Amsterdam: Elsevier. p. 105–202.

MENSING, S., S. LIVINGSTON, AND P. BARKER. 2006. Long-term fire history in Great Basin sagebrush reconstructed from macroscopic charcoal in spring sediments, Newark Valley, Nevada. *Western North American Naturalist* 66:64–77.

MILCHUNAS, D. G., O. E. SALA, AND W. K. LAUENROTH. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. *American Naturalist* 132:87–106.

MILLENNIUM ECOSYSTEM ASSESSMENT. 2005. Sustainable rangeland roundtable. Available at: http://www.millenniumassessment.org/en/index.aspx. Accessed 17 June 2010.

MILLER, R. F., S. T. KNICK, D. A. PYKE, C. W. MEINKE, S. E. HANSER, M. J. WISDOM, AND A. L. HILD. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. *In:* S. T. Knick and J. W. Connelly [EDS.]. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology. Vol. 38. Berkeley, CA, USA: University of California Press. p. 145–184.

MILLER, R. F., AND L. L. EDDLEMAN. 2001. Spatial and temporal changes of sage grouse habitat in the sagebrush biome. Corvallis, OR, USA: Oregon State University, Agricultural Experiment Station Bulletin 151. 35 p.

MILLER, R. F., AND E. K. HEYERDAHL. 2008. Fine-scale variation of historical fire regimes in sagebrush-steppe and juniper woodlands: an example from California, USA. *International Journal of Wildland Fire* 17:245–254.

MILLER, R. F., AND R. J. TAUSCH. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. *In:* K. Galley and T. Wilson [EDS.]. Fire Conference 2000: the First National Congress on Fire Ecology, Prevention and Management; 27 November–1 December 2000; San Diego, CA, USA. Tallahassee, FL, USA: Tall Timbers Research Station. p. 15–30.

NOY-MEIR, I. 1973. Desert ecosystems: environment and producers. *Annual Review of Ecology and Systematics* 4:25–52.

PASE, C. P., AND D. E. BROWN. 1994. Interior chaparral. *In:* D. E. Brown [ED.]. Biotic communities: southwestern United States and northwestern Mexico. Salt Lake City, UT, USA: University of Utah Press. p. 95–99.

PETERSON, G. D. 2002. Contagious disturbance, ecological memory, and the emergence of landscape pattern. *Ecosystems* 5:329–338.

REJMANEK, M. 1989. Invasibility of plant communities. *In:* J. A. Drake, F. Di Castri, R. H. Groves, F. J. Kruger, H. A. Mooney, M. Rejmanek, and M. H. Williamson [EDS.]. Ecology of biological invasion: a global perspective. New York, NY, USA: Wiley & Sons. p. 369–388.

RICHARDS, R. T., J. C. CHAMBERS, AND C. ROSS. 1998. Use of native plants on federal lands: policy and practice. *Journal of Range Management* 51:625–632.

ROGERS, G. F., AND M. K. VINT. 1987. Winter precipitation and fire in the Sonoran Desert. *Journal of Arid Environments* 13:47–52.

SALO, L. F. 2005. Red brome (*Bromus rubens* subsp. *madritensis*) in North America: possible modes for early introductions, subsequent spread. *Biological Invasions* 7:165–180.

SCHMID, M. K., AND G. F. ROGERS. 1988. Trends in fire occurrence in the Arizona upland subdivision of the Sonoran Desert, 1955 to 1983. *The Southwestern Naturalist* 33:437–444.

SUGIHARA, N. G., J. W. VAN WAGTENDONK, AND J. FITES-KAUFMAN. 2006. Fire as an ecological process. *In:* N. G. Sugihara, J. W. van Wagtendonk, K. E. Shaffer, J. A. Fites-Kaufman, and A. E. Thode [EDS.]. Fire in California's ecosystems. Berkeley, CA, USA: University of California Press. p. 38–74.

VAN DE KOPPEL, J., M. REITKERK, F. V. LANGEVELDE, L. KUMAR, C. A. KLAUSMIER, J. M. FRYXELL, J. W. HEARNE, J. V. ANDEL, N. D. RIDDER, A. SKIDMORE, L. STROOSNIJDER, AND H. T. PRINS. 2002. Spatial heterogeneity and irreversible vegetation change in semiarid grazing systems. *American Naturalist* 159:209–218.

WALKER, B. C., S. HOLLING, S. R. CARPENTER, AND A. KINZIG. 2004. Perspective: resilience, adaptability, and transformability in social-ecological systems. *Ecology and Society* 9(2):5. Available at: http://www.ecologyandsociety.org/vol9/iss2/art5.

WEBB, R. H., J. W. STEIGER, AND R. M. TURNER. 1987. Dynamics of Mojave Desert assemblages in the Panamint Mountains, California. *Ecology* 68:478–490.

WEST, N. E. 1983a. Intermountain salt-desert shrubland. *In:* N. E. West [ED.]. Temperate deserts and semi-deserts. Amsterdam, The Netherlands: Elsevier Publishing Company. p. 375–378.

WEST, N. E. 1983b. Great Basin-Colorado Plateau sagebrush semi-desert. *In:* N. E. West [ED.]. Temperate deserts and semi-deserts. Amsterdam, The Netherlands: Elsevier Publishing Company. p. 331–350.

WEST, N. E., AND J. A. YOUNG. 2000. Intermountain valleys and lower mountain slopes. *In:* M. B. Barbour and W. D. Billings [EDS.]. North American terrestrial vegetation. Cambridge, United Kingdom: Cambridge University Press. p. 256–284.

WHISENANT, S. G. 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. *In:* E. D. McArthur, E. M. Romney, S. D. Smith, and P. T. Tueller [EDS.]. Proceedings: Symposium on Cheatgrass Invasion, Shrub Die-Off and Other Aspects of Shrub Biology and Management 1989; Las Vegas, NV, USA. USDA General Technical Report INT-276. Ogden, UT, USA: Intermountain Research Station. p. 4–10.

WHISENANT, S. G. 1999. Restoring damaged wildlands: a process-oriented, landscape-scale approach. Cambridge, United Kingdom: Cambridge University Press. 312 p.

WILCOX, B. P., AND T. L. THUROW. 2006. Emerging issues in rangeland ecohydrology: vegetation change and the water cycle. *Rangeland Ecology & Management* 59:220–224.

WISDOM, M. J., AND J. C. CHAMBERS. 2009. Concepts for ecologically-based management of Great Basin shrublands. *Restoration Ecology* 17:740–749.

WRIGHT, H. A., AND A. W. BAILEY. 1982. Fire ecology, United States and southern Canada. New York, NY, USA: John Wiley & Sons. 528 p.

Guidelines to manage sage grouse populations and their habitats

John W. Connelly, Michael A. Schroeder, Alan R. Sands, and Clait E. Braun

Abstract The status of sage grouse populations and habitats has been a concern to sportsmen and biologists for >80 years. Despite management and research efforts that date to the 1930s, breeding populations of this species have declined throughout much of its range. In May 1999, the western sage grouse (C. urophasianus phaios) in Washington was petitioned for listing under the Endangered Species Act because of population and habitat declines (C. Warren, United States Fish and Wildlife Service, personal communication). Sage grouse populations are allied closely with sagebrush (Artemisia spp.). Despite the well-known importance of this habitat to sage grouse and other sagebrush obligates, the quality and quantity of sagebrush habitats have declined for at least the last 50 years. Braun et al. (1977) provided guidelines for maintenance of sage grouse habitats. Since publication of those guidelines, much more information has been obtained on sage grouse. Because of continued concern about sage grouse and their habitats and a significant amount of new information, the Western States Sage and Columbian Sharp-tailed Grouse Technical Committee, under the direction of the Western Association of Fish and Wildlife Agencies, requested a revision and expansion of the guidelines originally published by Braun et al. (1977). This paper summarizes the current knowledge of the ecology of sage grouse and, based on this information, provides guidelines to manage sage grouse populations and their habitats.

Key words Artemisia, Centrocercus urophasianus, guidelines, habitat, management, populations, sage grouse, sagebrush

The status of sage grouse populations and habitats has been a concern to sportsmen and biologists for >80 years (Hornaday 1916, Patterson 1952, Autenrieth 1981). Despite management and research efforts that date to the 1930s (Girard 1937), breeding populations of this species have declined by at least 17–47% throughout much of its range (Connelly and Braun 1997). In May 1999, the western sage grouse (*C. urophasianus phaios*) in Washington was petitioned for listing under the Endangered Species Act because of population and habitat declines (C. Warren, United States Fish and Wildlife Service, personal communication).

Sage grouse populations are allied closely with sagebrush (*Artemisia* spp.) habitats (Patterson 1952, Braun et al. 1977, Braun 1987). The dependence of sage grouse on sagebrush for winter habitat has been well documented (Eng and Schladweiler 1972, Beck 1975, Beck 1977, Robertson 1991). Similarly, the relationship between sagebrush

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Sage grouse on a nest with good shrub and herbaceous cover. The nest was successful.

habitats and sage grouse nest success has been described thoroughly (Klebenow 1969, Wallestad and Pyrah 1974, Wakkinen 1990, Connelly et al. 1991, Gregg et al. 1994). Despite the well-known importance of this habitat to sage grouse and other sagebrush obligates (Braun et al. 1976, Saab and Rich 1997), the quality and quantity of sagebrush habitats have declined for at least the last 50 years (Braun et al. 1976, Braun 1987, Swenson et al. 1987, Connelly and Braun 1997).

Braun et al. (1977) provided guidelines for maintenance of sage grouse habitats. Since publication of those guidelines, much more information has been obtained on relative size of sagebrush habitats used by these grouse (Connelly 1982, Connelly et al. 1988, Wakkinen et al. 1992), seasonal use of sagebrush habitats (Benson et al. 1991, Connelly et al. 1991), effects of insecticides on sage grouse (Blus et al. 1989), importance of herbaceous cover in breeding habitat (Wakkinen 1990, Connelly et al. 1991, Gregg 1991, Barnett and Crawford 1994, Drut et al. 1994*a*, Gregg et al. 1994), and effects of fire on their habitat (Hulet 1983; Benson et al. 1991; Robertson 1991; Fischer 1994; Fischer et al. 1996*a*, 1997; Pyle and Crawford 1996; Connelly et al. 2000*b*). Because of continued concern about sage grouse and their habitats and a significant amount of new information, the Western States Sage and Columbian Sharp-tailed Grouse Technical Committee, under the direction of the Western Association of Fish and Wildlife Agencies, requested a revision and expansion of the guidelines originally published by Braun et al. (1977). This paper summarizes the current knowledge of the ecology of sage grouse and, based on this information, provides guidelines to manage sage grouse populations and their habitats.

Population biology

Seasonal movements and home range

Sage grouse display a variety of annual migratory patterns (Beck 1975, Wallestad 1975, Hulet 1983, Berry and Eng 1985, Connelly et al. 1988, Wakkinen 1990, Fischer 1994). Populations may have: 1) distinct winter, breeding, and summer areas; 2) distinct summer areas and integrated winter and breeding areas; 3) distinct winter areas and integrated breeding and summer areas; or 4) well-integrated seasonal habitats (nonmigratory populations). Seasonal movements between distinct seasonal ranges may exceed 75 km (Dalke et al. 1963, Connelly et al. 1988), which complicates attempts to define populations. Thus, Connelly et al. (1988) suggested that sage grouse populations be defined on a temporal and geographic basis. Because of differences in seasonal movements among populations (Dalke et al. 1963, Wallestad 1975, Connelly et al. 1988, Wakkinen 1990), 3 types of sage grouse populations can

Sage grouse on a nest with poor shrub and herbaceous cover. This nest was unsuccessful. Photo by Jena Hickey.

Sage grouse on winter range. Note the relatively sparse cover; without snow, the canopy cover of sagebrush in this area exceeds 20%.

be defined: 1) nonmigratory, grouse do not make long-distance movements (i.e., >10 km one way) between or among seasonal ranges; 2) one-stage migratory, grouse move between 2 distinct seasonal ranges; and 3) 2-stage migratory, grouse move among 3 distinct seasonal ranges. Within a given geographic area, especially summer range, there may be birds that belong to more than one of these types of populations.

On an annual basis, migratory sage grouse populations may occupy areas that exceed 2,700 km^2 (Hulet 1983, Leonard et al. 2000). During winter, Robertson (1991) reported that migratory sage grouse in southeastern Idaho made mean daily movements of 752 m and occupied an area $\geq$140 km^2. For a nonmigratory population in Montana, Wallestad (1975) reported that winter home range size ranged from 11 to 31 km^2. During summer, migratory sage grouse in Idaho occupied home ranges of 3 to 7 km^2 (Connelly and Markham 1983, Gates 1983).

Despite large annual movements, sage grouse have high fidelity to seasonal ranges (Keister and Willis 1986, Fischer et al. 1993). Females return to the same area to nest each year (Fischer et al. 1993) and may nest within 200 m of their previous year's nest (Gates 1983, Lyon 2000).

Survival

Wallestad (1975) reported that annual survival rates for yearling and adult female sage grouse were 35 and 40%, respectively, for poncho-tagged birds. However, Zablan (1993) reported that survival rates for banded yearling and adult females in Colorado were similar and averaged 55%; survival rates for yearling and adult males differed, averaging 52 and 38%, respectively. In Idaho, annual survival of male sage grouse ranged from 46 to 54% and female survival from 68 to 85% (Connelly et al. 1994). Lower survival rates for males may be related to physiological demands because of sexual dimorphism and greater predation rates (Swenson 1986).

Reproduction

Bergerud (1988) suggested that most female tetraonids nest as yearlings. Although essentially all female sage grouse nested in Washington (Schroeder 1997), Connelly et al. (1993) reported that in Idaho up to 45% of yearling and 22% of adult female sage grouse do not nest each year. Gregg (1991) indicated that, of 119 females monitored through the breeding season in eastern Oregon, 26 (22%) did not nest. However, Coggins (1998) reported a 99% nest initiation rate for 3 years for the same population in Oregon. The differences may be related to improved range condition that resulted in better nutritional status of pre-laying hens (Barnett and Crawford 1994).

Estimates of sage grouse nest success throughout the species' range vary from 12 to 86% (Trueblood 1954, Gregg 1991, Schroeder et al. 1999). Nest success also may vary on an annual basis (Schroeder 1997, Sveum et al. 1998*a*). Wallestad and Pyrah (1974) observed greater nest success by adults than yearlings. However, significant differences in nest success between age groups have not been reported in other studies (Connelly et al. 1993, Schroeder 1997).

Clutch size of sage grouse is extremely variable and relatively low compared to other species of gamebirds (Edminster 1954, Schroeder 1997). Average clutch size for first nests varies from 6.0 to

Sage grouse nest. Photo by Jena Hickey.

9.5 throughout the species' range (Sveum 1995, Schroeder 1997). Greatest and least average clutch sizes have been reported in Washington (Sveum 1995, Schroeder 1997).

Renesting by sage grouse varies regionally from <20% (Patterson 1952, Eng 1963, Hulet 1983, Connelly et al. 1993) to >80% (Schroeder 1997). Despite regional variation, differences in renesting rates due to age have not been documented (Connelly et al. 1993, Schroeder 1997). Because of variation in nest initiation, success, and renesting rates, the proportion of females successfully hatching a brood varies between 15 and 70% (Wallestad and Pyrah 1974, Gregg et al. 1994). Despite this variation, sage grouse generally have low reproductive rates and high annual survival compared to most gallinaceous species (Zablan 1993, Connelly et al. 1994, Connelly and Braun 1997, Schroeder 1997, Schroeder et al. 1999).

Little information has been published on mortality of juvenile sage grouse or the level of production necessary to maintain a stable population. Among western states, long-term ratios have varied from 1.40 to 2.96 juveniles/hen in the fall; since 1985 these ratios have ranged from 1.21 to 2.19 (Connelly and Braun 1997). Available data suggest that a ratio $\geq$2.25 juveniles/hen in the fall should result in stable to increasing sage grouse populations (Connelly and Braun 1997, Edelmann et al. 1998).

Habitat requirements

Breeding habitats

Leks, or breeding display sites, typically occur in open areas surrounded by sagebrush (Patterson 1952, Gill 1965); these sites include, but are not limited to, landing strips, old lakebeds, low sagebrush flats and ridge tops, roads, cropland, and burned areas (Connelly et al. 1981, Gates 1985). Sage grouse males appear to form leks opportunistically at sites within or adjacent to potential nesting habitat. Although the lek may be an approximate center of annual ranges for nonmigratory populations (Eng and Schladweiler 1972, Wallestad and Pyrah 1974, Wallestad and Schladweiler 1974), this may not be the case for migratory populations (Connelly et al. 1988, Wakkinen et al. 1992). Average distances between nests and nearest leks vary from 1.1 to 6.2 km, but distance from lek of female capture to nest may be >20 km (Autenrieth 1981, Wakkinen et al. 1992, Fischer 1994, Hanf et al. 1994, Lyon 2000). Nests are placed independent of lek location (Bradbury et al. 1989, Wakkinen et al. 1992).

Habitats used by pre-laying hens also are part of the breeding habitat. These areas should provide a diversity of forbs high in calcium, phosphorus, and protein; the condition of these areas may greatly affect nest initiation rate, clutch size, and subsequent reproductive success (Barnett and Crawford 1994, Coggins 1998).

Most sage grouse nests occur under sagebrush (Patterson 1952, Gill 1965, Gray 1967, Wallestad and Pyrah 1974), but sage grouse will nest under other plant species (Klebenow 1969, Connelly et al. 1991, Gregg 1991, Sveum et al. 1998*a*). However, grouse nesting under sagebrush experience greater nest success (53%) than those nesting under other plant species (22%, Connelly et al. 1991).

Table 1. Habitat characteristics associated with sage grouse nest sites.

	Sagebrush		Grass		
State	Height[a](cm)	Coverage (%)[b]	Height(cm)	Coverage(%)[c]	Reference
Colo.	52				Petersen 1980
Id.		15		4	Klebenow 1969
Id.	58–79	23–38			Autenrieth 1981
Id.	71	22	18	3–10	Wakkinen 1990
Id.			19–23	7–9	Connelly et al. 1991
Id.	61		22	30	Fischer 1994
Id.		15–32	15–30		Klott et al. 1993
Id.	69	19	34	15	Apa 1998
Mont.	40	27			Wallestad 1975
Oreg.	80	20			Keister and Willis 1986
Oreg.		24	14	9–32	Gregg 1991
Wash.		20		51	Schroeder 1995
Wash.		19		32	Sveum et al. 1998a
Wyo.	36				Patterson 1952
Wyo.	29	24	15	9	Heath et al. 1997
Wyo.	31	25	18	5	Holloran 1999
Wyo.	33	26	21	11	Lyon 2000

[a] Mean height of nest bush.
[b] Mean canopy coverage of the sagebrush surrounding the nest.
[c] Some coverage estimates may include both grasses and forbs.

Mean height of sagebrush most commonly used by nesting grouse ranges from 29 to 80 cm (Table 1), and nests tend to be under the tallest sagebrush within a stand (Keister and Willis 1986, Wakkinen 1990, Apa 1998). In general, sage grouse nests are placed under shrubs having larger canopies and more ground and lateral cover as well as in stands with more shrub canopy cover than at random sites (Wakkinen 1990, Fischer 1994, Heath et al. 1997, Sveum et al. 1998*a*, Holloran 1999). Sagebrush cover near the nest site was greater around successful nests than unsuccessful nests in Montana (Wallestad and Pyrah 1974) and Oregon (Gregg 1991). Wallestad and Pyrah (1974) also indicated that successful nests were in sagebrush stands with greater average canopy coverage (27%) than those of unsuccessful nests (20%). Gregg (1991) reported that sage grouse nest success varied by cover type. The greatest nest success occurred in a mountain big sagebrush (*A. t. tridentata vaseyana*) cover type where shrubs 40–80 cm in height had greater canopy cover at the site of successful nests than at unsuccessful nests (Gregg 1991). These observations were consistent with the results of an artificial nest study showing greater coverage of medium-height shrubs improved success of artificial nests (DeLong 1993, DeLong et al. 1995).

Grass height and cover also are important components of sage grouse nest sites (Table 1). Grass associated with nest sites and with the stand of vegetation containing the nest was taller and denser than grass at random sites (Wakkinen 1990, Gregg 1991, Sveum et al. 1998*a*). Grass height at nests under non-sagebrush plants was greater ($P<0.01$) than that associated with nests under sagebrush, further suggesting that grass height is an important habitat component for nesting sage grouse (Connelly et al. 1991). Moreover, in Oregon, grass cover was greater at successful nests than at unsuccessful nests (Gregg 1991). Grass >18 cm in height occurring in stands of sagebrush 40–80 cm tall resulted in lesser nest predation rates than in stands with lesser grass heights (Gregg et al. 1994). Herbaceous cover associated with nest sites may provide scent, visual, and physical barriers to potential predators (DeLong et al. 1995).

Early brood-rearing areas occur in upland sagebrush habitats relatively close to nest sites, but movements of individual broods may vary (Connelly 1982, Gates 1983). Within 2 days of hatching, one brood moved 3.1 km (Gates 1983). Early brood-rearing habitats may be relatively open

Radiotelemetry and a pointing dog are used to capture sage grouse chicks for a research project in southeastern Idaho.

(about 14% canopy cover) stands of sagebrush (Martin 1970, Wallestad 1971) with ≥15% canopy cover of grasses and forbs (Sveum et al. 1998*b,* Lyon 2000). Great plant species richness with abundant forbs and insects characterize brood areas (Dunn and Braun 1986, Klott and Lindzey 1990, Drut et al. 1994*a*, Apa 1998). In Oregon, diets of sage grouse chicks included 34 genera of forbs and 41 families of invertebrates (Drut et al. 1994*b*). Insects, especially ants (Hymenoptera) and beetles (Coleoptera), are an important component of early brood-rearing habitat (Drut et al. 1994*b*, Fischer et al. 1996*a*). Ants and beetles occurred more frequently ($P=0.02$) at brood-activity centers compared to nonbrood sites (Fischer et al. 1996*a*).

Summer–late brood-rearing habitats

As sagebrush habitats desiccate, grouse usually move to more mesic sites during June and July (Gill 1965, Klebenow 1969, Savage 1969, Connelly and Markham 1983, Gates 1983, Connelly et al. 1988, Fischer et al. 1996*b*). Sage grouse broods occupy a variety of habitats during summer, including sagebrush (Martin 1970), relatively small burned areas within sagebrush (Pyle and Crawford 1996), wet meadows (Savage 1969), farmland, and other irrigated areas adjacent to sagebrush habitats (Connelly and Markham 1983, Gates 1983, Connelly et al. 1988). Apa (1998) reported that sites used by grouse broods had twice as much forb cover as independent sites.

Fall habitats

Sage grouse use a variety of habitats during fall. Patterson (1952) reported that grouse move from summer to winter range in October, but during mild weather in late fall, some birds may still use summer range. Similarly, Connelly and Markham (1983) observed that most sage grouse had abandoned summering areas by the first week of October. Fall movements to winter range are slow and meandering and occur from late August to December (Connelly et al. 1988). Wallestad (1975) documented a shift in feeding habits from September, when grouse were consuming a large amount of forbs, to December, when birds were feeding only on sagebrush.

Winter habitats

Characteristics of sage grouse winter habitats are relatively similar throughout most of the species' range (Table 2). Eng and Schladweiler (1972) and Wallestad (1975) indicated that most observations of radiomarked sage grouse during winter in Montana occurred in sagebrush habitats with >20% canopy cover. However, Robertson (1991) indicated that sage grouse used sagebrush habitats that had average canopy coverage of 15% and average height of 46 cm during 3 winters in southeastern Idaho. In Idaho, sage grouse selected areas with greater canopy cover of Wyoming big sagebrush (*A. t. wyomingensis*) in stands containing taller shrubs when compared to random sites (Robertson 1991).

Table 2. Characteristics of sagebrush at sage grouse winter-use sites.

	Canopy		
State	Coverage[a] (%)	Height[a] (cm)	Reference
Colo.		24–36[bd]	Beck 1977
Colo.		20–30[cd]	Beck 1977
Colo.	43[b]	34[b]	Schoenberg 1982
Colo.	37[c]	26[c]	Schoenberg 1982
Colo.	30–38[de]	41–54[de]	Hupp 1987
Id.	38[e]	56[e]	Autenrieth 1981
Id.	26[b]	29[b]	Connelly 1982
Id.	25[c]	26[c]	Connelly 1982
Id.	15	46	Robertson 1991
Mont.	27	25	Eng and Schladweiler 1972
Mont.	>20		Wallestad 1975
Oreg.	12–17[d]		Hanf et al. 1994

[a] Mean canopy coverage or height of sagebrush above snow.
[b] Males
[c] Females
[d] Ranges are given when data were provided for more than one year or area.
[e] No snow present when measurements were made or total height of plant was measured.

In Colorado, sage grouse may be restricted to <10% of the sagebrush habitat because of variation in topography and snow depth (Beck 1977, Hupp and Braun 1989). Such restricted areas of use may not occur throughout the species' range because in southeastern Idaho, severe winter weather did not result in the grouse population greatly reducing its seasonal range (Robertson 1991).

During winter, sage grouse feed almost exclusively on leaves of sagebrush (Patterson 1952, Wallestad et al. 1975). Although big sagebrush dominates the diet in most portions of the range (Patterson 1952; Wallested et al. 1975; Remington and Braun 1985; Welch et al. 1988, 1991), low sagebrush (*A. arbuscula*), black sagebrush (*A. nova*, Dalke et al. 1963, Beck 1977), fringed sagebrush (*A. frigida*, Wallestad et al. 1975), and silver sagebrush (*A. cana*, Aldridge 1998) are consumed in many areas depending on availability. Sage grouse in some areas apparently prefer Wyoming big sagebrush (Remington and Braun 1985, Myers 1992) and in other areas mountain big sagebrush (Welch et al. 1988, 1991). Some of the differences in selection may be due to preferences for greater levels of protein and the amount of volatile oils (Remington and Braun 1985, Welch et al. 1988).

Effects of habitat alteration

Range management treatments

Breeding habitat. Until the early 1980s, herbicide treatment (primarily with 2,4-D) was the most common method to reduce sagebrush on large tracts of rangeland (Braun 1987). Klebenow (1970) reported cessation of nesting in newly sprayed areas with < 5% live sagebrush canopy cover. Nesting also was nearly nonexistent in older sprayed areas containing about 5% live sagebrush cover (Klebenow 1970). In virtually all documented cases, herbicide application to blocks of sagebrush rangeland resulted in major declines in sage grouse breeding populations (Enyeart 1956, Higby 1969, Peterson 1970, Wallestad 1975). Effects of this treatment on sage grouse populations seemed more severe if the treated area was subsequently seeded to crested wheatgrass (*Agropyron cristatum*, Enyeart 1956).

Using fire to reduce sagebrush has become more common since most uses of 2,4-D on public lands were prohibited (Braun 1987). Klebenow (1972) and Sime (1991) suggested that fire may benefit sage grouse populations. Neither Gates (1983), Martin (1990), nor Bensen et al. (1991) reported adverse effects of fire on breeding populations of sage grouse. In contrast, following a 9-year study, Connelly et al. (1994, 2000*b*) indicated that prescribed burning of Wyoming big sagebrush during a drought period resulted in a large decline (>80%) of a sage grouse breeding population in southeastern Idaho. Additionally, Hulet (1983) documented loss of leks from fire and Nelle et al. (2000) reported that burning mountain big sagebrush stands had long-term negative impacts on sage grouse nesting and brood-rearing habitats. Canopy cover in mountain big sagebrush did not provide appropriate nesting habitat 14 years after burning (Nelle et al. 2000). The impact of fire on sage grouse populations using habitats dominated by silver sagebrush (which may resprout following fire) is unknown.

Cheatgrass (*Bromus tectrorum*) will often occupy sites following disturbance, especially burning (Valentine 1989). Repeated burning or burning in late summer favors cheatgrass invasion and may be a major cause of the expansion of this species (Vallentine 1989). The ultimate result may be a loss of the sage grouse population because of long-term conversion of sagebrush habitat to rangeland dominated by an annual exotic grass. However, this situation largely appears confined to the western portion of the species' range and does not commonly occur in Wyoming (J. Lawson, Wyoming Department of Game and Fish, personal communication).

Mechanical methods of sagebrush control have often been applied to smaller areas than those treated by herbicides or fire, especially to convert rangeland to cropland. However, adverse effects of this type of treatment on sage grouse breeding populations also have been documented. In Montana, Swenson et al. (1987) indicated that the number of breeding males declined by 73% after 16% of their study area was plowed.

Brood-rearing habitats. Martin (1970) reported that sage grouse seldom used areas treated with herbicides to remove sagebrush in southwestern Montana. In Colorado, Rogers (1964) indicated that an entire population of sage grouse appeared to emigrate from an area that was subjected to several years of herbicide application to remove sagebrush. Similarly, Klebenow (1970) reported that herbicide spraying reduced the brood-carrying capacity of an area in southeastern Idaho. However, application of herbicides in early spring to reduce sagebrush cover may enhance some brood-rearing habitats by increasing the amount of herbaceous plants used for food (Autenrieth 1981).

Fire may improve sage grouse brood-rearing habitat (Klebenow 1972, Gates 1983, Sime 1991), but until recently, experimental evidence was not available to support or refute these contentions (Braun 1987). Pyle and Crawford (1996) suggested that fire may enhance brood-rearing habitat in montane settings but cautioned that its usefulness requires further investigation. A 9-year study of the effects of fire on sage grouse did not support that prescribed fire, conducted during late summer in a Wyoming big sagebrush habitat, improved brood-rearing habitat for sage grouse (Connelly et al. 1994, Fischer et al. 1996*a*). Prescribed burning of sage grouse habitat did not increase amount of forbs in burned areas compared to unburned areas (Fischer et al. 1996*a*, Nelle et al. 2000) and resulted in decreased insect populations in the treated area compared to the unburned area. Thus, fire may negatively affect sage grouse brood-rearing habitat rather than improve it in Wyoming big sagebrush habitats (Connelly and Braun 1997), but its effect on grouse habitats in mountain big sagebrush communities requires further investigation (Pyle and Crawford 1996, Nelle et al. 2000).

Sage grouse often use agricultural areas for brood-rearing habitat (Patterson 1952, Wallestad 1975, Gates 1983, Connelly et al. 1988, Blus et al. 1989). Grouse use of these areas may result in mortality because of exposure to insecticides. Blus et al. (1989) reported die-offs of sage grouse that were exposed to methamidiphos used in potato fields and dimethoate used in alfalfa fields. Dimethoate is used commonly for alfalfa, and 20 of 31 radio-marked grouse (65%) died following direct exposure to this insecticide (Blus et al. 1989).

Winter habitat. Reduction in sage grouse use of an area treated by herbicide was proportional to the severity (i.e., amount of damage to sagebrush) of the treatment (Pyrah 1972). In sage grouse winter range, strip partial kill, block partial kill, and total kill of sagebrush were increasingly detrimental to sage grouse in Montana (Pyrah 1972) and Wyoming (Higby 1969).

In Idaho, Robertson (1991) reported that a 2,000-ha prescribed burn that removed 57% of the sagebrush cover in sage grouse winter habitat minimally impacted the sage grouse population. Although sage grouse use of the burned area declined following the fire, grouse adapted to this disturbance by moving 1 to 10 km outside of the burn to areas with greater sagebrush cover (Robertson 1991) than was available in the burned area.

Land use

Mining–energy development. Effects of mining, oil, and gas developments on sage grouse populations are not well known (Braun 1998). These activities negatively impact grouse habitat and populations over the short term (Braun 1998), but research suggests some recovery of populations following initial development and subsequent reclamation of the affected sites (Eng et al. 1979, Tate et al. 1979, Braun 1986). In Colorado, sage grouse were displaced by oil development and coal-mining activities, but numbers returned to pre-disturbance levels once the activities ceased (Braun 1987, Remington and Braun 1991). At least 6 leks in Alberta were disturbed by energy development and 4 were abandoned (Aldridge 1998). In Wyoming, female sage grouse captured on leks disturbed by natural gas development had lower nest-initiation rates, longer movements to nest sites, and different nesting habitats than hens captured on undisturbed leks (Lyon 2000). Sage grouse may repopulate an area following energy development but may not attain population levels that occurred prior to development (Braun 1998). Thus, short-term and long-term habitat loss appears to result from energy development and mining (Braun 1998).

Grazing. Domestic livestock have grazed over most areas used by sage grouse and this use is generally repetitive with annual or biennial grazing periods of varying timing and length (Braun 1998). Grazing patterns and use of habitats are often dependent on weather conditions (Valentine 1990). Historic and scientific evidence indicates that livestock grazing did not increase the distribution of sagebrush (Peterson 1995) but markedly reduced the herbaceous understory over relatively large areas and increased sagebrush density in some areas (Vale 1975, Tisdale and Hironaka 1981). Within the intermountain region, some vegetation changes from livestock grazing likely occurred because sagebrush steppe in this area did not evolve with intensive grazing by wild herbivores, as did the grassland prairies of central North America (Mack and Thompson 1982). Grazing by wild ungulates may reduce sagebrush cover (McArthur et al. 1988, Peterson 1995), and livestock grazing may result in high trampling mortality of sagebrush seedlings (Owens and Norton 1992). In Wyoming big sagebrush habitats, resting areas from livestock grazing may improve understory production as well as decrease sagebrush cover (Wambolt and Payne 1986).

There is little direct experimental evidence linking grazing practices to sage grouse population levels (Braun 1987, Connelly and Braun 1997). However, grass height and cover affect sage grouse nest site selection and success (Wakkinen 1990, Gregg 1991, Gregg et al. 1994, Delong et al. 1995, Sveum et al. 1998*a*). Thus, indirect evidence suggests grazing by livestock or wild herbivores that significantly reduces the herbaceous understory in breeding habitat may have negative impacts on sage grouse populations (Braun 1987, Dobkin 1995).

Miscellaneous activities. Construction of roads, powerlines, fences, reservoirs, ranches, farms, and housing developments has resulted in sage grouse habitat loss and fragmentation (Braun 1998). Between 1962 and 1997, >51,000 km of fence were constructed on land administered by the Bureau of Land Management in states supporting sage grouse populations (T. D. Rich, United States Bureau of Land Management, personal communication). Structures such as powerlines and fences pose hazards to sage grouse because they provide additional perch sites for raptors and because sage grouse may be injured or killed when they fly into these structures (Call and Maser 1985).

Weather

Prolonged drought during the 1930s and mid-1980s to early 1990s coincided with declining sage grouse populations throughout much of the species' range (Patterson 1952, Fischer 1994, Hanf et al. 1994). Drought may affect sage grouse populations by reducing herbaceous cover at nests and the quantity and quality of food available for hens and chicks during spring (Hanf et al. 1994, Fischer et al. 1996*a*).

Spring weather may influence sage grouse production. Relatively wet springs may result in increased production (Wallestad 1975, Autenrieth 1981). However, heavy rainfall during egg-laying or unseasonably cold temperatures with precipitation during hatching may decrease production (Wallestad 1975).

There is no evidence that severe winter weather affects sage grouse populations unless sagebrush cover has been greatly reduced or eliminated (Wallestad 1975, Beck 1977, Robertson 1991).

Predation

Over the last 25 years, numerous studies have used radiotelemetry to address sage grouse survival and nest success (Wallestad 1975; Hulet 1983; Gregg 1991; Robertson 1991; Connelly et al. 1993, 1994; Gregg et al. 1994; Schroeder 1997). Only Gregg (1991) and Gregg et al. (1994) indicated that predation was limiting sage grouse numbers, and their research suggested that low nest success from predation was related to poor nesting habitat. Most reported nest-success rates are >40%, suggesting that nest predation is not a widespread problem. Similarly, high survival rates of adult (Connelly et al. 1993, Zablan 1993) and older (>10 weeks of age) juvenile sage grouse indicate that population declines are not generally related to high levels of predation. Thus, except for an early study in Oregon (Batterson and Morse 1948), predation has not been identified as a major limiting factor for sage grouse (Connelly and Braun 1997).

Constructing ranches, farms, and housing developments has resulted in the addition of nonnative predators to sage grouse habitats, including dogs, cats, and red foxes (*Vulpes vulpes*; J. W. Connelly, Idaho Department of Fish and Game, unpublished data; B. L. Welch, United States Forest Service, personal communication) and may be responsible for increases in abundance of the common raven (*Corvus corax*, Sauer et al. 1997). Relatively high raven populations may decrease sage grouse nest success (Batterson and Morse 1948, Autenrieth 1981), but rigorous field studies using radiotelemetry do not support this hypothesis. Current work in Strawberry Valley, Utah, suggests that red foxes are taking a relatively high proportion of the population (Flinders 1999). This may become a greater problem if red foxes become well established throughout sage grouse breeding habitat.

Recommended guidelines

Sage grouse populations occupy relatively large areas on a year-round basis (Berry and Eng 1985, Connelly et al. 1988, Wakkinen 1990, Leonard et al. 2000), invariably involving a mix of ownership and jurisdictions. Thus, state and federal natural resource agencies and private landowners must coordinate efforts over at least an entire seasonal range to successfully implement these guidelines. Based on current knowledge of sage grouse population and habitat trends, these guidelines have been developed to help agencies and landowners effectively assess and manage populations, protect and manage remaining habitats, and restore damaged habitat. Because of gaps in our knowledge and regional variation in habitat characteristics (Tisdale and Hironaka 1981), the judgment of local biologists and quantitative data from population and habitat monitoring are necessary to implement the guidelines correctly. Further, we urge agencies to use an adaptive management approach (Macnab 1983, Gratson et al. 1993), using monitoring and evaluation to assess the success of implementing these guidelines to manage sage grouse populations.

Activities responsible for the loss or degradation of sagebrush habitats also may be used to restore these habitats. These activities include prescribed fire, grazing, herbicides, and mechanical treatments. Decisions on land treatments using these tools should be based on quantitative knowledge of vegetative conditions over an entire population's seasonal range. Generally, the treatment selected should be that which is least disruptive to the vegetation community and has the most rapid recovery time. This selection should not be based solely on economic cost.

Definitions

For the purpose of these guidelines, we define an occupied lek as a traditional display area in or adjacent to sagebrush-dominated habitats that has been attended by ≥2 male sage grouse in ≥2 of the previous 5 years. We define a breeding population as a group of birds associated with 1 or more occupied leks in the same geographic area separated from other leks by >20 km. This definition is somewhat arbitrary but generally based on maximum distances females move to nest.

Population management

1) Before making management decisions, agencies should cooperate to first identify lek locations and determine whether a population is migratory or nonmigratory. In the case of migratory populations, migration routes and seasonal habitats must be identified to allow for meaningful and correct management decisions.

2) Breeding populations should be assessed by either lek counts (census number of males attending leks) or lek surveys (classify known leks as active or inactive) each year (Autenrieth et al. 1982). Depending on number of counts each spring (Jenni and Hartzler 1978, Emmons and Braun

1984) and weather conditions when the counts were made, lek counts may not provide an accurate assessment of sage grouse populations (Beck and Braun 1980) and the data should be viewed with caution. Despite these shortcomings, lek counts provide the best index to breeding population levels and many long-term data sets are available for trend analysis (Connelly and Braun 1997).

3) Production or recruitment should be monitored by brood counts or wing surveys (Autenrieth et al. 1982). Brood counts are labor-intensive and usually result in inadequate sample size. Where adequate samples of wings can be obtained, we recommend using wing surveys to obtain estimates of sage grouse nesting success and juvenile:adult hen (including yearlings) ratios.

4) Routine population monitoring should be used to assess trends and identify problems for all hunted and nonhunted populations. Check stations, wing collections, and questionnaires can be used to obtain harvest information. Breeding population and production data (above) can be used to monitor nonhunted populations.

5) The genetic variation of relatively small, isolated populations should be documented to better understand threats to these populations and implement appropriate management actions (Young 1994, Oyler-McCance et al. 1999).

6) Hunting seasons for sage grouse should be based on careful assessments of population size and trends. Harvest should not be based on the observations of Allen (1954:43), who stated, "Our populations of small animals operate under a 1-year plan of decimation and replacement; and Nature habitually maintains a wide margin of overproduction. She kills off a huge surplus of animals whether we take our harvest or not." To the contrary, sage grouse tend to have relatively long lives with low annual turnover (Zablan 1993, Connelly et al. 1994) and a low reproductive rate (Gregg 1991, Connelly et al. 1993). Consequently, hunting may be additive to other causes of mortality for sage grouse (Johnson and Braun 1999, Connelly et al. 2000*a*). However, most populations appear able to sustain hunting if managed carefully (Connelly et al. 2000*a*).

7) If populations occur over relatively large geographic areas and are stable to increasing, seasons and bag limits can be relatively liberal (2- to 4-bird daily bag limit and a 2- to 5-week season) for hunting seasons allowing firearms (Braun and Beck 1985).

8) If populations are declining (for 3 or more consecutive years) or trends are unknown, seasons and bag limits should be generally conservative (1- or 2-bird daily bag limit and a 1-to 4-week season) for hunting seasons allowing firearms, or suspended (for all types of hunting, including falconry and Native American subsistence hunting) because of this species' population characteristics (Braun 1998, Connelly et al. 2000*a*).

9) Where populations are hunted, harvest rates should be 10% or less of the estimated fall population to minimize negative effects on the subsequent year's breeding population (Connelly et al. 2000*a*).

10) Populations should not be hunted where $\leq$300 birds comprise the breeding population (i.e., $\leq$100 males are counted on leks [C. E. Braun, Colorado Division of Wildlife, unpublished report]).

11) Spring hunting of sage grouse on leks should be discouraged or, if unavoidable, confined to males only during the early portion of the breeding season. Spring hunting is considered an important tradition for some Native American tribes. However, in Idaho, 80% of the leks hunted during spring in the early 1990s (n=5) had become inactive by 1994 (Connelly et al. 1994).

12) Viewing sage grouse on leks (and censusing leks) should be conducted so that disturbance to birds is minimized or preferably eliminated (Call and Maser 1986). Agencies should generally not provide all lek locations to individuals simply interested in viewing birds. Instead, 1 to 3 lek locations should be identified as public viewing leks, and if demand is great enough, agencies should consider erecting 2–3 seasonal blinds at these leks for public use. Camping in the center of or on active leks should be vigorously discouraged.

13) Discourage establishment of red fox and other nonnative predator populations in sage grouse habitats.

14) For small, isolated populations and declining populations, assess the impact of predation on survival and production. Predator control programs are expensive and often ineffective. In some cases, these programs may provide temporary help while habitat is recovering. Predator management programs also could be considered in areas where seasonal habitats are in good condition but their extent has been reduced greatly. However, predator management should be implemented only if the available data (e.g., nest success <25%, annual survival of adult hens <45%) support the action.

General habitat management

The following guidelines pertain to all seasonal habitats used by sage grouse:

1) Monitor habitat conditions and propose treatments only if warranted by range condition (i.e., the area no longer supports habitat conditions described in the following guidelines under habitat protection). Do not base land treatments on schedules, targets, or quotas.

2) Use appropriate vegetation treatment techniques (e.g., mechanical methods, fire) to remove junipers and other conifers that have invaded sage grouse habitat (Commons et al. 1999). Whenever possible, use vegetation control techniques that are least disruptive to the stand of sagebrush, if this stand meets the needs of sage grouse (Table 3).

3) Increase the visibility of fences and other structures occurring within 1 km of seasonal ranges by flagging or similar means if these structures appear hazardous to flying grouse (e.g., birds have been observed hitting or narrowly missing these structures or grouse remains have been found next to these structures).

4) Avoid building powerlines and other tall structures that provide perch sites for raptors within 3 km of seasonal habitats. If these structures must be built, or presently exist, the lines should be buried or poles modified to prevent their use as raptor perch sites.

Table 3. Characteristics of sagebrush rangeland needed for productive sage grouse habitat.

	Breeding		Brood-rearing		Winter [e]	
	Height (cm)	Canopy (%)	Height (cm)	Canopy (%)	Height (cm)	Canopy (%)
Mesic sites [a]						
Sagebrush	40–80	15–25	40–80	10–25	25–35	10–30
Grass–forb	>18 [c]	≥25 [d]	variable	>15	N/A	N/A
Arid sites [a]						
Sagebrush	30–80	15–25	40–80	10–25	25–35	10–30
Grass/forb	>18 [c]	≥15	variable	>15	N/A	N/A
Area [b]	>80		>40		>80	

[a] Mesic and arid sites should be defined on a local basis; annual precipitation, herbaceous understory, and soils should be considered (Tisdale and Hironaka 1981, Hironaka et al. 1983).

[b] Percentage of seasonal habitat needed with indicated conditions.

[c] Measured as "droop height"; the highest naturally growing portion of the plant.

[d] Coverage should exceed 15% for perennial grasses and 10% for forbs; values should be substantially greater if most sagebrush has a growth form that provides little lateral cover (Schroeder 1995)

[e] Values for height and canopy coverage are for shrubs exposed above snow.1

Breeding habitat management

For migratory and nonmigratory populations, lek attendance, nesting, and early brood rearing occur in breeding habitats. These habitats are sagebrush-dominated rangelands with a healthy herbaceous understory and are critical for survival of sage grouse populations. Mechanical disturbance, prescribed fire, and herbicides can be used to restore sage grouse habitats to those conditions identified as appropriate in the following sections on habitat protection. Local biologists and range ecologists should select the appropriate technique on a case-by-case basis. Generally, fire should not be used in breeding habitats dominated by Wyoming big sagebrush if these areas support sage grouse. Fire can be difficult to control and tends to burn the best remaining nesting and early brood-rearing habitats (i.e., those areas with the best remaining understory), while leaving areas with poor understory. Further, we recommend against using fire in habitats dominated by xeric mountain big sagebrush (*A. t. xericensis*) because annual grasses commonly invade these habitats and much of the original habitat has been altered by fire (Bunting et al. 1987).

Although mining and energy development are common activities throughout the range of sage grouse, quantitative data on the long-term effects of these activities on sage grouse are limited. However, some negative impacts have been documented (Braun 1998, Lyon 2000). Thus, these activities should be discouraged in breeding habitats, but when they are unavoidable, restoration efforts should follow procedures outlined in these guidelines.

Habitat protection

1) Manage breeding habitats to support 15–25% canopy cover of sagebrush, perennial herbaceous cover averaging ≥18 cm in height with ≥15% canopy cover for grasses and ≥10% for forbs and a diversity of forbs (Barnett and Crawford 1994, Drut et al. 1994*a*, Apa 1998) during spring (Table 3). Habitats meeting these conditions should have a high priority for wildfire suppression and should

not be considered for sagebrush control programs. Sagebrush and herbaceous cover should provide overhead and lateral concealment from predators. If average sagebrush height is >75 cm, herbaceous cover may need to be substantially greater than 18 cm to provide this protection. There is much variability among sagebrush-dominated habitats (Tisdale and Hironaka 1981, Hironaka et al. 1983), and some Wyoming sagebrush and low sagebrush breeding habitats may not support 25% herbaceous cover. In these areas, total herbaceous cover should be ≥15 % (Table 3). Further, the herbaceous height requirement may not be possible in habitats dominated by grasses that are relatively short when mature. In all of these cases, local biologists and range ecologists should develop height and cover requirements that are reasonable and ecologically defensible. Leks tend to be relatively open, thus cover on leks should not meet these requirements.

2) For nonmigratory grouse occupying habitats that are distributed uniformly (i.e., habitats have the characteristics described in guideline 1 and are generally distributed around the leks), protect (i.e., do not manipulate) sagebrush and herbaceous understory within 3.2 km of all occupied leks. For nonmigratory populations, consider leks the center of year-round activity and use them as focal points for management efforts (Braun et al. 1977).

3) For nonmigratory populations where sagebrush is not distributed uniformly (i.e., habitats have the characteristics described in guideline 1 but distributed irregularly with respect to leks), protect suitable habitats for ≤5 km from all occupied leks. Use radiotelemetry, repeated surveys for grouse use, or habitat mapping to identify nesting and early brood-rearing habitats.

4) For migratory populations, identify and protect breeding habitats within 18 km of leks in a manner similar to that described for nonmigratory sage grouse. For migratory sage grouse, leks generally are associated with nesting habitats but migratory birds may move >18 km from leks to nest sites. Thus, protection of habitat within 3.2 km of leks may not protect most of the important nesting areas (Wakkinen et al. 1992, Lyon 2000).

5) In areas of large-scale habitat loss (≥40% of original breeding habitat), protect all remaining habitats from additional loss or degradation. If remaining habitats are degraded, follow guidelines for habitat restoration listed below.

6) During drought periods (≥2 consecutive years), reduce stocking rates or change management practices for livestock, wild horses, and wild ungulates if cover requirements during the nesting and brood-rearing periods are not met. Grazing pressure from domestic livestock and wild ungulates should be managed in a manner that at all times addresses the possibility of drought.

Sage grouse just leaving a nest in good-condition breeding habitat in southwestern Idaho. Note the height of grass and herbaceous cover.

7) Suppress wildfires in all breeding habitats. In the event of multiple fires, land management agencies should have all breeding habitats identified and prioritized for suppression, giving the greatest priority to those that have become fragmented or reduced by >40% in the last 30 years.

8) Adjust timing of energy exploration, development, and construction activity to minimize disturbance of sage grouse breeding activities. Energy-related facilities should be located >3.2 km from active leks whenever possible. Human activities within view of or <0.5 km from leks should be minimized during the early morning and late evening when birds are near or on leks.

Habitat restoration

1) Before initiating vegetation treatments, quantitatively evaluate the area proposed for treatment to ensure that it does not have sagebrush and herbaceous cover suitable for breeding habitat (Table 3). Treatments should not be undertaken within sage grouse habitats until the limiting vegetation factor(s) has been identified, the proposed treatment is known to provide the desired vegetation response, and land-use activities can be managed after treatment to ensure that vegetation objectives are met.

2) Restore degraded rangelands to a condition that again provides suitable breeding habitat for sage grouse by including sagebrush, native forbs

(especially legumes), and native grasses in reseeding efforts (Apa 1998). If native forbs and grasses are unavailable, use species that are functional equivalents and provide habitat characteristics similar to those of native species.

3) Where the sagebrush overstory is intact but the understory has been degraded severely and quality of nesting habitat has declined (Table 3), use appropriate techniques (e.g., brush beating in strips or patches and interseed with native grasses and forbs) that retain some sagebrush but open shrub canopy to encourage forb and grass growth.

4) Do not use fire in sage grouse habitats prone to invasion by cheatgrass and other invasive weed species unless adequate measures are included in restoration plans to replace the cheatgrass understory with perennial species using approved reseeding strategies. These strategies could include, but are not limited to, use of pre-emergent herbicides (e.g., Oust®, Plateau®) to retard cheatgrass germination until perennial herbaceous species become established.

5) When restoring habitats dominated by Wyoming big sagebrush, regardless of the techniques used (e.g., prescribed fire, herbicides), do not treat >20% of the breeding habitat (including areas burned by wildfire) within a 30-year period (Bunting et al. 1987). The 30-year period represents the approximate recovery time for a stand of Wyoming big sagebrush. Additional treatments should be deferred until the previously treated area again provides suitable breeding habitat (Table 3). In some cases, this may take <30 years and in other cases >30 years. If 2,4-D or similar herbicides are used, they should be applied in strips such that their effect on forbs is minimized. Because fire generally burns the best remaining sage grouse habitats (i.e., those with the best understory) and leaves areas with sparse understory, use fire for habitat restoration only when it can be convincingly demonstrated to be in the best interest of sage grouse.

This breeding habitat is in poor condition because of a lack of understory.

Nest habitat is measured in Owyhee County, southwestern Idaho.

6) When restoring habitats dominated by mountain big sagebrush, regardless of the techniques used (e.g., fire, herbicides), treat ≤20% of the breeding habitat (including areas burned by wildfire) within a 20-year period (Bunting et al. 1987). The 20-year period represents the approximate recovery time for a stand of mountain big sagebrush. Additional treatments should be deferred until the previously treated area again provides suitable breeding habitat (Table 3). In some cases, this may take <20 years and in other cases >20 years. If 2,4-D or similar herbicides are used, they should be applied in strips such that their effect on forbs is minimized.

7) All wildfires and prescribed burns should be evaluated as soon as possible to determine whether reseeding is necessary to achieve habitat management objectives. If needed, reseed with sagebrush, native bunchgrasses, and forbs whenever possible.

8) Until research unequivocally demonstrates that using tebuthiuron and similar-acting herbicides to control sagebrush has no long-lasting negative impacts on sage grouse habitat, use these herbicides only on an experimental basis and over a sufficiently small area that any long-term negative impacts are negligible. Because these herbicides have the potential of reducing but not eliminating sagebrush cover within grouse breeding habitats, thus stimulating herbaceous development, their use as sage grouse habitat management tools should be examined closely.

John Crawford explains Oregon's sage grouse research program to field-trip attendees during a meeting of the Western States Sage and Columbian sharp-tailed Grouse Technical Committee.

Summer–late brood-rearing habitat management

Sage grouse may use a variety of habitats, including meadows, farmland, dry lakebeds, sagebrush, and riparian zones from late June to early November (Patterson 1952, Wallestad 1975, Connelly 1982, Hanf et al. 1994). Generally, these habitats are characterized by relatively moist conditions and many succulent forbs in or adjacent to sagebrush cover.

Habitat protection

1) Avoid land-use practices that reduce soil moisture effectiveness, increase erosion, cause invasion of exotic plants, and reduce abundance and diversity of forbs.

2) Avoid removing sagebrush within 300 m of sage grouse foraging areas along riparian zones, meadows, lakebeds, and farmland, unless such removal is necessary to achieve habitat management objectives (e.g., meadow restoration, treatment of conifer encroachment).

3) Discourage use of very toxic organophosphorus and carbamate insecticides in sage grouse brood-rearing habitats. Sage grouse using agricultural areas may be adversely affected by pesticide applications (Blus et al. 1989). Less toxic agrichemicals or biological control may provide suitable alternatives in these areas.

4) Avoid developing springs for livestock water, but if water from a spring will be used in a pipeline or trough, design the project to maintain free water and wet meadows at the spring. Capturing water from springs using pipelines and troughs may adversely affect wet meadows used by grouse for foraging.

Habitat restoration

1) Use brush beating or other mechanical treatments in strips 4–8 m wide in areas with relatively high shrub-canopy cover (≥35% total shrub cover) to improve late brood-rearing habitats. Brush beating can be used to effectively create different age classes of sagebrush in large areas with little age diversity.

2) If brush beating is impractical, use fire or herbicides to create a mosaic of openings in mountain big sagebrush and mixed-shrub communities used as late brood-rearing habitats where total shrub cover is ≥35%. Generally, 10–20% canopy cover of sagebrush and ≤25% total shrub cover will provide adequate habitat for sage grouse during summer.

3) Construct water developments for sage grouse only in or adjacent to known summer-use areas and provide escape ramps suitable for all avian species and other small animals. Water developments and "guzzlers" may improve sage grouse summer habitats (Autenrieth et al. 1982, Hanf et al. 1994). However, sage grouse used these developments infrequently in southeastern Idaho because most were constructed in sage grouse winter and breeding habitat rather than summer range (Connelly and Doughty 1989).

4) Whenever possible, modify developed springs and other water sources to restore natural free-flowing water and wet meadow habitats.

Winter habitat management

Sagebrush is the essential component of winter habitat. Sage grouse select winter-use sites based on snow depth and topography, and snowfall can affect the amount and height of sagebrush available to grouse (Connelly 1982, Hupp and Braun 1989, Robertson 1991). Thus, on a landscape scale, sage grouse winter habitats should allow grouse access to sagebrush under all snow conditions (Table 3).

Habitat protection

1) Maintain sagebrush communities on a landscape scale, allowing sage grouse access to sagebrush stands with canopy cover of 10–30% and heights of at least 25–35 cm regardless of snow cover. These areas should be high priority for wildfire suppression and sagebrush control should be avoided.

2) Protect patches of sagebrush within burned areas from disturbance and manipulation. These areas may provide the only winter habitat for sage grouse and their loss could result in the extirpation of the grouse population. They also are important

seed sources for sagebrush re-establishment in the burned areas. During fire-suppression activities do not remove or burn any remaining patches of sagebrush within the fire perimeter.

3) In areas of large-scale habitat loss ($\geq$40% of original winter habitat), protect all remaining sagebrush habitats.

Habitat restoration

1) Reseed former winter range with the appropriate subspecies of sagebrush and herbaceous species unless the species are recolonizing the area in a density that would allow recovery (Table 3) within 15 years.

2) Discourage prescribed burns >50 ha, and do not burn >20% of an area used by sage grouse during winter within any 20–30-year interval (depending on estimated recovery time for the sagebrush habitat).

Conservation strategies

We recommend that each state and province develop and implement conservation plans for sage grouse. These plans should use local working groups comprised of representatives of all interested agencies, organizations, and individuals to identify and solve regional issues (Anonymous 1997). Within the context of these plans, natural resource agencies should cooperate to document the amount and condition of sagebrush rangeland remaining in the state or province. Local and regional plans should summarize common problems to conserve sage grouse and general conditions (Table 3) needed to maintain healthy sage grouse populations. Local differences in conditions that affect sage grouse populations may occur and should be considered in conservation plans. Natural resource agencies should identify remaining breeding and winter ranges in Wyoming big sagebrush habitats and establish these areas as high priority for wildfire suppression. Prescribed burning in habitats that are in good ecological condition should be avoided. Protection and restoration of sage grouse habitats also will likely benefit many other sagebrush obligate species (Saab and Rich 1997) and enhance efforts to conserve and restore sagebrush steppe.

Although translocating sage grouse to historical range has been done on numerous occasions, few attempts have been successful (Musil et al. 1993, Reese and Connelly 1997). Thus, we agree with Reese and Connelly (1997) that translocation efforts should be viewed as only experimental at this time and not as a viable management strategy.

More information is needed on characteristics of healthy sagebrush ecosystems and the relationship of grazing to sage grouse production. Field experiments should be implemented to evaluate the relationship of grazing pressure (i.e., disturbance and removal of herbaceous cover) to sage grouse nest success and juvenile survival (Connelly and Braun 1997). The overall quality of existing sage grouse habitat will become increasingly important as quantity of these habitats decrease. Sage grouse populations appear relatively secure in some portions of their range and at risk in other portions. However, populations that have thus far survived extensive habitat loss may still face extinction because of a time lag between habitat loss and ultimate population collapse (Cowlishaw 1999).

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Literature cited

Aldridge, C. L. 1998. Status of the sage grouse (*Centrocercus urophasianus urophasianus*) in Alberta. Alberta Environmental Protection, Wildlife Management Division, and Alberta Conservation Association, Wildlife Status Report 13, Edmonton, Canada.

Allen, D. L. 1954. Our wildlife legacy. Funk and Wagnalls, New York, New York, USA.

Anonymous. 1997. Gunnison sage grouse conservation plan, Gunnison Basin, Colorado. Colorado Division of Wildlife, Fort Collins, USA.

Apa, A. D. 1998. Habitat use and movements of sympatric sage and Columbian sharp-tailed grouse in southeastern Idaho. Dissertation, University of Idaho, Moscow, USA.

Autenrieth, R. E. 1981. Sage grouse management in Idaho. Idaho Department of Fish and Game, Wildlife Bulletin 9, Boise, Idaho, USA.

Autenrieth, R. E., W. Molini, and C. E. Braun. 1982. Sage grouse management practices. Western States Sage Grouse Committee, Technical Bulletin 1, Twin Falls, Idaho, USA.

Barnett, J. F., and J. A. Crawford. 1994. Pre-laying nutrition of sage grouse hens in Oregon. Journal of Range Management 47: 114–118.

Batterson, W. M., and W. B. Morse. 1948. Oregon sage grouse. Oregon Game Commission Fauna Series 1, Portland, USA.

Beck, T. D. I. 1975. Attributes of a wintering population of sage grouse, North Park, Colorado. Thesis, Colorado State University, Fort Collins, USA.

Beck, T. D. I. 1977. Sage grouse flock characteristics and habitat selection during winter. Journal of Wildlife Management 41: 18–26.

Beck, T. D. I., and C. E. Braun. 1980. The strutting ground count: variation, traditionalism, management needs. Proceedings of the Western Association of Fish and Wildlife Agencies 60: 558–566.

Benson, L. A., C. E. Braun, and W. C. Leininger. 1991. Sage grouse response to burning in the big sagebrush type. Proceedings, issues and technology in the management of impacted western wildlife. Thorne Ecological Institute 5: 97–104.

Bergerud, A. T. 1988. Population ecology of North American grouse. Pages 578–648 *in* A. T. Bergerud and M. W. Gratson, editors. Adaptive strategies and population ecology of northern grouse. University of Minnesota, Minneapolis, USA.

Berry, J. D., and R. L. Eng. 1985. Interseasonal movements and fidelity to seasonal use areas by female sage grouse. Journal of Wildlife Management 49: 237–240.

Blus, L. J., C. S. Staley, C. J. Henny, G. W. Pendleton, T. H. Craig, E. H. Craig, and D. K. Halford. 1989. Effects of organophosphorus insecticides on sage grouse in southeastern Idaho. Journal of Wildlife Management 53: 1139–1146.

Bradbury, J. W., R. M. Gibson, C. E. McCarthy, and S. L. Vehrencamp. 1989. Dispersion of displaying male sage grouse. II. The role of female dispersion. Behavioral Ecology and Sociobiology 24: 15–24.

Braun, C. E. 1986. Changes in sage grouse lek counts with advent of surface coal mining. Proceedings, issues and technology in the management of impacted western wildlife. Thorne Ecological Institute 2: 227–231.

Braun, C. E. 1987. Current issues in sage grouse management. Proceedings of the Western Association of Fish and Wildlife Agencies 67: 134–144.

Braun, C. E. 1998. Sage grouse declines in western North America: what are the problems? Proceedings of the Western Association of State Fish and Wildlife Agencies 78: 139–156.

Braun, C. E., and T. D. I Beck. 1985. Effects of changes in hunting regulations on sage grouse harvest and populations. Pages 335–344 *in* S. L. Beasom and S. F. Roberson, editors. Game harvest management. Caesar Kleberg Wildlife Research Institute, Kingsville, Texas, USA.

Braun C. E., T. Britt, and R. O. Wallestad. 1977. Guidelines for maintenance of sage grouse habitats. Wildlife Society Bulletin 5: 99–106.

Braun C. E., M. F. Baker, R. L. Eng, J. W. Gashwiler, and M. H. Schroeder. 1976. Conservation committee report on effects of alteration of sagebrush communities on the associated avifauna. Wilson Bulletin 88: 165–171.

Bunting, S. C., B. M. Kilgore, and C. L. Bushey. 1987. Guidelines for prescribed burning sagebrush-grass rangelands in the northern great basin. United States Department of Agriculture, Forest Service, General Technical Report INT-231, Ogden, Utah, USA.

Call, M. W., and C. Maser. 1985. Wildlife habitats in managed rangelands—the great basin of southeastern Oregon. Sage grouse. United States Department of Agriculture, Forest Service, General Technical Report PNW-187, Portland, Oregon, USA.

Coggins, K. A. 1998. Sage grouse habitat use during the breeding season on Hart Mountain National Antelope Refuge. Thesis, Oregon State University, Corvallis, USA.

Commons, M. L., R. K. Baydack, and C. E. Braun. 1999. Sage grouse response to pinyon-juniper management. Pages 238–239 *in* S. B. Monson and R. Stevens, compilers. Proceedings: ecology and management of pinyon–juniper communities. United States Department of Agriculture, Forest Service, RMRS-P9, Fort Collins, Colorado, USA.

Connelly, J. W., Jr. 1982. An ecological study of sage grouse in southeastern Idaho. Dissertation, Washington State University, Pullman, USA.

Connelly, J. W., W. J. Arthur, and O. D. Markham. 1981. Sage grouse leks on recently disturbed sites. Journal of Range Management 52: 153–154.

Connelly, J. W., A. D. Apa, R. B. Smith, and K. P. Reese. 2000*a*. Effects of predation and hunting on adult sage grouse *Centrocercus urophasianus* in Idaho. Wildlife Biology 6: in press.

Connelly, J. W., and C. E. Braun. 1997. Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. Wildlife Biology 3/4: 123–128.

Connelly, J. W., H. W. Browers, and R. J. Gates. 1988. Seasonal movements of sage grouse in southeastern Idaho. Journal of Wildlife Management 52: 116–122.

Connelly, J. W., and L. A. Doughty. 1989. Sage grouse use of wildlife water developments in southeastern Idaho. Pages 167–173 *in* S. Stiver and G. Tsukomoto, editors. Symposium on wildlife water developments. Nevada Department of Fish and Game, Reno, USA.

Connelly, J. W., R. A. Fischer, A. D. Apa, K. P. Reese, and W. L. Wakkinen. 1993. Renesting of sage grouse in southeastern Idaho. Condor 95: 1041–1043.

Connelly, J. W., and O. D. Markham. 1983. Movements and radionuclide concentrations of sage grouse in southeastern Idaho. Journal of Wildlife Management 47: 169–177.

Connelly, J. W., K. P. Reese, R. A. Fischer, and W. L. Wakkinen. 2000*b*. Response of sage grouse breeding population to fire in southeastern Idaho. Wildlife Society Bulletin 28: 90–96.

Connelly, J. W., K. P. Reese, W. L. Wakkinen, M. D. Robertson, and R. A. Fischer. 1994. Sage grouse ecology report. Idaho Department of Fish and Game, Job Completion Report W-160-R-19, Subproject 9, Boise, Idaho, USA.

Connelly, J. W., W. L. Wakkinen, A. D. Apa, and K. P. Reese. 1991. Sage grouse use of nest sites in southeastern Idaho. Journal of Wildlife Management 55: 521–524.

Cowlishaw, G. 1999. Predicting the pattern of decline of African primate diversity: an extinction debt from historical deforestation. Conservation Biology 13: 1183–1193.

Dalke, P. D., D. B. Pyrah, D. C. Stanton, J. E. Crawford, and E. F. Schlatterer. 1963. Ecology, productivity, and management of sage grouse in Idaho. Journal of Wildlife Management 27: 810–841.

DeLong, A. K. 1993. Relationships between vegetative structure and predation rates of artificial sage grouse nests. Thesis, Oregon State University, Corvallis, USA.

DeLong, A. K., J. A. Crawford, and D. C. DeLong, Jr. 1995. Relationships between vegetational structure and predation of artificial sage grouse nests. Journal of Wildlife Management 59:88–92.

Dobkin, D. S. 1995. Management and conservation of sage grouse, denominative species for the ecological health of shrubsteppe ecosystems. United States Department of Interior, Bureau of Land Management, Portland, Oregon, USA.

Drut, M. S., J. A. Crawford, and M. A. Gregg. 1994*a*. Brood habitat use by sage grouse in Oregon. Great Basin Naturalist 54: 170–176.

Drut, M. S., W. H. Pyle, and J. A. Crawford. 1994*b*. Diets and food selection of sage grouse chicks in Oregon. Journal of Range Management 47:90–93.

Dunn, P. O., and C. E. Braun. 1986. Summer habitat use by adult female and juvenile sage grouse. Journal of Wildlife Management 50:228–235.

Edelmann, F. B., M. J. Ulliman, M. J. Wisdom, K. P. Reese, and J. W. Connelly. 1998. Assessing habitat quality using population fitness parameters: a remote sensing–GIS-based habitat-explicit population model for sage grouse (*Centrocercus urophasianus*). Idaho Forest, Wildlife and Range Experiment Station, Technical Report 25, Moscow, USA.

Edminster, F. C. 1954. American game birds of field and forest. Charles Scribner's Sons, New York, New York, USA.

Emmons, S. R., and C. E. Braun. 1984. Lek attendance of male sage grouse. Journal of Wildlife Management 48:1023–1028.

Eng, R. L. 1963. Observations on the breeding biology of male sage grouse. Journal of Wildlife Management 27:841–846.

Eng, R. L., and P. Schladweiler. 1972. Sage grouse winter movements and habitat use in central Montana. Journal of Wildlife Management 36:141–146.

Eng, R. L., E. J. Pitcher, S. J. Scott, and R. J. Greene. 1979. Minimizing the effect of surface coal mining on a sage grouse population by a directed shift of breeding activities. Pages 464–468 *in* G. A. Swanson, technical coordinator. The mitigation symposium: a national workshop on mitigating losses of fish and wildlife habitats. United States Department of Agriculture, Forest Service, General Technical Report RM-65, Fort Collins, Colorado, USA.

Enyeart, G. 1956. Responses of sage grouse to grass reseeding in the Pines area, Garfield County, Utah. Thesis, Utah State Agricultural College, Logan, USA.

Fischer, R. A. 1994. The effects of prescribed fire on the ecology of migratory sage grouse in southeastern Idaho. Dissertation, University of Idaho, Moscow, USA.

Fischer, R. A., A. D. Apa, W. L. Wakkinen, K. P. Reese, and J. W. Connelly. 1993. Nesting-area fidelity of sage grouse in southeastern Idaho. Condor 95:1038–1041.

Fischer, R. A., K. P. Reese, and J. W. Connelly. 1996*a*. An investigation on fire effects within xeric sage grouse brood habitat. Journal of Range Management 49:194–198.

Fischer, R. A., K. P. Reese, and J. W. Connelly. 1996*b*. Influence of vegetal moisture content and nest fate on timing of female sage grouse migration. Condor 98:868–872.

Fischer, R. A., K. P. Reese, and J. W. Connelly. 1997. Effects of prescribed fire on movements of female sage grouse from breeding to summer ranges. Wilson Bulletin 109:82–91.

Flinders, J. T. 1999. Restoration of sage grouse in Strawberry Valley, Utah, 1998–99 report. Utah Reclamation Mitigation and Conservation Commission, Progress Report. Brigham Young University, Provo, Utah, USA.

Gates, R. J. 1983. Sage grouse, lagomorph, and pronghorn use of a sagebrush grassland burn site on the Idaho National Engineering Laboratory. Thesis, Montana State University, Bozeman, USA.

Gates, R. J. 1985. Observations of the formation of a sage grouse lek. Wilson Bulletin 97:219–221.

Gill, R. B. 1965. Distribution and abundance of a population of sage grouse in North Park, Colorado. Thesis, Colorado State University, Fort Collins, USA.

Girard, G. L. 1937. Life history, habits, and food of the sage grouse, *Centrocercus urophasianus* Bonaparte. University of Wyoming, Publication 3, Laramie, USA.

Gratson, M. W., J. W. Unsworth, P. Zager, and L. Kuck. 1993. Initial experiences with adaptive resource management for determining appropriate antlerless elk harvest rates in Idaho. Transactions of the North American Wildlife and Natural Resources Conference 58:610–619.

Gray, G. M. 1967. An ecological study of sage grouse broods with reference to nesting, movements, food habits and sagebrush strip spraying in the Medicine Lodge drainage, Clark County, Idaho. Thesis, University of Idaho, Moscow, USA.

Gregg, M. A. 1991. Use and selection of nesting habitat by sage grouse in Oregon. Thesis, Oregon State University, Corvallis, USA.

Gregg, M. A., J. A. Crawford, M. S. Drut, and A. K. DeLong. 1994. Vegetational cover and predation of sage grouse nests in Oregon. Journal of Wildlife Management 58:162–166.

Hanf, J. M., P. A. Schmidt, and E. B. Groshens. 1994. Sage grouse in the high desert of central Oregon: results of a study, 1988–1993. United States Department of Interior, Bureau of Land Management, Series P-SG-01, Prineville, Oregon, USA.

Heath, B. J., R. Straw, S. H. Anderson, and J. Lawson. 1997. Sage grouse productivity, survival, and seasonal habitat use near Farson, Wyoming. Wyoming Game and Fish Department, Project Completion Report, Laramie, USA.

Higby, L. W. 1969. A summary of the Longs Creek sagebrush control project. Proceedings, Biennial Western States Sage Grouse Workshop 6:164–168.

Hironaka, M., M. A. Fosberg, and A. H. Winward. 1983. Sagebrush-grass habitat types of southern Idaho. Idaho Forest, Wildlife and Range Experiment Station, Bulletin 35, Moscow, Idaho, USA.

Holloran, M. J. 1999. Sage grouse (*Centrocercus urophasianus*) seasonal habitat use near Casper, Wyoming. Thesis, University of Wyoming, Laramie, USA.

Hornaday, W. T. 1916. Save the sage grouse from extinction, a demand from civilization to the western states. New York Zoological Park Bulletin 5:179–219.

Hulet, B. V. 1983. Selected responses of sage grouse to prescribed fire, predation, and grazing by domestic sheep in southeastern Idaho. Thesis, Brigham Young University, Provo, Utah, USA.

Hupp, J. W. 1987. Sage grouse resource exploitation and endogenous reserves in Colorado. Dissertation, Colorado State University, Fort Collins, USA.

Hupp, J. W., and C. E. Braun. 1989. Topographic distribution of sage grouse foraging in winter. Journal of Wildlife Management 53:823–829.

Jenni, D. A., and J. E. Hartzler. 1978. Attendance at a sage grouse lek: implications for spring censuses. Journal of Wildlife Management 42:46–52.

JOHNSON, K. H., AND C. E. BRAUN. 1999. Viability and conservation of an exploited sage grouse population. Conservation Biology 13:77–84.

KEISTER, G. P., AND M. J. WILLIS. 1986. Habitat selection and success of sage grouse hens while nesting and brooding. Oregon Department of Fish and Wildlife, Progress Report W-87-R-2, Subproject 285, Portland, Oregon, USA.

KLEBENOW, D. A. 1969. Sage grouse nesting and brood habitat in Idaho. Journal of Wildlife Management 33:649–661.

KLEBENOW, D. A. 1970. Sage grouse versus sagebrush control in Idaho. Journal of Range Management 23:396–400.

KLEBENOW, D. A. 1972. The habitat requirements of sage grouse and the role of fire in management. Tall Timbers Fire Ecology Conference 12:305–315.

KLOTT, J. H., AND F. G. LINDZEY. 1990. Brood habitats of sympatric sage grouse and Columbian sharp-tailed grouse in Wyoming. Journal of Wildlife Management 54:84–88.

KLOTT, J. H., R. B. SMITH, AND C. VULLO. 1993. Sage grouse habitat use in the Brown's Bench area of south-central Idaho. United States Department of Interior, Bureau of Land Management, Idaho State Office, Technical Bulletin 93–4, Boise, Idaho, USA.

LYON, A.G. 2000. The potential effects of natural gas development on sage grouse (*Centrocercus urophasianus*) near Pinedale, Wyoming. Thesis, University of Wyoming, Laramie, USA.

LEONARD, K. M., K. P. REESE, AND J. W. CONNELLY. 2000. Distribution, movements, and habitats of sage grouse *Centrocercus urophasianus* on the Upper Snake River Plain of Idaho: changes from the 1950's to the 1990's. Wildlife Biology 6: in press.

MACK, R. N., AND J. N. THOMPSON. 1982. Evolution in steppe with few large, hooved mammals. American Naturalist 119: 757–773.

MACNAB, J. 1983. Wildlife management as scientific experimentation. Wildlife Society Bulletin 11:397–401.

MARTIN, N. S. 1970. Sagebrush control related to habitat and sage grouse occurrence. Journal of Wildlife Management 34: 313–320.

MARTIN, R. C. 1990. Sage grouse responses to wildfire in spring and summer habitats. Thesis, University of Idaho, Moscow, USA.

MCARTHUR, E. D., A. C. BLAUER, AND S. C. SANDERSON. 1988. Mule deer-induced mortality of mountain big sagebrush. Journal of Range Management 41:114–117.

MUSIL, D. D., J. W. CONNELLY, AND K. P. REESE. 1993. Movements, survival, and reproduction of sage grouse translocated into central Idaho. Journal of Wildlife Management 57:85–91.

MYERS, O. B. 1992. Sage grouse habitat enhancement: effects of sagebrush fertilization. Dissertation, Colorado State University, Fort Collins, USA.

NELLE, P. J. , K. P. REESE, AND J. W. CONNELLY. 2000. The long-term effect of fire on sage grouse nesting and brood-rearing habitats on the Upper Snake River Plain. Journal of Range Management 53:586–591.

OWENS, M. K., AND B. E. NORTON. 1992. Interactions of grazing and plant protection on basin big sagebrush (*Artemisia tridentata* spp. *tridentata*) seedling survival. Journal of Range Management 45:257–262.

OYLER-MCCANCE, S. J., N. W. KAHN, K. P. BURNHAM, C. E. BRAUN, AND T. W. QUINN. 1999. A population genetic comparison of large and small-bodied sage grouse in Colorado using microsatellite and mitochondrial DNA markers. Molecular Ecology, 8: 1457–1465.

PATTERSON, R. L. 1952. The sage grouse in Wyoming. Sage Books, Denver, Colorado, USA.

PETERSEN, B. E. 1980. Breeding and nesting ecology of female sage grouse in North Park, Colorado. Thesis, Colorado State University, Fort Collins, USA.

PETERSON, J. G. 1970. The food habits and summer distribution of juvenile sage grouse in central Montana. Journal of Wildlife Management 34:147–155.

PETERSON, J. G. 1995. Ecological implications of sagebrush manipulation: a literature review. Montana Fish, Wildlife, and Parks, Helena, USA.

PYLE, W. H., AND J. A. CRAWFORD. 1996. Availability of foods of sage grouse chicks following prescribed fire in sagebrush–bitterbrush. Journal of Range Management 49:320–324.

PYRAH, D. B. 1972. Effects of chemical and mechanical sagebrush control on sage grouse. Montana Fish and Game Department, Job Completion Report W-105-R-6, Helena, USA.

REESE, K. P., AND J. W. CONNELLY. 1997. Translocations of sage grouse *Centrocercus urophasianus* in North America. Wildlife Biology 3/4:235–241.

REMINGTON, T. E., AND C. E. BRAUN. 1985. Sage grouse food selection in winter, North Park, Colorado. Journal of Wildlife Management 49:1055–1061.

REMINGTON, T. E., AND C. E. BRAUN. 1991. How surface coal mining affects sage grouse, North Park, Colorado. Proceedings, Issues and Technology in the Management of Impacted Western Wildlife. Thorne Ecological Institute 5:128–132.

ROBERTSON, M. D. 1991. Winter ecology of migratory sage grouse and associated effects of prescribed fire in southeastern Idaho. Thesis, University of Idaho, Moscow, USA.

ROGERS, G. E. 1964. Sage grouse investigations in Colorado. Colorado Game, Fish, and Parks Department, Technical Publication 16, Denver, Colorado, USA.

SAAB, V. A., AND T. D. RICH. 1997. Large-scale conservation assessment for neotropical migratory land birds in the interior Columbia River Basin. United States Department of Agriculture, Forest Service, General Technical Report PNW-GTR-399, Portland, Oregon, USA.

SAUER, J. R., J. E. HINES, J. FALLON AND G. GOUGH. 1997. The North American breeding bird survey, results and analysis 1996–98. Version 98.1. United States Geological Survey, Patuxent Wildlife Research Center, Laurel, Maryland, USA.

SAVAGE, D. E. 1969. Relation of sage grouse to upland meadows in Nevada. Nevada Fish and Game Commission, Job Completion Report, Project W-39-R-9, Job 12, Reno, Nevada, USA.

SCHOENBERG, T.J. 1982. Sage grouse movements and habitat selection in North Park, Colorado. Thesis, Colorado State University, Fort Collins, USA.

SCHROEDER, M. A. 1995. Productivity and habitat use of sage grouse in north-central Washington. Washington Department of Fish and Wildlife, Job Progress Report Project W-96-R, Olympia, USA.

SCHROEDER, M. A. 1997. Unusually high reproductive effort by sage grouse in a fragmented habitat in north-central Washington. Condor 99:933–941.

SCHROEDER, M. A., J. R. YOUNG, AND C. E. BRAUN. 1999. Sage grouse (*Centrocercus urophasianus*). Pages 1–28 *in* A. Poole and F. Gill, editors. The birds of North America, No. 425. The Birds of North America, Philadelphia, Pennsylvania, USA.

SIME, C. A. 1991. Sage grouse use of burned, non-burned, and seeded vegetation communities on the Idaho National

Engineering Laboratory, Idaho. Thesis, Montana State University, Bozeman, USA.

SVEUM, C. M. 1995. Habitat selection by sage grouse hens during the breeding season in south-central Washington. Thesis, Oregon State University, Corvallis, USA.

SVEUM, C. M., W. D. EDGE, AND J. A. CRAWFORD. 1998*a*. Nesting habitat selection by sage grouse in south-central Washington. Journal of Range Management 51: 265–269.

SVEUM, C. M., J. A. CRAWFORD, AND W. D. EDGE. 1998*b*. Use and selection of brood-rearing habitat by sage grouse in south-central Washington. Great Basin Naturalist 58: 344–351.

SWENSON, J. E. 1986. Differential survival by sex in juvenile sage grouse and gray partridge. Ornis Scandinavica 17: 14–17.

SWENSON, J. E., C. A. SIMMONS, AND C. D. EUSTACE. 1987. Decrease of sage grouse *Centrocercus urophasianus* after plowing of sagebrush steppe. Biological Conservation 41: 125–132.

TATE, J., JR., M. S. BOYCE, AND T. R. SMITH. 1979. Response of sage grouse to artificially created display ground. Pages 464–468 *in* G. A. Swanson, technical coordinator. The mitigation symposium: a national workshop on mitigating losses of fish and wildlife habitats. United States Department of Agriculture, Forest Service, General Technical Report RM-65, Fort Collins, Colorado, USA.

TISDALE, E. W., AND M. HIRONAKA. 1981. The sagebrush-grass region: a review of the ecological literature. Idaho Forest, Wildlife, and Range Experiment Station, Bulletin 33, Moscow, USA.

TRUEBLOOD, R. W. 1954. The effect of grass reseeding in sagebrush lands on sage grouse populations. Thesis, Utah State Agricultural College, Logan, USA.

VALE, T. R. 1975. Presettlement vegetation in the sagebrush-grass area of the Intermountain West. Journal of Range Management 28: 32–36.

VALLENTINE, J. F. 1989. Range development and improvements. Third edition. Academic, San Diego, California, USA.

VALLENTINE, J. F. 1990. Grazing management. Academic, San Diego, California, USA.

WAKKINEN, W. L. 1990. Nest site characteristics and spring–summer movements of migratory sage grouse in southeastern Idaho. Thesis, University of Idaho, Moscow, USA.

WAKKINEN, W. L., K. P. REESE, AND J. W. CONNELLY. 1992. Sage grouse nest locations in relation to leks. Journal of Wildlife Management 56:381–383.

WALLESTAD, R. O. 1971. Summer movements and habitat use by sage grouse broods in central Montana. Journal of Wildlife Management 35: 129–136.

WALLESTAD, R. O. 1975. Life history and habitat requirements of sage grouse in central Montana. Montana Fish and Game Department, Technical Bulletin, Helena, USA.

WALLESTAD, R. O., AND D. B. PYRAH. 1974. Movement and nesting of sage grouse hens in central Montana. Journal of Wildlife Management 38: 630–633.

WALLESTAD, R. O., AND P. SCHLADWEILLER. 1974. Breeding season movements and habitat selection of male sage grouse. Journal of Wildlife Management 38: 634–637.

WALLESTAD, R. O., J. G. PETERSON, AND R. L. ENG. 1975. Foods of adult sage grouse in central Montana. Journal of Wildlife Management 39: 628–630.

WAMBOLT, C. L., AND G. F. PAYNE. 1986. An 18-year comparison of control methods for Wyoming big sagebrush in southwestern Montana. Journal of Range Management 39: 314–319.

WELCH, B. L., J. C. PEDERSON, AND R. L. RODRIQUEZ. 1988. Selection of big sagebrush by sage grouse. Great Basin Naturalist 48: 274–279.

WELCH, B. L., F. J. WAGSTAFF, AND J. A. ROBERSON. 1991. Preference of wintering sage grouse for big sagebrush. Journal of Range Management 44: 462–465.

YOUNG, J. R. 1994. The influence of sexual selection on phenotypic and genetic divergence among sage grouse populations. Dissertation, Purdue University, West Lafayette, Indiana, USA.

ZABLAN, M. A. 1993. Evaluation of sage grouse banding program in North Park, Colorado. Thesis, Colorado State University, Fort Collins, USA.

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USDA United States Department of Agriculture

Science Framework for Conservation and Restoration of the Sagebrush Biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions

Part 2. Management Applications

Forest Service

Rocky Mountain Research Station

General Technical Report
RMRS-GTR-389

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Abstract

The Science Framework is intended to link the Department of the Interior's Integrated Rangeland Fire Management Strategy with long-term strategic conservation and restoration actions in the sagebrush biome. The focus is on sagebrush (*Artemisia* spp.) ecosystems and sagebrush dependent species with an emphasis on Greater sage-grouse (*Centrocercus urophasianus*). Part 1 of the Science Framework, published in 2017, provides the scientific information and decision-support tools for prioritizing areas for management and determining effective management strategies across the sagebrush biome. Part 2, this document, provides the management considerations for applying the information and tools in Part 1. Part 2 is intended to facilitate implementation of resource management priorities and use of management strategies that increase ecosystem resilience to disturbance and resistance to nonnative invasive annual grasses. The target audience of Part 2 is field managers, resource specialists, and regional and national-level managers. The topics addressed in this volume include adaptive management and monitoring, climate adaptation, wildfire and vegetation management, nonnative invasive plant management, application of National Seed Strategy concepts, livestock grazing management, wild horse and burro considerations, and integration and tradeoffs. Geospatial data, maps, and models for the Science Framework are provided through the U.S. Geological Survey's ScienceBase database and Bureau of Land Management's Landscape Approach Data Portal. The Science Framework is intended to be adaptive and will be updated as additional data become available on other values and species at risk. It is anticipated that the Science Framework will be widely used to: (1) inform emerging strategies to conserve sagebrush ecosystems, sagebrush dependent species, and human uses of the sagebrush system; and (2) assist managers in prioritizing and planning on-the-ground restoration and mitigation actions across the sagebrush biome.

Keywords: sagebrush habitat, Greater sage-grouse, resilience, resistance, conservation, restoration, monitoring, adaptive management, climate adaptation, wildfire, nonnative invasive plants, National Seed Strategy, livestock grazing, wild horses and burros

Front cover photo. Sagebrush ecosystem in the Toiyabe Range, Nevada (photo: Jeanne Chambers, USDA Forest Service). Inset: Greater sage-grouse chick (photo: USDOI Fish and Wildlife Service).

Rear cover photo. Tail feathers of a Greater sage-grouse (photo: USDOI Fish and Wildlife Service).

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Contents

1. OVERVIEW OF THE SCIENCE FRAMEWORK

Jeanne C. Chambers, Karen L. Prentice, and Michele R. Crist

Introduction

The Science Framework is part of an unprecedented conservation effort underway across 11 States in the western United States to address threats to sagebrush (*Artemisia* spp.) ecosystems and the species that depend on them. Sagebrush ecosystems provide a large diversity of habitats and support more than 350 species of vertebrates (Suring et al. 2005). These ecosystems currently make up only about 59 percent of their historical area, and the primary patterns, processes, and components of many sagebrush ecosystems have been significantly altered since Euro-American settlement in the mid-1800s (Knick et al. 2011; Miller et al. 2011). The primary threats to sagebrush ecosystems are well recognized and include large-scale wildfire, invasion of exotic annual grasses, conifer expansion, energy development, conversion to cropland, and urban and exurban development (Coates et al. 2016; Davies et al. 2011; Knick et al. 2011; USDOI FWS 2013). The continued loss and fragmentation of sagebrush habitats has placed many species at risk, including Greater sage-grouse (*Centrocercus urophasianus*; hereafter, GRSG), which has been considered for listing under the Endangered Species Act several times (USDOI FWS 2010, 2015) and whose status will be reevaluated in 2020 (USDOI FWS 2015).

The Science Framework was developed to provide a transparent, ecologically defensible approach for making policy and management decisions to reduce threats to sagebrush ecosystems and GRSG across multiple scales. It is directly linked to U.S. Department of the Interior directives and ongoing multi-partner conservation efforts (table 1.1).

The Science Framework represents a paradigm shift for agencies and managers in sagebrush ecosystems. Recent research has provided the basis for characterizing sagebrush ecosystems according to their ecological resilience to disturbance and resistance to invasive annual grasses (Chambers et al. 2014a,b, 2017b; Maestas et al. 2016). This has enabled development of approaches that couple information on resilience and resistance with knowledge of GRSG habitat and threats to sagebrush ecosystems in order to prioritize conservation actions based not only on species habitat requirements but also on the likely response of that habitat to disturbances and management actions (Chambers et al. 2014c, 2016, 2017a; Ricca et al. 2018). New geospatial data and analytical approaches provide the capacity to prioritize management actions to conserve and restore sagebrush ecosystems at much larger scales than in the past. Managing multiple resources across scales and surface land management jurisdictions in an integrated and collaborative manner is becoming common practice for agencies managing sagebrush ecosystems.

Top left: Mule deer walking through sagebrush (photo: USDOI National Park Service). Top right: Badger near its burrow (photo: USDOI Fish and Wildlife Service). Middle left: Burrowing owls near their burrow (photo: USDOI Fish and Wildlife Service). Middle right: Common sagebrush lizard on a rock (photo: commons.wikimedia.org). Bottom left: Pygmy rabbit hiding underneath sagebrush in snow (photo: USDOI Fish and Wildlife Service). Bottom middle: Sagebrush sparrow on a sagebrush plant (photo: S. Richards). Bottom right: Male Hera buckmoth on a sagebrush plant (photo: USDA Forest Service).

Table 1.1—Key directives, science information, and conservation and restoration strategies for the sagebrush biome.

Title	Description	Cooperators
An Integrated Rangeland Fire Management Strategy: Final Report to the Secretary of the Interior (IRFMS)	Longer-term actions to implement policies and strategies for preventing and suppressing rangeland fire and restoring rangeland landscapes affected by fire in the Western United States. Section 7b(iv) called for development of a Conservation and Restoration Strategy for sagebrush ecosystems that considered emerging science and included a baseline assessment, conceptual models, and other components necessary to provide an overarching strategy for "on the ground" restoration actions and provide a foundation for adaptive management and budget prioritization.	U.S. Department of the Interior (DOI) (USDOI 2015)
Science Framework for Conservation and Restoration of the Sagebrush Biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions Part 1. Science Basis and Applications	Scientific information and decision-support tools to: (1) facilitate prioritization of areas for conservation and restoration management actions; (2) inform budget prioritization of management actions; and (3) inform management strategies across scales and ownerships. Developed per IRFMS, Section 7b (iv). Builds on prior interagency work that developed a strategic, multi-scale approach to manage threats to sagebrush ecosystems and sage-grouse using resilience and resistance concepts (Chambers et al. 2014a, 2016).	State and Federal agencies (Chambers et al. 2017a)
Science Framework for Conservation and Restoration of the Sagebrush Biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions Part 2. Management Applications	Guidance for applying the scientific information and decision-support tools in Part 1 of the Science Framework in order to: (1) implement resource management priorities at large, landscape scales; and (2) use management strategies that increase ecosystem resilience to disturbance and resistance to nonnative invasive plant species across scales. Developed per IRFMS, Section 7b (iv).	State and Federal agencies (Crist et al. this volume)
Sagebrush Science Initiative	A collaborative effort to identify and fill the highest priority gaps in scientific knowledge needed to effectively conserve sagebrush dependent species and the sagebrush habitats they depend on.	Fish and Wildlife Service, Western Association of Fish and Wildlife Agencies (WAFWA), Bureau of Land Management (WAFWA lead; in progress)
Sagebrush Conservation Strategy Developed to meet the requirements of IRFMS, Section 7b (iv) in collaboration with the Sagebrush Science Initiative	A comprehensive, collaborative strategy to conserve sagebrush, sagebrush dependent species, and human uses of sagebrush ecosystems that builds on the resilience and resistance concepts, threat assessments, and habitat prioritization methods described in the Science Framework. This broad strategy will provide for voluntary conservation measures for managing and conserving sagebrush ecosystems, and is intended to provide an inclusive "all-hands, all-lands" approach.	State and Federal agencies, nongovernmental organizations, universities (WAFWA lead; in progress)
Secretarial Order 3362: Improving Habitat Quality in Western Big-Game Winter Range and Migration Corridors	Guidance to conserve and restore priority winter range and migration corridors for elk, mule deer, and pronghorn, as identified by State and tribal wildlife agency partners. DOI agencies will work with State, tribal, and other Federal partners such as USDA Forest Service to restore habitats, minimize disturbance, and use other site-specific management to conserve these areas. Much of the habitat for these three species is within the sagebrush biome.	DOI agencies, State agencies, WAFWA, USDA Forest Service (USDOI 2018)

The Science Framework uses a multi-scale approach to inform management decisions and actions. It applies the best available information on resilience and resistance to invasive annual grasses, GRSG habitat, and threats to sagebrush ecosystems to: (1) inform strategic management and conservation investments at broad scales (ecoregion or GRSG Management Zone to sagebrush biome), and (2) determine appropriate management strategies at local (field office or district) scales. An integrated monitoring and adaptive management approach is recommended to reduce the uncertainty in the effectiveness of management actions over time by improving both management objectives and strategies (Allen et al. 2011; Thompson et al. 2013). Syntheses of the best available science and considerations of the tradeoffs involved in making decisions facilitate development of appropriate management objectives and strategies in planning processes as well as alternatives for National Environmental Policy Act (NEPA) analyses.

Part 1 of the “Science Framework for Conservation and Restoration of the Sagebrush Biome: Linking the Department of the Interior’s Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions” focuses on the **science basis and applications** for protecting, conserving, and restoring sagebrush ecosystems and GRSG habitat (Chambers et al. 2017a; hereafter, Part 1). Scientific information and decision-support tools are provided to: (1) assist in prioritizing areas for conservation and restoration management actions, (2) inform budget prioritization of management actions, and (3) inform management strategies across scales and ownerships.

Part 2 focuses on **management considerations and tradeoffs** for Part 1 and emphasizes adaptive management. The information in this volume can be used to apply the scientific information and decision-support tools in Part 1 in order to: (1) implement resource management priorities at large, landscape scales; and (2) use management strategies that increase ecosystem resilience to disturbance and resistance to nonnative invasive plant species across spatial scales. The concepts and approaches that form the basis for Parts 1 and 2 of the Science Framework are briefly reviewed in this section. The applications of these concepts and approaches are described in sections 2 through 8 and focus on key resource management topics, including adaptive management and monitoring, climate adaptation, wildfire and vegetation management, nonnative invasive plant management, application of National Seed Strategy concepts, livestock grazing management, and wild horse and burro considerations. Section 9 discusses integration of the management strategies for the different topics, and the associated tradeoffs involved in managing for diverse resources across large landscapes.

The Science Framework was developed to be used by resource specialists and practitioners at field and regional management levels, while providing a broader context for regional and national-level managers. Although the focus is largely on the sagebrush biome and GRSG, the information and tools provided allow managers to address other resource values and at-risk species as the necessary geospatial data are developed.

Concepts and Approaches Used in the Science Framework

The Science Framework provides the information and tools to address the primary threats to sagebrush ecosystems at geographical scales relevant to management. The threats addressed in the Science Framework were identified in the Sage-Grouse Conservation Objectives Team Final Report (USDOI FWS 2013) and reflect the threats to sagebrush ecosystems in general. These threats are consistent with those included in the Greater Sage-Grouse Monitoring Framework developed by the Interagency Greater Sage-Grouse Disturbance and Monitoring Subteam (IGSDMS 2014) and the State Wildlife Action Plans, which were prepared for the purpose of maintaining the health and diversity of wildlife within the State and reducing the need for future listings under the Endangered Species Act. In addition to these previously identified threats, climate adaptation is addressed in the Science Framework and climate adaptation strategies are provided.

The Science Framework includes three scales that inform different aspects of planning and implementation: (1) the sagebrush biome scale, where consistent data for the range of sagebrush and GRSG can inform budget prioritization; (2) the mid-scale (ecoregions and Management Zones), where assessments are typically conducted to inform budget prioritization and develop priority planning areas; and (3) the local scale, where local data and expertise are used to select project sites and determine appropriate management strategies and treatments within priority planning areas (table 1.2). At the mid-scale, a crosswalk is provided between U.S. Environmental Protection Agency ecoregions (USEPA 2016) and sage-grouse Management Zones (Stiver et al. 2006) (fig. 1.1). This approach aligns with the Sage-grouse Habitat Assessment Framework (Johnson 1980; Stiver et al. 2015).

Table 1.2—Scales and areas included in the strategic approach for managing threats to sagebrush ecosystems, sage-grouse, and other sagebrush obligate species as well as the data, tools, models, and processes considered at each scale or area.

Area	Geographic scale	Map extent	Data, tools, models	Process
Sagebrush biome and multiple Management Zones (MZs)	Broad	West-wide	Habitat Soils Population data and models Priority resource data Fire and other threat data Climate change projections	Budget prioritization for rangewide consistency
Sage-grouse MZs and ecoregions	Mid	State or national forest	Above, plus: Assessments and planning documents Regional data and models Regional tools	Assessments at ecoregion or MZ scale for prioritization of management actions
Local planning areas	Local	District, field office, or project area	Above, plus: Local data and information	Selection of treatment types within prioritized project areas

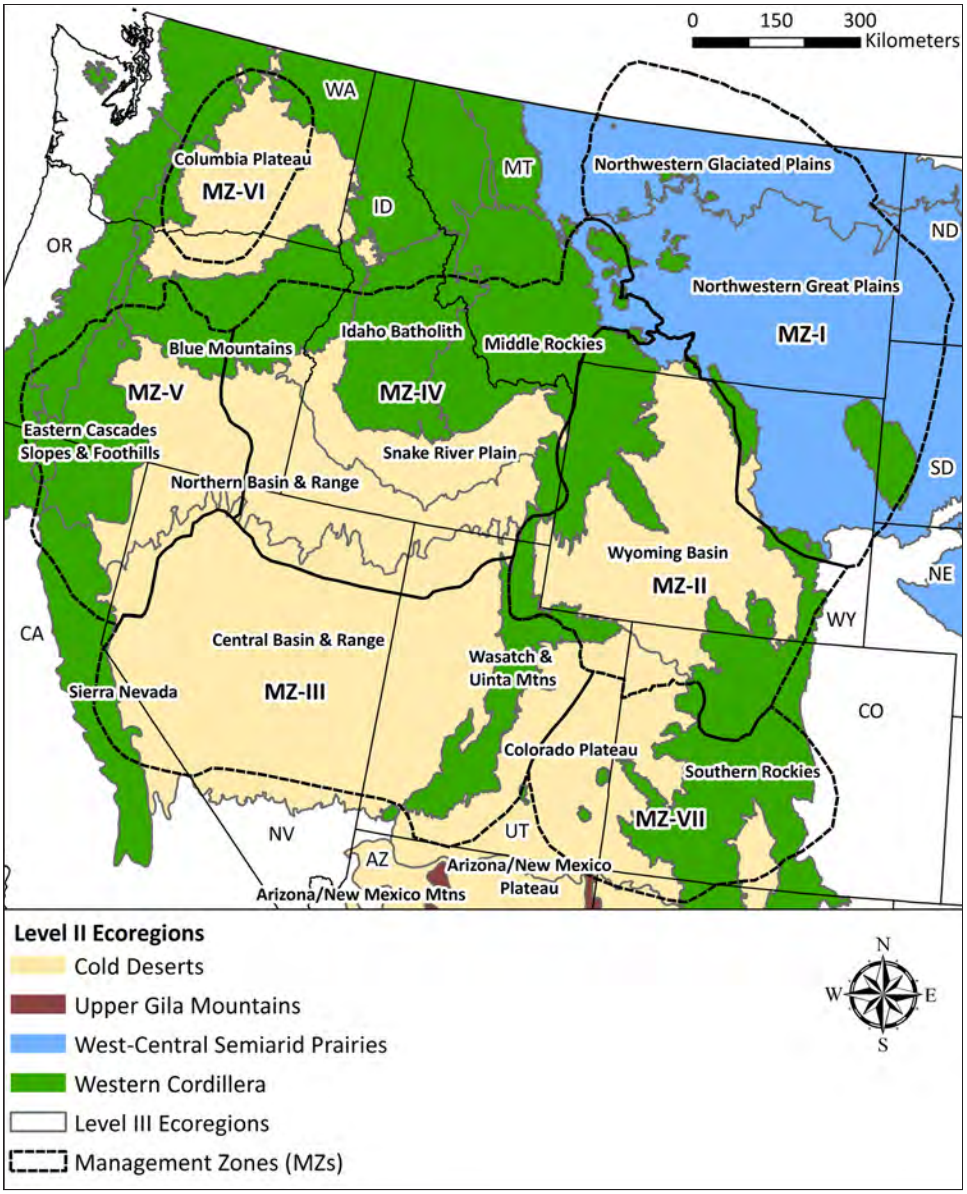

Figure 1.1—A crosswalk between level II and level III ecoregions (USEPA 2016) and sage-grouse Management Zones (MZs; Stiver et al. 2006) (Chambers et al. 2017a, fig. 1).

The Science Framework uses an approach that is based on current understanding of ecosystem resilience to disturbance and resistance to nonnative invasive plants in sagebrush ecosystems. Resilient ecosystems have the capacity to **reorganize and regain** their basic characteristics when altered by stressors such as invasive plants and disturbances such as improper livestock grazing and altered fire regimes (Angler and Allen 2016; Holling 1973). Species resilience refers to the ability of a species to recover from stressors and disturbances (USDOI FWS 2013), and is closely linked to ecosystem resilience. Resistant ecosystems have the capacity to ***retain*** their fundamental structure, processes, and functioning when exposed to stresses, disturbances, or invasive species (Angeler and Allen 2016; Folke et al. 2004). Resistance to invasion by nonnative plants is increasingly important in sagebrush ecosystems; it is a function of the abiotic and biotic attributes and ecological processes of an ecosystem that limits the population growth of an invading species (D'Antonio and Thomsen 2004). A detailed explanation of the factors that influence resilience and resistance in sagebrush ecosystems is found in Chambers et al. (2014a). Definitions of the terms used in this document are in Appendix 1.

Management focused on ecosystem resilience and resistance can help sustain local communities by ensuring that ecosystem services, such as water for human consumption and agricultural use, forage for livestock, and recreational opportunities, are maintained or improved over time. The resilience of socioeconomic systems, threats to those systems, and current capacities to implement management actions to address those threats are a separate aspect of developing an approach for conservation and restoration of the sagebrush biome and are addressed elsewhere.

The approach used in the Science Framework is intended to help prioritize areas for management and determine the most appropriate management strategies. The Science Framework is based on: (1) the likely response of an area to disturbance or stress due to threats, management actions, or a combination thereof (i.e., resilience to disturbance and resistance to invasion by nonnative plants); (2) the capacity of an area to support target species or resources; and (3) the predominant threats. It uses a mid-scale approach and has six steps.

- Identify focal species or resources and key habitat indicators for the assessment area, and then delineate their distribution or area using the best information available. For GRSG, this currently includes the modeled breeding habitat probabilities and the population index (Doherty et al. 2016). Information and tools are provided to allow managers to address other resource values and at-risk species as geospatial data for those values and species become available.
- Develop an understanding of ecosystem resilience to disturbance and resistance to nonnative invasive plants for the assessment area. At landscape scales, resilience and resistance to invasive annual grasses, which are a primary cause of altered fire regimes and habitat degradation in sagebrush ecosystems, are closely linked to soil temperature and moisture regimes (Chambers et al. 2014a,b; 2017b). Thus, soil temperature and moisture regimes are used to quantify and map resilience and resistance to invasive annual grasses (Maestas et al. 2016). More detailed information on soil characteristics and ecological site descriptions help managers to step-down generalized vegetation dynamics, including resilience and resistance concepts, to local scales.
- Integrate ecosystem resilience to disturbance and resistance to invasive annual grasses with species or resource habitat requirements and develop a decision matrix that can be used to spatially link ecosystem resilience and resistance, habitat requirements, and management strategies (table 1.3).
- Assess the key threats in the assessment area using geospatial data and maps.
- Prioritize areas for management in the assessment area using geospatial data and maps of species or resource habitat requirements, such as the breeding habitat probabilities for GRSG, resilience to disturbance and resistance to invasive annual grasses, and the key threats (fig. 1.2).
- Determine the most appropriate management strategies for areas prioritized for targeted conservation and restoration management actions based on habitat characteristics, relative resilience to disturbance and resistance to invasive annual grasses, and the predominant threats. The management strategies are developed in collaboration with stakeholders and are reconciled with socioeconomic and budgetary considerations. Other priority resources are considered such as special status plant species.

Table 1.3—Sage-grouse habitat, resilience and resistance matrix based on resilience and resistance concepts from Chambers et al. (2014a,b) and GRSG breeding habitat probabilities from Doherty et al. (2016). Rows show the ecosystem's relative resilience to disturbance and resistance to invasive annual grasses (1 = high resilience and resistance, 2 = moderate resilience and resistance, 3 = low resilience and resistance). Resilience and resistance categories were derived from soil temperature and moisture regimes (Chambers et al. 2017a [Part1], Appendix 2; Maestas et al. 2016) and relate to the sagebrush ecological types in Part 1, table 6. Columns show the landscape-scale GRSG breeding habitat probability based on Part 1, table 7 (A = 0.25 to <0.5 probability; B = 0.5 to <0.75 probability; C = ≥0.75 probability). Use of the matrix is explained in Part 1, section 7.4. Potential management strategies for persistent ecosystem threats, anthropogenic threats, and climate change are in table 1.4.

Landscape-Scale Sage-Grouse Breeding Habitat Probability

Ecosystem Resilience to Disturbance and Resistance to Invasion	**Low (0.25 to <0.5 probability)**	**Moderate (0.5 to <0.75 probability)**	**High (≥0.75 probability)**
	Landscape context is likely to be limiting habitat suitability. If limiting factors are within management control, significant restoration may be needed. These landscapes may still be important for other seasonal habitat needs or connectivity.	Landscape context may be affecting habitat suitability and could be aided by restoration. These landscapes may be at higher risk of becoming unsuitable with additional disturbances that degrade habitat.	Landscape context is highly suitable to support breeding habitat. Management strategies to maintain and enhance these landscapes have a high likelihood of benefiting sage-grouse.
High	**1A**	**1B**	**1C**
	Potential for favorable perennial herbaceous species recovery after disturbance without seeding is typically high. Risk of invasive annual grasses becoming dominant is relatively low. EDRR can be used to address problematic invasive plants. Tree removal can increase habitat availability and connectivity in expansion areas. Seeding/transplanting success is typically high.		
Moderate	**2A**	**2B**	**2C**
	Potential for favorable perennial herbaceous species recovery after disturbance without seeding is usually moderately high, especially on cooler and moister sites. Risk of invasive annual grasses becoming dominant is moderate, especially on warmer sites. EDRR can be used to address problematic invasive plants in many areas. Tree removal can increase habitat availability and connectivity in expansion areas. Seeding/transplanting success depends on site characteristics, and more than one intervention may be required, especially on warmer and drier sites. Recovery following inappropriate livestock use depends on site characteristics and management.		
Low	**3A**	**3B**	**3C**
	Potential for favorable perennial herbaceous species recovery after disturbance without seeding is usually low. Risk of invasive annual grasses becoming dominant is high. EDRR can be used to address problematic invasive plants in relatively intact areas. Seeding/transplanting success depends on site characteristics, extent of annual invasive plants, and post-treatment precipitation, but is often low. More than one intervention likely will be required. Recovery following inappropriate livestock use is unlikely without active restoration.		

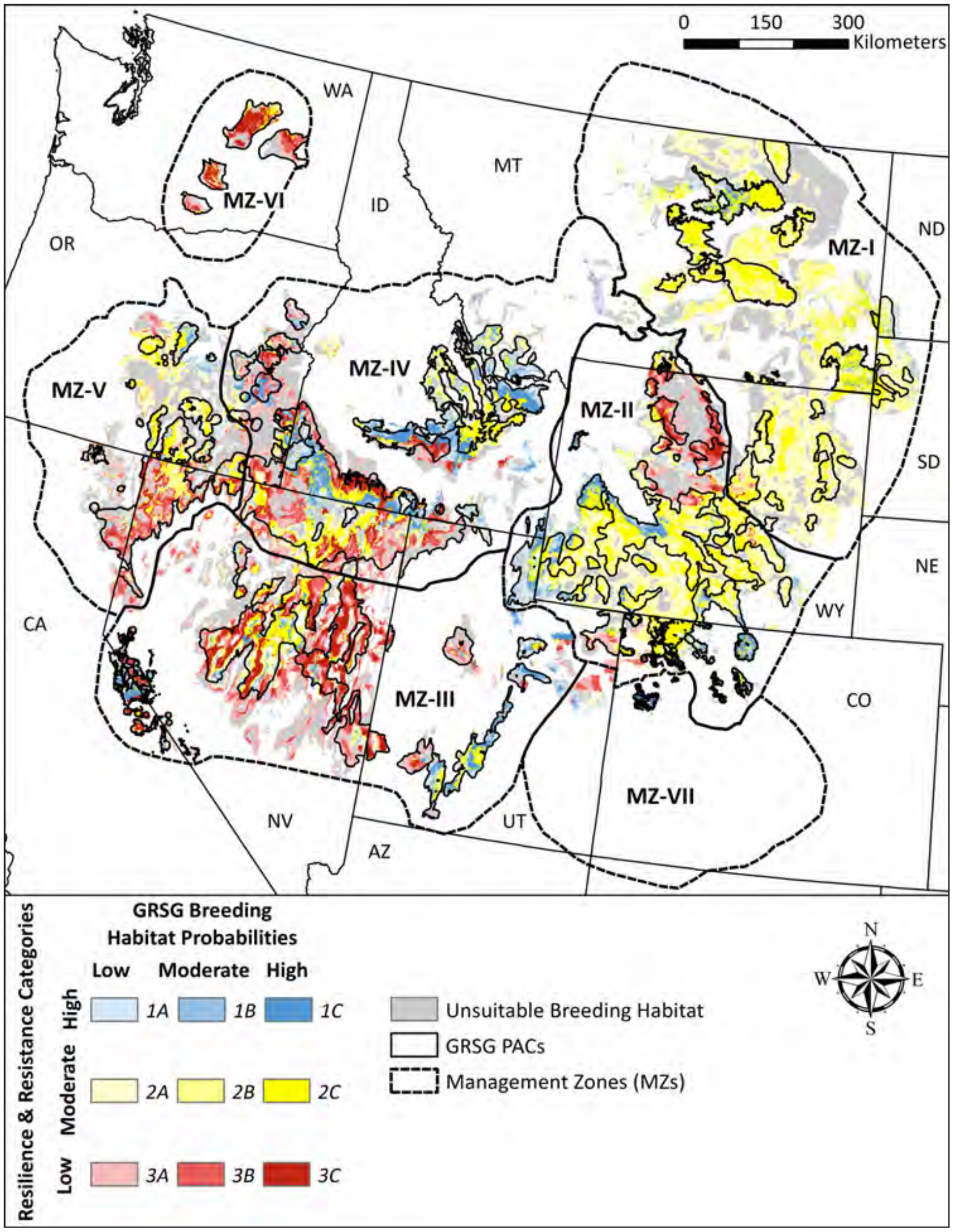

Figure 1.2—Greater sage-grouse (GRSG) breeding habitat probabilities based on 2010–2014 lek data (Doherty et al. 2016) intersected with resilience and resistance categories developed from soil temperature and moisture regimes (Chambers et al. 2017a). This map provides a spatial depiction of the sage-grouse habitat, resilience and resistance matrix (Chambers et al. 2017a, fig. 38).

These six steps help identify priority areas for management and overarching management strategies for the assessment area. Key aspects of the approach are the sage-grouse habitat resilience and resistance matrix (table 1.3) and linked management strategies for addressing threats to sagebrush ecosystems (table 1.4). To step down ecoregion or Management Zone priorities to the local scale, managers and stakeholders are engaged to: (1) refine priorities and management strategies based on higher resolution geospatial products, additional species information, and local knowledge, including traditional ecological knowledge; (2) select specific project areas; and (3) identify opportunities to leverage partner resources.

Table 1.4—Management strategies for persistent ecosystem threats, climate change, and land use and development threats. Recommendations are provided for prioritizing and targeting strategies based on cells in the sage-grouse habitat, resilience and resistance matrix (table 1.3). Threats and strategies are cross-cutting and affect multiple program areas. While many of the strategies fall under the broad umbrella of vegetation management, a coordinated and integrated approach is likely to be used in addressing threats. For example, it is expected that many agency program areas, such as nonnative invasive plant management, fuel management, range management, and wildlife, will contribute to strategies that use vegetation manipulation to address persistent ecosystem and anthropogenic threats.

Threat—Nonnative Plant Invasive Species

Management strategies

- Apply integrated vegetation management practices to manage nonnative invasive plant species, using an interdisciplinary and coordinated approach in designing and implementing projects and treatments.
  - Prioritize areas where management resources are likely to be available to ensure successful management in the long term.
- Use resilience and resistance categories and knowledge of invasive plant distributions to select appropriate management approaches.
  - Protect high quality (relatively weed-free) sagebrush communities with moderate to high sage-grouse habitat probabilities (cells 1B, 1C, 2B, 2C, 3B, 3C):
    - Focus on preventing introduction and establishment of invasive plant species, especially in low resistance areas with high susceptibility to annual grass invasion (in and adjacent to cells 3B, 3C);
    - Avoid seeding introduced forage species (e.g., crested wheatgrass, smooth brome) in postfire rehabilitation or restoration in moderate to high resilience and resistance areas because these species can dominate sagebrush communities; and
    - Practice Early Detection and Rapid Response (EDRR) approaches for emerging invasive species of concern (in and adjacent to cells 1B, 1C, 2B, 2C, 3B, 3C).
  - Where weed populations already exist, seek opportunities to maximize treatment effectiveness by prioritizing restoration within relatively intact sagebrush communities (cells 1B, 1C, 2B, 2C, 3B, 3C). Restoration is likely to be easier at locations in cooler and moister ecological types with higher resilience and resistance.
    - Prioritize sites with sufficient native perennial herbaceous species to respond to release from invasive plant competition;
    - Manage grazing to reduce invasive species and promote native perennial grasses. In the West-Central Semiarid Prairies and other cool and moist areas, manage grazing to reduce crested wheatgrass, Kentucky bluegrass, smooth brome, and other introduced forage species and to promote native cool season perennial grasses (see grazing strategies).
  - Restrict spread of large weed infestations located in lower breeding habitat probability areas (cells 1A, 2A, 3A) to prevent compromising adjacent higher quality habitats (cells 1B, 1C, 2B, 2C, 3B, 3C).

Threat—Conifer Expansion

Management strategies

- Addressing localized conifer expansion requires an interdisciplinary approach and necessarily involves multiple program areas.
  - Apply integrated vegetation management practices to treat conifer expansion, using an interdisciplinary approach in designing projects and treatments.
  - Focus tree removal on early to mid-phase (e.g., Phases I, II) conifer expansion into sagebrush ecological sites to maintain shrub/herbaceous cover.
  - Use prescribed burning cautiously and selectively in moderate to high resilience/resistance (cells 1A, 1B, 2A, 2B) to control conifer expansion.
  - Prioritize for treatment:
    - Areas with habitat characteristics that can support sage-grouse with moderate to high resilience and resistance (cells 1B, 1C, 2B, 2C), especially near leks. (Note: Cells 3B and 3C are generally too warm and dry to support conifers.)
    - Areas where conifer removal will provide connectivity between sagebrush habitats.
    - Areas where sufficient native perennial grasses and forbs exist to promote recovery and limit increases in invasive plant species.

Threat—Wildfire

Management strategies

The wildfire threat is generally addressed through fire operations, fuel management (mechanical treatments, prescribed burning, chemical and seeding treatments), and postfire rehabilitation.

Fire Operations: Protection of areas supporting sagebrush is important for maintaining sagebrush habitat. The types and locations of GRSG habitats have been incorporated into decision support, dispatch, and initial attack procedures, and represent key considerations for fire managers.

(Continued)

Table 1.4—(Continued).

If resources become limiting, consider the following prioritization:

- Fire suppression—typically shifts from low to moderate priority when resilience and resistance categories shift from high to moderate, but varies with large fire risk and landscape condition (cells 1B, 1C, 2B, 2C). In low resilience and resistance areas, the priority shifts from moderate to high as sage-grouse habitat probability increases (cells 3B, 3C). Scenarios requiring high priority may include:
  - Areas of sagebrush that bridge large, contiguous expanses of sagebrush and that are important for providing habitat connectivity;
  - Areas where sagebrush communities have been successfully reestablished through seedings or other rehabilitation investments; and
  - All areas during critical fire weather conditions, where fire growth may move into valued sagebrush communities. These conditions may be identified by a number of products including, but not limited to: Predictive Services National 7-Day Significant Fire Potential products; National Weather Service Fire Weather Watches and Red Flag Warnings; and fire behavior analyses and local fire environment observations.

Fuel Management: Fuel management is a subset of vegetation management. Fuel management activities include treatments that mitigate wildfire risk, modify fire behavior, improve resilience to disturbance and resistance to invasive annual grasses, and protect and restore habitat. Mechanical treatments are typically applied to reduce fuel loading, modify fire behavior, augment fire suppression efforts, or alter species composition consistent with land use plan objectives. Roadside fuel breaks are applied most commonly in MZ III, IV, and V. Prescribed burning is one form of fuel management that may be used to improve habitat conditions or create fuel conditions that limit future fire spread in areas with moderate to high resilience and resistance, but should be considered only after consultation with local biologists and land managers. Chemical and seeding treatments are conducted to reduce invasive plants and change species composition to native, more fire resistant species, or a combination thereof, where native perennial grasses and forbs are depleted. When setting priorities for fuel management, consider the following.

<u>Mechanical Treatments—Conifer Removal</u>

- Conifer removal conducted to decrease woody fuels and reduce the loss of large, contiguous sagebrush stands are high priority in areas with high GRSG breeding habitat probabilities and moderate to high resilience and resistance (cells 1B, 1C, 2B, 2C), and shift to low in areas with low breeding habitat probabilities (cells 1A, 2A). In these areas, the focus is primarily on conifer expansion areas with sufficient native perennial understory species for recovery.
- Management activities may include:
  - Tree removal in early to mid-phase (Phases I, II) postsettlement conifer stands to maintain shrub/herbaceous cover and reduce fuel loads;
  - Tree removal in later phase (Phase III) postsettlement conifer stands to reduce risks of large or high severity fires; and
  - Herbicide, seeding associated with mechanical treatments, or both, to reduce invasive species and restore native perennial herbaceous species where native perennial species are depleted.

<u>Mechanical Treatments—Fuel Breaks</u>

Fuel breaks are strategically placed treatments where vegetation is modified in order to change fire behavior, making fire control efforts safer or more effective. Common types of fuel breaks include road maintenance/roadside disking (brown strips), mowed fuel breaks, and vegetative fuel breaks (green strips).

- In areas of low resilience and resistance, fuel breaks may increase in priority as sage-grouse habitat probability increases (cells 3B, 3C). Repeated treatments may be necessary to maintain functional fuel breaks.
- Key management considerations for the design and placement of fuel breaks:
  - Implement where fire managers believe they will benefit suppression efforts;
  - Design at large landscape scales, providing multiple options for fire managers;
  - Design collaboratively with interdisciplinary specialists, private landowners, fire response partners, and other agencies;
  - Include plans for long-term monitoring and maintenance;
  - Design to minimize habitat impacts, including nonnative invasive species introduction and spread, while maximizing potential fire management benefits.
- Key ecological considerations for the design and placement of fuel breaks:
  - Design fuel breaks in an interdisciplinary setting which addresses the need, cumulative effects, alternative treatments, and possible undesired results;
  - Consider ecosystem resilience and resistance and place fuel breaks to minimize catastrophic ecological state changes;
  - Include conservation buffers around sagebrush leks, habitat fragmentation thresholds, and minimum habitat patch sizes;
  - Include the influence on habitat connectivity between seasonal sage-grouse habitats;
  - Follow technical guidance related to recommended design features (see Maestas et al. 2016a).

(Continued)

Table 1.4—(Continued).

Prescribed Fire

Prescribed fire to address the threat of wildfire includes burning to reduce woody biomass resulting from treatments, to control conifer expansion, to reduce hazardous fuels, and to create fuel breaks which augment fire suppression efforts. When setting priorities for prescribed fire, consider the following:

- Consider alternatives to prescribed burning where other treatment alternatives may meet management objectives.
- In low resilience and resistance areas, consider prescribed fire only after consultation with local biologists and land managers and when:
  - Site information, such as state-and-transition models, affirm that the postburn trajectory will lead to functioning sagebrush communities. Most low resilience and resistance areas that receive <12 in/yr (30 cm/yr) of precipitation do not respond favorably to burning (see Miller et al. 2014).
  - Burning is part of multi-stage restoration projects where burning is required to remove biomass following chemical treatments for site preparation or for improved chemical applications.
  - Monitoring data validates that the preburn composition will lead to successful, native plant dominance post-burn
- Use prescribed fire cautiously and selectively in moderate to high resilience and resistance areas, after consulting with local biologists and land managers and assessing site recovery potential and other management options based on the following:
  - Preburn community composition;
  - Probability of invasive species establishment or spread;
  - Historical fire regime, and patch size/pattern to be created by burning;
  - Wildfire risk and desired fuel loading to protect intact sagebrush; and
  - Alternative treatments that may meet objectives.

Chemical Treatment of Nonnative Invasive Plant Species and Seeding

Chemical treatments and seedings are used to decrease invasive species composition and increase native species dominance in areas where native perennial grasses and forbs are insufficient for site recovery. Chemical and seeding treatments may be selectively applied in conjunction with prescribed fire or mechanical treatments. Typically, these treatments are in response to clear evidence of a nonnative invasive species threat. Areas of higher priority for chemical and seeding treatments:

- Lower resistance and resilience cells (2A, 2B, 3A, 3B) lacking the ability for natural recovery;
- Recently disturbed areas where recovery will not occur without chemical or seeding treatments;
- Areas where investments have been made and objectives cannot be attained without chemical or seeding treatments.

Postfire Rehabilitation: General considerations for prioritization of postfire rehabilitation efforts are:

- Priority generally increases as resilience and resistance decrease and habitat probability for sage-grouse increases. High priorities include areas of low to moderate resilience and resistance that (1) lack sufficient native perennial grasses and forbs to recover on their own and (2) have nearby areas still supporting sage-grouse habitat (cells 2B, 2C, 3B, 3C). Areas of low habitat probability for sage-grouse (cells 2A, 3A) are generally lower priority but may become higher priority in areas that support other resource values or that increase connectivity for GRSG populations.
- Areas of higher priority across all cells include:
  - Areas where prefire perennial herbaceous cover, density, and species composition is inadequate for recovery (see Miller et al. 2015);
  - Areas where seeding or transplanting sagebrush is needed to maintain habitat connectivity for sage-grouse;
  - Areas threatened by nonnative invasive plants; and
  - Steep slopes and soils with erosion potential.

Threat—Sagebrush Reduction

Management strategies

- Avoid intentional sagebrush removal (either prescribed fire or mechanical removal) across all areas in the West-Central Semiarid Prairies due to relatively limited sagebrush availability and extended periods of recovery in the region. Many areas are characterized by moderate to moderately low resilience and resistance, and many sagebrush species lack the capacity to resprout.
- Use caution when attempting to increase herbaceous perennials by reducing sagebrush dominance through mechanical or chemical treatments in general.
  - Lower resistance and resilience areas are prone to annual grass increases and potential dominance if invasive annual grasses exist in the area before treatment.
  - Pretreatment densities of 2 to 3 native perennial bunch grasses per square meter are often necessary for successful increases in perennial herbaceous plants and for suppression of invasive annual grasses after treatment in lower resistance and resilience areas (Miller et al. 2014, 2015).

(Continued)

Table 1.4—(Continued).

Threat—Climate Change

Management strategies

- Continue to use best management practices where effects of climate change and its interactions with stressors are expected to be relatively small and knowledge and management capacity are high.
- Consider proactive management actions to help ecosystems transition to new climatic regimes where climate change and stressor interactions are expected to be severe.
- Practice drought adaptation measures such as reduced grazing during droughts, conservation actions to facilitate species persistence, and seeding and transplanting techniques more likely to work during drought. Consider developing formal drought management plans for livestock grazing.
- Anticipate and respond to species declines such as may occur on the southern or warmer edges of their geographic range.
- Favor genotypes for seeding and out-planting that are better adapted to future conditions because of pest resistance, broad tolerances, or other characteristics.
- Increase diversity of plant materials for restoration activities to provide those species or genotypes likely to succeed.
- Protect future-adapted regeneration from inappropriate livestock grazing.
- Monitor transition zones between climatic regimes (the edges) to provide advanced warning of range shifts. Plant community shifts that affect management decisions often occur between Major Land Resource Areas or level III ecoregions.

Threat—Cropland Conversion

Management strategies

- Secure Conservation Easements to maintain existing sagebrush grasslands and sage-grouse habitat and prevent conversion to tillage agriculture. Prioritize all areas supporting moderate to high sage-grouse habitat probability (cells 1B, 1C, 2B, 2C, 3B, 3C) in locations where tillage risk is elevated (see Sage Grouse Initiative, Cultivation Risk layer).
- Secure term leases (e.g., 30 years) to maintain existing sagebrush grasslands and sage-grouse habitat and prevent conversion to tillage agriculture as a secondary strategy to Conservation Easements. Prioritize all areas supporting moderate to high sage-grouse habitat probability (cells 1B, 1C, 2B, 2C, 3B, 3C) especially in locations where tillage risk is elevated (see SGI Cultivation Risk layer).
- Offer alternatives to farming on expired USDA Conservation Reserve Program (CRP) lands through Federal and State programs. Prioritize lands in and around intact habitats (cells 1B, 1C, 2B, 2C, 3B, 3C).
- Encourage enrollment in the USDA CRP or similar programs to return tilled lands to perennial plant communities supporting mixtures of grasses, forbs, and sagebrush where there are benefits to sage-grouse. Prioritize lands in and around intact habitats (cells 1B, 1C, 2B, 2C, 3B, 3C).

Threat—Energy Development

Management strategies

- Avoid development, if feasible, in areas with high breeding habitat probability for sage-grouse and high sagebrush cover (cells 1C, 2C, 3C) and steer development in non-habitat areas (1A, 2A, 3A).
- Minimize habitat fragmentation in areas with moderate and high breeding habitat probabilities for sage-grouse (cells 1B, 2B, 3B, 1C, 2C, 3C).
- For disturbances that remove vegetation and cause soil disturbance, minimize and mitigate impacts (topsoil banking, certified weed-free [including annual bromes] seed mixes, appropriate seeding technologies, and monitoring). Plan for multiple restoration interventions in areas with low resilience and resistance (cells 3B, 3C).
- Minimize or co-locate energy transport corridors (e.g., roads, pipelines, transmission lines) and limit vehicle access, where feasible.
- Maintain resilience and resistance of existing patches of sagebrush habitat by aggressively managing weeds that may require the following management practices (especially important in low resilience and resistant areas—cells 3A, 3B, 3C):
  - Implement a weed management plan that addresses management actions specific to a project area;
  - Use certified weed-free (including annual bromes) gravel and fill material;
  - Assess and treat weed populations, if necessary, prior to surface disturbing activities;
  - Remove mud, dirt, and plant parts from construction equipment;
  - Address weed risk and spread factors in travel management plans;
  - Ensure timely establishment of desired native plant species on reclamation sites;
  - Use locally adapted native seed, whenever possible;
  - Intensively monitor reclamation sites to ensure seeding success, determine presence of weeds, and implement corrective actions as necessary;
  - Use mulch, soil amendments, or other practices to expedite reclamation success when necessary; and
  - Ensure weeds are controlled on stockpiled topsoil.

(Continued)

Table 1.4—(Continued).

Threat—Urban and Exurban Development

Management Strategies

- Secure conservation easements to maintain existing sagebrush stands and sage-grouse habitat. Prioritize areas with high habitat probability for sage-grouse and high sagebrush cover (cells 1C, 2C, 3C).
- Encourage the protection of existing sage-grouse habitat through appropriate land use planning and Federal land sale policies. Steer development toward non-habitat (cells 1A, 2A, 3A) where habitat is unlikely to become suitable through management.

Threat—Livestock Grazing

Management strategies

- Manage livestock grazing to maintain a balance of native perennial grasses (warm or cool season species, or a combination, as described in Ecological Site Descriptions for that area), forbs, and biological soil crusts to allow natural regeneration and to maintain resilience and resistance to invasive plants. Ensure strategies prevent degradation and loss of native cool-season grasses in particular. Areas with low to moderate resilience and resistance may be particularly vulnerable (cells 2A, 2B, 2C, 3A, 3B, 3C).
- Implement grazing strategies that incorporate periodic deferment from use during the critical growth period, especially for cool season grasses, to ensure maintenance of a mixture of native perennial grasses. This strategy is important across all sites, but particularly essential on areas with low to moderate resilience and resistance supporting sage-grouse habitat (cells 2B, 2C, 3B, 3C).
- Ensure grazing strategies are designed to promote native plant communities and decrease nonnative invasive plants. In ephemeral drainages and higher precipitation areas in the West-Central Semiarid Prairies that receive more summer moisture and have populations of nonnative invasive plant species, too much rest may inadvertently favor species such as field brome, Kentucky bluegrass, and smooth brome. Adjustments in timing, duration, and intensity of grazing may be needed to reduce these species.

To support use of the Science Framework, geospatial data, maps, and models are provided through the Bureau of Land Management's (BLM's) Landscape Approach Data Portal (https://landscape.blm.gov/geoportal/catalog/main/home.page) and U.S. Geological Survey's (USGS's) ScienceBase database (https://www.sciencebase.gov/catalog/). USGS is developing a visualization tool that supports use of this information and that when completed will be accessible through the Landscape Approach Data Portal and ScienceBase database.

Updates to the Science Framework

The Science Framework, both Part 1, **science basis and applications,** and Part 2, **management considerations,** is intended to be adaptive and will be updated to highlight potential management considerations as new science and information on focal species and habitats become available. The mechanism for providing updates is being developed and is likely to include Fact Sheets and webinars developed with partner research and management agencies and organizations. Updates will be linked to periodic updates of the Western Association of Fish and Wildlife Agencies' (WAFWA's) Sagebrush Science Initiative and Sagebrush Conservation Strategy (table 1.1). Updates will be numbered to show the relationship to Part 1, Part 2, and the broader Sagebrush Conservation Strategy and will be housed on the BLM's Landscape Approach Data Portal, the Great Basin Fire Science Exchange website (http://greatbasinfirescience.org/), and USGS's ScienceBase database.

Updates to the Science Framework are expected to address the sagebrush biome, mid-, and local scales and may include new information, science, and analyses that were not included in this version. Updates to the Science Framework could be informed by State Heritage databases and the results of new research conducted as part of implementation of the Actionable Science Plan (IRFMSASPT 2016) and other ongoing research efforts. The State Wildlife Action Plans provide a resource for more detailed information for the Science Framework at the State level, while the Science Framework provides a resource for Wildlife Action Plan revisions by the individual States. Science synthesized to support the WAFWA Sagebrush Conservation Strategy or during development of NEPA analyses to support management decisions could also be considered for inclusion.

References

Allen, C.R.; Fontaine, J.J..; Pope, K.L.; [et al.]. 2011. Adaptive management for a turbulent future. Journal of Environmental Management. 92: 1339–1345.

Angeler, D.G.; Allen, C.R. 2016. Quantifying resilience. Journal of Applied Ecology. 53: 617–624.

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017a. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications. Gen. Tech. Rep. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p.

Chambers, J.C.; Beck, J.L.; Campbell, S.; [et al.]. 2016. Using resilience and resistance concepts to manage threats to sagebrush ecosystems, Gunnison sage-grouse, and Greater sage-grouse in their eastern range: A strategic multi-scale approach. Gen. Tech. Rep. RMRS-GTR-356. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 145 p.

Chambers, J.C.; Bradley, B.A.; Brown, C.A.; [et al.]. 2014a. Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. invasion in the cold desert shrublands of western North America. Ecosystems. 17: 360–375.

Chambers, J.C.; Maestas, J.D.; Pyke, D.A.; [et al.]. 2017b. Using resilience and resistance concepts to manage persistent threats to sagebrush ecosystems and Greater sage-grouse. Rangeland Ecology and Management. 70: 149–164. https://www.treesearch.fs.fed.us/pubs/53742

Chambers, J.C.; Miller, R.F.; Board, D.I.; [et al.]. 2014b. Resilience and resistance of sagebrush ecosystems: Implications for state and transition models and management treatments. Rangeland Ecology and Management. 67: 440–454.

Chambers, J.C.; Pyke, D.A.; Maestas, J.D.; [et al.]. 2014c. Using resistance and resilience concepts to reduce impacts of annual grasses and altered fire regimes on the sagebrush ecosystem and sage-grouse—A strategic multi-scale approach. Gen. Tech. Rep. RMRS-GTR-326. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 73 p. https://www.treesearch.fs.fed.us/pubs/46329

Coates, P.S.; Ricca, M.A.; Prochazka, B.G.; [et al.]. 2016. Wildfire, climate, and invasive grass interactions negatively impact an indicator species by reshaping sagebrush ecosystems. Proceedings of the National Academy of Science. 113: 12745–12750.

D'Antonio, C.M.; Thomsen, M. 2004. Ecological resistance in theory and practice. Weed Technology. 18: 1572–1577.

Davies, K.W.; Boyd, C.S.; Beck, J.L.; [et al.]. 2011. Saving the sagebrush sea: An ecosystem conservation plan for big sagebrush plant communities. Biological Conservation. 144: 2573–2584.

Doherty, K.E.; Evans, J.S.; Coates, P.S.; [et al.]. 2016. Importance of regional variation in conservation planning: A range-wide example of the Greater sage-grouse. Ecosphere. 7: Article e01462.

Endangered Species Act of 1973; 16 U.S.C. 1531-1536, 1538-1540. https://www.gpo.gov/fdsys/pkg/USCODE-2012-title16/html/USCODE-2012-title16-chap35-sec1531.htm. [Accessed May 24, 2018].

Folke, C.; Carpenter, S.; Walker, B.; [et al.]. 2004. Regime shifts, resilience, and biodiversity in ecosystem management. Annual Review of Ecology and Systematics. 35: 557–581.

Holling, C.S. 1973. Resilience and stability in ecological systems. Annual Review of Ecology and Systematics. 4: 1–23.

Interagency Greater Sage-Grouse Disturbance and Monitoring Subteam [IGSDMS]. 2014. The greater sage-grouse monitoring framework. Washington, DC: U.S. Department of the Interior, Bureau of Land Management; U.S. Department of Agriculture, Forest Service, Interagency Greater Sage-Grouse Disturbance and Monitoring Subteam. https://eplanning.blm.gov/epl-front-office/projects/lup/21152/48421/52584/GRSG-FINAL-Monitoring_Framework_20140530.pdf. [Accessed Sept. 12, 2018].

Integrated Rangeland Fire Management Strategy Actionable Science Plan Team [IRFMSASPT]. 2016. The Integrated Rangeland Fire Management Strategy Actionable Science Plan. Washington, DC: U.S. Department of the Interior, Integrated Rangeland Fire Management Strategy Actionable Science Plan Team. 128 p. http://integratedrangelandfiremanagementstrategy.org/wp-content/uploads/2016/10/IRFMS_Actionable_Science_Plan.pdf. [Accessed Sept. 12, 2018].

Johnson, D.H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology. 61: 65–71.

Knick, S.T.; Hanser, S.E.; Miller, R.F.; [et al.]. 2011. Ecological influence and pathways of land use in sagebrush. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 203–251.

Maestas, J.D.; Campbell, S.B.; Chambers, J.C.; [et al.]. 2016. Tapping soil survey information for rapid assessment of sagebrush ecosystem resilience and resistance. Rangelands. 38: 120–128.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2014. A field guide to selecting the most appropriate treatments in sagebrush and pinyon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322-rev. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 68 p.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2015. A field guide for rapid assessment of post-wildfire recovery potential in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-338. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 70 p.

Miller, R.F.; Knick, S.T.; Pyke, D.A.; [et al.]. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. In: Knick, S.T.; Connelly, J.W. eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 145–185.

Ricca, M.A.; Coates, P.S.; Gustafson, K.B.; [et al.]. 2018. Using indices of species distribution, resilience, and resistance as an ecological currency for conservation planning of greater sage-grouse. Ecological Applications. 28(4): 878–896.

Stiver, S.J.; Rinkes, E.T.; Naugle, D.E.; [et al.], eds. 2015. Sage-grouse habitat assessment framework: a multiscale assessment tool. Technical Reference 6710-1. Denver, CO: U.S. Department of the Interior, Bureau of Land Management; Boise, ID: Western Association of Fish and Wildlife Agencies. https://www.fs.fed.us/sites/default/files/sage-grouse-habitat-assessment-framework.pdf. [Accessed Sept. 12, 2018].

Stiver, S.J.; Apa, A.D.; Bohne, J.R.; [et al.]. 2006. Greater sage-grouse comprehensive conservation strategy. Cheyenne, WY: Western Association of Fish and Wildlife Agencies. 442 p. https://wdfw.wa.gov/publications/01317/. [Accessed Sept. 12, 2018].

Suring, L.H.; Rowland, M.M.; Wisdom, M.J. 2005. Identifying species of conservation concern. In: Wisdom, M.J.; Rowland, M.M.; Suring, L.H., eds. Habitat threats in the sagebrush ecosystem: Methods of regional assessment and applications in the Great Basin. Lawrence, KS: Alliance Communications Group: 150–162.

Thompson, M.P.; Marcot, B.G.; Thompson, F.R., III; [et al.]. 2013. The science of decision making: Applications for sustainable forest and grassland management in the national forest system. Gen. Tech. Rep. GTR-WO-88. Washington, DC: U.S. Department of Agriculture, Forest Service. https://www.fs.usda.gov/treesearch/pubs/44326. [Accessed Sept. 16, 2018].

U.S. Department of the Interior [USDOI]. 2015. An integrated rangeland fire management strategy. Final Report to the Secretary of the Interior. Washington, DC: U.S. Department of the Interior. https://www.forestsandrangelands.gov/rangeland/documents/IntegratedRangelandFireManagementStrategy_FinalReportMay2015.pdf. [Accessed Sept. 12, 2018].

U.S. Department of the Interior [USDOI]. 2018. Secretarial Order No. 3362. Improving habitat quality in western big-game winter range and migration corridors. Washington, DC: U.S. Department of the Interior. https://www.doi.gov/sites/doi.gov/files/uploads/so_3362_migration.pdf. [Accessed Sept. 18, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2010. Endangered and threatened wildlife and plants; 12-month findings for petitions to list the greater sage-grouse (*Centrocercus urophasianus*) as threatened or endangered: Federal Register, 75, 13909–14014. https://www.federalregister.gov/articles/2010/03/23/2010-5132/endangered-and-threatened-wildlife-and-plants-12-month-findings-for-petitions-to-list-the-greater. [Accessed Sept. 19, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2013. Greater sage-grouse (*Centrocercus urophasianus*) conservation objectives: Final Report. Denver, CO: U.S. Department of the Interior, Fish and Wildlife Service. 91 p. https://www.fws.gov/greatersagegrouse/documents/COT-Report-with-Dear-Interested-Reader-Letter.pdf. [Accessed Sept. 19, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2015. Endangered and threatened wildlife and plants; 12-month finding on a petition to list the Greater sage-grouse (*Centrocercus urophasianus*) as an endangered or threatened species. Proposed Rule. Fed. Register 80, 59858-59942. http://www.gpo.gov/fdsys/pkg/FR-2015-10-02/pdf/2015-24292.pdf. [Accessed Sept. 19, 2018].

U.S. Environmental Protection Agency [EPA]. 2016. Western Ecology Division. Level III and IV Ecoregions of the Continental United States. Washington, DC: U.S. Environmental Protection Agency. https://www.epa.gov/eco-research/ecoregions. [Accessed Sept. 12, 2018].

2. ADAPTIVE MANAGEMENT AND MONITORING

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Introduction

Monitoring programs designed to track ecosystem changes in response to both stressors and disturbances use repeated observations of ecosystem attributes. Such programs can increase our understanding of how interactions among resilience to disturbance, resistance to invasive species, and "change agents" including management actions influence resource conditions (or status) and trends and outcomes of conservation and restoration actions. This type of monitoring information provides the basis for adaptive management. The overarching goal of an integrated monitoring and adaptive management program is to reduce the uncertainty in the effectiveness of management actions over time by improving management objectives and strategies to increase the effectiveness of those actions.

An integrated monitoring and adaptive management program includes a series of steps that are repeated over time and are designed to facilitate "learning by doing" (fig. 2.1). A structured decisionmaking process may be useful for developing meaningful objectives, and can aid land managers and stakeholders in examining the context, options, and probable outcomes of decisions through an explicit and repeatable process (Allen et al. 2011; Marcot et al. 2012; Thompson et al. 2013). The first step, **assessment**, involves defining the problem, identifying objectives, and determining evaluation criteria. In the second step, **design**, the alternatives are defined, the consequences and key uncertainties are identified, and tradeoffs are evaluated. Next, the preferred alternative is identified, and the decision is made to **implement** the preferred alternative and management action(s).

Figure 2.1—The primary components of the adaptive management cycle.

Top left: Assessment, Inventory, and Management (AIM) meeting (photo: Emily Karchergis, USDOI Bureau of Land Management). Middle left: Mark Szcztpinski using telemetry to track the movements of Greater sage-grouse (photo: Kenton Rowe. Montana Fish, Wildlife and Parks). Bottom left: Digging a soil pit and describing the soils (photo: Emily Karchergis, USDOI Bureau of Land Management). Right: Monitoring vegetation (photo: Emily Karchergis, USDOI Bureau of Land Management).

Text Box 2.1—Components of Monitoring Objectives

An example monitoring objective is:

- Maintain sagebrush cover greater than 15 percent and less than 25 percent across 70 percent of the sage-grouse nesting and early brood-rearing habitats in the assessment area.

Monitoring objectives should identify:

- The indicator(s) that will be monitored;
  - In this example, the indicator would be ***sagebrush cover***.
- Quantitative benchmark(s) for each indicator;
  - In this example, a range of values from ***15 to 25 percent*** sagebrush cover across ***70 percent of the sage-grouse brood-rearing habitat*** in the assessment area would be used.
- A timeframe for evaluating the indicator(s);
  - In this example, the timeframe is likely to be determined by the ***life of the management plan or strategy.*** However, projects and treatments may have a finite timeframe.
- The geographic scale(s) (likely local to mid-scale) over which the monitoring results will be reported (e.g., treatment area, land use planning area).
  - In this example, the scale would be ***sage-grouse nesting and brood-rearing habitat*** in the assessment area.

For more detailed information, refer to part B of table 2.1

Monitoring is the fourth step and is key to adaptive management. The information from a long-term monitoring program is used to **evaluate** ecological status and trends and whether or not management objectives are being met. That information is then used to **adjust**, as necessary, the management action(s) to meet management objectives. A well-designed and rigorous monitoring program has many components (table 2.1) Together these components are used to estimate the proportion of an area that is or is not meeting certain objectives or standards, and provide an unbiased estimate of environmental conditions and changes for ecosystems, species, and populations. Describing the likely data analysis techniques can help ensure that the sampling design will produce meaningful results.

Elzinga et al. (1998) describe how to establish a monitoring program for plant populations and Hayward and Suring (2013) describe this process for wildlife habitat monitoring. These sources provide the necessary information for developing monitoring programs for other types of resources. Definitions related to developing a monitoring program are in Appendix 1.

Monitoring is most effective for adaptive management when the objectives are clearly defined and are consistent with the broader management objectives for the resource. Text box 2.1 provides an example of a monitoring objective. To determine whether the objectives are being met, specific indicators are identified that can be measured and can account for changes in the resource within a realistic timeframe and budget given the site potential and spatial scale of the area being managed (table 2.1).

Benchmarks are indicator values, or ranges of values that establish desired conditions and are meaningful for management. Benchmarks are used to compare observed indicator values to desired conditions. For example, achieving a benchmark value of plant density may tell the practitioner that a seeding project was successful; failure to achieve it may prompt a reevaluation of seeding methods.

Benchmarks for a given indicator may vary for sites with different biophysical characteristics and ecological potential (e.g., ecological site types). Thus, it may be necessary to group benchmarks for areas with different characteristics within a project area and to include the proportion of the landscape that is required to meet a given benchmark. Without appropriate benchmarks, such values lack context and cannot be used to assess condition or the attainment of management objectives.

Table 2.1—Components of a monitoring program based on Elzinga et al. (1998) and Goldstein et al. (2013).

A. Complete Background Tasks

1. Compile and review existing information on the ecosystems, species, and populations. Ecological models of the relationships among ecosystem or habitat characteristics, species abundance, and management effects can help in developing monitoring objectives and improve interpretation and application of the data.
2. Review existing planning documents describing management objectives, including benchmarks or desired conditions, and planned management actions.
3. Prioritize the ecosystems, species, and populations to be monitored based on existing assessments. These priorities may require periodic reassessment due to changes in threats, management, conflicts, and the interests of outside parties.
4. Assess the resources available for monitoring, including management support, priorities, and people and equipment available.
5. Determine the scale of interest for the monitoring effort, such as the sagebrush biome, the range of a species, certain ecological types, or local scales (e.g., populations in certain management units).
6. Determine the type and intensity of monitoring based on the management objectives.
7. Ensure adequate review of the proposed monitoring program by higher level management and by individuals working in relevant disciplines. For larger programs or highly controversial ecosystems, species, and populations, a team may need to be assembled.

B. Develop Monitoring Objectives

1. Develop monitoring objectives that are consistent with the management objectives.
2. Select indicators that can be used to identify the status and trends of a resource or the effectiveness of a management action.
3. Identify the indicators that are most sensitive and appropriate for measuring status and trends or change toward the management goals or benchmarks.
4. Specify the amount and direction of change that is desired or that can be tolerated for each indicator. This science-based value may include a percentage change, or a target or threshold value.
5. Specify a biologically meaningful timeframe for monitoring, considering the indicators selected, to measure ecosystem and species responses following a management action.
6. Specify the management responses needed if monitoring indicates that the management objectives have or have not been met.

C. Design the Monitoring Methodology

1. Develop the sampling objectives.
2. Determine and map the area to be monitored.
3. Define the sampling unit for each indicator that will be measured.
4. Determine the method of sampling unit placement within the monitoring area. An unbiased estimate of resource status and trends can be gained by incorporating randomization into sampling designs.
5. Determine biologically meaningful monitoring durations, intervals, and frequencies.
6. Design the data sheets for the indicator to be measured.
7. Describe the likely data analyses for the different indicators.
8. Identify the necessary resources required to implement the monitoring plan.
9. Write a monitoring plan that has sufficient details for the monitoring to be repeated over time.

F. Implement Monitoring

1. Collect the data at specified intervals using trained personnel.
2. Analyze the data that are collected after each measurement cycle.
3. Describe what if any monitoring triggers have been passed, or what if any benchmarks have been met during the monitoring cycle.
4. Evaluate monitoring methods, costs, sample sizes, and relevance after each measurement cycle. Conducting a trial run or pilot study can expose problems and allow adjustments in the methodology to increase monitoring effectiveness.

G. Manage, Store, and Report Data

1. Ensure that the data for each measurement cycle are complete, entered into standardized databases, verified, and backed up.
2. Analyze all data collected over the reporting period.
3. Review the results for potential issues with either the data collection protocols or the amount and direction of change occurring in the indicator variables.
4. Compile the data and analyses into reports. For data collected over longer time periods, reports should be developed at regular intervals.

H. Apply Results of Monitoring in an Adaptive Management Context

1. Use monitoring results to adjust priority areas for programs of work and resource allocation.
2. Use monitoring results to inform revisions of Land Use Plans and Amendments.
3. Use monitoring results to assess the effectiveness of management strategies and treatment methods and to guide revisions in these as needed.

Monitoring benchmarks can be established based on the management objectives and current ecological site potential of the area (text box 2.2). For example, the Bureau of Land Management (BLM) has set a number of benchmarks for sagebrush cover and other vegetation characteristics in order to maintain habitat for Greater sage-grouse (*Centrocercus urophasianus*; hereafter, GRSG) (e.g., Stiver et al. 2015). Ecological site descriptions and state-and-transition models provide information on the current ecological states and the likely effects of stressors, disturbances, and management actions and can be used to help determine appropriate management objectives (see text box 7.2) and set meaningful benchmarks.

Environmental **thresholds** (conditions sufficient to modify ecosystem structure and function beyond the limits of ecological resilience that result in transition to alternative states [Briske et al. 2008]) are necessary to provide a clear path for management options or alternatives under adaptive management. Knowledge, or estimates, of environmental **thresholds** is important for establishing monitoring triggers. **Triggers** are levels of environmental conditions that can provide an early warning of possible thresholds and of management changes that may be necessary to maintain the desired environmental conditions (Briske et al. 2008; Goldstein et al. 2013).

Monitoring of the indicators must be repeated over sufficient, predetermined time intervals to detect changes and trends in resource status at the spatial scale of management interest. After each measurement cycle is complete, the data are entered into standardized databases, verified, and backed up. **Analyzing** the monitoring data to assess whether the management objective has been achieved or any thresholds have been crossed is the fifth step in an adaptive management program.

The final step is either continuing or changing management at the scale necessary to achieve the desired response or condition. Natural resource decisions are often complex and made with uncertainty, yet managers and biologists are expected to effectively justify and communicate their decisions. In the context of Part 1 of the Science Framework (Chambers et al. 2017; hereafter, Part 1), monitoring results can be used to adjust priority areas for programs of work and budget allocation, to inform efforts such as Federal land use plans (LUPs) and State Wildlife Action Plan revisions, to assess the effectiveness of management strategies and treatment methods, and to guide improvements.

Text Box 2.2—Information to Consider for Establishing Benchmarks

Sources of information and data that can be used to develop benchmarks in an interdisciplinary team environment to build consensus include:

- Policy (e.g., sage-grouse habitat standards, State water quality standards)
- Ecological site descriptions or state-and-transition models
- Comparable monitoring efforts (e.g., baseline Assessment, Inventory, and Monitoring [AIM] data)
- Scientific literature (e.g., sage-grouse habitat assessments)
- Predicted natural conditions (e.g., ecological models)
- Best professional judgment (e.g., considering local knowledge and best available science together)
- Paired reference sites

Overview of the Types of Monitoring

Monitoring can be divided into two categories. The first category describes the ecological status and trends of management areas, and the second category evaluates how well management objectives are being met in project areas. For the purposes of this document, we define “treatments” as site-specific management actions that directly influence one or more of the four ecosystem attributes that are defined in the next paragraph (e.g., biotic integrity can be influenced by juniper and piñon removals, fuel treatments, or GRSG population size). “Projects” can encompass multiple treatments and may relate to broader-scale landscape objectives. “Management action” is a general term that includes active treatments, but may also include actions such as changing management of livestock grazing or recreational uses.

Regardless of the category of monitoring, four ecosystem attributes are important to monitor for determining ecosystem status of an individual management unit (local scale), an ecoregion or Management Zone (mid-scale), or the sagebrush biome (broad scale). Because these attributes are difficult to measure directly, they must be tracked through multiple indicators (Herrick et al. 2010, 2017).

Soil Stability and Health. Soil is the basic foundation of terrestrial ecosystems. Thus, the attributes of soil stability and soil health (quality) are critical elements for sustaining plant, animal, fungal, and microbial functions.

Hydrologic Function. Hydrologic function of terrestrial systems is closely linked to soil stability and quality. All land types (upland, wetland, and riparian ecosystems) are important for maintaining the capture, storage, and release of water.

Water Flow and Quality. Lentic (still water) and lotic (moving water) ecosystems have unique functions as basic resources for biotic integrity, but their capacity to function properly (e.g., recharge and discharge of water to or from the soil) may be linked to other attributes such as soil stability (e.g., sedimentation) or hydrologic function.

Biotic Integrity. Biotic integrity of the plant, animal, fungal, and microbial components of the ecosystem, whether on land or in water, is closely linked to resilience to disturbance and resistance to invasion. This may often include composition, structure, and function of the community or ecosystem.

Monitoring Ecological Status and Trends (Condition and Change)

Status and trends monitoring aims to understand the current condition of natural resources (status) as well as changes in resource condition over time (trends). This type of monitoring informs adaptive management decisionmaking by revealing whether any triggers or benchmarks in soil stability and health, hydrologic function, water flow and quality, and biotic integrity have been reached and whether subsequent management actions are necessary. Status and trends monitoring in sagebrush ecosystems can address questions about the quality and quantity of habitat, the spatial distribution of observed changes, and when possible, **why** resource conditions are changing over time (see *Validation Monitoring*). Such monitoring is often a subset of a larger program or inventory aimed at a broad set of resources within a particular land ownership or jurisdiction. Ideally, by using standardized protocols across land ownership or jurisdictional boundaries, data can be aggregated to understand changes at multiple scales (Rowland and Vojta 2013). Monitoring may be intensified in areas where more information is needed such as in high-priority GRSG habitat and areas with low resilience and resistance (table 1.3: cells 3B, 3C). Causal associations between resource conditions and drivers of change, such as land management decisions or climate change, can be determined by evaluating information from status and trends monitoring along with spatial information about those drivers and reference or control sites.

An unbiased estimate of resource status and trends can be gained by incorporating randomization into sampling designs across an area of interest and keeping track of other potential influences on monitoring results, such as different detection levels, observers, and environmental conditions, which can be accounted for in the analysis. Finally, this type of monitoring can provide information at multiple scales of interest.

Several monitoring programs have been developed to address status and trends of resources, including the BLM's Assessment, Inventory, and Monitoring (AIM) and the Natural Resources Conservation Service's National Resources Inventory (NRI), both of which use common indicators and protocols; the Forest Service's Forest Inventory and Analysis (FIA) program; and the national Landscape Monitoring Framework, which is part of BLM's AIM strategy. Although AIM and NRI use different measurement techniques from FIA, the sample designs allow for analyses that cross administrative boundaries, provided that appropriate analytical methods are implemented (Patterson et al. 2014). Regional and finer scale monitoring efforts are also implemented through BLM AIM, the National Park Service Inventory and Monitoring Program, National Inventory and Monitoring Initiative (I&M) managed by the National Wildlife Refuge System, and other efforts. These types of monitoring efforts are the recommended means of understanding status and trends of GRSG habitat (e.g., Stiver et al. 2015; USDOI 2014).

Monitoring to Evaluate Management Objectives

To evaluate whether management objectives are being met, measurements can be conducted at local, mid-, and broad scales. The types of monitoring typically used to monitor management objectives, **implementation**, **effectiveness**, and **validation**, are described next.

Implementation Monitoring

Implementation monitoring determines whether planned management decisions, actions, and treatments have been implemented, and whether standards outlined within planning documents were followed or modified. The BLM and Forest Service report on the actions implemented that are described in their LUPs and that relate to decisions aimed at conserving, improving, or restoring sagebrush habitats (USDOI 2014). Initially, this type of monitoring is conducted by planning units. However, given some consistencies in management objectives across planning unit boundaries, this level of monitoring can often be scaled up to the mid-scale.

Effectiveness Monitoring

Effectiveness monitoring assesses the condition of a management action's outcome. Success is typically achieved by meeting predetermined treatment objectives that can be measured against baseline or reference conditions determined by status and trends monitoring, or another desired condition or benchmark as stipulated in the treatment objectives (table 2.1). As an example, effectiveness monitoring may be conducted at the project scale when expanding juniper and piñon or nonnative invasive plants are removed to restore GRSG habitat. Monitoring indicators, such as landscape cover of trees, and the appropriate benchmarks can be used to evaluate whether the effort has reduced tree cover below the response threshold (e.g., less than X% cover across Y% of the monitoring area, which varies regionally) (Baruch-Mordo et al. 2013). Pretreatment levels (baseline) of nonnative invasive plants can be compared to posttreatment levels of perennial native grasses and forbs (e.g., Chambers et al. 2014). If radio-marked GRSG are being monitored in the area of the treatment, the subsequent space or habitat use can be monitored and used to evaluate the efficacy of the treatment. The effectiveness of multiple projects or treatments within the mid-scale can help determine the effectiveness of the management objectives contained within a LUP or other guiding management document. Appropriate landscape-level indicators tied to project objectives provide the opportunity to assess the effectiveness of efforts in achieving conservation goals at the broad scale. This type of monitoring also lends itself to evaluating the effectiveness of and potential benefit achieved from mitigation efforts.

Validation Monitoring

Validation monitoring uses an experimental approach to determine whether the observed outcome is due to the management action. This requires treating some areas and leaving some areas untreated to serve as "controls" for the treated areas, as is done in research and management projects like the Sagebrush Treatment Evaluation Project (http://www.sagestep.org/). The untreated areas are compared to the treated areas to determine whether they differ in meeting the stated objectives. For example, after a wildfire in a Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) ecosystem with low to moderate resilience and resistance, restoration efforts might focus on seeding Wyoming big sagebrush and native perennial bunchgrasses in a randomly selected sample of potential treatment sites. After X years of monitoring ("X" is equal to the time stated in the objectives statement), cover of native perennial bunchgrasses and stem density of sagebrush are measured to determine whether they are trending toward the desired management objective. If the treated sites have higher cover of native perennial bunchgrasses and stem density of sagebrush than the untreated sites, then the management treatment was successful. If the cover and stem density are similar between treated sites and untreated (or control) sites, then the outcome may be attributed to natural successional processes. Due to its relatively high costs and complexity, validation monitoring is most likely to occur at the local scale rather than at mid- or broad scales.

A combination of these monitoring approaches can ensure that management objectives are achieved at multiple spatial scales and that the observed outcome is due to the treatment. These different types of monitoring provide important feedbacks for adaptive management and thus provide further support for incorporating monitoring strategies into the planning or development phase of any project or treatment, including budget planning. Archiving data collected through implementation, effectiveness, and validation monitoring in tools, such as the Land Treatment Digital Library for the BLM (Pilliod and Welty 2013) and the Conservation Efforts Database (USDOI FWS 2014), and analyzing the status and trends can allow managers to learn from past treatments and decide on appropriate management actions in the future.

Standardization of Indicators and Protocols

Adoption of a standardized set of indicators and protocols for collecting indicator data will allow a wide range of users (i.e., managers, landowners, interested public, and researchers) to compare data collected in different areas and for different objectives. The NRCS and BLM currently use common protocols for national and regional monitoring of many rangeland vegetation and soil indicators (Herrick et al. 2010, 2017; Toevs et al. 2011). The Forest Service recently released protocols for standardized wildlife habitat monitoring (Rowland and Vojta 2013), which rely primarily on existing, commonly used sampling methods and datasets. The Integrated Rangeland Fire Management Strategy (IRFMS) (USDOI 2015) provides guidance for working out some of the differences among protocols and indicators to reduce conflicts.

Measuring standardized indicators with consistent protocols allows ground-based data to be scaled-up from local to mid-scales through ground-truthing and validation with remotely sensed data. Provided that data are collected using a randomized sampling design with known methods of stratification, level of effort, and other parameters, data collected from each location or landscape can be weighted in a statistically sound manner and combined with similar data in other areas to obtain cross-site or cross-landscape comparisons with spatial relevance and known levels of error (Patterson et al. 2014).

Rule sets for making data collection decisions are necessary to ensure precise measurement among different field crews (Rowland and Vojta 2013). Herrick et al. (2005) illustrate how rule sets are stipulated. BLM's AIM and NRCS's NRI both use rule sets to standardize measurement decisions. No one rule set is perfect, but rule sets provide a means for collecting consistent data among different observers.

Linking Resilience and Resistance Concepts and Monitoring

Monitoring landscape heterogeneity over time can provide a clearer understanding of how sagebrush dominated landscapes are changing in response to natural ecosystem processes, anthropogenic disturbances, and management actions. Relative resilience to disturbance and resistance to invasive annual grasses influence the responses of sagebrush ecosystems to threats such as wildfire, land uses, and development. Information on resilience and resistance can provide an additional data layer in monitoring programs that can be used to help understand the changes in ecosystem status and trends and the effectiveness of management treatments at broad, mid-, and local scales. The relationships among resilience and resistance, as indicated by soil temperature and moisture regimes, the predominant sagebrush ecological types, and the responses of those ecological types to both disturbance and management, can be used to inform monitoring designs, to help develop benchmarks and triggers for changes in management, and to determine appropriate changes in management strategies and treatments (Part 1, section 6).

By stratifying monitoring across resilience and resistance categories, the range of potential responses to management actions can be captured. Even if a monitoring program is already in place, including resilience and resistance as a factor in the analyses may still provide useful information and context on the effects of resilience and resistance given adequate sample sizes in the different categories.

Generalized state-and-transition models developed for the dominant ecological types in both the western and eastern parts of the sagebrush biome and GRSG range, provide information on the alternative states for these types, the effects of ecosystem threats and management actions on these states, and the potential restoration pathways (Part 1, Appendices 5 and 6). Examples of how to apply resilience and resistance concepts are provided for areas with different ecological types and threats (Part 1, section 9.2).

Using the Science Framework Approach to Inform Monitoring

The Science Framework, Part 1 gives an approach for prioritizing areas for management and determining effective management strategies based on: (1) the predominant threats, (2) the likely response of an area to disturbance or stress due to threats or management actions (i.e., resilience to disturbance and resistance to invasive annual grasses), and (3) the capacity of an area to support target species or resources.

The geospatial data layers and analyses used in the approach are described in Part 1, sections 8.1 and 8.2, and can be used to help design monitoring programs and interpret monitoring results. Analyses are generally conducted at the mid-scale because of similarities in ecoregional climate, soil properties, resilience to disturbance, and resistance to invasive annual grasses. Key data layers include resilience and resistance as indicated by soil temperature and moisture regimes, GRSG breeding habitat probabilities, habitats of other sagebrush dependent species, and the primary threats for the ecoregions or Management Zones. At the mid- to local scale higher resolution geospatial data that are specific to the assessment area (i.e., the best available data) are used in the analyses. Interpretations of these analyses for monitoring programs, based on the Science Framework approach for GRSG (tables 1.3, 1.4), follow a similar approach and can be used for other species at risk as well as priority resources.

Monitoring areas of high GRSG breeding habitat probability (table 1.3: cells 1C, 2C, 3C) provides information on whether these areas are retaining their composition, structure, and function as GRSG habitat. Protective management is used to retain resilience and resistance in these areas. Monitoring for status and trends and using the Early Detection and Rapid Response approach (EDRR) (USDOI 2016) for nonnative invasive plants can help ensure that invasive plants do not increase and thereby degrade these high value sites. Monitoring areas of low resilience and resistance with high GRSG breeding habitat probabilities is especially important because these areas are at high risk of habitat loss from wildfire and potential for conversion to invasive annual grasses (table 1.3: cell 3C). Regardless of an area's resilience and resistance, implementation and effectiveness monitoring are used to assess treatment outcomes and determine whether follow-up management is needed.

Areas with moderate breeding habitat probabilities are a focus for habitat improvements (table 1.3: cells 1B, 2B, 3B). Treated areas within GRSG habitat are often moderate to high priority for monitoring because habitat improvements resulting from treatments could translate into increased use or improved demographic indices (e.g., population trends, survival), or both, for GRSG. Treated areas typically undergo EDRR, implementation, and effectiveness monitoring to ensure that the treatments were implemented as planned, objectives of the management action(s) are met, and an understanding of the effectiveness of the outcome is gained (Mulder et al. 1999; Noss and Cooperrider 1994).

Monitoring areas with low GRSG breeding habitat probabilities and low resistance and resilience can provide information on continued changes in composition, structure, and function, but is generally lower priority unless other at-risk species or management concerns are identified in these areas (table 1.3: cells 1C, 2C, 3C). Areas of low resilience and resistance and with low breeding habitat probabilities that are currently dominated by invasive annual grasses may be given the lowest priority for monitoring (table 1.3: cell 3A). These areas of invasive annual grasses have gaps in function and structure, which can hinder management efforts toward reference conditions. This reduces the number of adaptive management options.

Monitoring Change in Landscape Status and Trend

Landscape monitoring is an important aspect of land management that provides a way to examine the big picture—it gives information on ecosystem processes, habitat characteristics, and species distributions and movements that operate beyond the scope of management units and land ownership boundaries. This type of monitoring can also provide information on the landscape characteristics of areas with different resilience and resistance and the response of these areas to ecosystem threats and management actions. There are several types of indicators (e.g., indicators developed to map broad spatial patterns for different vegetation types) that can be used to monitor landscapes and evaluate: (1) change in environmental conditions and ecosystem structure, process, and function; (2) cumulative effects of management activities; and (3) crossing of thresholds over broad areas. These indicators can measure physical characteristics on the ground and connect them to ecological processes. They may also be used as surrogates for environmental conditions that cannot be measured directly. Typically, these types of indicators are calculated using spatial data within a specified assessment area (e.g., ecoregion, Management Zone, jurisdictional boundary). The resulting measurements from monitoring these indicators may differ based on the size of the assessment (broad, mid-, and local). Thus, it is important to measure the appropriate indicators at the appropriate resolution and scale to provide comprehensive, integrated monitoring for the scale of interest.

Landscape Indicators

There are certain indicators useful for monitoring and quantifying landscape heterogeneity and change at multiple scales. Examples of indicators that can be monitored and quantified across an assessment area to identify natural and human-caused change over time are: percent cover of the vegetation types occurring across the assessment area, the average cover of all vegetation or habitat patch size, patch size coefficient of variation, the average and range of distance to neighboring patches, vegetation or habitat patch richness, and patch edge contrast or density (Cushman et al. 2008, 2013a,b; Goldstein et al. 2013). These indicators measure various aspects of landscape structure, but when analyzed together can offer a comprehensive evaluation of change in landscape pattern, land cover class conversion, and fragmentation across the assessment area. For example, an aggregate of local-scale monitoring data and remote sensing data (e.g., National Gap Analysis [GAP], Landscape Fire and Resource Management Planning Tools [LANDFIRE], National Land Cover Database [NLCD], Geospatial Multi-Agency Coordination [GeoMAC] Wildland Fire Support Tools) can be examined to quantify sagebrush landscape pattern, heterogeneity, and change over time independently or relative to other landscape class mean patch sizes. These indicators, when evaluated within or across land cover classes, quantified over specific time intervals, provide a measure of how sagebrush patches have changed (expanded or contracted) in response to natural ecosystem processes, anthropogenic disturbances, and management actions over time.

Depending on the management question, distance to neighboring vegetation patches may increase or decrease over time. This indicator combined with other landscape indicators (e.g., change in average sagebrush patch size) will help provide information on whether the assessment area is meeting management objectives and benchmarks and avoiding triggers. For instance, an increase in the average nearest neighboring patch distance along with a decrease in the average sagebrush patch size over time typically indicates an increase in fragmentation of sagebrush across the assessment area. In contrast, a decrease in distance to neighboring sagebrush patches combined with an increase in average sagebrush patch size may indicate successful restoration and a decrease in fragmentation across the assessment area. The landscape indicators monitored should be identified carefully and should address the management objectives. The use of consistent landscape indicators across jurisdictional boundaries will improve our understanding of overall landscape change at the biome scale as well as provide the information needed by land management agencies to understand how management practices are effective in meeting management goals.

Landscape Monitoring of Habitats

Habitats are spatially structured, forming patterns at multiple scales. These patterns may influence wildlife behavior and use of space and influence population dynamics and community structure (Johnson et al. 1992). For all species, habitat must have sufficient size and proximity of resource patches to: (1) support reproduction, (2) facilitate dispersal, and (3) maintain metapopulation structure (if that is a characteristic of the species) (Cushman et al. 2013a). To monitor landscape-level changes within the sagebrush ecosystem with a focus on wildlife-specific species indicator data, landscape indicators can be used to quantify how habitat changes over time in response to management decisions and natural ecosystem processes. For example, much information is available on landscape indicators for GRSG, such as habitat intactness (Aldridge et al. 2008; Wisdom et al. 2011); breeding habitat probability (Doherty et al. 2016); landscape genetics (Cushman et al. 2013b; Row et al. 2015); habitat patch size, habitat connectivity, and networks; ecological minimums (thresholds) (Crist et al. 2015; Knick and Hanser 2011; Meinke et al. 2009); edge effects (Coates et al. 2014; Howe et al. 2014); and distance to water (Donnelly et al. 2016). Goldstein et al. (2013) provide an example monitoring plan for GRSG habitat monitoring at multiple scales, with sagebrush patch size, sagebrush canopy cover, and habitat connectivity selected as landscape-level habitat monitoring indicators. Spatial data from remote sensing efforts (e.g., NLCD, LANDFIRE, GeoMAC, GAP), along with monitoring data collected on the ground, can be used to analyze these indicators and quantify the amount of habitat area and connectivity lost or gained due to habitat conversion or natural succession (Goldstein et al. 2013).

Disturbance, Reclamation, and Restoration

Tracking and measuring the influence of persistent ecosystem and anthropogenic threats, separately and in combination at broad scales, can provide useful information on whether or not management objectives for sagebrush ecosystems are met. Overlaying information on resilience and resistance can aid in the interpretation of management outcomes. For example, the ability to achieve successful reclamation and subsequent restoration will differ for ecosystems with different resilience and resistance. Monitoring can help inform where to prioritize management and conservation actions, what to expect under certain measured conditions, and what the best indicators of overall management effectiveness are.

Classifying habitat restoration, vegetation treatments for fuel management, and other types of vegetation treatments separately from land cover classifications used in vegetation mapping (e.g., Homer et al. 2015) can allow these treatments to be monitored and evaluated over time at the landscape scale. This can provide the basis for determining whether an area has recovered, whether benchmarks (or triggers) at the landscape level (ecosystem or species-specific) have been exceeded, and whether management actions are needed. For example, triggers associated with habitat thresholds, such as mean distance to, and density of, oil and gas wells (Doherty et al. 2008; Holloran et al. 2005; Lyon and Anderson 2003; Naugle et al. 2011; Walker et al. 2007), have guided science-based land use and management decisions in recently amended BLM and Forest Service LUPs, and some State plans.

Recent work has shown variation in threshold responses to disturbance, such as canopy cover and the human disturbance index, across the different sage-grouse Management Zones, indicating that a one-size-fits-all approach to setting thresholds is seldom appropriate (Doherty et al. 2016). These authors (Doherty et al. 2016, p. 23) stated that "when potential for conflict is high and thresholds are extrapolated into novel landscapes, clearly defined adaptive management goals and monitoring systems would be prudent." This recommendation highlights the tension between using research conducted in small parts of the sagebrush biome and the extrapolation of those results to new areas to justify the claim of treatment effectiveness in other parts of the area. This emphasizes the need to have monitoring systems in place to understand whether the results are applicable in the ecological context of the system in which the treatments are occurring. Information on resilience and resistance has provided the means for developing appropriate management strategies based on the likely response of ecosystems to both disturbance and management actions. Monitoring ecosystem threats and land use and development threats at the same time will aid in determining the effectiveness of on-the-ground conservation actions, understanding the reasons for changes in the landscape, and designing more effective management strategies.

Linking Efforts to Identify GRSG Population and Habitat Thresholds

Certain population response thresholds have been defined for managing GRSG habitat within State and Federal plans and in the scientific literature (Doherty et al. 2016; Knick et al 2013; Manier et al. 2014). Disturbance data collected at the project scale can be aggregated within habitat management designations across a landscape. These data can be used to determine whether adaptive management triggers associated with thresholds (such as disturbance caps and limitations of disturbance density specified in the Federal LUPs, and some State plans) have been met or exceeded that prompt actions or decisions by the appropriate agencies responsible for land management. By building on the GRSG Monitoring Framework (IGSDMS 2014) and the Sage-Grouse Habitat Assessment Framework (Stiver et al. 2015), adaptive management triggers tied to population levels or GRSG habitat, or both, have been developed for each LUP. For GRSG, individual and population responses to road densities, oil and gas densities, and other factors (Knick and Hanser 2011; Knick et al. 2013; Manier et al. 2014) are available and can be assessed to gain a better understanding of habitat and GRSG population conditions relative to these specified thresholds as well as offer a more of the landscape-level perspective.

Establishing a robust monitoring program or strategy that informs clearly defined management objectives is paramount to a meaningful adaptive management process. Monitoring the outcomes of management actions allows land managers and resource specialists to gain the necessary knowledge and information to locate treatments and projects in areas where they are more likely to be effective. Monitoring outcomes is essential for understanding the effectiveness of management actions in sustaining sagebrush ecosystems over time. In the aggregate, these efforts can improve resilience and resistance across the sagebrush biome and increase the return on conservation investments.

References

Aldridge, C.L.; Nielsen, S.E.; Beyer, H.L.; [et al.]. 2008. Range-wide patterns of greater sage-grouse persistence. Diversity and Distributions. 14: 983–994.

Allen, C.R.; Fontaine, J.J..; Pope, K.L.; [et al.]. 2011. Adaptive management for a turbulent future. Journal of Environmental Management. 92: 1339–1345.

Baruch-Mordo, S.; Evans, J.S.; Severson, J.P.; [et al.]. 2013. Saving sage-grouse from the trees: A proactive solution to reducing a key threat to a candidate species. Biological Conservation. 167: 233–241.

Briske, D.D.; Bestelmeyer, B.T.; Stringham, T.K., [et al.]. 2008. Recommendations for development of resilience-based state-and-transition models. Rangeland Ecology and Management. 61(4): 359–367.

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications. Gen. Tech. Rep. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p.

Chambers, J.C.; Miller, R.F.; Board, D.I.; [et al.]. 2014. Resilience and resistance of sagebrush ecosystems: Implications for state and transition models and management treatments. Rangeland Ecology and Management. 67: 440–454.

Coates, P.S.; Howe, K.B.; Casazza, M.L.; [et al.]. 2014. Landscape alterations influence differential habitat use of nesting buteos and ravens within sagebrush ecosystems: Implications for transmission line development. Condor: Ornithological Applications. 116: 341–356.

Crist, M.R.; Knick, S.T.; Hanser, S.E. 2015. Range-wide network of priority areas for Greater sage-grouse—A design for conserving connected distributions or isolating individual zoos? Open-file Report 2015-1158. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 34 p.

Cushman, S.A.; Landguth, E.L.; Flather, C.H. 2013a. Evaluating population connectivity for species of conservation concern in the American Great Plains. Biodiversity and Conservation. 22: 2583–2605.

Cushman, S.A.; McGarigal, K.; Neel, M.C. 2008. Parsimony in landscape metrics: Strength, universality, and consistency. Ecological Indicators. 8: 691–703.

Cushman, S.A.; Mersmann, T.J.; Moisen, G.G.; [et al.]. 2013b. Chapter 5. Using habitat models for habitat mapping and monitoring. In: Rowland, M.M.; Vojta, C.D., tech. eds. A technical guide for monitoring wildlife habitat. Gen. Tech. Rep. WO-GTR-89. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 14 p.

Doherty, K.E.; Evans, J.S.; Coates, P.S.; [et al.]. 2016. Importance of regional variation in conservation planning: A rangewide example of the Greater sage-grouse. Ecosphere 7: e01462.

Doherty, K.E.; Naugle, D.E.; Walker, B.L.; [et al.]. 2008. Greater sage-grouse winter habitat selection and energy development. Journal of Wildlife Management. 72: 187–195.

Donnelly, J.P.; Naugle, D.E.; Hagen, C.A.; [et al.]. 2016. Public lands and private waters: Scarce mesic resources structure land tenure and sage-grouse distributions. Ecosphere. Article 7: e01208.

Elzinga, C.L.; Salzer, D.W.; Willoughby, J.W. 1998. Measuring and monitoring plant populations. BLM Tech. Ref. 1730-1. Denver, CO: U.S. Department of the Interior, Bureau of Land Management, National Applied Resource Sciences Center. 477 p. http://www.blm.gov/nstc/library/pdf/MeasAndMon.pdf. [Accessed May 22, 2018].

Goldstein, M.I.; Suring, L.H.; Vojta, C.D.; [et al.]. 2013. Chapter 10. Developing a habitat monitoring program: Three examples from National Forest planning. In: Rowland, M.M.; Vojta, C.D., tech. eds. A technical guide for monitoring wildlife habitat. Gen. Tech. Rep. WO-GTR-89. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 74 p.

Hayward, G.D.; Suring, L.H. 2013. Chapter 2. Selection of key habitat attributes for monitoring. In: Rowland, M.M.; Vojta, C.D., tech. eds. A technical guide for monitoring wildlife habitat. Gen. Tech. Rep. WO-GTR-89. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 20 p.

Herrick, J.E.; Van Zee, J.W.; Havstad, K.M.; [et al.]. 2005. Monitoring manual for grassland, shrubland and savanna ecosystems. Volumes 1 and 2. Las Cruces, NM: U.S. Department of Agriculture, Agricultural Research Service, Jornada Experimental Range.

Herrick, J.E.; Van Zee, J.W.; McCord, S.E.; [et al.]. 2017. Monitoring manual for grassland, shrubland, and savanna ecosystems. 2nd ed. Volume I: Core methods. Las Cruces, NM: U.S. Department of Agriculture, Agricultural Research Service, Jornada Experimental Range.

Herrick, J.E.; Wills, S.; Karl, J.; [et al.]. 2010. Terrestrial indicators and measurements: Selection process and recommendations. Washington, DC: U.S. Department of Agriculture and U.S. Department of the Interior. https://jornada.nmsu.edu/files/AIM_Terrestrial_Indicators_Selection.pdf. [Accessed May 23, 2018].

Holloran, M.J.; Heath, B.J.; Lyon, A.G.; [et al.]. 2005. Greater sage-grouse nesting habitat selection and success in Wyoming. Journal of Wildlife Management. 69: 638–649.

Homer, C.G.; Xian, G; Aldridge, C.L.; [et al.]. 2015. Forecasting sagebrush ecosystem components and greater sage-grouse habitat for 2050: Learning from past climate patterns and Landsat imagery to predict the future. Ecological Indicators. 55: 131–141.

Howe, K.B.; Coates P.S.; Delehanty, D.J. 2014. Selection of anthropogenic features and vegetation characteristics by nesting common ravens in the sagebrush ecosystem. Condor: Ornithological Applications. 116: 35–49.

Interagency Greater Sage-Grouse Disturbance and Monitoring Subteam [IGSDMS]. 2014. The greater sage-grouse monitoring framework. Washington, DC: U.S. Department of the Interior, Bureau of Land Management; U.S. Department of Agriculture, Forest Service, Interagency Greater Sage-Grouse Disturbance and Monitoring Subteam. https://eplanning.blm.gov/epl-front-office/projects/lup/21152/48421/52584/GRSG-FINAL-Monitoring_Framework_20140530.pdf. [Accessed Sept. 12, 2018].

Johnson, A.R.; Wiens, J.A.; Milne, B.T; [et al.]. 1992. Animal movements and population-dynamics in heterogeneous landscapes. Landscape Ecology. 7: 63–75.

Knick, S.T.; Hanser, S.E. 2011. Connecting pattern and process in Greater sage-grouse populations and sagebrush landscapes. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 383–406.

Knick, S.T.; Hanser, S.E.; Preston, K.L. 2013. Modeling ecological minimum requirements for distribution of Greater sage-grouse leks: Implications for population connectivity across their western range, U.S.A. Ecology and Evolution. 3: 1539–1551.

Lyon, A.G.; Anderson, S.H. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin. 31: 486–491.

Manier, D.J.; Bowen, Z.H.; Brooks, M.L.; [et al.]. 2014. Conservation buffer distance estimates for Greater sage-grouse—A review. Open-File Report 2014-1239. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 14 p. https://doi.org/10.3133/ofr20141239. [Accessed May 22, 2018].

Marcot, B.G.; Thompson, M.P.; Runge, M.C.; [et al.]. 2012. Recent advances in applying decision science to managing national forests. Forest Ecology and Management. 285: 123–132.

Meinke, C.W.; Knick, S.T.; Pyke, D.A. 2009. A spatial model to prioritize sagebrush landscapes in the Intermountain West (U.S.A.) for restoration. Restoration Ecology. 17: 652–659.

Mulder, B.S.; Noon, B.R.; Spies, T.A.; [et al.], tech. coords. 1999. The strategy and design of the effectiveness monitoring program for the Northwest Forest Plan. Gen. Tech. Rep. PNW-GTR-437. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 138 p.

Naugle, D.E.; Doherty, K.E.; Walker, B.L.; [et al.]. 2011. Energy development and Greater sage-grouse. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 489–504.

Noss, R.F.; Cooperrider, A.Y. 1994. Saving nature's legacy. Washington, DC: Island Press. 433 p.

Patterson, P.L.; Alegria, J.; Jolley, L.; [et al.]. 2014. Multi-agency Oregon pilot: Working towards a National Inventory and Assessment of rangelands using onsite data. Gen. Tech. Rep. RMRS-GTR-317. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Pilliod, D.S.; Welty, J.L. 2013. Land Treatment Digital Library. U.S. Geological Survey Data Series 806. http://ltdl.wr.usgs.gov/. [Accessed May 22, 2018].

Row, J.R.; Oyler-McCance, S.J.; Fike, J.A.; [et al.]. 2015. Landscape characteristics influencing the genetic structure of Greater sage-grouse within the stronghold of their range: A holistic modeling approach. Ecology and Evolution. 5: 1955–1969.

Rowland, M.M.; Vojta, C.D., tech. eds. 2013. A technical guide for monitoring wildlife habitat. Gen. Tech. Rep. GTR-WO-89. Washington, DC: U.S. Department of Agriculture, Forest Service. https://www.fs.usda.gov/treesearch/pubs/45213. [Accessed Sept. 16, 2018].

Stiver, S.J.; Rinkes, E.T.; Naugle, D.E.; [et al.], eds. 2015. Sage-grouse habitat assessment framework: a multiscale assessment tool. Tech. Ref. 6710-1. Denver, CO: U.S. Department of the Interior, Bureau of Land Management and Western Association of Fish and Wildlife Agencies. https://www.fs.fed.us/sites/default/files/sage-grouse-habitat-assessment-framework.pdf. [Accessed Sept. 16, 2018].

Thompson, M.P.; Marcot, B.G.; Thompson, F.R., III; [et al.]. 2013. The science of decision making: applications for sustainable forest and grassland management in the national forest system. Gen. Tech. Rep. GTR-WO-88. Washington, DC: U.S. Department of Agriculture, Forest Service. https://www.fs.usda.gov/treesearch/pubs/44326. [Accessed Sept. 16, 2018].

Toevs, G.R.; Karl, J.W.; Taylor, J.J.; [et al.]. 2011. Consistent indicators and methods and a scalable sample design to meet assessment, inventory, and monitoring information needs across scales. Rangelands. 33: 14–20.

U.S. Department of the Interior [USDOI]. 2014. The Greater sage-grouse monitoring framework. 48 p. Washington, DC: U.S. Department of the Interior. https://eplanning.blm.gov/epl-front-office/projects/lup/21152/48421/52584/GRSG-FINAL-Monitoring_Framework_20140530.pdf. [Accessed May 22, 2018].

U.S. Department of the Interior [USDOI]. 2015. SO-3336: The final report: An integrated rangeland fire management strategy. Final Report to the Secretary of the Interior. Washington, DC: U.S. Department of the Interior. https://www.forestsandrangelands.gov/rangeland/documents/IntegratedRangelandFireManagementStrategy_FinalReportMay2015.pdf. [Accessed May 22, 2018].

U.S. Department of the Interior [USDOI]. 2016. Safeguarding America's lands and waters from invasive species: A national framework for early detection and rapid response. Washington, DC: U.S. Department of the Interior. 55 p. https://www.doi.gov/sites/doi.gov/files/National%20EDRR%20Framework.pdf. [Accessed May 22, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2014. Conservation Efforts Database. Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service. https://conservationefforts.org/. [Accessed Sept. 15, 2018].

Walker, B.L.; Naugle, D.E.; Doherty, K.E. 2007. Greater sage-grouse population response to energy development and habitat loss. Journal of Wildlife Management. 71: 2644–2654.

Wisdom, M.J.; Meinke, C.W.; Knick, S.T.; [et al.]. 2011. Factors associated with extirpation of sage-grouse. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 451–474.

3. CLIMATE ADAPTATION

Jeanne C. Chambers, Louisa Evers, and Linda A. Joyce

Introduction

Management actions that enable adaptation to climate change and promote resilience to disturbance are becoming increasingly important in the sagebrush biome. In recent decades temperatures have increased, growing seasons have lengthened, and in many areas the timing and amount of precipitation has changed (Chambers et al. 2017 [hereafter, Part 1], section 4; Kunkel et al. 2013a,b,c). Global climate change models are used to project future changes in temperature and precipitation based on relative concentration pathways of likely emissions of carbon dioxide (CO_2) and other trace gases and information on the Earth's surfaces and oceans. These models project continued temperature increases and additional changes in precipitation throughout the remainder of the century, although the magnitude and rate of change differ based on the relative concentration pathway used (Part 1, section 4; Kunkel et al. 2013a,b,c).

Continued changes in climate are likely to influence the distributions of native species (Bradley 2010; Homer et al. 2015; Schlaepfer et al. 2012c; Still and Richardson 2015), invasive annual grasses (Bradley et al. 2016), fire regimes (Abatzoglou and Kolden 2013; Littell et al. 2009; Westerling et al. 2014), and insects and disease (Bentz et al. 2016). Snowpacks are declining in many areas (Mote and Sharp 2016), droughts are becoming more severe (Cook et al. 2015; Prein et al. 2016), and the length of the fire season and duration of extreme fire weather is increasing (Abatzoglou and Kolden 2013; Littell et al. 2009; Westerling et al. 2014; but see also McKenzie and Littell 2017). Reducing ecosystem vulnerability, or the degree to which a system is susceptible to the adverse effects of climate change, including climate variability and extremes (IPCC 2014), will require scientific guidance and agency direction to enable climate adaptation planning and implementation across scales.

Climate adaptation, the process of adjusting to actual or expected changes in climate, is an important consideration when developing management strategies in the face of climate change. The focus of climate adaptation is to moderate or avoid harm or to exploit beneficial opportunities (IPCC 2014). Adaptation can be **incremental**, where the objective is to maintain the integrity of a system or process at a given scale. Climate scientists anticipate that climate will continue to change throughout the 21st century due to continued accumulation of greenhouse gases in the atmosphere. As the climate warms, ecosystems may not persist in their current locations. Thus, adaptation can also be **transformational**, where actions focus on changing the fundamental attributes of a system in response to climate and its effects (IPCC 2014). Mitigation of climate change is another approach to managing climate change that is based on reducing the sources or enhancing the storage of greenhouse gases (IPCC 2014). This section focuses on incremental and transformational adaptation actions that can enhance the resilience of sagebrush systems. It also reviews the available information on the effects of management actions on carbon storage.

Top: Road to Nixon, Nevada, sunrise (photo by Nolan Preece, used with permission). Middle right: Dr. Matt Germino illustrating a weather station on the Soda Fire in SE Idaho (photo: U.S. Geological Survey). Middle left: A common garden study for assessing the importance of local adaptation in sagebrush (photo: USDA Forest Service). Bottom: Planting sagebrush seedlings after a wildfire (photo: USDA Forest Service). Bottom inset. Sagebrush transplant (photo: Stacy Moore, Institute for Applied Ecology).

Climate Adaptation and Resilience Management

Concepts

Managing natural resources within the context of climate adaptation is consistent with the approach described in Part 1 of the Science Framework, but requires the necessary flexibility to modify management actions as environmental conditions change. Widely used concepts for addressing adaptation in use by the Fish and Wildlife Service (FWS) (USDOI FWS 2010), the Forest Service (USDA FS 2011), and their partners focus on climate resistance, resilience, response, and realignment strategies (Halofsky et al. 2018a,b). Resistance strategies aim to increase the capacity of ecosystems to retain their fundamental structure, processes, and functioning despite climate-related stressors such as drought, wildfire, insects, and disease. These types of strategies may offer only short-term solutions, but often describe the intensive and localized management of rare and isolated species (Heller and Zavaleta 2009). Strategies to increase ecosystem resilience aim to minimize the severity of climate change impacts by reducing vulnerability and increasing the capacity of ecosystem elements to adapt to climate change and its effects. Response strategies seek to facilitate large-scale ecological transitions in response to changing environmental conditions and may include realignment or the use restoration practices to ensure persistence of ecosystem processes and functions in a changing climate.

These concepts of climate resistance, resilience, and response apply to many management and land ownership contexts and can be used to help determine appropriate climate adaptation strategies. Using these concepts to manage for changes in climate involves examining whether current assumptions about the effects of weather and climate on environmental responses and underlying assumptions about the expected result of management actions are still viable in a changing environment. Examples are ecological site descriptions and state-and-transition models in which the reference state often serves as the management target (fig. 3.1) (Bestelmeyer et al. 2009; Briske et al. 2005; Caudle et al. 2013). While managers can use historical data to help understand ecosystem response to environmental changes (e.g., Swetnam et al. 1999), it is important to recognize that the relationship between climate and ecosystem response will shift over time with continued warming. Consequently, managing for historical conditions may not maintain ecological sustainability (goods and services, values, biological diversity) into the future and management actions should be planned accordingly (Hobbs et al. 2009; Millar et al. 2007).

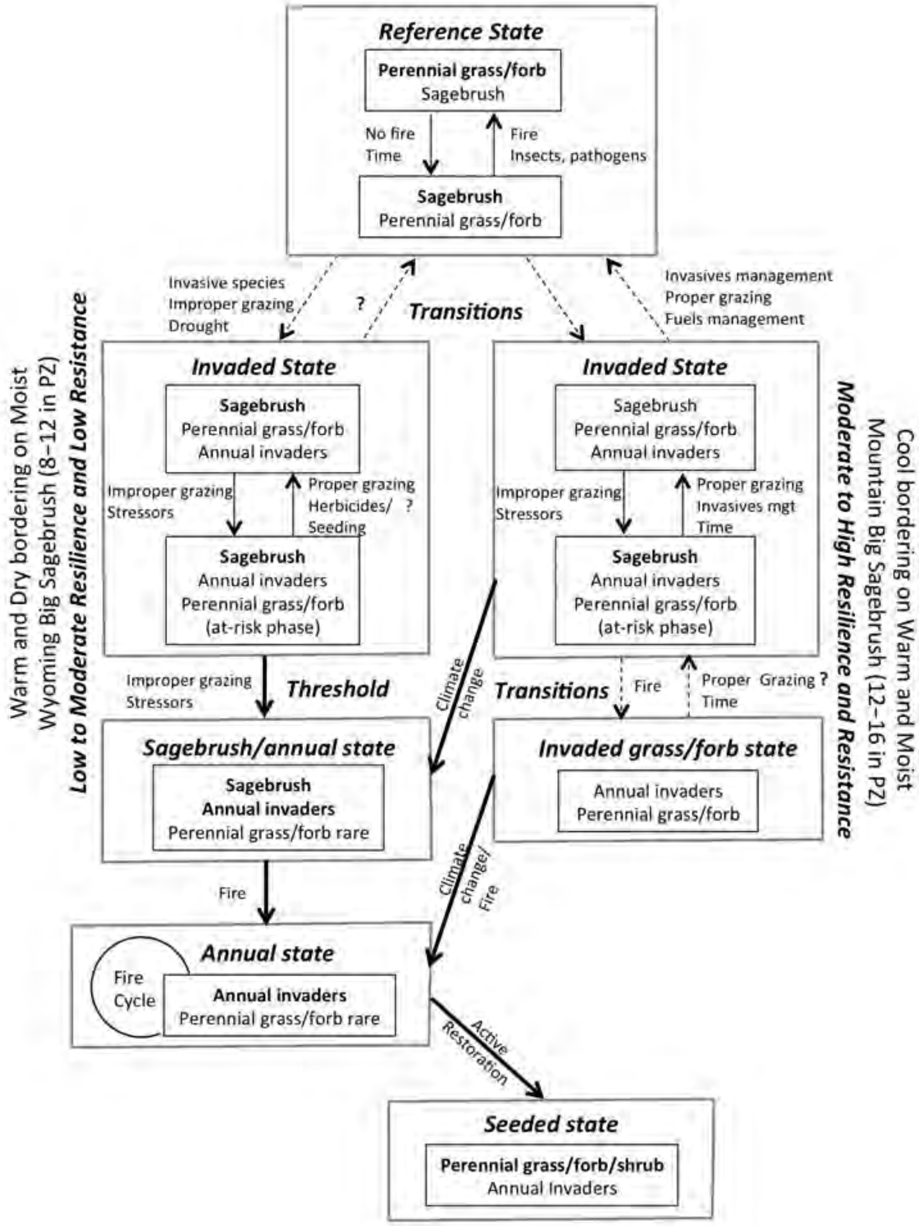

Figure 3.1—Generalized conceptual model showing the states, transitions, and thresholds for relatively warm and dry Wyoming big sagebrush ecosystems with low to moderate resilience and low resistance to cheatgrass and cool and moist mountain big sagebrush ecosystems with moderate resilience and resistance in the Cold Deserts (Chambers et al. 2017, Appendix 6). Reference state: Vegetation dynamics are similar for both types. Perennial grass/forb increases due to disturbances that decrease sagebrush, and sagebrush increases with time after disturbance. Invaded state: An invasive seed source, improper grazing, stressors such as drought, or a combination thereof, trigger a transition to an invaded state. Perennial grass/forb decreases, and both sagebrush and invaders increase with improper grazing and stressors, resulting in an at-risk phase in both types. Proper grazing, invasive species management, and fuel treatments may restore perennial grass and decrease invaders in relatively cool and moist Wyoming big sage and in mountain big sage types with adequate grass/forb, but return to the reference state is likely only for mountain big sage types. Sagebrush/annual state: In the Wyoming big sagebrush type, improper grazing and stressors trigger a threshold to sagebrush/annual dominance. Annual state: Fire, disturbances, or management treatments that remove sagebrush result in dominance of annuals. Perennial grass is rare, and repeated fire causes further degradation. Seeded state: Active restoration results in dominance of perennial grass/forb/shrub. Treatment effectiveness and return to the annual state are related to site conditions, posttreatment weather, and seeding mixture. Invaded grass/forb state: In the mountain big sagebrush type, fire results in a transition to annual invaders and perennial grass/forb. Proper grazing and time may result in return to the invaded state given adequate perennial grass/forb. Increases in climate suitability for cheatgrass and other annual invaders may shift vegetation dynamics of cooler and moister mountain big sagebrush ecosystems toward those of warmer and drier Wyoming big sagebrush ecosystems. Although not shown here, woodland expansion and infill in mountain big sagebrush sites with conifer potential can result in transition to woodland-dominated or eroded states, leading to crossing of biotic and abiotic thresholds (adapted from Chambers et al. 2014b).

Climate Adaptation Strategies

Due to uncertainty about exactly what the future will look like, planning for multiple possibilities and using adaptive management principles is essential. Adaptive management uses the best available information for helping ecosystems and the plant and animal species they support to adapt to inevitable changes in climate (Millar et al. 2007). Climate adaptation strategies for the sagebrush biome are in table 3.1. The specific approaches for sagebrush ecosystems build on the sage-grouse habitat resilience and resistance matrix (table 1.3) and the sagebrush ecosystem management strategies (table 1.4).

Climate adaptation strategies incorporate multiple scales and focus on preventing the loss of ecosystem services by maintaining and enhancing ecosystem processes, functional attributes, and feedbacks (table 3.1). For example, the extent and connectivity of intact sagebrush ecosystems provide a buffer that facilitates species adaptation and movement in response to climate change as well as to the increasing effects of human development and land use (e.g., Knick et al. 2011, 2013; Millar et al. 2007). Maintaining intact and connected sagebrush ecosystems is based on developing public land use plans (LUPs) and policies that reduce the impact of existing ecological, land use, and development stressors on these ecosystems at broad (sagebrush biome and multiple Management Zones) to mid- (Greater sage-grouse [*Centrocercus urophasianus*; hereafter, GRSG] Management Zone and ecoregion) scales. It also involves strategic placement of conservation easements to prevent conversion to tillage agriculture and anthropogenic developments and to maintain existing connectivity at mid- to local (district, field office, or project level) scales.

Many climate adaptation strategies work together to accrue multiple ecosystem benefits. Maintaining or enhancing key plant structural and functional groups is central to most climate adaptation strategies. Certain plant structural and functional groups are critical for stabilizing hydrologic and geomorphic processes, promoting desired successional processes, and lowering risk of conversion to invasive annual grasses following disturbances that remove native vegetation (Pyke 2011). Postfire rehabilitation and restoration activities can increase ecosystem capacity to absorb change by using functionally diverse species mixtures and including plant materials from across a greater geographic range that considers current climate and near-future climate (next 20 to 30 years) (table 3.1) (Butler et al. 2012; Finch et al. 2016). Favoring existing genotypes that are better adapted to future conditions because of broad tolerances to disturbances, drought adaptations, pest resistance, or other characteristics can also increase adaptive capacity (table 3.1) (Butler et al. 2012; Finch et al. 2016). Where shown to be successful, assisted migration, or the purposeful movement of individuals or propagules of a species to facilitate or mimic natural range expansion or long-distance gene flow within the current range, may facilitate community adjustments (Buchorava 2017). Implementing these strategies requires developing the necessary research and management capacity to forecast changes in ecological conditions and species distributions and to better understand ecosystem and species response to changes in climate at mid- to local scales.

Table 3.1–Climate change adaptation strategies for the sagebrush biome. General strategies are based on Millar et al. (2007, 2012) and Butler et al. (2012). Specific approaches for sagebrush ecosystems build on the sage-grouse habitat, resilience and resistance matrix (table 1.3) and management strategies for persistent ecosystem threats and land use and development threats (table 1.4). Resistance = R1; Resilience = R2; Response = R3.

Sustain fundamental ecological conditions (R1, R2, R3)
- Maintain or restore soil quality and nutrient cycling by reevaluating the timing and intensity of land use practices such as livestock grazing
- Maintain or restore hydrologic and geomorphic processes following stress and disturbance

Reduce the impact of existing ecological, land use, and development stressors (R1, R2, R3)
- Develop appropriate policies, land use plans, and project plans to protect sagebrush habitat and prevent fragmentation
- Secure conservation easements to prevent conversion to tillage agriculture, housing developments, and other land conversions, and maintain existing connectivity

Promote landscape connectivity (R2, R3)
- Reduce juniper and piñon expansion to maintain connectivity among sage-grouse and sagebrush dependent species populations and facilitate seasonal movements
- Suppress fires that occur under more severe burning conditions in targeted areas where altered fuel beds facilitate large and severe fires, increase landscape fragmentation, and impede dispersal, establishment, and persistence of native plants and animals
- Manage landscapes to create or enhance permeability and increase the ability of sagebrush dependent species to move between individual Priority Areas for Conservation or Biologically Significant Units

Maintain or create refugia (R1)
- Identify and maintain ecosystems that: (1) are on sites that may be better buffered against climate change and short-term disturbances, and (2) contain communities and species that are at risk across the greater landscape
- Prioritize and protect existing populations on unique sites
- Prioritize and protect sensitive or at-risk species or communities
- Establish artificial reserves for at-risk and displaced species

Reduce the risk of wildfires that result in abrupt transitions to novel states (R1, R2)
- Reduce fuel loads and fuel continuity to (1) decrease fire size, alter burn patterns, decrease perennial grass mortality, and maintain landscape connectivity; (2) decrease competitive suppression of native perennial grasses and forbs by woody species, including sagebrush; and thus (3) lower the longer-term risk of dominance by invasive annual grasses and other invaders
- Use mechanical treatments (e.g., cutting, mastication) to reduce woody fuels in juniper and piñon expansion areas with moderate to high resilience that have little or no presence of invasive annual grasses and sufficient perennial grasses and forbs to promote recovery
- Use prescribed fire to create fuel mosaics and promote successional processes in sagebrush and juniper and piñon expansion areas with moderate to high resilience that have little or no presence of invasive annual grasses and sufficient perennial grasses and forbs to promote recovery
- Use herbicide applications and appropriately timed livestock grazing to reduce cheatgrass fuels in sagebrush ecosystems where they have potential to increase perennial grasses and forbs
- Suppress wildfires in moderate and especially low resilience and resistance sagebrush-dominated areas to prevent conversion to invasive annual grass states and thus maintain ecosystem connectivity, ecological processes, and ecosystem services
- Suppress wildfires adjacent to or within recently restored ecosystems to promote recovery and increase capacity to absorb future change
- Use fuel breaks in carefully targeted locations along existing roads where they can aid fire suppression efforts and have minimal effects on ecosystem processes (Maestas et al. 2016)

Reduce the risk of nonnative invasive plant species introduction, establishment, and spread (R1, R2, R3)
- Limit anthropogenic activities that facilitate invasion processes including surface disturbances, altered nutrient dynamics, and invasion corridors
- Use Early Detection and Rapid Response (USDOI 2016) for emerging invasive species of concern to prevent invasion and spread
- Manage livestock grazing to promote native perennial grasses and forbs that compete effectively with invasive plants
- Actively manage invasive plant infestations using integrated management approaches such as chemical treatment of invasives and seeding of native perennials from climatically appropriate seed sources

(Continued)

Table 3.1—(Continued).

Maintain or enhance key structural and functional groups (R1, R2, R3)

- Manage grazing by livestock and wild horse and burro populations to maintain soil and hydrologic functioning and capacity of native perennial herbaceous species, especially perennial grasses, to effectively compete with invasive plant species
- Manage grazing by livestock and wild horse and burro populations to maintain riparian-wetland functioning, streambank and floodplain stability, and vegetation sufficient to dissipate flood energy, promote infiltration, minimize erosion, and compete with invasive plant species
- Reduce conifer expansion to prevent high severity fires and maintain native perennial herbaceous species that can stabilize geomorphic and hydrologic processes and minimize invasions
- Restore disturbed areas with functionally diverse mixtures of native perennial herbaceous species and shrubs with climatically appropriate seed sources and with capacity to persist and stabilize ecosystem processes under altered disturbance regimes and in a warming environment

Enhance genetic diversity (R2, R3)

- Use seeds, germplasm, and other genetic material from across a greater geographic range based on current climate and near-future (next ~20–30 years) climate considerations
- Favor existing genotypes that are better adapted to future conditions because of pest resistance, broad tolerances, or other characteristics
- Increase diversity of nursery stock to provide those species or genotypes likely to succeed

Facilitate community adjustments through species transitions (R3)

- Monitor both native and invasive species at range margins to provide advanced warning of range shifts
- Investigate assisted migration options—the purposeful movement of individuals or propagules of a species to facilitate or mimic natural range expansion or long-distance gene flow within the current range—in areas with high rates of climate change

Plan for and respond to disturbance (R3)

- Practice drought adaptation measures, such as altered grazing seasons or reduced grazing during droughts, and implement conservation actions to facilitate species persistence
- Identify current and potential future areas where snowpack cover and duration are declining in order to manage to reduce other current stressors
- Anticipate and respond to species declines such as may occur on the southern or warmer edges of their geographic range by including plant materials from neighboring climate types in seed and planting mixes
- Leverage topographic features (landforms) that retain soil moisture longer for restoration activities (Bainbridge 2007)
- Favor or restore native species that are expected to be better adapted to the future range of climatic and site conditions
- Protect future-adapted restoration and reclamation seedings from inappropriate livestock grazing and wild horse and burro populations
- Avoid seeding introduced forage species such as crested wheatgrass that outcompete native species (Davies et al. 2013; Lesica and Deluca 1996)

Management and research studies coupled with landscape monitoring can provide the basis for developing cost-effective and feasible management strategies for adapting to climate change. Carefully designed management and research studies implemented in the near future may increase our understanding of viable approaches for adaptation measures, such as appropriate grazing regimes for drought conditions, conservation actions to facilitate species persistence during climate warming, seeding and transplanting techniques during drought, and identification of species and ecotypes that can be used successfully in assisted migration. Monitoring to detect the rates and magnitudes of change occurring within the context of adaptive management can identify both populations and habitats that are declining (Carwardine et al. 2011; Field et al. 2004), as well as new or novel combinations of species that constitute a functioning ecosystem under climate change. Increased understanding of both the changes occurring and viable strategies for addressing those changes may reduce uncertainty and provide direction for adaptive management strategies (Hobbs et al. 2009).

A participatory scenario planning process may be one approach to help identify relevant adaptation strategies in the context of adaptive management (Cross et al. 2013; Star et al. 2016; USDA FS 2012; USDOI NPS 2013). Participants can use climate change projections and associated natural resource models to depict both the amount of change and the degree of uncertainty (Star et al. 2016). Decisionmakers, stakeholders, and experts can work together to identify the most relevant and uncertain drivers of system change, which often include sociopolitical and socioeconomic factors. They can then develop a shared understanding of future climate scenarios that is likely to lead to broader support for suggested adaptation strategies (Star et al. 2016). To date, scenario planning has been more commonly used in nongovernmental organizations and local government planning.

Prioritizing Management Actions and Determining Appropriate Management Strategies

Assessing ongoing and projected climate change using the best available data is integral to evaluating priority areas for management at mid-scales and determining appropriate management treatments at local scales. In the context of the Science Framework, the effects of changes in climate on species and ecosystems can be addressed similarly to other persistent ecosystem threats such as wildfire and invasive annual grasses (see Part 1, section 8; table 3.1, this volume). For GRSG and other at-risk species and resources, the process involves overlaying key data layers in a geospatial analysis to both visualize and quantify: (1) species locations and abundances, (2) the probability that an area has suitable habitat, (3) the likely response to disturbance or management treatments, and (4) the dominant threats including projected climate change.

Geospatial analyses with overlays of key data layers can: (1) help evaluate the level of risk to vegetation types and species to climate change, (2) target areas for adaptive management, and (3) determine the most appropriate types of management actions. Key data layers include projected changes in climate variables (Part 1, section 8). Land managers can use these layers to assess the rate and magnitude of change projected for the assessment area. Other important layers are projections for changes in individual plant species (e.g., Bradley et al. 2016; Homer et al. 2015; Still and Richardson 2015) and vegetation types (e.g., Rehfeldt et al. 2012; Schlaepfer et al. 2012c) under different climate change scenarios. In addition, climate change vulnerability analyses of key ecological and socioeconomic resources (water, fisheries, vegetation and disturbance, wildlife, recreation, infrastructure, cultural heritage, and ecosystem services) are available for the Intermountain Region (Halofsky et al. 2018b) and Northern Rocky Mountain Region (Halofsky et al. 2018a). Additional websites and resources for climate change are in Appendix 2.

Climate change projections can be factored into prioritizing areas for management within assessment areas (Part 1, section 8) by considering the following factors.

- Continued changes in climate (i.e., increases in temperature and shifts in precipitation timing and amount) and the associated effects are expected to be relatively small within the next decade or two. Areas can be prioritized for management that provide suitable habitat and support species populations at mid-scales, and management practices can be adapted to build resilience to changes in climate into sagebrush ecosystems at local scales (table 3.1). Monitoring can provide critical information on changes in species and ecosystems resulting from climate changes that allows managers to take advantage of opportunities to facilitate transitions to systems that will be better adapted in the long term.

- Changes in climate and the interactions of these changes with other threats are already documented and are expected to be large (e.g., rapid warming events, uncertainty of snowpack, extreme drought) in the next few decades (table 3.1). The impacts of changes in climate on plant community composition and vegetation types will be most evident following major disturbances, such as wildfires, that occur at an ecotone between different vegetation types or on warmer, drier sites. In this case, more proactive adaptation strategies may be necessary to facilitate community adjustments and species persistence. These may include favoring or restoring native species that have been shown to be better adapted to the future range of climatic and site conditions and to have acceptable effects on biotic interactions and ecosystem process (Bucharova 2017). The use of assisted migration to address changes in climate suitability will require additional research and management guidelines to evaluate the potential positive as well as negative effects of purposeful species movements (Bucharova 2017).

Key Topics in Climate Adaptation

Across much of the sagebrush biome, climate change is resulting in a warmer and drier environment (Kunkel et al. 2013a,b,c). In turn, the warmer, drier conditions are resulting in increasing magnitude and frequency of droughts (Cook et al. 2015; Prein et al. 2016), increasing dust in the atmosphere (Livneh et al. 2015; Painter et al. 2012; Steltzer et al. 2009), and in most areas, decreasing snowpacks (Mote and Sharp 2016). Several studies indicate that the length of the fire season and duration of extreme fire weather also are increasing (Abatzoglou and Kolden 2013; Littell et al. 2009; Westerling et al. 2014). These changes are projected to have significant effects on ecosystem processes, species distributions, and community composition (e.g., Blumenthal et al. 2016; Schlaepfer et al. 2012c). Changing climate conditions can also influence the abundance and spread of insects and diseases and increase the stress levels of host species, making them more susceptible to the effects of insects and diseases and causing higher mortality (IPCC 2014). Developing an understanding of the changes that are occurring is essential for evaluating the effects of ongoing management actions and determining effective adaptation strategies (text box 3.1).

Drought

From a meteorological perspective drought is defined as the accumulated imbalance between the supply of water and the demand for water by plants, animals, the atmosphere, the soil column, and humans (Kunkel et al. 2013a,b). Drought can also be defined from other perspectives including hydrologic (e.g., streamflow), agricultural (e.g., ecosystem productivity), or socioeconomic (Luce et al. 2016). Determining whether a drought is occurring can take a relatively longer time for areas where the effects of drought may accumulate slowly, such as forests and sagebrush ecosystems. Ecological indicators of drought exist for rangelands and can be listed sequentially: Water shortages stress plants and animals, vegetation production is reduced, plant mortality increases, plant cover is reduced, amount of bare ground increases, soil erosion becomes more

Text Box 3.1—Monitoring Climate Change Effects

Long-term monitoring results can be used to track changes in species and ecosystems induced by the effects of climate change. At the biome to mid-scale, remote sensing can be used to detect changes in environmental conditions, such as the duration of snowpacks and seasonal soil moisture availability, and the effects on ecosystems, such as changes in plant phenology and productivity. Remote sensing can also be used to monitor changes in persistent ecosystem threats, such as plant invasions and wildfire patterns. Information on the rates and direction of change across the sagebrush biome can be used to prioritize resource allocation for management of invasive species, wildfire and vegetation, and wild horses and burros. It can also be used to determine where to target adaptation strategies to maintain landscape connectivity, ecosystem redundancy, and refugia.

Combining ground-based monitoring with remote sensing can help scale-up results to assess which species and ecosystems may be most vulnerable to climate change. Focusing monitoring efforts on climate transition zones and areas projected to exhibit rapid change (e.g., rapid warming events, loss of snowpack, extreme drought) can provide much needed information on climate change effects. Information on these changes can be used to identify effective adaptation strategies, such as maintaining or enhancing key structural and functional groups, increasing genetic diversity, facilitating community adjustments through species transitions, and planning for and responding to disturbance. Monitoring following changes in management or after treatments can be used to verify the effectiveness of management strategies designed to help ecosystems transition to the new climatic conditions.

prevalent, habitat and food resources for wildlife are reduced, wildlife mortality increases, rangeland fires may increase, some insect pests and invasive weeds may increase, forage value and livestock carrying capacity decrease, and then, economic depression in the agricultural sector sets in (Finch et al. 2016).

Drought adaptation measures with shorter-term and longer-term horizons have been identified for rangelands and forests across the western United States (see Briske et al. 2015; Finch et al. 2016; Joyce et al. 2013). Planning for a drought involves developing a drought management plan (UNL-NDMC 2012; examples available at http://drought.unl.edu/ranchplan/WriteaPlan.aspx). Management actions vary regionally and reflect the resources available to cope with drought. In general, the goal is to minimize the risk of environmental degradation and loss of ecosystem function. Planning for adaptation actions is most successful if coordinated across all land ownerships and if management plans consider the next drought as well as the current drought and its aftermath (Finch et al. 2016).

Current management actions may need to be reexamined with the onset of drought. For example, adaptation actions with respect to livestock management during the drought include: reducing stocking rate to allow plant recovery; using fencing and other developments to manage livestock distribution; using drought-resistant restoration species; using drought-adapted stock; adjusting season of use; implementing a deferred grazing system; developing, restoring, or reclaiming water sources; providing shade structures for livestock; reducing the time livestock graze a specific grazing unit; increasing the time between periods of grazing (rest); and testing new techniques for responding to drought.

With respect to restoration, climate and weather models are now available that can be used to help inform the timing of planting (Hardegree et al. 2012). Under certain conditions, it may be beneficial to delay planting and shift the focus to restoring areas with less desirable species. For example, implementing measures to control crested wheatgrass (*Agropyron cristatum*) during dry years and seeding native grass in wetter years may result in more effective restoration in the West-Central Semiarid Prairies (Bakker et al. 2003). To mitigate the longer-term impact of drought or other abiotic stressors, plant material selection should consider the adaptive capacity of different species and genetic variation within species (Richardson et al. 2012). Assisted migration can be considered for areas where high rates of climate change are expected and the likelihood of success has been evaluated (table 3.1). These decisions will be critical given the potential for increased frequency and duration of drought in the future.

Snowpack and Dust

Total snowfall has been declining precipitously in the West since the 1920s (Kunkel et al. 2009). Maximum seasonal snow depth declined from winter 1960–1961 to winter 2014–2015 across North America, and other studies showed declines in snow cover as well (Kunkel et al. 2016). A recent analysis of April snowpack data, which are used extensively for spring streamflow forecasting, indicated declines at more than 90 percent of the sites when measured from 1955 to 2016 (Mote and Sharp 2016). The average change across all sites amounted to about a 23-percent decline in snow water equivalent. These decreases were observed throughout the western United States, with the most prominent declines in Washington, Oregon, and the northern Rockies (Mote and Sharp 2016).

Decreases in snowpack may not affect overall patterns of soil water availability if precipitation that arrives during the cold season simply switches from snow to rain (Schlaepfer et al. 2012a). However, increases in soil temperature and associated decreases in soil water availability due to longer growing seasons and higher evapotranspiration may influence plant species establishment and survival and thus community composition (Palmquist et al. 2016a,b).

Drought, wildfire, and agricultural activities in the western United States contribute to dust in the atmosphere, which settles on snow-covered areas in the winter. Over the last decade, the number of dust-on-snow events increased in the Colorado Rocky Mountains (Painter et al. 2007; Toepfer et al. 2006). Dust-on-snow events reduce duration of snow cover (Painter et al. 2007), increase rate of snowmelt associated with more extreme dust deposition, and produce earlier peak stream flows of 1 to 3 weeks (Livneh et al. 2015; Painter et al. 2012; Steltzer et al. 2009). As a result of these dust-on-snow events, snow chemistry increases in pH, calcium content, and acid neutralizing capacity with more pronounced effects at upper elevations than lower elevation forested sites (Rhoades et al. 2010).

Effects of decreasing snowpack on sagebrush ecosystems will be widespread, but are likely to be most significant in areas with measurable changes in the amount and duration of snowpack. The most vulnerable areas are likely to be those that previously retained snow cover for all or most of the winter, or where winter snowpack was critical to recharge deep soil water. Adaptation strategies specific to these areas have not been developed (but see David 2013). However, identifying these areas and managing them to sustain ecological functions and reduce the impact of existing ecological, land use, and development stressors can facilitate adaptation (table 3.1). Monitoring these areas for changes in soil moisture and temperature and in species composition can provide information on (1) establishment and spread of nonnative invasive plant species and the need for intervention and (2) the need for community adjustments through species transitions.

Fire Regimes

Higher temperatures associated with climate change have been linked to increases in fire size and longer fire seasons and durations of extreme fire weather in forested ecosystems (Westerling et al. 2014). Although some have suggested that these relationships also exist for the western portion of the sagebrush biome, recent analyses of LANDFIRE data (1984–2014) for the Basin and Range, Snake River Plain, and Columbia Plateau ecoregions (fig. 1.1) fail to show significant changes in number of large fires per year, total fire area per year, or 90th percentile large fire size per year. However, these analyses do point toward increasing total fire area and 90th percentile fire size (Dennison et al. 2014). In addition, analyses of fire patterns in juniper and piñon land cover types show that the fire season started earlier and ended later in the Basin and Range ecoregions over the same 30-year study period (1984–2014) (Board et al. 2018).

Both temperature and amount and seasonality of precipitation influence fire regimes. In the Basin and Range, Snake River Plain, and Columbia Plateau ecoregions most precipitation arrives as winter snow and rain, and woody species, such as sagebrush, tend to dominate vegetation communities (Part 1, section 4). In these areas, most fires burn in July and August. Fire intensities are typically moderate to high and extreme fire weather can result in extensive fire spread (Brown 1982; Romme et al. 2009). In contrast, the Northwestern Plains and portions of the Wyoming Basin and Southern Rockies receive more summer precipitation and most fires burn earlier in the year. These areas have higher relative abundance of grasses and usually exhibit moderate fire spread (Brown 1982; Romme et al. 2009). Fire regimes are further influenced by fire season length, which varies from about 90 days per year in cooler and moister ecoregions to more than 135 days per year in warmer and drier ecoregions (Board et al. 2018). Changes in amount and seasonality of precipitation may cause shifts in relative abundances of woody species and grasses and thus live fuel moisture dynamics, which will affect fire behavior. Further increases in temperature without compensating increases in precipitation, especially during the growing season, will continue to cause greater aridity, longer fire seasons, and more extreme fire weather across much of the sagebrush biome (Dai 2013).

Changes in precipitation due to climate change may have very different effects than changes in temperature on the locations and characteristics of wildfires in sagebrush ecosystems, and these effects are likely to differ within and across ecoregions. The relationships between precipitation and fire exhibit high regional variability due to the heterogeneity of topography, climate, soils, vegetation, and land use (Littell et al. 2009; Pilliod et al. 2017). In general, warmer and drier areas characterized by Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) at lower elevations have the potential for large fires to burn every summer, but are **fuel limited** and do not always have enough fuel to burn (Abatzoglou and Kolden 2013; Littell et al. 2009; Westerling et al. 2014). These areas often require 1 or more years with above-normal precipitation to create sufficient fuel for large wildfires (Crimmins and Comrie 2004; Littell et al. 2009; Pilliod et al. 2017; Westerling et al. 2014). At higher elevations, temperatures become cooler, precipitation usually increases, and ecosystems become increasingly **energy limited** in that they have enough fuel to support fires every summer, but may not be dry enough to burn (Abatzoglou and Kolden 2013; Littell et al. 2009; Westerling et al. 2014). Mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) and mountain shrub communities exhibit these characteristics. These areas often require warmer and drier conditions to decrease fuel moisture sufficiently for large wildfires to burn.

Invasive annual grasses are influencing both the areas burned and fire size through the invasive annual grass-fire cycle, primarily in relatively warm and dry areas, where most precipitation arrives in winter and spring (Balch et al.

2013; Pilliod et al. 2017). These grasses increase fine fuels and fuel continuity and thus fire frequency and extent (Balch et al. 2013; Brooks et al. 2004). A 1- to 3-year lag effect of precipitation on both area burned and number of fires in landscapes dominated by cheatgrass (*Bromus tectorum*) is typical (Pilliod et al. 2017). Changes in fire regimes due to invasive annual grasses are most evident in the Snake River Plain and Northern Great Basin (Balch et al. 2013; Pilliod et al. 2017), but these species are projected to expand northward and upwards in elevation with climate warming (Bradley et al. 2016) and are increasing in the eastern portion of the range (Knight et al. 2014; Lauenroth et al. 2014).

Wildfire and vegetation management plays a key role in enhancing resilience to disturbance and resistance to invasive annual grasses in the face of climate change (tables 1.3, 1.4). Primary objectives are to reduce ecosystem vulnerability, increase the capacity of ecosystems to adapt to climate change and its effects, and facilitate species and plant community transitions in response to changing environmental conditions. This entails: (1) reducing fuel loads and continuity to decrease fire severity or extent, or both; (2) lowering competitive suppression of perennial herbaceous species, which largely determine resilience to wildfire and resistance to invasion; and (3) using postfire revegetation to design vegetation communities that maintain higher live fuel moisture and have lower fuel bed continuity and packing ratios (a measure of fuel bed compactness or the fraction of fuel bed volume that is occupied by fuel).

Fuel management to reduce fuel loads and continuity focuses on areas with increased woody fuels (sagebrush or juniper [*Juniperus* spp.] and piñon [*Pinus* spp.]) or fine fuels (grasses and forbs), or both. Woody fuel loading and fine fuel loading interact with fire weather to influence the propensity for wildfires, and decreases in fuel loads can lower the likelihood of wildfires over a range of fire weather conditions (fig. 3.2). A variety of treatments exist to reduce woody fuels, including sagebrush mowing; juniper and piñon cutting, shredding, and mastication; and prescribed fire (table 1.4 and section 4). Similarly, treatments exist to reduce fine fuels, such as herbicide applications and appropriately timed livestock grazing in areas dominated by cheatgrass (Strand et al. 2014; table 1.4 and section 5, this volume). The use of fuel breaks in carefully targeted locations can aid fire suppression efforts (Maestas et al. 2016). For treatments to maintain or increase resilience to wildfire as the climate changes, it is necessary to ensure that sufficient perennial herbaceous species exist before treatment to promote ecosystem recovery and that treatments do not introduce or lower resistance to invasive plants (Chambers et al. 2014a,b). Use of traditional phenological knowledge from Native Americans regarding the appropriate timing of treatments, including use of prescribed fire, shows promise for achieving desired conditions (Armatas et al. 2016; Huffman 2013).

Managing for fuel beds with high temporal and spatial variability could increase resilience to wildfire (Abatzoglou and Kolden 2013; Kay 1995; Littell et al. 2009). This could include treatments that increase sagebrush patch size and variability in gap size (the distances between shrubs and grasses) (Kay 1995). Patch burning to increase vegetation heterogeneity is increasingly used in the U.S. Great Plains, southern Africa, and Australia (e.g., Bird et al. 2013; Brockett et al. 2001; Fuhlendorf et al. 2017; Ricketts and Sandercock 2016; Voleti et al. 2014). It may be possible to create fuel bed heterogeneity in sagebrush ecosystems by conducting patch-scale burns in early spring or late fall to remove conifers and shrubs in ecosystems with moderate to high resilience (e.g., Davies et al. 2008; Pyle and Crawford 1996; Trauernicht et al. 2015).

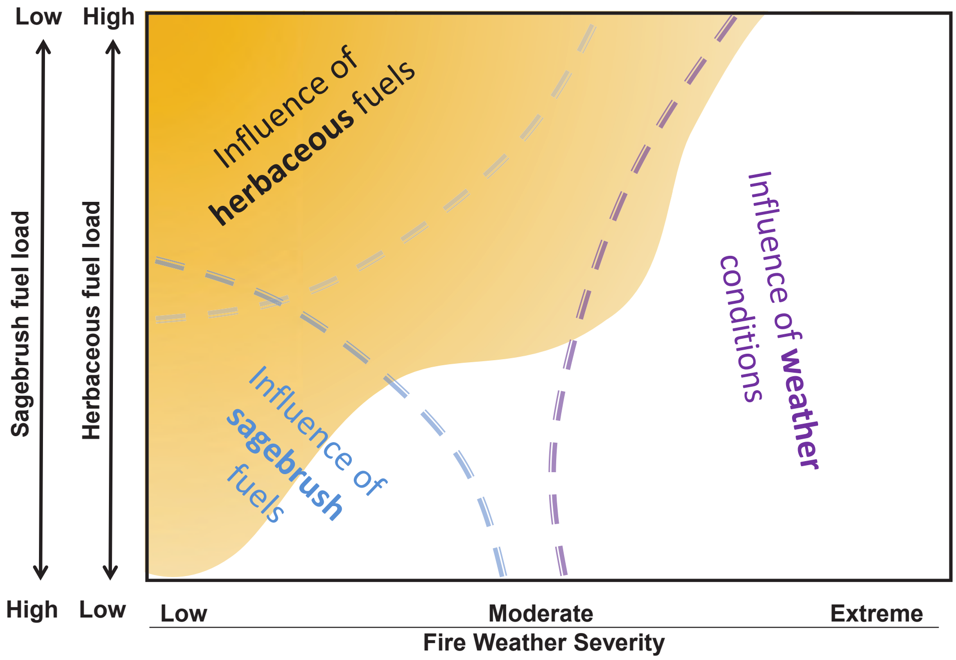

Figure 3.2—The interaction of herbaceous and sagebrush fuels with fire weather severity. In this conceptual model, fuel composition is displayed on the y-axis and fire weather condition is displayed on the x-axis. Low fire weather severity is characterized by high fuel moistures, high relative humidity, low temperature, and low wind speeds, while extreme fire weather is characterized by the opposite conditions. As woody fuel loading or fine fuel loading, or both, increases, fuel packing ratios become more optimal, fuel continuity increases, and less severe fire weather is required for large wildfires. Annual grasses fill interspaces between native fuels (shrubs and bunchgrasses) and are particularly problematic. However, progressive increases in sagebrush or juniper and piñon stand density also lower the severity of fire weather required for large wildfires. Reductions in fuel loads can decrease the likelihood of large wildfires over a range of fire weather conditions. However, extreme fire weather conditions, which are projected to increase in the future, can override the influence of fuel loads and continuity (figure modified from Strand et al. 2014).

Post-wildfire revegetation provides an opportunity to establish vegetation communities with high fuel bed heterogeneity that may be more resilient to wildfire. Resilience to wildfire could be increased by restoring or maintaining plant communities that maintain higher live fuel moisture during dry periods or drought through differences in the relative proportions of herbaceous vegetation to shrubs, varying phenologies and water use patterns, and differences in the cure rate of grasses and forbs (Kay 1995; Palmquist et al. 2016a,b; Schlaepfer et al. 2012b). Also, fuel bed continuity and packing ratio could be decreased by seeding native plant species with growth forms and structures (e.g., size of stems, distance between stems) that are not conducive to carrying fire, even when cured. Most native forbs and some rhizomatous grasses, such as western wheatgrass (*Pascopyrum smithii*), have these properties.

Monitoring the responses of sagebrush ecosystems to wildfire as the climate changes can help inform adaptive management strategies (text box 3.1). At broad scales monitoring changes in wildfire patterns in relation to habitats of species at risk and other resource values can help prioritize resource allocation for preparedness, prevention, suppression, and postfire rehabilitation. At mid- to local scales, information on changes in wildfire area burned and size for specific ecological types or ecological sites that characterize ecoregions provides the basis for adjusting preparedness, prevention, and suppression management strategies over time (section 4). Large changes in species composition and decreased resistance to invasive plants, particularly invasive annual grasses, indicate decreased resilience to wildfire and the need to modify postwildfire rehabilitation strategies.

Changes in Species Distributions and Community Composition

The changes in precipitation and temperature regimes occurring as a result of climate warming are projected to have large consequences for species distributions and, because individual species differ in their climatic requirements, for community composition (Part 1, section 5.2). The distribution of species such as big sagebrush is projected to move to the north and upward in elevation (Bradley 2010; Homer et al. 2015; Schlaepfer et al. 2012c; Still and Richardson 2015). For juniper and piñon species, habitat with suitable climate is projected to move north and upslope with principal gains in Colorado and southwest Wyoming and losses in the Southwest (Rehfeldt et al. 2006, 2012). Cheatgrass is likely to spread upward in elevation while red brome (*Bromus rubens*) moves northward or increases its abundance in the Cold Deserts and Colorado Plateau, or both (Bradley et al. 2016). Decreases in average summer precipitation or prolonged summer droughts could enable cheatgrass invasion into sagebrush ecosystems that are currently more resistant to invasion and resilient to fire disturbance (Bradley et al. 2016; Mealor et al. 2013), such as the northern mixed-grass prairie, allowing it to more successfully colonize what is currently considered a largely invasion-resistant grassland (Blumenthal et al. 2016).

Climate adaptation strategies for the sagebrush biome are designed to facilitate adaptation of species and communities to a warming climate and to reduce the risk of nonnative invasive plant species introduction, establishment, and spread. An understanding of the rates and magnitude of projected change (see Part 1, Appendix 3) can help managers to prioritize areas for different types of management actions (table 3.1). Areas that are likely to support big sagebrush ecosystems in the future may be good candidates for proactive weed and fire management. Areas that may become more suitable for big sagebrush over time may be candidates for assisted migration during restoration activities. Areas that are unlikely to support big sagebrush ecosystems in the future require careful evaluation to determine the types of ecosystems they are likely to support and whether they merit investment in conservation and restoration resources. Careful assessment of connectivity requirements for sagebrush-dependent species to survive and persist as the climate changes can help inform decisions about where to place limited conservation and restoration resources (Part 1, Appendix 9).

Successful adaptation will include monitoring along climate transition zones to detect changes in both soil temperature and moisture regimes and species composition. Consideration of scale will ensure that planning at broad scales promotes strategies such as landscape connectivity, ecosystem redundancy, and refugia, and that planning at more local scales promotes strategies such as maintaining or enhancing key structural and functional groups, increasing genetic diversity, facilitating community adjustments through species transitions, and planning for and responding to disturbance.

Insects and Disease

Major insect pests and diseases affecting plant and sagebrush dependent wildlife species are poorly identified and studied in sagebrush ecosystems. For example, Aroga moth (*Aroga websteri*), or sagebrush defoliator, is a native moth that experiences periodic outbreaks over large areas affecting sagebrush and sage-grouse habitat quality and quantity. West Nile virus (*Flavivirus* spp.) is a recently established disease in the western hemisphere with potential to greatly reduce many avian species populations such as GRSG.

Outbreaks of the native Aroga moth can damage and kill sagebrush over local to mid-scales, although the only documented outbreaks to date have been in the Cold Deserts in the western part of the sagebrush biome. Anecdotal evidence from the northern Great Basin indicates that Aroga moth outbreaks can be associated with years that have much larger than average fires (Tony Svejcar, retired Rangeland Scientist and Research Leader, Burns, OR, personal communication, 2012). Outbreaks are associated with warm conditions from mid-May through mid-June, during the first and second instar development, followed by high precipitation in June and July, during the fourth and fifth instar development (Bolshakova 2013; Bolshakova and Evans 2016). Peak larval abundance occurs around 239 degree-days (accumulated since January 1 using a base temperature of 5 °C [41 °F]), so managers can track degree-days and monitor larval populations to determine when an outbreak is possible (Bolshakova and Evans 2016). How changes in climate may alter the likelihood of such outbreaks is unclear. Outbreaks may occur at the same frequency but earlier in the year as conditions warm, or the frequency may decline due to the combination of warming temperatures and changes in precipitation timing.

Higher moth survival and abundance are also associated with northerly aspects at mid-elevation, suggesting that sagebrush canopy cover may play an as-yet poorly understood role in outbreaks (Bolshakova and Evans 2014). These sites typically experience lower daily and annual temperature fluctuation, greater snow accumulation, and slower snowmelt, thereby creating more favorable conditions for moth larvae and adults (Bolshakova and Evans 2014). More homogeneous stands of sagebrush may serve as epicenters for outbreaks (Bolshakova 2013; Bolshakova and Evans 2014), suggesting that enhancing heterogeneity of sagebrush cover may limit the size and impact of future outbreaks.

Sage-grouse mortality from West Nile virus typically occurs between mid-May and mid-September with peak mortality in July and August (Walker and Naugle 2011), which are also the warmest and driest months. Sage-grouse frequently use ponds, springs, and other standing water sources during hot weather, which are the same sites used by *Culex tarsalis*, the primary mosquito species that transmits West Nile virus to birds (Schrag et al. 2010; Walker and Naugle 2011). Increasing storm intensity that results in more runoff than infiltration, and the potential need to develop additional water sources for domestic and wild ungulates or for irrigation, could result in creating new or enhancing existing breeding sites for *C. tarsalis* mosquitoes. Where West Nile virus is present, fencing or other modifications to watering sites to limit trampling by livestock, wild horses and burros, and wild ungulates can reduce the number of potential *Culex* mosquito breeding sites (NTT 2011, p. 61). Ponds and tanks can be constructed or modified to discourage breeding mosquitoes (Doherty 2007; Walker and Naugle 2011).

Greenhouse Gas Emissions and Carbon Storage

Actions taken to maintain or enhance the resilience of sagebrush ecosystems to disturbance have implications for greenhouse gas emissions and carbon storage. Semiarid ecosystems strongly influence the trend and interannual variability in the global carbon balance, in part due to widespread woody species expansion and high interannual variability in temperature and precipitation (Ahlström et al. 2015). In wetter years, semiarid systems are typically carbon sinks, while in drier years they tend to be carbon sources because respiration exceeds photosynthesis. In more-or-less average years, semiarid systems tend to be more carbon neutral with uptake by photosynthesis roughly equal to release by respiration (Ahlström et al. 2015; Svejcar et al. 2008).

Actions intended to avoid or halt the spread of invasive annual grasses by increasing resilience to disturbance and resistance to invasion and by restoring invaded sites to sagebrush communities would enhance carbon storage and reduce potential greenhouse gas emissions at all scales. In sagebrush ecosystems most carbon is stored belowground in the roots (Rau et al. 2011a). Conversion of native sagebrush ecosystems to annual grassland converts a greenhouse gas sink into a greenhouse gas source with reductions in aboveground and especially belowground carbon storage (Bradley et al. 2006; Germino et al. 2016; Rau et al. 2011a).

Juniper and piñon expansion and infill in sagebrush ecosystems increase aboveground carbon storage many-fold due to the large increase in biomass, but the impacts belowground are not well understood (Rau et al. 2011b, 2012). Once aboveground tree cover equals 50 percent, resilience to disturbance and resistance to invasive annual grasses drop, and the site may become susceptible to invasive annual grasses after fire (Rau et al. 2012) or other stand-replacing disturbances. The tree cover at which this reduction occurs may be lower on less productive sites.

Further, juniper and piñon expansion and infill reduce total soil nitrogen, which has long-term adverse implications for carbon storage in deep soil, where the carbon pool is very stable (Rau et al. 2012). Juniper and piñon expansion and infill can lengthen fire return intervals but greatly increase the biomass consumed during fire in comparison to sagebrush dominated ecosystems. Consequently, the science is unclear as to the long-term tradeoffs in potential greenhouse gas emissions. Even though the increase in biomass from tree cover would seem more consistent with increasing carbon storage, over the longer term it may be less sustainable than maintaining or restoring sites to sagebrush ecosystems. Short-term greenhouse gas emissions and reductions in carbon storage from projects intended or designed to reduce juniper and piñon expansion and restore sage-grouse habitat are acceptable tradeoffs (CEQ 2016, p. 18). Management objectives to increase carbon storage that are consistent with maintaining habitat and key ecosystem functions will be most beneficial in the long term.

References

Abatzoglou, J.T.; Kolden, C.A. 2013. Relationships between climate and macroscale area burned in the western United States. International Journal of Wildland Fire. 22: 1003–1020.

Ahlström, A.; Raupach, M.R.; Schurgers, G.; [et al.]. 2015. The dominant role of semi-arid ecosystems in the trend and variability of the land CO2 sink. Science. 348: 895–899.

Armatas, C.A.; Venn, T.J.; McBride, B.B.; [et al.]. 2016. Opportunities to utilize traditional phenological knowledge to support adaptive management of social-ecological systems vulnerable to changes in climate and fire regimes. Ecology and Society. 21: 16.

Bainbridge, D.A. 2007. A guide for desert and dryland restoration: New hope for arid lands. Covelo, CA: Island Press. 416 p.

Bakker, J.D.; Wilson, S.D.; Christian, J.M.; [et al.]. 2003. Contingency of grassland restoration on year, site, and competition from introduced grasses. Ecological Applications. 13: 137–153.

Balch, J K.; Bradley, B.A.; D'Antonio, C.M.; [et al.]. 2013. Introduced annual grass increases regional fire activity across the arid western USA (1980–2009). Global Change Biology. 19(1): 173–183.

Bentz, B.J.; Duncan, J.P.; Powell, J.A. 2016. Elevational shifts in thermal suitability for mountain pine beetle population growth in a changing climate. Forestry. 89: 271-283.

Bestelmeyer, B.T.; Tugel, A.J.; Peacock, G.L.J.; [et al.]. 2009. State-and-transition models for heterogeneous landscapes: A strategy for development and application. Rangeland Ecology and Management. 62: 1–15.

Bird, R.B.; Tayor, N.; Codding, B.F.; [et al.]. 2013. Niche construction and dreaming logic: Aboriginal patch mosaic burning and varanid lizards (*Varanus gouldii*) in Australia. Proceedings of the Royal Society B. 280: 20132297.

Blumenthal, D.M.; Kray, J.A.; Ortmans, W.; [et al.]. 2016. Cheatgrass is favored by warming but not CO2 enrichment in a semi-arid grassland. Global Change Biology. 9: 3026–3038.

Board, D.I.; Chambers, J.C.; Miller, R.M.; [et al.]. 2018. Fire patterns in piñon and juniper land cover types in the semiarid Western U.S. from 1984 through 2013. RMRS-GTR-372. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 57 p. https://www.fs.usda.gov/treesearch/pubs/55663. [Accessed May 23, 2018].

Bolshakova, V.L. 2013. Causes and consequences of local variability in *Aroga websteri* Clarke abundance over space and time. Dissertation. Logan, UT: Utah State University. 153 p. https://digitalcommons.usu.edu/etd/2014/. [Accessed May 23, 2018].

Bolshakova, V.L.; Evans, E.W. 2014. Spatial and temporal dynamics of *Aroga* moth (Lepidoptera: Gelechiidae) populations and damage to sagebrush in shrub steppe across varying elevation. Environmental Entomology. 43: 1475–1484.

Bolshakova, V.L.; Evans, E.W. 2016. Pheonology of the sagebrush defoliating moth *Aroga websteri* (Lepidopters:Gelechiidae), with application to population irruptions. Annals of the Entomological Society of America. 109: 424–431.

Bradley, B.A. 2010. Assessing ecosystem threats from global and regional change: Hierarchical modeling of risk to sagebrush ecosystems from climate change, land use and invasive species in Nevada, USA. Ecography. 33: 198–208.

Bradley, B.A.; Curtis, C.A.; Chambers, J.C. 2016. *Bromus* response to climate and projected changes with climate change. In: Germino, M.J.; Chambers, J.C.; Brown, C.S., eds. Exotic brome-grasses in arid and semiarid ecosystems of the western U.S. New York, NY: Springer: 257–274.

Bradley, B.A.; Houghton, R.A.; Mustard, J.F.; [et al.]. 2006. Invasive grass reduces aboveground carbon stocks in shrublands of the western US. Global Change Biology. 12: 1815–1822.

Briske, D.D.; Fuhlendorf, S.D.; Smeins, F.E. 2005. State-and-transition models, thresholds, rangeland health: A synthesis of ecological concepts and perspectives. Rangeland Ecology and Management. 58: 1–10.

Briske, D.D.; Joyce, L.A.; Polley, H.W.; [et al.]. 2015. Climate change adaptation on rangelands: Linking regional exposure with diverse adaptive capacity. Frontiers in Ecology and Environment. 13: 249–256.

Brockett, B.H.; Biggs, H.C.; van Wilgen, B.W. 2001. A patch mosaic burning system for conservation areas in southern Africa savannas. International Journal of Wildland Fire. 10: 169–183.

Brooks, M.L.; D'Antonio, C.M.; Richardson, D.M.; [et al.]. 2004. Effects of invasive alien plants on fire regimes. BioScience. 54(7): 677–688.

Brown, J.K. 1982. Fuel and fire behavior prediction in big sagebrush. Res. Pap. INT-RP-290. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 10 p.

Bucharova, A. 2017. Assisted migration within species range ignores biotic interactions and lacks evidence. Restoration Ecology. 25: 14–18.

Butler, P.R.; Swanston, C.W.; Janowiak, M.K.; [et al.]. 2012. Adaptation strategies and approaches. In: Swanston, C.W.; Janowiak, M.K., eds. Forest adaptation resources: Climate change tools and approaches for land managers. Gen. Tech. Rep. NRS-GTR-87. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station: 15–34.

Carwardine, J.; O'Connor, T.; Legge, S.; [et al.]. 2011. Priority threat management to protect Kimberley wildlife. CSIRO Tech. Rep. Brisbane, Australia: CSIRO Ecosystem Sciences. csiro:EP102445. https://doi.org/10.4225/08/584ee8578e288

Caudle, D.; DiBenedetto, J.; Karl, M.; [et al.]. 2013. Interagency ecological site handbook for rangelands. Washington, DC: U.S. Department of the Interior, Bureau of Land Management; U.S. Department of Agriculture, Forest Service and Natural Resources Conservation Service. http://directives.sc.egov.usda.gov/viewerFS.aspx?hid=33150. [Accessed May 23, 2018].

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p.

Chambers, J.C.; Bradley, B.A.; Brown, C.A.; [et al.]. 2014a. Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. invasion in the cold desert shrublands of western North America. Ecosystems. 17: 360–375.

Chambers, J.C.; Miller, R.F.; Board, D.I.; [et al.]. 2014b. Resilience and resistance of sagebrush ecosystems: Implications for state and transition models and management treatments. Rangeland Ecology and Management. 67: 440-454.

Cook, B.I.; Ault, T.R.; Smerdon, J.E. 2015. Unprecedented 21st century drought risk in the American Southwest and Central Plains. Science Advances. 1: e1400082.

Council on Environmental Quality [CEQ]. 2016. Memorandum for heads of federal departments and agencies. Issued Aug. 1, 2016. Washington, DC: Council on Environmental Quality. 34 p.

Crimmins, M.A.; Comrie, A.C. 2004. Interactions between antecedent climate and wildfire variability across southeastern Arizona. International Journal of Wildland Fire. 13(4): 455–466.

Cross, M.S.; McCarthy, P.D.; Garfin, G.; [et al.]. 2013. Accelerating adaptation of natural resource management to address climate change. Conservation Biology. 27: 4–13.

Dai, A. 2013. Increasing drought under global warming in observations and models. Nature Climate Change. 3: 52–58.

David, E. 2013. Innovative snow harvesting technology increases vegetation establishment success in native sagebrush ecosystem restoration. Plant and Soil. 373: 843–856.

Davies, K.W.; Sheley, R.L.; Bates, J.D. 2008. Does fall prescribed burning of *Artemisia tridentata* steppe promote invasion or resistance to invasion after a recovery period? Journal of Arid Environments. 72: 1076–1085.

Dennison, P.E.; Brewer, S.C.; Arnold, J.D.; [et al.]. 2014. Large wildfire trends in the western United States, 1984–2011. Geophysical Research Letters. 41: 2928–2933.

Doherty, M.K. 2007. Mosquito populations in the Powder River Basin, Wyoming: A comparison of natural, agricultural and effluent coal-bed natural gas aquatic habitats. Thesis. Bozeman, MT: Montana State University. https://scholarworks.montana.edu/xmlui/bitstream/handle/1/1184/DohertyM1207.pdf?sequence=1. [Accessed May 23, 2018].

Field, S.A.; Tyre, A.J.; Jonzen, N.; [et al.]. 2004. Minimizing the cost of environmental management decisions by optimizing statistical thresholds. Ecology Letters. 7: 669–675.

Finch, D.M.; Pendleton, R.L.; Reeves, M.C.; [et al.]. 2016. Rangeland drought: Effects, restoration, and adaptation. In: Vose, J.M.; Clark, J.S.; Luce, C.H.; [et al.], eds. Effects of drought on forests and rangelands in the United States: A comprehensive science synthesis. Gen. Tech. Rep. WO-GTR-93b. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office: 155–194.

Fuhlendorf, S.D.; Hovick, T.J.; Elmore, R.D.; [et al.]. 2017. A hierarchical perspective to woody plant encroachment for conservation of prairie-chickens. Rangeland Ecology and Management. 70: 9–14.

Germino, M.J.; Belnap, J.; Stark, J.M.; [et al.]. 2016. Ecosystem impacts of exotic annual invaders in the genus *Bromus*. In: Germino, M.J.; Chambers, J.C.; Brown, C.S., eds. Exotic brome-grasses in arid and semiarid ecosystems of the western U.S. New York, NY: Springer: 61–95.

Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K; [et al.], eds. 2018a. Climate change vulnerability and adaptation in the Northern Rocky Mountains [Parts 1 and 2]. Gen. Tech. Rep. RMRS GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 475 p.

Halofsky, J.E.; Peterson, D.L.; Ho, J.J.; [et al.], eds. 2018b. Climate change vulnerability and adaptation in the Intermountain Region [Parts 1 and 2]. Gen. Tech. Rep. RMRS-GTR-375. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 509 p.

Hardegree, S.P.; Schneider, J.M.; Moffet, C.A. 2012. Weather variability and adaptive management for rangeland restoration. Rangelands. 34: 53–56.

Heller, N.E.; Zavaleta, E.S. 2009. Biodiversity management in the face of climate change: A review of 22 years of recommendations. Biological Conservation. 142: 14–32.

Hobbs, R.J.; Higgs, E.; Harris, J.A. 2009. Novel ecosystems: Implications for conservation and restoration. Trends in Ecology and Evolution. 24: 599–605.

Homer, C.G.; Xian, X.; Aldridge, C.L.; [et al.]. 2015. Forecasting sagebrush ecosystem components and Greater sage-grouse habitat for 2050: Learning from past climate patterns and Landsat imagery to predict the future. Ecological Indicators. 55: 131–145.

Huffman, M.R. 2013. The many elements of traditional fire knowledge: Synthesis, classification, and aids to cross-cultural problem solving in fire-dependent systems around the world. Ecology and Society. 18: 3.

Intergovernmental Panel on Climate Change [IPCC]. 2014. Climate change 2014: Impacts, adaptation, and vulnerability. In: Barros, V.R.; Field, C.B.; Dokken, D.J.; [et al.], eds. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. New York, NY: Cambridge University Press, 688 p.

Joyce, L.A.; Briske, D.D.; Brown, J.R.; [et al.]. 2013. Climate change and North American rangelands: Assessment of mitigation and adaptation strategies. Rangeland and Ecology Management. 66: 512–528.

Kay, C.E. 1995. Aboriginal overkill and native burning: Implications for modern ecosystem management. Western Journal of Applied Forestry. 10: 121–126.

Knick, S.T.; Hanser, S.E.; Miller, R.F.; [et al.]. 2011. Ecological influence and pathways of land use in sagebrush. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 203–251.

Knick, S.T.; Hanser, S.E.; Preston, K.L. 2013. Modeling ecological minimum requirements for distribution of Greater sage-grouse leks: Implications for population connectivity across their western range, U.S.A. Ecology and Evolution. 3: 1539–1551.

Knight, D.H.; Jones, G.P.; Reiners, W.A.; [et al.]. 2014. Mountains and plains: The ecology of Wyoming landscapes. Second edition. New Haven, CT: Yale University Press. 404 p.

Kunkel, K.E.; Palecki, M.; Ensor, L.; [et al.]. 2009. Trends in twentieth-century U.S. snowfall using a quality-controlled dataset. Journal of Climate. 26: 33–44.

Kunkel, K.E.; Robinson, D.A.; Champion, S.; [et al.]. 2016. Trends and extremes in northern hemisphere snow characteristics. Current Climate Change Reports. 2: 65–73.

Kunkel, K.E.; Stevens, L.E.; Stevens, S.E.; [et al.]. 2013a. Regional climate trends and scenarios for the U.S. national climate assessment. Part 4. Climate of the U.S. Great Plains. NOAA Technical Report, NESDIS 142-4. Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration. 82 p.

Kunkel, K.E.; Stevens, L.E.; Stevens, S.E.; [et al.]. 2013b. Regional climate trends and scenarios for the U.S. national climate assessment. Part 5. Climate of the Southwest U.S. NOAA Technical Report NESDIS 142-5. Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration. 79 p.

Kunkel, K.E.; Stevens, L.E.; Stevens, S.E.; [et al.]. 2013c. Regional climate trends and scenarios for the U.S. National Climate Assessment. Part 6. Climate of the Northwest U.S. NOAA Technical Report NESDIS 142-6. Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration. 83 p.

Lauenroth, W.K.; Schlaepfer, D.R.; Bradford, J.B. 2014. Interactions between climate and habitat loss effects on biodiversity: A systematic review and meta-analysis. Ecosystems. 17: 1469–1479.

Lesica, P.; Deluca, T.H. 1996. Long-term harmful effects of crested wheatgrass on Great Plains grassland ecosystems. Journal of Soil and Water Conservation. 51: 408–409.

Littell, J.S.; McKenzie, D.; Peterson, D.L.; [et al.]. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications. 19: 1003–1021.

Livneh, B.; Deems, J.S.; Buma, B.; [et al.]. 2015. Catchment response to bark beetle outbreak and dust-on-snow in the Colorado Rocky Mountains. Journal of Hydrology. 523: 196–210.

Luce, C.H.; Pederson, N.; Campbell, J.; [et al.]. 2016. Characterizing drought for forested landscapes and streams. In: Vose, J.M.; Clark, J.S.; Luce, C.H.; [et al.], eds. Effects of drought on forests and rangelands in the United States: A comprehensive science synthesis. Gen. Tech. Rep. WO-GTR-93b. Washington, DC: U.S. Department of Agriculture, Forest Service: 14–48.

Maestas, J.; Pellant, M.; Okeson, L.; [et al.]. 2016. Fuel breaks to reduce large wildfire impacts in sagebrush ecosystems. Plant Materials Tech. Note No. 66. Boise, ID: U.S. Department of Agriculture, Natural Resources Conservation Service. 30 p.

McKenzie, D.; Littell, J.S. 2017. Climate change and the eco-hydrology of fire: Will area burned increase in a warming western USA? Ecological Applications. 27: 26–36.

Mealor, B.A.; Mealor, R.D.; Kelley, W.K.; [et al.]. 2013. Cheatgrass management handbook: Managing an invasive annual grass in the Rocky Mountain Region. Laramie, WY: University of Wyoming; Fort Collins, CO: Colorado State University. 131 p.

Millar, C.I.; Skog, K.E.; McKinley, D.C.; [et al.]. 2012. Chapter 4. Adaptation and mitigation. In: Vose, J.M.; Peterson, D.L.; Patel-Weynand, T., eds. Effects of climatic variability and change on forest ecosystems: A comprehensive science synthesis for the U.S. forest sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Millar, C.I; Stephenson, N.L.; Stephens, S.L. 2007. Climate change and forests of the future: Managing in the face of uncertainty. Ecological Applications. 17: 2145–2151.

Mote, P.W.; Sharp, D. 2016. 2016 update to data originally published in: Mote, P.W.; Hamlet, A.F.; Clark, M.P.; [et al.]. 2005. Declining mountain snowpack in Western North America. B. American Meteorological. Society. 86: 39–49.

Painter, T.H.; Barnett, A.P.; Landry, C.C.; [et al.]. 2007. Impact of disturbed desert soils on duration of mountain snow cover. Geophysical Research Letters. 34: L12502.

Painter, T.H.; Skiles, S.M.; Deems, J.S.; [et al.]. 2012. Dust radiative forcing in snow of the Upper Colorado River Basin: 1. A 6 year record of energy balance, radiation, and dust concentrations. Water Resources Research. 48: W07521.

Palmquist, K.A.; Schlaepfer, D.R.; Bradford, J.B.; [et al.]. 2016a. Mid-latitude shrub steppe plant communities: Climate change consequences for soil water resources. Ecology. 97: 2342–2354.

Palmquist, K.A.; Schlaepfer, D.R.; Bradford, J.B.; [et al.]. 2016b. Spatial and ecological variation in dryland ecohydrological responses to climate change: Implications for management. Ecosphere. 7: e01590

Pilliod, D.S.; Welty, J.L.; Arkle, R.S. 2017. Refining the cheatgrass-fire cycle in the Great Basin: Precipitation timing and fine fuel composition predict wildfire trends. Ecology and Evolution. 7: 8126-8151.

Prein, A.F.; Holland, G.J.; Rasmussen, R.M.; [et al.]. 2016. Running dry: The U.S. Southwest's drift into a drier climate state. Geophysical Research. 43: 1272–1279.

Pyke, D.A. 2011. Restoring and rehabilitating sagebrush habitats. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 531–548.

Pyle, W.H.; Crawford, J.A. 1996. Availability of foods of sage grouse chicks following prescribed fire in sagebrush-bitterbrush. Journal of Range Management. 49: 320–324.

Rau, B.M.; Johnson, D.W.; Blank, R.R.; [et al.]. 2011a. Transition from sagebrush steppe to annual grass (*Bromus tectorum*): Influence on belowground carbon and nitrogen. Rangeland Ecology and Management. 64: 139–147.

Rau, B.M.; Johnson, D.W.; Blank, R.R.; [et al.]. 2011b. Woodland expansion's influence on belowground carbon and nitrogen in the Great Basin U.S. Journal of Arid Environments. 75: 827–835.

Rau, B.M.; Tausch, R.; Reiner, A.; [et al.]. 2012. Developing a model framework for predicting effects of woody expansion and fire on ecosystem carbon and nitrogen in a pinyon-juniper woodland. Journal of Arid Environments. 76: 97–104.

Rehfeldt, G.E.; Crookston, N.L.; Enz-Romero, C.M.; [et al.]. 2012. North American vegetation model for land-use planning in a changing climate: A solution to large classification problems. Ecological Applications. 22: 119–141.

Rehfeldt, G.E.; Crookston, N.L.; Warwell, M.; [et al.]. 2006. Empirical analyses of plant-climate relationships for the western United States. Journal of Plant Science. 167: 1123–1150.

Rhoades, C.C.; Elder, K.; Green, E. 2010. The influence of an extensive dust event on snow chemistry in the Southern Rocky Mountains. Arctic, Antarctic, and Alpine Research. 42: 98–105.

Richardson, B.A.; Shaw, N.L.; Pendleton, R.L. 2012. Chapter 4. Plant vulnerabilities and genetic adaptation. In: Finch, D.M., ed. Climate change in grasslands, shrublands, and deserts of the interior American West: A review and needs assessment. Gen. Tech. Rep. RMRS-GTR-285. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 48–59.

Ricketts, A.M.; Sandercock, B.K. 2016. Patch-burn grazing increases habitat heterogeneity and biodiversity of small mammals in managed rangelands. Ecosphere. 7: e01431.

Romme, W.H.; Allen, C.D.; Bailey, J.D.; [et al.]. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon–juniper vegetation of the western United States. Rangeland Ecology and Management. 62: 203–222.

Sage-grouse National Technical Team [NTT]. 2011. A report on national Greater sage-grouse conservation measures. 74 p. https://eplanning.blm.gov/epl-front-office/projects/lup/9153/39961/41912/WySG_Tech-Team-Report-Conservation-Measure_2011.pdf. [Accessed May 22, 2018].

Schlaepfer, D.R.; Lauenroth, W.K.; Bradford, J.B. 2012a. Consequences of declining snow accumulation for water balance of mid-latitude dry regions. Global Change Biology. 18: 1988–1997.

Schlaepfer, D.R.; Lauenroth, W.K.; Bradford, J.B. 2012b. Ecohydrological niche of sagebrush ecosystems. Ecohydrology. 5: 453–466.

Schlaepfer, D.R.; Lauenroth, W.K.; Bradford, J.B. 2012c. Effects of ecohydrological variables on current and future ranges, local suitability patterns, and model accuracy in big sagebrush. Ecography. 35: 374–384.

Schrag, A.; Konrad, S.; Miller, S.; [et al.]. 2010. Climate-change impacts on sagebrush habitat and West Nile virus transmission risk and conservation implications for Greater sage-grouse. GeoJournal. 76: 561–575.

Star, J.; Rowland, E.L.; Black, M.E.; [et al.]. 2016. Supporting adaptation decisions through scenario planning: Enabling the effective use of multiple methods. Climate Risk Management. 13: 88–94.

Steltzer, H.; Landry, C.; Painter, T.H.; [et al.]. 2009. Biological consequences of earlier snowmelt from desert dust deposition in alpine landscapes. Proceedings of the National Academy of Science. 106: 11629–11634.

Still, S.M.; Richardson, B.A. 2015. Projections of contemporary and future climate niche for Wyoming big sagebrush (*Artemisia tridentata* subsp. *wyomingensis*): A guide for restoration. Natural Areas Journal. 35: 30–43.

Strand, E.K.; Launchbaugh, K.L.; Limb, R.; [et al.]. 2014. Livestock grazing effects on fuel loads for wildland fire in sagebrush dominated ecosystems. Journal of Rangeland Applications. 1: 35–57.

Svejcar, T.; Angell, R.; Bradford, J.A.; [et al.]. 2008. Carbon fluxes on North American rangelands. Rangeland Ecology and Management. 61: 465–474.

Swetnam, T.W.; Allen, C.D.; Betancourt, J.L. 1999. Applied historical ecology: Using the past to manage for the future. Ecological Applications. 9: 1189–1206.

Toepfer, S.; Borgeson, L.; Edgerly, B.; [et al.]. 2006. The spatial distribution and impact on avalanche conditions of a dust-on-snow event in the Colorado Rocky Mountains. In: International snow science workshop; 2006 October 1–6; Telluride, CO: 913–916. http://arc.lib.montana.edu/snow-science/objects/issw-2006-913-916.pdf. [Accessed May 22, 2018].

Trauernicht, C.; Brook, B.W.; Murphy, B.P.; [et al.]. 2015. Local and global pyrogeographic evidence that indigenous fire management creates pyrodiversity. Ecology and Evolution. 5: 1908–1918.

University of Nebraska-Lincoln, National Drought Mitigation Center (UNL-NDMC). 2012. Managing drought risk on the ranch: A planning guide for Great Plains ranchers. Lincoln, NE: University of Nebraska, National Drought Mitigation Center. 39 p. http://drought.unl.edu/ranchplan/WriteaPlan.aspx. [Accessed May 22, 2018].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2011. National roadmap for responding to climate change. FS-957b. Washington, DC: U.S. Department of Agriculture, Forest Service. 32 p. http://www.fs.fed.us/climatechange/pdf/Roadmapfinal.pdf. [Accessed May 22, 2018].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2012. Future of America's Forest and Rangelands: Forest Service 2010 Resources Planning Act Assessment. Gen. Tech. Rep. WO-GTR-87. Washington, DC: U.S. Department of Agriculture, Forest Service. 198 p.

U.S. Department of the Interior [USDOI]. 2016. Safeguarding America's lands and waters from invasive species: A national framework for early detection and rapid response. Washington, DC: U.S. Department of the Interior. 55 p. https://www.doi.gov/sites/doi.gov/files/National%20EDRR%20Framework.pdf. [Accessed May 22, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2010. Rising to the urgent challenge: Strategic plan for responding to accelerating climate change. Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service. 36 p. https://www.fws.gov/home/climatechange/pdf/CCStrategicPlan.pdf. [Accessed May 22, 2018].

U.S. Department of the Interior, National Park Service [USDOI NPS]. 2013. Using scenarios to explore climate change: A handbook for practitioners. Fort Collins, CO: U.S. Department of the Interior, National Park Service Climate Change Response Program. https://www.nps.gov/subjects/climatechange/upload/CCScenariosHandbookJuly2013.pdf. [Accessed May 22, 2018].

Voleti, R.; Winter, S.L.; Leis, S. 2014. Patch burn-grazing: An annotated bibliography. Papers in Natural Resources 462. 12 p. http://digitalcommons.unl.edu/natrespapers/462. [Accessed May 22, 2018].

Walker, B.L.; Naugle, D.E. 2011. West Nile virus ecology in sagebrush habitat and impacts on Greater sage-grouse populations. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 127–142.

Westerling, A.; Brown, T.; Schoennagel, T.; [et al.]. 2014. Briefing: Climate and wildfire in Western U.S. forests. In: Sample, V.A.; Bixler, R.P., eds. Forest conservation and management in the Anthropocene: Conference proceedings. Proceedings RMRS-P-71. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 81–102.

4. WILDLAND FIRE AND VEGETATION MANAGEMENT

Michele R. Crist, Jeanne C. Chambers, and Jonathon A. Skinner

Introduction

Wildland fire has always been an important ecosystem process across the sagebrush biome. Recently, the scale of sagebrush ecosystem loss and fragmentation has increased due to a combination of uncharacteristic wildland fire, invasive annual grasses, juniper *(Juniperus* spp.) and piñon *(Pinus* spp.) expansion, and anthropogenic land use and development. A strategic approach to wildland fire and vegetation management is now required that focuses available resources in the places that will maximize conservation return on investment. Wildland fire management integrated with vegetation management (fuel reduction and ecosystem restoration) has the potential to increase that return on investment by enhancing the resilience of native sagebrush ecosystems to stress and disturbance and resistance to invasive annual grasses. Similarly, vegetation management along with postfire restoration helps maintain functionally diverse plant communities with the capacity to persist and stabilize ecosystem processes under altered disturbance regimes. When placed in the context of large landscapes, these actions collectively are part of a strategy to maintain the necessary ecosystem processes and connectivity that allow ecosystems and species to adapt to increasing pressure from anthropogenic land use and development and fluctuations in climate.

Managing for Wildland Fire-Resilient Ecosystems

An understanding of the links among ecosystem resilience to disturbance and resistance to invasive annual grasses, priority areas and habitats for management, and the predominant threats is useful for effectively targeting wildland fire and vegetation management actions. Definitions of wildland fire and related terms are in Appendix 1. In the context of the Science Framework, wildland fire has varying negative and positive effects on sagebrush communities, depending on a site's relative resilience to disturbance and resistance to invasive annual grasses (see Chambers et al. 2017 [hereafter, Part 1], sections 5.1 and 6). Geospatial analyses can be used to assess the relative resilience and resistance of areas that support species or resources at risk. They also can be used to assess the probability of wildland fire occurring within these areas and the interactions of fire with resilience and resistance in sagebrush habitats (see tables 1.3, 1.4; Part 1, sections 8 and 9).

Top: Aerial drop of fire retardant onto a wildfire (photo: USDOI Bureau of Land Management). Middle left: Fire crew on fire line (photo: USDOI Bureau of Land Management). Middle right: Planting sagebrush and other native plants after a fire on land managed by BLM (photo: Tetona Dunlap, Courtesy of TIMES-NEWS, magicvalley.com). Middle center: Removing juniper by cutting the trees with chainsaws (photo: Jeremy Roberts, Conservation media). Bottom: Mowed fuel break along road (photo: USDOI Bureau of Land Management).

Identifying Greater sage-grouse (*Centrocercus urophasianus*; hereafter, GRSG) habitats at risk from wildland fire involves overlaying key data layers to both visualize and quantify: (1) the likely response of the area to either fire or management treatments (i.e., an area's resilience and resistance to invasive annual grasses), (2) the probability that an area has suitable GRSG breeding habitat and supports GRSG populations, and (3) the exposure to dominant threats. Using geospatial analysis to quantify areas within different resilience and resistance to invasive annual grasses and habitat categories, along with different burn probabilities, by ecoregion, Management Zone (fig. 1.1), or Priority Areas of Conservation within Management Zones for GRSG, provides additional information for prioritization.

A wildland fire risk assessment was conducted using GIS data layers to understand how resilience to disturbance and resistance to invasive annual grasses may inform wildland fire management related to preparedness, suppression, fuel management, and postfire restoration within GRSG habitat across the sagebrush biome (Part 1, Appendix 10). Three GIS datasets were used: burn probability (Short et al. 2016); GRSG breeding habitat probabilities (Doherty et al. 2016); and resilience and resistance as indicated by soil temperature and moisture regimes (Maestas et al. 2016b). The wildland fire risk assessment spatially identifies areas where ecosystem resilience and resistance interact and where sagebrush and GRSG habitats are at highest risk from fire across the sagebrush biome and current GRSG range (fig. 4.1). The wildland fire risk assessment is useful to: (1) evaluate the level of fire risk to vegetation types and species, (2) target areas for fire management, and (3) determine the most appropriate types of fire management actions based on an ecosystem's resilience to fire and resistance to invasive annual grasses. Incorporating spatial information on invasive annual grass occurrence, juniper and piñon expansion, and threatened and endangered species in the risk assessment can further inform the type of management actions and the allocation of budgets at broad (biome) and mid- (ecoregion or Management Zone) scales, as well as local (project or site) scales. Note that in the eastern part of the sagebrush biome, invasive annual grass/fire cycles are an emerging problem (Baker 2011; Floyd et al. 2004, 2006; Mealor et al. 2012, 2013) that modeled burn probabilities, based on historical burn areas, do not illustrate well.

Broad- to Mid-Scale Considerations

Wildland Fire Preparedness, Suppression, and Prevention

Optimizing wildland fire preparedness and suppression response is highly complex and considers fire danger, availability of suppression resources, access to and remoteness of the fire, and many other ecological, social, political, and economic variables. Federal land management agencies and their partners are starting to incorporate sagebrush conservation into wildland fire management decisions across the sagebrush biome. Fire operations and integrated vegetation management programs, coupled with fire simulation modeling, contribute to a strategic, landscape approach based on the likelihood of wildland fire occurrence and potential fire behavior (Finney et al. 2010; Oregon Department of Forestry 2013). Numerous factors influence the placement of fire management resources, including safety, climate, weather, human values, infrastructure, and natural resource considerations. In the sagebrush biome, the Integrated Rangeland Fire Management Strategy (IRFMS) (USDOI 2015) directs fire managers to assess preparedness and suppression responses based on the location of GRSG habitats and populations, resilience and resistance information, and other factors.

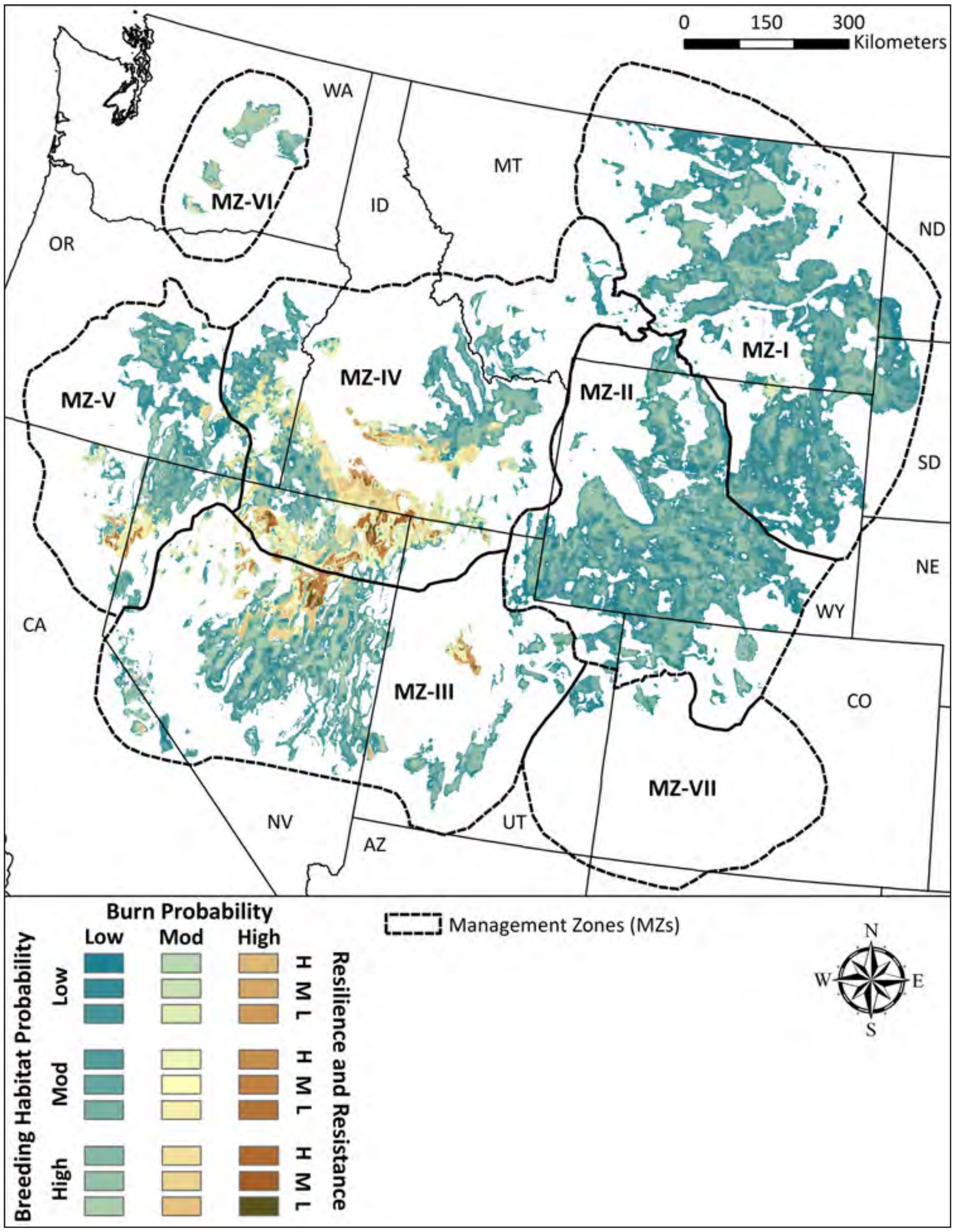

Figure 4.1—Wildland fire risk map (Chambers et al. 2017, Appendix 10; Crist et al. 2016) depicting 27 different combinations of Greater sage-grouse breeding habitat probability (Doherty et al. 2016), resilience and resistance (Maestas et al. 2016b), and large fire probability (Short et al. 2016).

Table 4.1—Considerations for prioritizing wildfire operations response to wildfires burning in GRSG habitat. These consideration are consistent with tables 1.3 and 1.4.

- In general, areas that support medium to high GRSG breeding habitat probabilities (or other important resources) and have moderate to high wildfire risk are higher priorities for preparedness and suppression efforts, especially in low resilience and resistance categories (figs. 4.1, 4.2).
- Areas with moderate and, especially, high resilience and resistance often have the potential to recover through successional processes without management intervention (table 1.3: cells 1B, 1C, 2B, 2C; fig. 4.3). Wildfire suppression priority typically increases from low to moderate as resilience and resistance decreases from high to moderate.
- Areas adjacent to high to moderate priority habitats may be places to focus wildfire operations activities to protect priority habitats from burning during wildfire events, especially areas with low resilience and resistance that have converted to annual grasses and are prone to frequent wildfires (table 1.3: cells 1A, 2A, 3A; fig. 4.4).
- Areas with low resilience and resistance often lack the potential to recover without significant intervention. Wildfire suppression priority typically increases from moderate to high as GRSG breeding habitat probabilities and population abundances increase from moderate to high (table 1.3: cells 3B, 2C; fig. 4.2). Cheatgrass land cover layers can help identify these areas.
- Newly rehabilitated areas and areas that provide sagebrush habitat connectivity are conservation priorities and considered fire suppression priorities. Sagebrush land cover layers can help identify these areas.
- Managing wildfires in sagebrush habitats in high resilience and resistance juniper and piñon expansion areas can be part of a vegetation management strategy where: (1) weather and fuel conditions allow for managing the wildfire within acceptable limits to values at risk, (2) high priority GRSG breeding habitats and the associated populations are not at risk from loss, and (3) sufficient perennial native grasses and forbs exist to promote recovery.

The Science Framework and the GRSG wildland fire risk assessment provide a spatial framework and management considerations for prioritizing fire suppression efforts for GRSG habitats and populations (table 4.1; fig. 4.1). Geospatial datasets and the mapping process are detailed in Part 1, sections 8 and 9. This information, combined with many other risk factors, such as the wildland-urban interface, is used in the decisionmaking processes for preparing and responding to wildland fires across the Nation. Differences in environmental characteristics, resource values, predominant threats, and management strategies are included to further refine prioritizations across the sagebrush biome. For rapid response in GRSG habitat, combining results of the wildland fire risk assessment (table 4.1; figs. 4.1, 4.2, 4.3, 4.4) with National Interagency Fire Center (NIFC) Predictive Services 7-day potential fire forecasts informs where to pre-position fire crews, equipment, and aircraft in areas predicted to experience fire ignitions and large fire growth.

The mapping products described earlier are used to identify suppression priorities for GRSG and their habitats and to respond to incidents and assign resources at broad- and mid-scales. Fire managers can distribute the wildland fire risk assessment and other geospatial data layers to dispatch offices, incident commanders, fire crew bosses, and other fire responders. Recently, cooperators contributing to suppressing fire in sagebrush habitats include rural, city, and State agencies as well as Rangeland Fire Protection Associations. Sharing these mapping resources may help coordinate and improve initial attack effectiveness during periods of increased fire activity.

In fire preparedness and suppression efforts, the road network is a key element for quick wildland fire response. It also functions as a fuel break network by disrupting fuel continuity across large scales (Agee et al. 2000; Narayanaraj and Wimberly 2013; Syphard et al. 2011). Travel and recreation planning processes identify a minimum road network needed to maintain access for all aspects of land management. The geospatial data layers from the Science Framework, Part 1 are useful for identifying priorities for road maintenance and updates to standards in travel and recreation management planning efforts. Prioritizing roads in travel planning for fire management access and maintenance that are near GRSG habitat areas, at high risk of fire, and characterized by low resilience and resistance to invasive annual grasses will contribute to an effective response to fire (fig. 4.1).

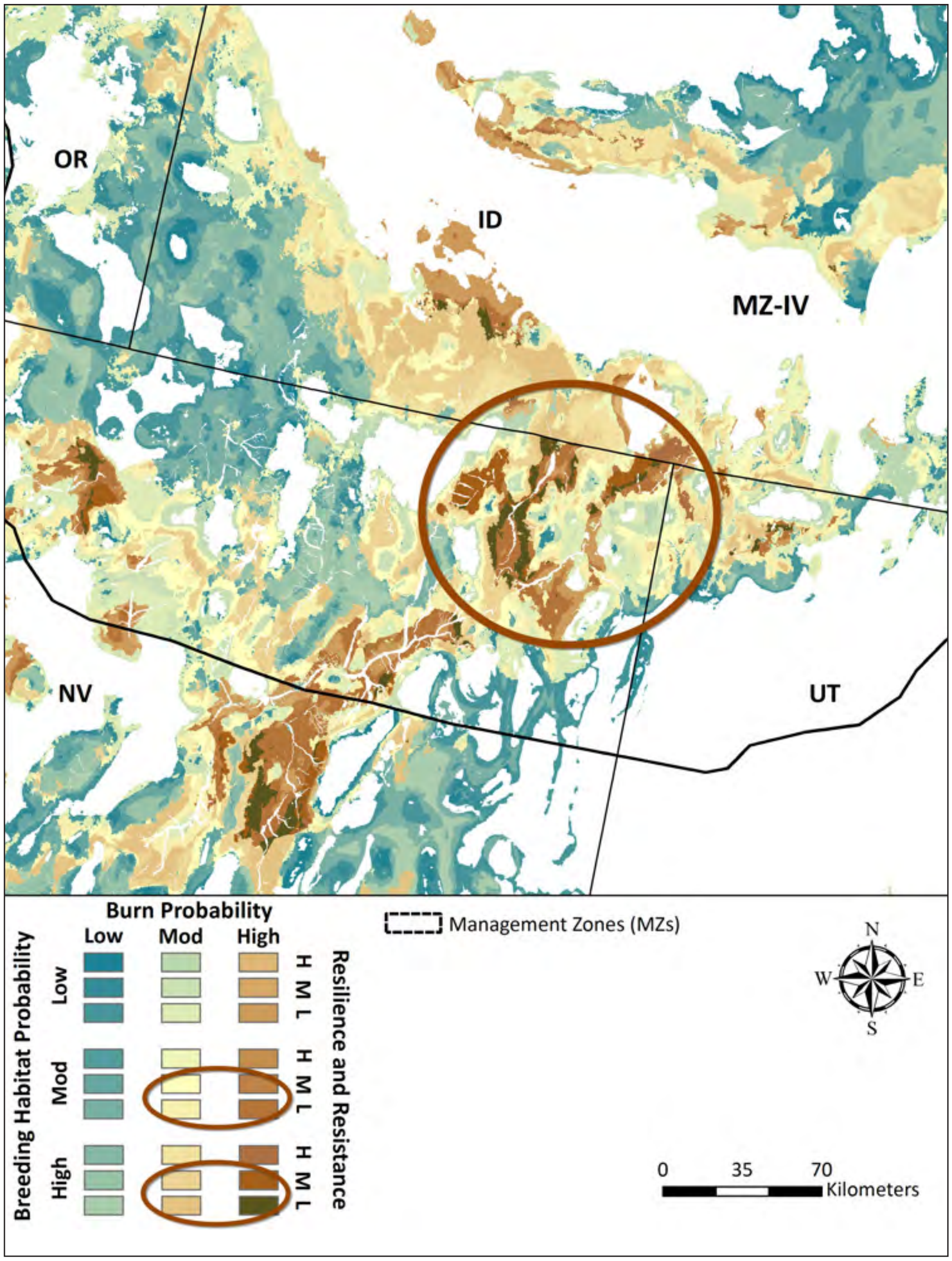

Figure 4.2—Wildland fire risk map (Chambers et al. 2017, Appendix 10; Crist et al. 2016), where circles depict areas of high to moderate burn probability, high to moderate GRSG habitat probabilities, and low to moderate resilience and resistance. High priorities for management are placing fuel reduction treatments or fuel breaks strategically around GRSG habitats, implementing fire prevention strategies, conducting postfire rehabilitation, and monitoring for spread of nonnative annual grasses. See table 1.3: cells 2B, 2C, 3B, 3C; and table 1.4.

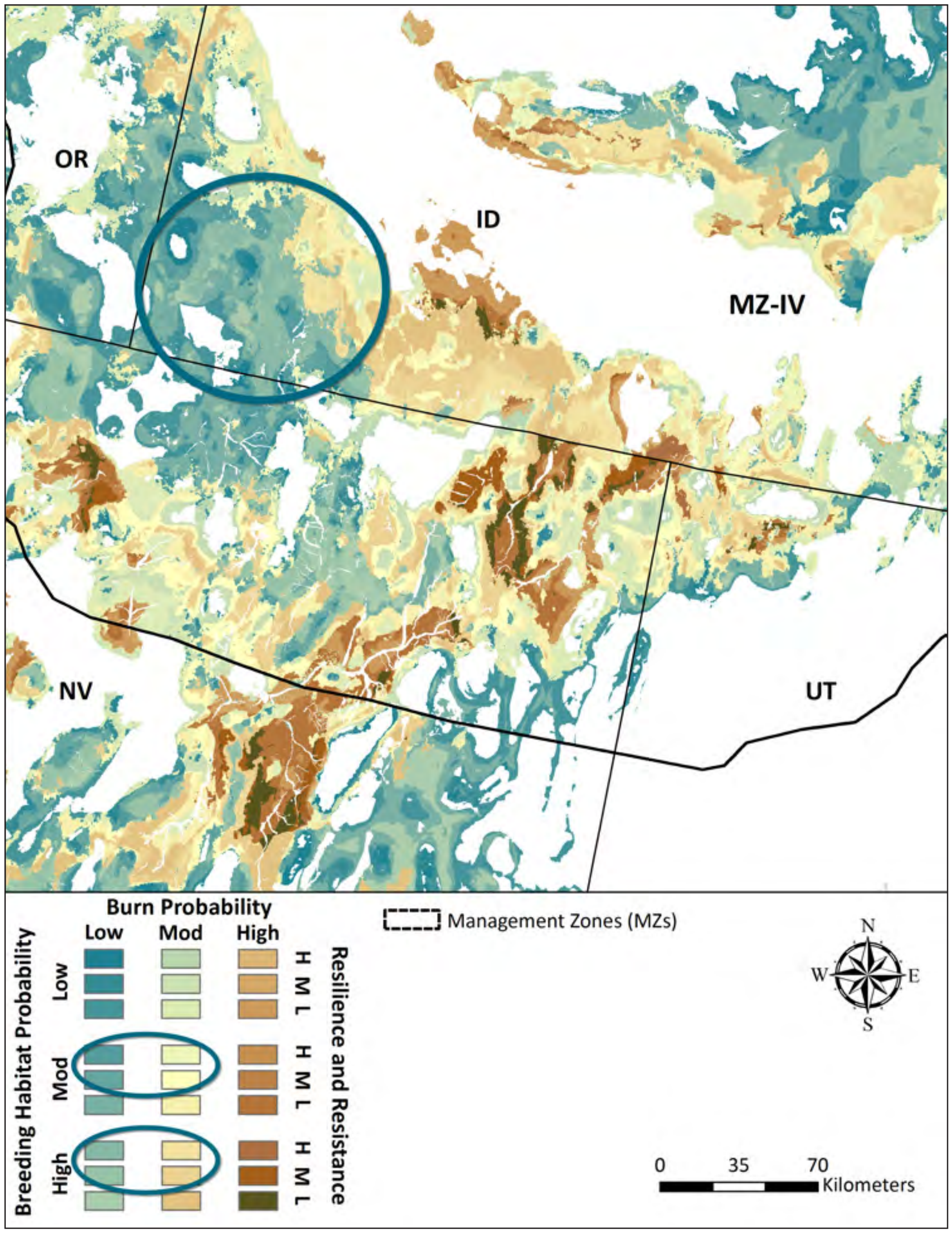

Figure 4.3—Wildland fire risk map (Chambers et al. 2017, Appendix 10; Crist et al. 2016), where circles depict areas of low to moderate burn probability, high to moderate GRSG habitat probabilities, and high to moderate resilience and resistance. High priorities for management are removing juniper and piñon in expansion areas, allowing natural recovery after fire without intervention, and monitoring for new invasions of nonnative annual grasses and changes in fire frequencies. See table 1.3: cells 1B, 1C, 2B, 2C; and table 1.4.

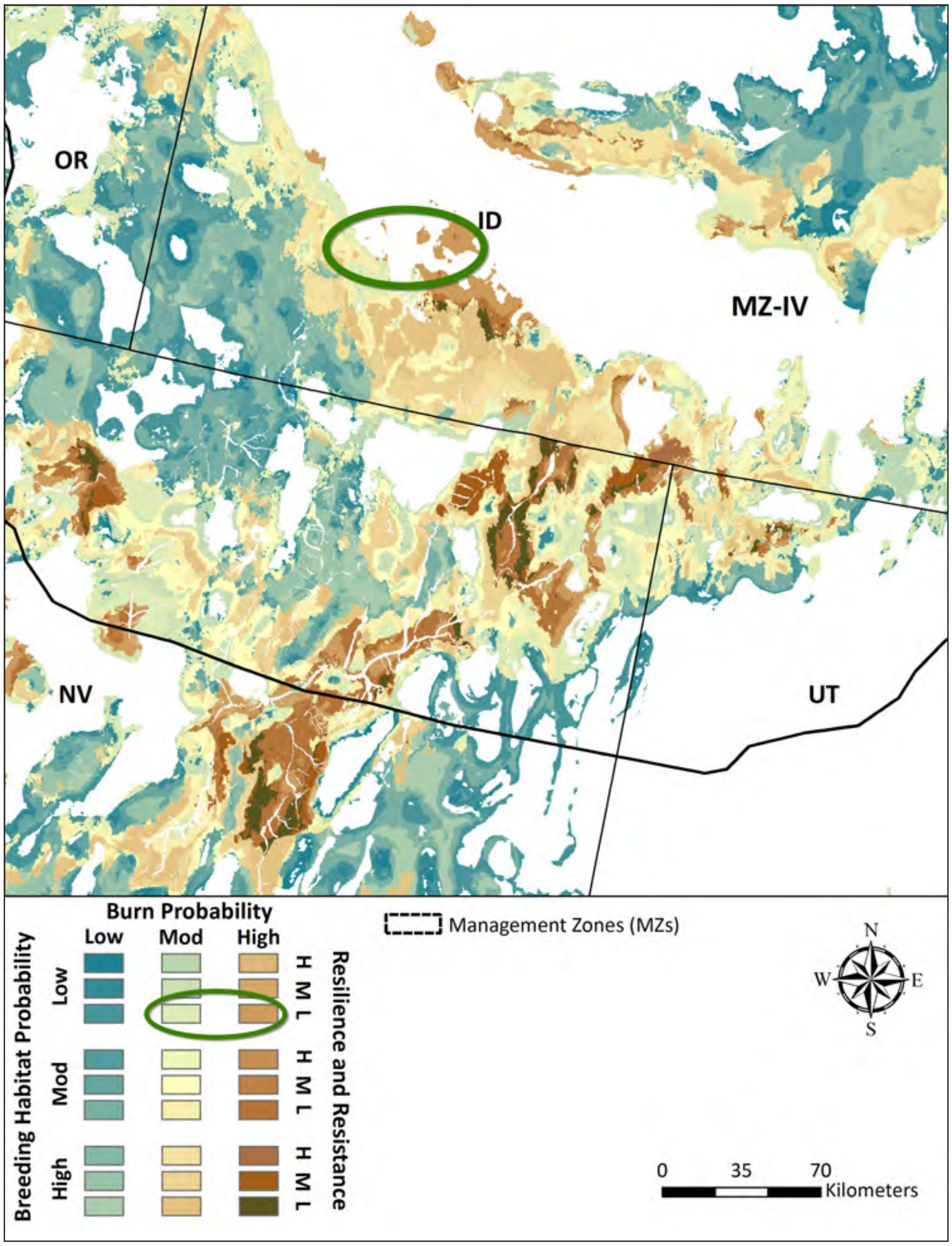

Figure 4.4—Wildland fire risk map (Chambers et al. 2017, Appendix 10; Crist et al. 2016), where circles depict areas of high to moderate burn probability, high to moderate GRSG habitat probabilities, and low to moderate resilience and resistance. High priorities for management are fuel reduction and fuel breaks, fire prevention strategies, and monitoring for changing conditions. See table 1.3: cells 2A, 3A; and table 1.4.

Wildland Fire Prevention—Human ignitions account for thousands of wildland fires each year across the western United States and well over half of all wildland fires annually (NIFC 2017). Many of these fires occur near wildland-urban interface areas and require a substantial fire suppression response. These fires can take firefighting resources away from fires occurring in sagebrush habitat and other high-value resource areas, especially when multiple fire starts occur during high-wind or lightning events. Areas at most risk from human-caused fires are sagebrush ecosystems with low resilience to fire and resistance to invasive annual grasses located near the wildland-urban interface. These human-caused fires tend to ignite easily and spread quickly, and they are difficult to control, especially in areas where continuous fuel from invasive annual grasses are present. Once fires start, options to protect and rehabilitate these sagebrush ecosystems are limited. Increases in invasive annual grasses post-fire are typical, which result in more human-caused fire ignitions and increase fire suppression costs over time. This annual grass/fire cycle could be disrupted with an effective fire prevention program that reduces human-caused fires in sagebrush ecosystems. Targeted fire prevention efforts that include **education, engineering,** and **enforcement** actions are proven to be successful in preventing human-caused ignitions.

The first step in fire prevention is using **education** to create awareness of new and common human fire causes, and inform citizens of the wildland fire risk and consequences to priority sagebrush areas that many native plant and wildlife species, and human communities, depend on. Analysis of the causes for human-ignited fires helps identify the main factors in human ignitions such as who started the fires, what caused the fires, and where and when the fires typically started for a specified area. This information combined with GIS spatial overlays of wildland fire risk based on resilience and resistance and frequency of human-starts (e.g., fig. 4.1) will help to spatially identify the design of educational campaigns, specifically, the locations and audiences that most benefit from protecting sagebrush habitats. Partnerships developed with interest groups, industries, and communities are important to foster an informed public that understands fire risk. To be effective over the long term, education efforts must move from awareness-building to providing specific information on fire safety measures that prevent ignitions by humans, such as proper fire safety procedures for agricultural or debris burning and not parking on dry grass on hot dry windy days.

Engineering actions taken to prevent wildland fires include working with power companies to ensure poles and transmission lines are constructed and maintained properly, especially in areas where repeated failures occur and ignite fires. Engineering also includes designing and maintaining recreation sites to ensure they are void of flammable vegetation that can ignite from human activities. These fire prevention measures are critical and can have an immediate and direct impact in decreasing the number of human-caused fires. Overlaying GIS datasets on locations of transmission corridors and recreation sites with the wildland fire risk for GRSG habitats depicted in figure 4.1, can determine places to prioritize these types of actions that can both decrease human ignitions and reduce fire risk to high quality sagebrush habitats at mid-scales.

Enforcement of fire safety laws and regulations is a must for an effective fire prevention program. Rigorous wildland fire investigations and cost recovery programs should determine fire origin and cause and pursue cost recovery or criminal penalties when appropriate. Fire investigations can help managers learn the cause of human-ignited wildland fires and design and implement fire prevention strategies. An aggressive cost recovery program can be an effective deterrent to human-caused fires, especially for repeat offenders. When covered by the media, cost recovery helps make the public aware of the consequences of starting a wildland fire.

Vegetation Management and Postfire Recovery

The IRFMS establishes key objectives for vegetation management and postfire rehabilitation. Meeting objectives for vegetation management includes improving the prioritization and siting of fuel reduction and management opportunities and ecosystem restoration projects. Considerations for postfire rehabilitation objectives include promoting long-term restoration efforts and natural recovery, updating prioritization criteria, and incorporating science to promote resilience to fire and resistance to invasive annual grasses. Integral to these objectives are considerations of sagebrush habitat in general, GRSG habitat, ecosystem resilience and resistance, and persistent ecosystem threats, such as fire, the current distribution and abundance of invasive annual grasses, and juniper and piñon expansion.

The Science Framework provides a spatial framework and management considerations for prioritizing vegetation management efforts for GRSG habitats and populations similar to those provided for fire suppression efforts (fig. 4.1, table 4.2). Geospatial datasets and the mapping process for prioritization are detailed in Part 1, sections 8 and 9. For mid-scale assessments conducted at the regional level, information on other resource values and the predominant threats are incorporated and the best available data are used. Depending on data availability, other data layers to consider are land cover of invasive annual grasses and juniper and piñon, habitats of other sagebrush dependent species and their movement or migration corridors, and other values at risk such as endangered plant species.

Vegetation Management—Strategic placement of vegetation management projects across large landscapes is an important step to mitigate the collective effects of wildland fires over broad spatial and temporal extents and help conserve sagebrush ecosystem patterns and processes (table 4.2). Assessments for prioritizing fuel reduction and restoration activities should consider potential fire behavior and spread, habitat fragmentation thresholds (Crist et al. 2015; Knick et al. 2013; Manier et al. 2014; Shinneman et al. 2018), minimum habitat patch sizes to support sagebrush dependent species, and corridors and movement pathways between seasonal and dispersal habitats. This information can help target fuel reduction and restoration actions to maintain or increase connected sagebrush areas while increasing capacity to protect areas at high risk from fire.

From a wildland fire behavior perspective, the siting of vegetation management projects should take into account the likelihood of fire spread around large sagebrush-dominated patches to reduce the potential for unwanted fire behavior or effects. In arid sagebrush and woodland ecosystems, increased continuity of invasive annual grass cover, such as cheatgrass (*Bromus tectorum*), can inhibit the natural recovery of native vegetation after fire. Once cheatgrass distribution moves from patchy to continuous, the invasive/fire cycle can lead to more frequent and larger fires, favoring cheatgrass dominance across broad areas. Where GRSG population densities are high and sagebrush ecosystems are intact but at risk of invasive annual grasses, strategically placed fuel reduction treatments may help maintain landscape and habitat resilience to fire (Gray and Dickson 2016). For example, relatively intact sagebrush patches may be located next to large patches of annual invasive grasses with a high likelihood of igniting and facilitating the spread of fire into the larger landscape. Sites already dominated by annual grasses that are low value GRSG habitat should be priorities for pre-positioning fire resources and proactive fuel management practices such as fuel breaks and green strips to avoid future spread into higher-value habitat in the surrounding landscape. More information on fuel break design is offered in *Local Scale Considerations*.

Table 4.2—Considerations for prioritizing vegetation management activities in areas that differ in resilience and resistance and GRSG breeding habitat probabilities. These consideration are consistent with tables 1.3 and 1.4.

- In general, areas that support medium to high GRSG breeding habitat probabilities or other important resources and have moderate to high fire risk (figs. 4.1, 4.2) are higher priorities for vegetation management.
- Areas with moderate and, especially, high resilience and resistance to invasive annual grasses often respond favorably to vegetation management projects (table 1.3: cells 1B, 1C, 2B, 2C; fig. 4.3). The risk of invasive annual grasses increases as resilience and resistance decrease.
  - Focusing tree removal in Phase I to Phase II juniper and piñon expansion areas in or adjacent to areas with high GRSG habitat breeding probabilities and populations (especially near leks) will help maintain resilience and resistance and provide necessary connectivity between sagebrush habitats. Treatment areas should contain sufficient native perennial forbs and grasses to promote recovery.
  - Prescribed fires may also be considered for reducing juniper and piñon expansion in areas with high resilience and resistance to invasive annual grasses. Important management considerations include: (1) timing the fire when weather and fuel conditions allow for managing the fire with acceptable implications to values at risk, (2) selecting areas where high priority GRSG populations and corresponding habitats would not be negatively impacted, and (3) ensuring that sufficient native grasses and forbs exist for recovery.
- Areas with low resilience and resistance to invasive annual grasses typically are more challenging to restore and take a longer time to respond to vegetation management treatments (table 1.3: cells 3B, 3C; fig. 4.2). The risk of invasive annual grasses increases as resilience and resistance decrease.
  - High quality GRSG breeding habitats with moderate to high fire risk and low resilience and resistance may be prioritized for wildfire protection activities but should not be prioritized for vegetation management activities that could degrade habitat quality and connectivity.
  - Areas of low breeding habitat quality in and adjacent to areas with high GRSG breeding habitat probabilities, moderate to high fire risk, and lower resilience and resistance may have higher priorities for fuel breaks (Maestas et al. 2016a).
  - Sagebrush reduction (prescribed fire, mechanical removal, chemical treatment) requires caution and is generally not recommended (table 1.4; also see Beck et al. 2012; Chambers et al. 2014; Davies et al. 2012).
  - Prescribed fire is also used occasionally in conjunction with other treatments to reduce invasive perennials and annual grasses as part of a sagebrush ecosystem restoration strategy. Similar management considerations as stated above should be evaluated when deciding to use this tool in these areas.
- In general, areas that support moderate to high GRSG breeding habitat probabilities or other important resources and have low to moderate resilience and resistance are priorities for postfire rehabilitation (fig. 4.2). In many cases, areas of high or moderate resilience and resistance that are relatively cool and moist recover without management intervention and are lower priorities for postfire rehabilitation (fig. 4.3).

When considering juniper and piñon removal treatments, the broader context of longer-term trends in wildland fire activity, past conifer removals, bark beetles, and climate is helpful in evaluating the need for management treatments (Allen et al. 2015; Arendt and Baker 2013; Board et al. 2018; Romme et al. 2009). Expansion of juniper and piñon woodlands into sagebrush ecosystems has occurred due to favorable climate periods for tree establishment, increases in carbon dioxide, fire suppression, and livestock grazing (Miller et al. 2011, 2013; Romme et al. 2009). This expansion, however, is not uniform across the sagebrush biome; some areas show substantial expansion and other regions show minimal to no expansion and infilling (Manier et al. 2005; Romme et al. 2009) and even declines (Arendt and Baker 2013). While rates of juniper and piñon expansion have slowed in recent decades due to less favorable climatic conditions, fewer suitable sites for tree establishment, and an increase in wildland fire and bark beetle activity in some regions (Breshears et al. 2005; Miller et al. 2008; Romme et al. 2009), infilling of trees appears to continue in expansion areas, most noticeably in the Great Basin (Miller et al. 2008). In general, early- to mid-phase (i.e., phases I and II; see Appendix 1 for definitions) juniper and piñon that have expanded into occupied GRSG breeding habitat with high to moderate resilience to fire and resistance to invasive annual grasses can be considered for removal treatments (table 1.3: cells 1B, 1C, 2A, 2B). Treatments should be conducted in areas with sufficient native perennial grasses and forbs to promote recovery and low risk of increases in invasive annual grasses (see table 1.4). Prescribed fire can be used selectively in consultation with wildlife and habitat managers. Posttreatment grazing deferral is essential to allow recovery of native grasses and forbs and reduce the risk of invasive plants.

Postfire recovery—Large wildland fires occur across environmental gradients and thus the areas burned often differ in their relative resilience and resistance to invasive annual grasses. An understanding of the areas' environmental conditions, dominant vegetation types pre-fire, and disturbance history provides the necessary information to evaluate differences in resilience and resistance, and identify areas where management actions have a higher likelihood of success for restoring ecosystem processes. In addition, this type of approach ensures that the limited rehabilitation funds are placed in the appropriate areas.

In areas with lower resilience and resistance, sagebrush restoration after a wildland fire can take several decades and presents a serious challenge for managers seeking to maintain stable populations of sagebrush dependent wildlife. Strategic placement of postfire recovery efforts to expand sagebrush patch refugia (unburned islands within a burned area) and reconnect these sagebrush patches to intact areas of sagebrush outside of burned areas will help restore large and contiguous sagebrush patches needed by GRSG and other sagebrush dependent species (Pyke 2011; Pyke et al. 2015a,b; Williams et al. 2011). Establishing patches of diverse native forbs, along with bunchgrasses and shrubs, within burned areas can increase the distribution and diversity of forbs, which serve as a foundational building block for resilient sagebrush systems. Seeding sagebrush around existing sagebrush patches can help increase connectivity for many sagebrush dependent species. This type of strategic restoration mimics natural succession where fire-tolerant plants generally resprout and fire-intolerant plants like sagebrush establish from the available seedbank or from seeds that disperse into the disturbed area from nearby unburned patches (Baker 2006; Meyer 1994; Meyer and Monsen 1990; Monsen et al. 2004; Pyke 2011; Rottler et al. 2015). This seeding strategy also addresses funding shortfalls that may not allow for seeding a diverse mixture of forbs, bunchgrasses, and sagebrush across an entire burned area.

Adaptive Management and Monitoring in Wildland Fire Management

Monitoring provides critical information on the effectiveness of management actions, including fuel management and postfire restoration treatments (see section 2). Monitoring data at broad and mid-scales should be used to evaluate changes in (1) vegetation, fuel, and fire characteristics; and (2) ecosystem response to management actions implemented to address ecosystem threats such as invasive annual grasses and juniper and piñon expansion (text box 4.1). Fire-related monitoring indicators are being identified and developed for agency monitoring programs in order to measure the effectiveness of wildland fire and vegetation management in decreasing the current trend of uncharacteristic fire in sagebrush ecosystems (e.g., the Bureau of Land Management's [BLM's] Assessment Inventory and Monitoring [AIM] and the Forest Service's Forest Inventory and Analysis [FIA] programs). Incorporating monitoring results into future assessments will provide information on where fuel reduction and restoration efforts have been successful and where changes in management strategies are needed (e.g., Knutson et al. 2014). This information should be used in an adaptive management context to determine shifts in fire management priorities and reallocate resources.

Climate Adaptation and Wildland Fire Management

Given climate variability and longer fire seasons across the western United States, resilience and resistance concepts offer a proactive approach for decreasing current trends of more frequent and large, uncharacteristic fires and for maintaining resilient ecosystems (see section 3). Wildland fire risk

Text Box 4.1—Monitoring to Inform Wildland Fire and Vegetation Managements

Monitoring is an important component of effective wildland fire and vegetation management programs and has two primary purposes. First, monitoring provides information on changes in vegetation, fuels, and fire characteristics over time that can be used to adapt fire management. Monitoring survey plots (e.g., the Bureau of Land Management's Assessment Inventory and Monitoring [AIM], the Natural Resources Conservation Service's National Resources Inventory [NRI], and the Forest Service's Forest Inventory and Analysis [FIA] program) and remote sensing data can provide information on the extent and relative abundance of woody and herbaceous plants and any transitions between dominance of woody plants and herbaceous species (especially highly flammable invasive annual grasses) that occur over time. This information is useful for pre-positioning fire-fighting resources and developing fuel treatments that address different types of fuel or build-up of fuel.

Second, monitoring provides information on the effectiveness of management treatments. Success is typically achieved by meeting predetermined treatment objectives that are measured against baseline or reference conditions or another desired condition or benchmark. Effectiveness monitoring may be conducted at the project scale following postfire rehabilitation to restore GRSG habitat. Monitoring indicators, such as establishment or cover of grasses, forbs, and shrubs, increases in invasive annual grasses, and the appropriate benchmarks, can be used to evaluate whether the effort has increased the cover of either the seeded species or invasive annual grasses above a response threshold. Results of this effectiveness monitoring are used to evaluate both the effects of site conditions on treatment success and the need for follow-up management.

assessments help identify where climate, weather patterns, and land uses contribute to increases in large, severe fires and conversion to new alternative states (Abatzaglou and Kolden 2013; Littell et al. 2009; Miller et al. 2008; Westerling et al. 2006). Identifying areas where sagebrush is projected to persist through time under differing climate scenarios can help identify sagebrush habitats in need of prioritization for protection, or management actions that maintain or improve their current habitat quality.

Local Scale Considerations

Wildland Fire Preparedness, Suppression, and Prevention

The key to effective local wildland fire management is strategic placement of fuel reduction and restoration projects in relation to fire risk and fire suppression resources for the upcoming fire season. The combination of these efforts is integral to improving the chances of reducing fire size and effects during suppression efforts. Local fire suppression priorities are developed by resource and fire managers before the fire season. Primary considerations are burn probabilities, ecosystem resilience to fire and resistance to invasive annual grasses, locations of completed vegetation and fuel reduction projects, and key habitats. For maximum effectiveness, this information should be integrated into preplanned dispatch procedures used to allocate fire suppression resources during the fire season across jurisdictional units. By using this information, local fire managers can determine where ecological benefits may or may not be achieved when managing wildland fire and where to prioritize suppression resources to protect sagebrush habitats at risk. For example, suppressing fires adjacent to or within recently restored ecosystems may promote recovery and increase capacity to absorb future changes in conditions. Additionally, wet weather years followed by dry or normal years can result in significant changes in fuel loads over time. During these climate cycles, information and maps on the changes in wildland fire risk can help inform decisions about where fire suppression strategies can best mitigate the effects of fire on key habitats.

In wildland fire suppression, tactics used when managing a fire can have major consequences for the resultant burned area, including larger final fire extents. Practices such as burning out unburned patches of sagebrush and placement of indirect fireline reduce the opportunity to maintain sagebrush seeding sources that are already established (Murphy et al. 2013). Management practices recommended to help preserve large patches of sagebrush habitat during fire incidents include: (1) extinguishing fire edges and hotspots within the burn perimeter, especially around unburned islands; (2) applying suppression strategies and tactics that retain large interior islands of unburned sagebrush within the burn perimeter; (3) considering direct rather than indirect line when locating firelines, as safety and fire behavior allow; and (4) when safety is not an issue, directing suppression efforts to the front of a fire.

Based on wildland fire weather forecasts, suppression resources are commonly staged or "pre-positioned" in anticipation of fire occurrence at certain fire weather thresholds. "Severity" funding is provided to units having high wildland fire danger based on local forecasts and conditions to obtain additional resources for initial attack. Fire operation units can acquire more aviation resources, engines, crews, and other assets to protect key GRSG habitats when known weather events or high fire danger conditions are anticipated. Data and maps contained in the Part 1 of the Science Framework and the wildland fire risk assessment (fig. 4.1) can be used to prioritize and allocate severity funding to jurisdictions that have large areas of sagebrush and GRSG habitat at risk of loss from fire.

Wildland Fire Prevention—Human-caused ignitions can have devastating effects on sagebrush landscapes, especially those with low resilience to wildland fire. Preventive actions are generally more effective when tailored and delivered at the local level such as field offices or communities surrounding BLM districts and national forests. Spatial analyses that factor in wildland fire risk along with identified causes and locations of wildland fire ignitions from local communities can be used to design fire prevention strategies. These strategies can specifically target the local causes for human-caused ignitions at sites close to or within the wildland-urban interface. Data from the Department of the Interior, Wildland Fire Management Information (WFMI) system from 1997 through 2016 identify the most common human causes (e.g., target shooting) of BLM fires that burn sagebrush habitat. While each area has a unique set of wildland fire causes, two common examples of human-caused fires, along with ways to reduce ignitions, are:

- Powerlines—Though some powerline failures will always occur, others are preventable with proper ground clearance around power poles and transmission lines and improved maintenance of powerlines to prevent failures. Working with Federal realty specialists to ensure fire prevention measures are included in Land Use Authorizations, such as rights-of-way, can also be an effective way to reduce ignitions. This is especially important in sagebrush ecosystems characterized as having low fire resilience or high priority GRSG habitats. Wildland fire prevention partnerships with power companies and other utilities can help reduce the number of failures and wildland fire starts by entering into joint inspection programs on transmission lines with a history of starting wildland fires or adopting wildland fire prevention measures during construction, maintenance, and repair activities.
- Vehicles—Roadside ignitions are common in areas with hot dry fine fuels near highways and major roads. Working with State transportation departments

to reduce flammable vegetation along highway corridors has been shown to reduce the number of ignitions occurring when vehicles pull off into fine, dry grass on the side of the road and when improperly maintained trailers break down or drag trailer parts or chains that ignite fires.

Many social science studies conducted over the past several years have focused on the public's perception of wildland fire risk and what motivates the public to take action, especially at the community level. A common finding is that, while general awareness campaigns are effective to help the public understand risk from wildland fire, awareness does not necessarily lead to action. Awareness campaigns are more effective when agencies use face-to-face meetings and two-way conversations with the publics they serve to build relationships and trust. Time as well as commitment from management, fire and resource staff, and partners is needed to communicate fire prevention strategies and messages.

Partnerships, agreements, and sound fire investigation and prevention programs at the local level are critical to reduce human-caused wildland fires each year. For example, public and private organizations such as power and railroad companies who use, or operate on, public lands have a vested interest in preventing fires and should be approached as partners to limit fire ignitions. Fire prevention measures can be incorporated into land use authorizations, and relationships can be forged to address recurring fire ignitions associated with a given land use. Though it may take years to cultivate such relationships, it is a critical step in moving toward real action, such as burying a transmission line that has caused wildland fires or removing flammable vegetation along a railroad right-of-way.

While not all human-caused wildland fires can be prevented, many can and are being prevented through an informed citizenry that understands fire risk and is taking precautions with activities that may start a fire.

Vegetation Management and Postfire Recovery

Vegetation management (fuel reduction and restoration treatments) and postfire rehabilitation activities influence the structure and composition of vegetation communities at the project scale and are intended to maintain or increase ecosystem resilience to disturbance and resistance to invasive annual grasses. The primary objective of fuel reduction treatments is removing or modifying vegetation in order to reduce fuel loads and decrease fire size and severity. Objectives of both vegetation management and postfire treatments are to maintain or increase native perennial grasses and forbs and thus recovery potential, lower the longer-term risk of increases in nonnative invasive plants, and increase soil stability and reduce erosion.

Vegetation Management—For sagebrush ecosystems exhibiting juniper and piñon expansion and infill, Miller et al. (2014) provide a framework for selecting treatment areas and methods based on resilience and resistance concepts. Specific criteria for determining suitable sites and treatments are based on: (1) ecological site characteristics, (2) the phase of juniper and piñon expansion, (3) temperature and moisture regimes, and (4) the relative abundance, type, and fire tolerance of the native perennial grasses and forbs. Other factors to be considered in treatment design include: (1) sagebrush ecosystem response to past tree removals, (2) past and current management actions, (3) variation in long-term weather patterns (e.g., warmer temperatures and less precipitation; see section 3), (4) presence and relative abundances of invasive annual grasses, and (5) tradeoffs for sharply declining populations of juniper and piñon dependent species (e.g., pinyon jay [*Gymnorhinus cyanocephalus*]). Tree removal in phases I and II to reclaim sagebrush habitat results in the removal of a biologically valuable part of the juniper and piñon woodland and sagebrush interface for other species habitats (e.g., pinyon jay; mule deer [*Odocoileus hemionus*]) (Gillihan 2005; Sauer et al. 2014). Surveying these sites for all declining wildlife populations before selecting sites for treatment, designing tree removals that mimic stand structure after natural disturbance such as fire (e.g., maintaining mature juniper and piñon and creating convoluted edges and small openings in mature woodland stands), avoiding sharp edges between sagebrush and juniper and piñon stands, and monitoring can help mitigate the effects of treatments on juniper and piñon associated and dependent species (Gillihan 2005).

For sagebrush ecosystems with significant cheatgrass cover, fuel reduction treatments are aimed at reducing the continuity of cheatgrass cover. The objective is to reduce fuel connectivity and slow or stop fire spread between cheatgrass patches and into intact native vegetation. Current methods for reducing cheatgrass fuel are detailed in section 5.

Roads play a significant role in influencing wildland fire ignition and control at the local scale. Wildland fire boundaries tend to occur near roads because roads provide access for fire suppression. Additionally, roads act as fuel breaks because the road footprint is vegetation free, providing a no-burn zone that reduces the spread of fire (Narayanaraj and Wimberly 2011, 2013; Price and Bradstock 2010; Syphard et al. 2011). In sagebrush ecosystems, fuel reductions have used roadsides to create linear fuel breaks that disrupt fuel continuity by reducing fuel accumulation (Maestas et al. 2016a; Shinneman et al. 2018). Removal of vegetation can vary (e.g., 50 feet to 0.25 mile [15–400 meters]) based on landscape conditions, fire spotting potential, and expected flame length. Fuel breaks are intended to reduce fire intensity, rates of fires spread, and flame length. Fire managers believe that they enhance firefighter access, improve response times, and provide safe and strategic anchor points for wildland firefighting activities (e.g., back burning) (Moriarti et al. 2015). Linear fuel breaks also may help to slow or stop human-caused fires ignited along roads, thereby reducing the risk of fire spread along roadsides into adjacent lands (Naravanaraj and Wimberly 2012, 2013).

While anecdotal evidence suggests that properly designed fuel breaks help with fire operations, the ecological and economic consequences of linear fuel breaks are relatively unknown (Shinneman et al. 2018). Because linear fuel breaks are located along roads, they may serve as conduits for invasive plant species, increase fragmentation of wildlife habitat, disrupt wildlife movement pathways, and increase predation on sagebrush obligates (Coates et al. 2014; Shinneman et al. 2018). As a result, the area influenced by roads and fuel breaks (e.g., edge effects) is likely to be markedly larger than the area covered by roads and fuel breaks themselves (Forman 2003; Forman and Alexander 1998; Naravanaraj and Wimberly 2013). For example, nonnative plants that invade along roads frequently create a source of combustible fuel (Arienti et al. 2009; D'Antonio and Vitousek 1992; Parendes and Jones 2000; Trombulak and Frissell 2000). Removal of native vegetation along roads can increase establishment and spread of invasive plants from the fuel break into the interior of large sagebrush patches. Subsequently, fuel breaks, if not monitored and maintained, may contribute to a greater incidence of human-caused fires near roads (Arienti et al. 2009; Syphard et al. 2007, 2008; Yang et al. 2007, 2008a,b).

In designing linear fuel breaks, Gray and Dickson (2016) and Shinneman et al. (2018) suggest using fire simulation modeling to help identify strategic places for placing fuel breaks by projecting their effectiveness in altering fire behavior and assessing utility and safety for firefighting activities. Combining these results with species habitat maps can also help to identify where fuel break placement should be avoided to maintain intact habitat and habitat connectivity. Considering the width of fuel breaks (including the width of the road) is important in assessing potential fragmentation effects on wildlife. For example, herbicide treatments of less than 30 meters (100 feet) wide help avoid negative effects on sagebrush dependent passerine birds (Best 1972). Once strategic places for fuel breaks have been identified, Shinneman et al. (2018) proposed that fuel break design along roadsides could include alternating strips of altered and undisturbed sagebrush rather than continuous altered strips along the entire length of a road. This type of design could be based on current knowledge of fire probability, habitat disturbance, fragmentation, and edge-effects to help maintain the overall integrity of sagebrush habitat in that area.

Assessments of soil characteristics and precipitation are helpful in determining which species are best suited to plant in fuel breaks (Maestas et al. 2016a). Species such as forage kochia (*Bassia prostrata* ssp.) and crested wheatgrass (*Agropyron cristatum*) or a mix of nonnative grasses are widely used to seed fire-resistant green strips and prevent soil erosion in fuel breaks. However, seeding introduced species has drawbacks (see section 6). For example, forage kochia is documented to spread outside of seeded areas (Gray and Muir 2013) and compete with slickspot peppergrass (*Lepidium papilliferum*), which is listed under the Endangered Species Act (Pellant 2004). At the same time, introduced species may establish quickly, outcompete invasive annual grasses, and persist without the need for repeated seedings dependent on environmental conditions.

Native perennial grasses and forbs are emerging as another viable alternative and have potential to be used more widely because: (1) native grasses and forbs with low stature, such as Sandberg bluegrass (*Poa secunda*), can compete well with invasive annual grasses and reduce fine fuels, fuel heights, fuel loadings, and fuel continuity; (2) many native grasses and forbs are drought tolerant and local seed sources may establish better on dry sites than forage kochia and crested wheatgrass; (3) many native grasses and forbs are tolerant of disturbance; and (4) the potential for spread into adjacent areas is not problematic (Gray and Muir 2013). Opportunities exist to test native plants that have the characteristics desired for fuel break plantings such as low stature, rapid establishment, competitive with invasive plants, remain green during the dry season, and fire tolerance. Other techniques for creating fuel breaks include modifying existing roadbeds with mowing, herbicide applications, intensive grazing, conifer removal, or prescribed fire to reduce vegetation (Moriarti et al. 2015). For fuel breaks to meet the intended purpose, the cost of monitoring and annual maintenance of fuel breaks should be analyzed, planned for, and incorporated into annual budgets upfront so that fuel breaks are maintained for safe fire operations and have minimal impacts (e.g., spread of invasive plants) to the sagebrush habitats they are designed to protect. Continual monitoring of fuel breaks is needed to determine the most appropriate strategy (timing, methods, additional seedings) for maintaining fuel breaks and assessing their potential for use in fire suppression activities every season.

Postfire Recovery—Miller et al. (2015) and Pyke et al. (2015a,b) provide frameworks for evaluating resilience to disturbance and resistance to invasive annual grasses of postfire sites in the Great Basin. They make recommendations for postfire recovery methods based on ecological site characteristics that can be modified for the eastern portion of the sagebrush biome (see Part 1, Appendix 5). The decision to seed postfire is based on rapid assessments of the ecological sites within the project area. Information on temperature and moisture regimes, preburn vegetation (including sagebrush species), perennial grasses and forbs, invasive annual grasses, and fire severity is used to rate the relative resilience and resistance of the ecological site(s). Specific criteria for determining the need to seed and appropriate seeding methods are provided based on temperature and moisture regimes and the relative abundance and type of native perennial grasses and forbs and invasive annual grasses.

In general, sites with higher resilience and resistance to invasive annual grasses (table 1.3: cells 1A, 1B, 1C) are more likely to recover without seeding than lower resilience and resistance sites (table 1.3: cells 3A, 3B, 3C) (Miller et al. 2015). If native perennial grasses and forbs are sufficient to promote recovery after fire, seeding is not needed. If native perennial grasses and forbs were depleted or absent before the fire or invasive annual grasses were abundant, seeding is likely to be needed, along with commensurate posttreatment management strategies such as grazing deferment or changes in season of use, to protect the restoration investment. Areas with severely depleted native species and abundant invasive annual grasses may require integrated management approaches that include herbicide application prior to seeding.

An understanding of resilience and resistance to invasive annual grasses as indicated by precipitation and temperature regimes can inform seeding decisions in vegetation management and postfire rehabilitation. Key considerations in determining seed mixes are selecting genetically appropriate native seed, compatibility of species in a seed mix, planting season, and appropriate seeding rates, techniques, and practices (see section 6). Nonnative species or aggressive native cultivars are often seeded in postfire recovery efforts because many germinate and establish quickly, are less expensive than native species, provide livestock forage, and compete with nonnative invasive species (Brooks and Pyke 2001; Davison and Smith 2005; Monaco et al. 2003; Pellant 1994; Pyke and McArthur 2002; Richards et al. 1998).

In the last two decades native seeds have become more readily available, the tradeoffs between seeding native and nonnative species are better understood, and resource managers are using more native species in fuel management and postfire recovery applications (see section 6). For sites with moderate to high resilience and resistance to invasive annual grasses where seeding is needed or sites with low resilience and resistance with low invasive plant densities pre-fire, native cultivars should be the preferred option given management concerns and the long-term challenges of seeding introduced species. For burned areas with low resilience and resistance to invasive annual grasses that had a low density of native species and high density of invasive plants pre-fire, native or introduced species—or a combination of both—may help minimize risk of a state shift to nonnative annual grass dominance depending on site characteristics and seeded species. In areas with low to moderate resistance to invasive annual grasses, nonnative invasive plant management is also an important consideration in postfire restoration efforts. Information for integrating nonnative invasive plant management into postfire restoration is in section 5.

Monitoring Vegetation Treatments

Monitoring to evaluate site recovery after fuel treatments and postfire rehabilitation provides the necessary information to determine whether management objectives were met and whether treated sagebrush ecosystems have recovered a composition, structure, and function that is sustainable over time (see section 2). Monitoring results can also identify areas where further restoration or adaptations to management strategies are needed to help lower wildland fire risk (text box 4.1).

Conclusions

Western sagebrush ecosystems continue to be threatened by larger and more frequent wildland fires that often result in the loss of large swaths of sagebrush and facilitate invasion by nonnative annual grasses. Longer fire seasons combined with warmer temperatures, failure to alter grazing regimes in response to climatic variability, and declines in ecological conditions are exacerbating the spread of invasive annual grasses to climatically suitable areas across the sagebrush biome. This ongoing spread of invasive plants is likely to increase fire frequency and extent in areas that currently do not experience a lot of fire. Natural recovery times and current management practices cannot keep up with the expanding invasive annual grass/fire cycle and some areas may have crossed thresholds of no return. In response, sagebrush obligate species that serve as indicators of ecosystem conditions, along with many other sagebrush obligates, are declining throughout the sagebrush biome (Coates et al. 2016).

This accelerated invasive annual grass/fire cycle needs to receive greater focus in sagebrush ecosystem conservation efforts. To help sustain ecosystems as well as transition and adapt to a changing climate, this section offers multi-scaled management approaches for wildland fire prevention, suppression, vegetation management, and postfire recovery that are prioritized based on resilience and resistance concepts. The integration of these approaches with those offered in the sections on climate adaptation (section 3), grazing (section 7), and seeding strategies (section 6) can help determine where investments are most likely to be successful in addressing uncharacteristic fire cycles and restoring sagebrush habitats. Consistency in these management approaches, to the extent possible, is key and can be achieved through collaboration and partnerships across jurisdictional boundaries, agencies, and disciplines. Changes in budgeting and policy structures are needed to increase flexibility, provide for quicker responses to disturbances, and allow longer implementation times to support restoration and climate adaptation opportunities. To help these ecosystems adapt to landscape changes in the future, we need increased efforts and focus on: (1) outreach to the public with prevention strategies targeting the causes of human-ignited fires and spread of invasive plants; (2) strategic fuel reduction and invasive plant treatments to help address climate adaptation, uncharacteristic fire cycles, and spread of invasive plants; (3) seeding strategies that mimic natural recovery, increase connectivity, and allow for natural transitions and climate adaptation; and (4) best management practices in fire suppression efforts to retain sagebrush.

References

Abatzoglou, J.T.; Kolden, C.A. 2013. Relationships between climate and macroscale area burned in the western United States. International Journal of Wildland Fire. 22: 1003–1020.

Agee, J.K.; Bahro, B.B.; Finney, M.A.; [et al.]. 2000. The use of shaded fuelbreaks in landscape fire management. Forest Ecology and Management. 127: 55–66.

Allen, C.D.; Breshears, D.D.; McDowell, N.G. 2015. On underestimation of global vulnerability to tree mortality and forest die-off from hotter drought in the Anthropocene. Ecosphere. 6: 129.

Arendt, P.A.; Baker, W.L. 2013. Northern Colorado Plateau piñon-juniper woodland decline over the past century. Ecosphere. 4: 1–30.

Arienti, M.C.; Cumming, S.G.; Krawchuk, M.A.; [et al.]. 2009. Road network density correlated with increased lightning fire incidence in the Canadian western boreal forest. International Journal of Wildland Fire. 18: 970–982.

Baker, W.H. 2006. Fire and restoration of sagebrush ecosystems. Wildlife Society Bulletin. 34: 177–185.

Baker, W.H. 2011. Pre–Euro-American and recent fire in sagebrush ecosystems. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 185–201.

Beck, J.L.; Connelly, J.W.; Wamboldt, C.L. 2012. Consequences of treating Wyoming big sagebrush to enhance wildlife habitats. Rangeland Ecology and Management. 65: 444–455.

Best, L.B. 1972. First-year effects of sagebrush control on two sparrows. Journal of Wildlife Management. 36: 534–544.

Board, D.I.; Chambers, J.C.; Miller, R.M.; [et al.]. 2018. Fire patterns in piñon and juniper land cover types in the semiarid western US from 1984 through 2013. RMRS-GTR-372. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 57 p.

Breshears, D.D.; Cobb, N.S.; Rich, P.M.; [et al.]. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences of the United States of America. 102: 15144–15148.

Brooks, M.L.; Pyke, D.A. 2001. Invasive plants and fire in the deserts of North America. In: Galley, K.M.; Wilson, T.P., eds. Proceedings of the invasive species workshop: The role of fire in the control and spread of invasive species. Fire Conference 2000: The First National Congress on Fire Ecology, Prevention, and Management.; 2000 November 27–December 1; San Diego, CA. Misc. Publ. No. 11, Tallahassee, FL: Tall Timbers Research Station: 1–14.

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p.

Chambers, J.C.; Miller, R.F.; Board, D.I.; [et al.]. 2014. Resilience and resistance of sagebrush ecosystems: Implications for state and transition models and management treatments. Rangeland Ecology and Management. 67: 440–454.

Coates, P.S.; Howe, K.B.; Casazza, M.L.; [et al.]. 2014. Landscape alterations influence differential habitat use of nesting buteos and ravens within sagebrush ecosystem—Implications for transmission line development. The Condor. 116: 341–356.

Coates, P.S.; Ricca, M.A.; Prochazka, B.G.; [et al.]. 2016. Wildfire, climate, and invasive grass interactions negatively impact an indicator species by reshaping sagebrush ecosystems. Proceedings of the National Academy of Sciences of the United States of America. 113: 12745-12750.

Crist, M.R.; Knick, S.T.; Hanser, S.E. 2015. Range-wide network of priority areas for Greater sage-grouse—A design for conserving connected distributions or isolating individual zoos? Open-file Report 2015-1158. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 34 p.

D’Antonio, C.M.; Vitousek, P.M. 1992. Biological invasions by exotic grasses, the grass fire cycle, and global change. Annual Review of Ecology and Systematics. 23: 63–87.

Davies, K.W.; Bates, J.D.; Nafus, A.M. 2012. Mowing Wyoming big sagebrush communities with degraded herbaceous understories: Has a threshold been crossed? Rangeland Ecology & Management. 65: 498–505.

Davison, J.; Smith, E. 2005. Living with fire, crested wheatgrass: Hero or villain in reclaiming disturbed rangelands. Fact Sheet 96-53. Reno, NV: University of Nevada-Reno, Cooperative Extension. 4 p.

Doherty, K.E.; Evans, J.S.; Coates, P.S.; [et al.]. 2016. Importance of regional variation in conservation planning: A range-wide example of the Greater sage-grouse. Ecosphere. 7: e01462.

Endangered Species Act of 1973; 16 U.S.C. 1531-1536, 1538-1540. https://www.gpo.gov/fdsys/pkg/USCODE-2012-title16/html/USCODE-2012-title16-chap35-sec1531.htm. [Accessed May 24, 2018].

Finney, M.A.; McHugh, C.W.; Grenfell, I. 2010. Continental-scale simulation of burn probabilities, flame lengths, and fire size distributions for the United States. In: Viegas, D.X., ed. Fourth international conference on forest fire research; 2010 November 13–18; Coimbra, Portugal. Associacao para o Desenvolvimento da Aerodinamica Industrial. 12 p.

Floyd, M.L.; Hanna, D.; Romme, W.H.; [et al.]. 2004. Predicting and mitigating weed invasions to restore natural post-fire succession in Mesa Verde National Park, Colorado, USA. International Journal of Wildland Fire. 15: 247–259.

Floyd, M.L.; Hanna, D.; Romme, W.H. 2006. Historical and recent fire regimes in piñon–juniper woodlands on Mesa Verde, Colorado, USA. Forest Ecology and Management. 198: 269–289.

Forman, R.T.T. 2003. Road ecology: Science and solutions. Washington, DC: Island Press.

Forman, R.T.T.; Alexander, L.E. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics. 29: 207–231.

Gillihan, S.W. 2005. Sharing the land with pinyon-juniper birds. Salt Lake City, UT: Partners in Flight Western Working Group. https://birdconservancy.org/wp-content/uploads/2015/08/PJ-manual-Nov-08-low-res.pdf. [Accessed May 22, 2018].

Gray, E.C.; Muir, P.S. 2013. Does Kochia prostrata spread from seeded sites? An evaluation from southwestern Idaho, USA. Rangeland Ecology and Management. 66: 191–203.

Gray, M.E.; Dickson, B.G. 2016. Applying fire connectivity and centrality measures to mitigate the cheatgrass-fire cycle in the arid West, USA. Landscape Ecology. 31: 1681–1696.

Knick, S.T.; Hanser, S.E.; Preston, K.L. 2013. Modeling ecological minimum requirements for distribution of Greater sage-grouse leks: Implications for population connectivity across their western range, U.S.A. Ecology and Evolution. 3: 1539–1551.

Knutson, K.C.; Pyke, D.A.; Wirth, T.A.; [et al.]. 2014. Long-term effects of reseeding after wildfire on vegetation composition in the Great Basin shrub steppe. Journal of Applied Ecology. 51(5): 1414–1424.

Littell, J.S.; McKenzie, D.; Peterson, D.L.; [et al.]. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications. 19: 1003–1021.

Maestas, J.; Pellant, M.; Okeson, L.; [et al.]. 2016a. Fuel breaks to reduce large wildfire impacts in sagebrush ecosystems. Plant Materials Technical Note No. 66. Boise, ID: U.S. Department of Agriculture, Natural Resources Conservation Service. http://www.sagegrouseinitiative.com/wp-content/uploads/2016/03/idpmctn16_tn66fuelbreaks-1.pdf. [Accessed May 22, 2018].

Maestas, J.D.; Campbell, S.B.; Chambers, J.C.; [et al.]. 2016b. Tapping soil survey information for rapid assessment of sagebrush ecosystem resilience and resistance. Rangelands. 38: 120–128.

Manier, D.J.; Aldridge, C.L.; O'Donnell, M.; [et al.]. 2014. Human infrastructure and invasive plant occurrence across rangelands of Southwestern Wyoming, USA. Rangeland Ecology and Management. 67: 170–172.

Manier, D.J.; Hobbs, N.T.; Theobold, D.M.; [et al.]. 2005. Canopy dynamics and human caused disturbance on a semi-arid landscape in the Rocky Mountains, USA. Landscape Ecology. 20: 1–17.

Mealor, B.A.; Cox, S.; Booth, D.T. 2012. Post fire downy brome (*Bromus tectorum*) invasion at high elevations in Wyoming. Invasive Plant Science and Management. 5: 427–435.

Mealor, B.A.; Mealor, R.D.; Kelley, W.K.; [et al.]. 2013. Cheatgrass management handbook: Managing an invasive annual grass in the Rocky Mountain Region. Laramie, WY: University of Wyoming; Fort Collins, CO: Colorado State University. 131 p.

Meyer, S.E. 1994. Germination and establishment ecology of big sagebrush: Implications for community restoration. In: Monsen, S.B.; Kitchen, S.G., comps. Proceedings—Ecology and management of annual rangelands; 1992 May 18–21; Boise, ID. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 244–251.

Meyer, S.E.; Monsen, S.B. 1990. Seeding equipment effects on establishment of big sagebrush on mine disturbance. In: Fifth Billing's symposium on disturbed land rehabilitation. Vol. 1. Reclamation Research Unit Publ. 9003. Bozeman, MT: Montana State University: 192–199.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2014. A field guide to selecting the most appropriate treatments in sagebrush and pinyon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322-rev. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 70 p.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2015. A field guide for rapid assessment of post-wildfire recovery potential in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-338. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 68 p.

Miller, R.F.; Chambers, J.C.; Pyke, D.A.; [et al.]. 2013. A review of fire effects on vegetation and soils in the Great Basin Region: Response and ecological site characteristics. Gen. Tech. Rep. RMRS-GTR-308. Fort Collins, CO: Department of Agriculture, Forest Service, Rocky Mountain Research Station. 126 p.

Miller, R.F.; Knick, S.T.; Pyke, D.A.; [et al.]. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 145–185.

Miller, R.F.; Tausch, R.J.; McArthur, E.D.; [et al.]. 2008. Age structure and expansion of piñon-juniper woodlands: A regional perspective in the Intermountain West. Res. Pap. RMRS-RP-69. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Monsen, S.B.; Stevens, R.; Shaw, N.L., eds. 2004. Restoring western ranges and wildlands. Gen. Tech. Rep. RMRS-GTR-136-vol-1. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 121–154.

Monaco, T.A.; Waldron, B.L.; Newhall, R.L.; [et al.]. 2003. Re-establishing perennial vegetation in cheatgrass monocultures. Rangelands. 25: 26–29.

Moriarti, K.; Okeson, L.; Pellant, M. 2015. Fuel breaks that work. Great Basin Fact Sheet Series. No. 5. Reno, NV: Great Basin Fire Science Exchange. https://www.sagegrouseinitiative.com/wp-content/uploads/2015/07/5_GBFS_Fuel-Breaks.pdf. [Accessed May 22, 2018].

Murphy, T.; Naugle, D.E.; Eardley, R.; [et al.]. 2013. Trial by fire: Improving our ability to reduce wildfire impacts to sage-grouse and sagebrush ecosystems through accelerated partner collaboration. Rangelands. 35: 2–10.

Narayanaraj, G.; Wimberly, M.C. 2011. Influences of forest roads on the spatial pattern of wildfire boundaries. International Journal of Wildland Fire. 20: 792–803.

Narayanaraj, G.; Wimberly, M.C. 2012. Influences of forest roads on the spatial patterns of human- and lightning-caused wildfire ignitions. Applied Geography. 32: 878–888.

Narayanaraj, G.; Wimberly, M.C. 2013. Influences of forest roads and their edge effects on the spatial pattern of burn severity. International Journal of Applied Earth Observation and Geoinformation. 23: 62–70.

National Interagency Fire Center [NIFC]. 2017. Fire information and statistics. Boise, ID: National Interagency Fire Center. https://www.nifc.gov/fireInfo/fireInfo_statistics.html. [Accessed May 23, 2018].

Oregon Department of Forestry. 2013. West wide wildfire risk assessment final report prepared for Oregon Department of Forestry. Western Forestry Leadership Coalition and Council of Western State Foresters. Salem, OR: Oregon Department of Forestry. 105 p. http://www.odf.state.or.us/gis/data/Fire/West_Wide_Assessment/WWA_FinalReport.pdf. [Accessed May 22, 2018].

Parendes, L.A.; Jones, J.A. 2000. Role of light availability and dispersal in exotic plant invasion along roads and streams in the H.J. Andrews Experimental Forest, Oregon. Conservation Biology. 14: 64–75.

Pellant, M. 1994. History and applications of the Intermountain Greenstripping Program. In: Monsen,S.B.; Kitchen, S.G., comps. Proceedings-symposium on ecology and management of annual rangelands; 1992 May 18–21; Boise, ID. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 63–68.

Pellant, M. 2004. Greenstripping with forage kochia. In: Proceedings: Forage kochia workshop and tour; 2004 November 9–10. Logan, UT: Utah State University: 52–56.

Price, O.F.; Bradstock, R. 2010. The effect of fuel age on the spread of fire in sclerophyll forest in the Sydney region of Australia. International Journal of Wildland Fire. 19: 35–45.

Pyke, D.A. 2011. Restoring and rehabilitating sagebrush habitats. In: Knick, S.T.; Connelly, J. W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 531–548.

Pyke, D.A.; Chambers, J.C.; Pellant, M.; [et al.]. 2015a. Restoration handbook for sagebrush steppe ecosystems with emphasis on Greater sage-grouse habitat. Part 1. Concepts for understanding and applying restoration. Circular 1416. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 44 p. https://doi.org/10.3133/cir1416.

Pyke, D.A.; Knick, S.T.; Chambers, J.C.; [et al.]. 2015b. Restoration handbook for sagebrush steppe ecosystems with emphasis on Greater sage-grouse habitat. Part 2. Landscape level restoration decisions. Circular 1418. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 21 p. https://doi.org/10.3133/cir1418.

Pyke, D.A.; McArthur, T.O. 2002. Emergency fire rehabilitation of BLM lands in the Intermountain West: Revegetation and monitoring. Interim report to the BLM. Corvallis, OR: U.S. Department of the Interior, U.S. Geological Survey, Forest and Rangeland Ecosystem Science Center.

Richards, R.T.; Chambers, J.C.; Ross, C. 1998. Use of native plants on federal lands: Policy and practice. Journal of Range Management. 51: 625–632.

Romme, W.H.; Allen, C.D.; Bailey, J.D.; [et al.]. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation in the western United States. Rangeland Ecology and Management. 62: 203–222.

Rottler, C.M.; Noseworthy, C.E.; Fowers, B.; [et al.]. 2015. Effects of conversion from sagebrush to non-native grasslands on sagebrush-associated species. Rangelands. 3: 1–6.

Sauer, J. R.; Hines, J.E.; Fallon, J.E.; [et al.]. 2014. The North American breeding bird survey, results and analysis 1966–2013. Version 01.30.2015. Laurel, MD: USGS Patuxent Wildlife Research Center. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprd3847466.pdf. [Accessed May 22, 2018].

Shinneman, D.J.; Aldridge, C.L.; Coates, P.S.; [et al.]. 2018. A conservation paradox in the Great Basin—Altering sagebrush landscapes with fuel breaks to reduce habitat loss from wildfire. Open-File Report 2018–1034. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 70 p. https://doi.org/10.3133/ofr20181034. [Accessed May 22, 2018].

Short, K.C.; Finney, M.A.; Scott, J.H.; [et al.]. 2016. Spatial dataset of probabilistic wildfire risk components for the conterminous United States. Fort Collins, CO: Forest Service Research Data Archive. https://doi.org/10.2737/RDS-2016-0034. [Accessed May 22, 2018].

Syphard, A.D.; Keeley, J.D.; Brennan, T.J. 2011. Factors affecting fuel break effectiveness in the control of large fires on the Los Padres National Forest, California. International Journal of Wildland Fire. 20: 764–775.

Syphard, A.D.; Radeloff, V.C.; Keeley, J.E.; [et al.]. 2007. Human influence on California fire regimes. Ecological Applications. 17: 1388–1402.

Syphard, A.D.; Radeloff, V.C.; Keuler, N.S.; [et al.]. 2008. Predicting spatial patterns of fire on a southern California landscape. International Journal of Wildland Fire. 17: 602–613.

Trombulak, S.C.; Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology. 14: 18–30.

U.S. Department of the Interior [USDOI]. 2015. An Integrated Rangeland Fire Management Strategy. Final Report to the Secretary of the Interior. Washington, DC: U.S. Department of the Interior. https://www.forestsandrangelands.gov/rangeland/documents/IntegratedRangelandFireManagementStrategy_FinalReportMay2015.pdf. [Accessed Sept. 12, 2018].

Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; [et al.]. 2006. Warming and earlier spring increases western U.S. forest wildfire activity. Science. 313: 940–943.

Wildland Fire Management Information System [WFMI]. 2018. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, National Interagency Fire Center. https://wfmi.nifc.gov/cgi/WfmiHome.cgi. [Accessed May 23, 2018].

Williams, M.I.; Thurow, T.L.; Paige, G.B.; [et al.]. 2011. Sagebrush-obligate passerine response to ecological site characteristics. Natural Resource Environmental Issues. 16: 1–9.

Yang, J.; He, H.S.; Shifley, S.R.; [et al.]. 2007. Spatial patterns of modern period human-caused fire occurrence in the Missouri Ozark Highlands. Forest Science. 53: 1–15.

Yang, J.; He, H.S.; Shifley, S.R. 2008a. Spatial controls of occurrence and spread of wildfires in the Missouri Ozark Highlands. Ecological Applications. 18: 1212–1225.

Yang, J.; He, H.S.; Sturtevant, B.R.; [et al.]. 2008b. Comparing effects of fire modeling methods on simulated fire patterns and succession: A case study in the Missouri Ozarks. Canadian Journal of Forest Research. 38: 1290–1302.

5. INVASIVE PLANT MANAGEMENT

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Introduction

One of the most significant stressors to the sagebrush biome is expansion and dominance of nonnative ecosystem-transforming species, particularly invasive annual and perennial plants. Presidential Executive Orders 13112 and 13751 define an invasive species as "a non-native organism whose introduction causes or is likely to cause economic or environmental harm, or harm to human, animal, or plant health." The use of the term "invasive species" requires two basic criteria to be met: (1) the species is alien, nonnative, or exotic to the ecosystem in question; and (2) the species has been documented as causing harm as noted in the definition. In addition, invasive annual and perennial plant species are categorized as either regulated species (nonnative species regulated under State or Federal noxious weed laws), or unregulated species (nonnative species which may pose a threat but have not been officially designated as regulated or restricted under State or Federal law). Cheatgrass (*Bromus tectorum*), for example, is not a Federally designated noxious weed, nor a State-designated noxious weed in many western States, but there are other State and local restrictions associated with this species in some areas.

Based on this definition, the labeling of a species as invasive requires closely examining both the origin and the effects of the species. Native species that may influence management objectives within a particular ecosystem would not be defined as invasive. For example, juniper (*Juniperus* spp.) and piñon pine (*Pinus* spp.) expansion into sagebrush ecosystems is a natural process resulting from a variety of factors (Miller et al. 2013; Romme et al. 2009) (see section 4). Yet unlike native juniper and piñon pine expansion, the establishment and spread of invasive plants, such as cheatgrass, Dyer's woad (*Isatis tinctoria*), and many other high-risk invasive perennial and annual plants, is not a natural ecosystem process in the sagebrush biome. There are important differences in the short- and long-term impacts to the sagebrush biome from invasive plant species compared to native species. Each invasive plant carries a different level of risk and properly describing these stressors helps managers to more effectively focus their restoration and management activities across the landscape.

Many invasive plants respond positively to ecosystem disturbance (e.g., human development, improper grazing practices, wildfires) and spread through various pathways and vectors, such as roads, trails, and vehicles (Pollnac et al. 2012; Trombulak and Frissell 2000); transmission corridors; and fuel breaks. Invasive plant species can colonize new areas rapidly, even areas that are somewhat ecologically intact. Once established, invasive plant species can continue to

Top: Spraying invasive plants with herbicides using backpack sprayers (photo: USDOI National Park Service). Middle left: Dalmation toadflax (Cal-IPC.org.; photo by Joe DiTomaso). Middle center: Spotted knapweed (photo: Alaska Plant Materials Center, State of Alaska). Middle right: Rush skeletonweed (photo: Washington State Noxious Weed Control Board). Bottom: Constantia Fire, Long Valley, California (photo: Nolan Preece, used with permission).

spread across the landscape where suitable conditions exist. Invasive plant species can become ecologically dominant, creating near-monocultures that result in reduced wildlife habitat, recreational opportunities, livestock forage, and altered fire regimes (Pyke et al. 2016). For example, even after disturbances are removed, invasive annual grasses (e.g., cheatgrass) can remain dominant. Native species may show little recovery even decades later (Keeley et al. 2003; Stromberg and Griffin 1996; Stylinski and Allen 1999) due to seed limitations of native species (Seabloom et al. 2003) and adverse interactions among invasive and native plants at the seed and seedling stages (DiVittorio et al. 2007). The complete elimination of invasive annual grasses is unlikely in these areas as the exotic annuals are highly competitive with native species for limiting resources (HilleRisLambers et al. 2010). This type of ecosystem conversion to invasive plants degrades ecosystem function by affecting geomorphic processes, hydrology, nutrient cycling, community structure, composition, productivity, and regeneration of native species (Germino et al. 2016).

The magnitude of the risk or impact that invasive plants pose to sagebrush ecosystems varies and depends on site conditions and the species' characteristics. Invasive annual grasses, most notably cheatgrass, medusahead (*Taeniatherum caput-medusae*), and red brome (*Bromus rubens*) are arguably the most widespread ecosystem disruptors across the sagebrush biome. Yet many other invasive species are also responsible for environmental impacts to sagebrush ecosystems (Ielmini et al. 2015) and new invaders (Appendix 3) continue to add to the existing management burden. For example, leafy spurge (*Euphorbia esula*) disperses into riparian and wet meadow areas important to Greater sage-grouse (*Centrocercus urophasianus*; hereafter, GRSG) brood-rearing habitat. Tap-rooted invasive plants, such as spotted knapweed (*Centaurea maculosa*), Russian knapweed (*Acroptilon repens*), and yellow salsify (*Tragopogon dubius*), spread into upland sagebrush ecosystems, especially in areas that experience heavy livestock grazing and other disturbances (Hill et al. 2006; Prevey et al. 2010). Additionally, species such as Dalmatian toadflax (*Linaria dalmatica*) are spreading into moister areas throughout the sagebrush biome (Ielmini et al. 2015).

Land managers are tasked with controlling various species of invasive plants, but limited resources are available for invasive plant management. Invasive species ranking systems (e.g., USDOI FWS 2018) can assist land managers in ranking invasive plant species for level of threat, feasibility of control, and degree of negative impact, but this information is lacking for several species. Therefore, land managers face difficult decisions regarding how to use limited resources and whether to target high-risk pathways and vectors of invasion for efficiency; focus on specific invasive plant patches that are feasible to control, such as Early Detection and Rapid Response (EDRR) programs for targeted species; or treat the periphery of a large invasion to slow and contain the spread. The need to manage multiple invasive plant species while considering ecological impacts and social and political priorities often results in significant challenges in determining how to partition resources for invasive plant management. Achieving long-term ecosystem conservation and restoration goals for invasive plant-dominated landscapes requires a substantial increase in invasive plant management capacity and the management flexibility to better align invasive species management and native plant restoration activities. It also requires innovative approaches that capitalize on the targeted ecosystem's resilience to disturbance and resistance to invasive plant invasion.

Integrating Resilience and Resistance Concepts into Invasive Plant Species Management

An understanding of ecosystem resilience to disturbance and resistance to invasive plants can be used to help prioritize invasive plant management and determine effective management strategies. Resistance to invasive plants is of particular relevance to this section. The resistance of an ecosystem to an invasive plant is a function of (1) the suitability of the ecosystem's climate and soils for establishment and persistence of the invasive plant, and (2) the capacity of the native plant community to prevent increases in the invasive plant's population through factors such as competition, herbivory, and ability of native plants to adapt to environmental conditions (Chambers et al. 2014a). Soil temperature and moisture regimes are a primary determinant of a species' ability to establish and persist in a given ecosystem and are an important indicator of ecosystem resistance to invasive plants, such as invasive annual brome grasses (Brooks et al. 2016; Chambers et al. 2016). In areas with suitable climate and soils for invasion, increases in invasive plant populations are strongly influenced by interactions with the native perennial plant community. Disturbances or management activities that reduce abundance of native perennial grasses and biological soil crusts and increase the distances between perennial grasses often are associated with higher resource availability and increased competitive ability of invasive annual grasses (Chambers et al. 2007; Collins and Uno 1985; Reisner et al. 2013; Roundy et al. 2014; Salo 2005) and invasive forbs like spotted knapweed (Willard et al. 1988). Reductions in native perennial grasses and herbaceous species coupled with increases in invasive plants can decrease the resilience of an ecosystem or its capacity to recover following disturbances such as wildfire.

The following questions identify the basic invasive plant management information needs for informing management decisions in the context of resilience to disturbance and resistance to invasive plants:

- Where are the priority areas for management, how are they defined (e.g., GRSG habitat, mule deer [*Odocoileus hemionus*] wintering habitat, particular allotment, community at risk of wildfire), and where can resources be leveraged with partners and stakeholders for the greatest chance of success (e.g., relative resilience to disturbance and resistance to invasive plants)?
- What is the current state of invasion and how great is the risk for new invasion of priority management areas (e.g., areas of low resilience and resistance to invasive plants, significant disturbance levels, high density of vectors, other invasions in the area)?
- Which management strategies (e.g., prevention, EDRR, eradication, suppression, containment, or restoration) are feasible and within the level of return for investment desired for a particular site. For example, containment may be the only feasible strategy for a site with low resilience and resistance that is dominated by invasive annual grasses.
- Which tool(s) is most appropriate for the site condition and level of invasion (e.g., herbicide application on a new invasion for eradication, biocontrol for suppression when several hundred acres are infested, and restoration or postfire rehabilitation for low to moderate levels of infestation in areas with moderate to high resilience and resistance to invasive annual grasses)?
- Is a monitoring plan in place to determine whether the management objective was achieved and the invasion threat reduced, and whether subsequent treatments are needed?

Broad- to Mid-Scale Considerations

Using the Science Framework Approach to Inform Invasive Species Management

Many invasive plants, such as invasive annual grasses, represent persistent ecosystem threats (Chambers et al. 2017a) and are widely distributed across the sagebrush biome. The extensive nature of the invasion threat and limited resources for invasive plant management preclude addressing invasive species across the entire biome. Part 1 of the Science Framework provides an approach that uses assessments at the mid-scale to help prioritize areas for management and determine effective management strategies (Chambers et al. 2017a; hereafter, Part 1). Although the approach was developed with a focus on invasive annual grasses, it is applicable to other invasive plants where information exists on the environmental characteristics necessary for their establishment, growth and reproduction, and persistence. This approach is based on: (1) the likely response of an area to disturbance or stress due to threats and management actions (i.e., resilience to disturbance and resistance to invasive annual grasses), (2) the capacity of an area to support target species or resources, and (3) the predominant threats. A geospatial process is used that involves overlaying key data layers including resilience and resistance to invasive annual grasses as indicated by soil temperature and moisture regimes (Maestas et al. 2016), GRSG breeding habitat probabilities (Doherty et al. 2016) or habitats of other sagebrush dependent species, and the primary threats for the ecoregions or Management Zones in the assessment (Part 1, sections 8.1 and 8.2).

Geospatial data on invasive plant species distribution and abundance can be used in conjunction with other threats in the analyses to: (1) evaluate the level of risk of vegetation types and communities to invasion, (2) further refine target areas for management, and (3) determine the most appropriate type of management actions (e.g., Part 1, section 9.2.2, example 2: southwestern Wyoming). Data layers or methods for remotely sensing invasive plants exist for cheatgrass in portions of the Cold Deserts (Boyte and Wylie 2015, 2016; Boyte et al. 2017), spotted knapweed and babysbreath (*Gypsophila paniculata*) (Lass et al. 2005), and rush skeleton weed (*Chondrilla juncea*) (Kesoju et al. 2015). Data layers on roads and other vectors can be used to evaluate the level of risk for future spread of the invasives. Data on interacting threats (e.g., wildfire) can help provide an understanding of the patterns and spread of the invasive plant. Available data layers to consider are in Part 1, section 8.1 and Appendix 8.

The GRSG habitat resilience and resistance matrix (table 1.3) illustrates an area's relative resilience to disturbance and resistance to invasive annual grasses in relation to its probability of providing breeding habitat for GRSG. This matrix, along with table 5.1, provides a decision-support tool that helps to prioritize areas for invasive plant management actions and develop effective management strategies. Management strategies to address the predominant threats for sagebrush ecosystems including invasive plants are found in table 1.4 and table 5.2. The maps and analyses that managers derive from the geospatial approach described in the Science Framework are used along with table 1.3 to prioritize areas for management actions and develop management strategies.

Coordination and Collaboration

Coordination and collaboration provide an effective, strategic approach for managing invasive plant threats across land ownerships and jurisdictions by developing shared priorities and leveraging resources. Collaborative spatial analyses conducted with partners and stakeholders can help identify the extent and scope of invasive plants and identify priority areas for management. A participatory process guided by common, strategic approaches can be used to prioritize what, where, how, when, and by whom actions are implemented at the project level (Beier et al. 2016).

Areawide invasive plant management coordination provides an opportunity for diverse interests and multiple stakeholders to work collaboratively across the landscape to prevent and control nonnative plant invasions, and accomplish mutually beneficial landscape restoration goals. Coordination among stakeholders is critical when there are limited resources, and when management activities are redundant, are not in alignment with partners, or conflict with recommended invasive plant management strategies. One mechanism to increase coordination and collaboration is to develop and participate in local organizations that integrate noxious weed management resources across jurisdictional boundaries and benefit entire communities. An example is Cooperative Weed Management Area (CWMA) partnerships, voluntary organizations that increase communication, share resources, and ultimately increase capacity to manage the invasive plant threat and meet restoration goals. For instance, the Utah-Idaho CWMA partnership has treated medusahead by burning prior to spring herbicide application. The partnership worked with over 200 landowners for more than 10 years to control invasive plants (http://www.utahweed.org/PDF/U&ICWMA.pdf). Several resources for establishing a CWMA are provided online (e.g., http://www.weedcenter.org/management/ guidelines/tableofcontents.html and http://invasivespecies.idaho.gov/2017-cost-share-app); an example of a CWMA is at: http://www.utahweed.org/cwma.htm. Although there is no single model, most functional and effective CWMAs have adequate and sustainable funding, strong core leadership, and clearly defined boundaries and management roles. They often include a diversity of private, county, State, Federal, and tribal members.

CWMAs are established in many areas in the West to address invasive plant management issues. In the sagebrush biome, full geographic coverage of CWMA partnerships would be advantageous in preventing management and coordination gaps across the broader landscape. CWMA membership is difficult to sustain as financial limitations are increasing across rural land ownerships in most regions of the West. Although funding has drastically declined over the last several years, the National Fish and Wildlife Foundation, in cooperation with the Federal Interagency Committee for the Management of Noxious and Exotic Weeds, established the Pulling Together Initiative grant program (http://www.nfwf.org/pti/Pages/home.aspx) in part to encourage the development and sustainability of CWMAs across the United States (FICMNEW 1998). This national grant program is vital in supporting establishment and sustainability of local CWMAs. If financial support continued and increased, the Pulling Together Initiative grant program could expand the establishment and functional effectiveness of CWMAs across the sagebrush biome.

CWMAs could be strategically located to maximize their ability to address the full range of invasive plant species threats in the highest priority areas and to maximize restoration effectiveness. However, CWMAs have not consistently been invited, encouraged, or financially supported to become involved in setting management priorities for sagebrush conservation or invasive plant management within fire and fuel management planning. In some cases, CWMAs are hampered because of policy or procedural roadblocks that prevent establishment of formal agreements with the CWMAs or transfer of Federal or State funds to either the group or individual members within the CWMA. These roadblocks should be evaluated for a more responsive approach through governmental and nongovernmental coordination groups, such as State and county weed management agencies, interagency State-Federal coalitions, or other authorities.

A web-based networking system to connect the activities of individual CWMAs and share information across the sagebrush biome could be established and supported through partnerships with State agriculture departments, Landscape Conservation Cooperatives, Federal land management agencies, tribes, and other stakeholders in the public and private sector. Various programs exist for reporting noxious weed infestation (e.g., Early Detection and Distribution Mapping System [EDDMapS; http://www.eddmaps.org/]). However, State and Federal agencies differ in their level of compliance and consistency for sharing data and utilizing a centralized clearinghouse of invasive plant species occurrence data. Federal, State, and county agencies, nongovernmental organizations, and researchers interested in using these data are working together to address these needs (e.g., Western Governors Association Invasive Species Initiative, North American Invasive Species Management Association, EDDMapS, several western States).

Prevention, Early Detection, and Rapid Response

Prevention is the key to a successful invasive species program as it ensures that the management burden is not continually increased as a result of new invasions (table 5.1). Prevention is generally low cost and has a high return on investment because preventive measures are less costly than funding efforts to control infestations over multiple years. Identifying invasion-free areas allows land managers to focus resources where they are most needed and will have the greatest chance of success. Coordination with partners can help identify invasion-free areas across regions by conducting collaborative monitoring inventories and surveys (Mealor et al. 2013; Rew and Pokorny 2006). Uninvaded areas at a higher risk of invasion, such as those with low resilience and resistance to invasive plants or higher amounts of disturbance, should be considered for frequent monitoring to help keep them invasion free (tables 5.1, 5.2).

Geospatial analyses of the distribution and abundance of invasive plants can help identify uninvaded areas and other areas at increased risk for invasion. Data layers may include current invasion extent, resilience and resistance to invasive annual grasses (Part 1, fig. 33), vectors such as roads (Part 1, fig. 20), and disturbances such as oil and gas wells (Part 1, fig. 16), human development (Part 1, fig. 18), and wildfires (Part 1, fig. 34). Distinguishing between **surveyed uninvaded** areas and **unsurveyed** areas when recording occurrence of invasive plants and analyzing their distribution is necessary to evaluate management and monitoring efforts in uninvaded areas and determine future actions.

Prevention strategies help minimize the risk of expansion of invaded areas and maintain connectivity of intact, uninvaded areas; these strategies should be applied across the sagebrush biome. Considering consequences for new invasions when implementing management and development activities in invasion-free areas can help prevent invasion. For example, using certified weed-free straw, hay, and gravel for development or restoration projects is critical to prevent unintended introductions (table 5.1). The Great Basin portion of the sagebrush biome has substantial areas with low resilience and resistance to invasive annual grasses that are now invaded by these annual grasses. In contrast, the eastern portion of the biome contains large areas of moderate to high resistance to invasive annual grasses. However, uninvaded areas in the eastern portion of the range, especially those with lower resilience and resistance, are still at risk and should be identified for prevention strategies to keep “clean areas clean” and avoid large-scale invasion and dominance of invasive annual grasses as in the Great Basin. Both the Great Basin and the eastern portion of the range also have other invasive plants, such as medusahead, ventenata (*Ventenata dubia*), leafy spurge, and Russian knapweed, that should be monitored for expansion and prevented from further spread. Stringent triage measures based on impact and risk need to be developed for these species to assist with prevention.

Table 5.1—General management strategies for cheatgrass and other invasive plants based on the invasion state with an estimate of cost:benefit of return depending on site and neighboring conditions. Management strategies are based on the level of invasion for cheatgrass, but many of the concepts also apply to annual and perennial invasive forbs. The strategies for invasive plant management are prevention, Early Detection and Rapid Response (EDDR) (USDOI 2016), eradication, suppression, containment, and restoration. The invasion state is adapted from Mealor et al. (2013). Several best management practices for prevention were adapted from Cal-IPC (2012).

	Invasion state				
	Invasion free	**Trace (1–5%) with perennials**	**Mild (6–25%) with perennials**	**Moderate (26–50%) with perennials missing**	**Invasion dominated state**
Management strategies based on invasion level	**Prevention** • Manage for sufficient density and cover of native perennial grasses and forbs and biological soil crusts • Monitor high-risk priority areas for new invaders • Use certified weed-free straw, hay, mulch, and gravel for development or restoration • Avoid use of invasive species in fuel breaks • Minimize road and infrastructure development and disturbance • Clean clothing, footwear, equipment, and vehicle of invasive plant material for land or fire management activities (Cal-IPC 2012, Checklist E) • Provide training on invasive plant awareness • Incorporate invasive plant information and management into Fire Incident Action Plans	**Prevention** • Manage for sufficient density and cover of native perennial grasses and forbs and biological soil crusts • Limit soil disturbance and revegetate bare soil post-fire **EDRR** • Early Detection monitoring • Rapid Response treatment of new invasions **Eradication** • Consistent and multiple year treatments with monitoring • Promote desirable, native vegetation	**Prevention** • Manage for native perennial grasses and forbs and prevent further disturbance of biological soil crusts • Limit soil disturbance and revegetate bare soil post-fire **Suppression/ Containment** • Implement control treatments • Seed post-treatment and implement restoration where appropriate • Monitor for invasive plants post-fire and post-treatment • Monitor and manage invasive plants to maintain fuel management sites **Restoration** • Monitor and maintain desirable vegetation • Identify native seed sources • Consider revegetation after invasive plant or fire management	**Prevention** • Manage for native perennial grasses and forbs and prevent further disturbance of biological soil crusts **Suppression/ Containment** • Monitor for invasives post-fire with restoration • Locate fire lines to reduce additional disturbance and invasive plant spread where feasible **Restoration** • Implement restoration with seeding in areas lacking perennial grasses and forbs	**Containment or restoration** • For areas with high fire probability, consider fuel breaks adjacent to, not intersecting, relatively uninvaded areas and restored areas • Consider significant and consistent control treatments for high priority areas or areas adjacent to uninvaded areas • Consider restoration when invasion-dominated site is located between intact sagebrush habitat patches or between high priority areas
Cost:Benefit	Low cost:Highest return	Low cost:Very high return	Moderate cost:High return	Moderate to high cost:High return	High cost:Moderate return

Table 5.2—Management strategies for cheatgrass and other invasive plants based on the area's relative resilience to disturbance, resistance to invasive annual grasses, and the invasion state. Management strategies are based on the level of invasion for cheatgrass, but many of the concepts also apply to annual and perennial invasive forbs. The invasion state is adapted from Mealor et al. (2013); resilience and resistance (R&R) categories are based on Chambers et al. (2017a).

	Invasion free	Trace (1–5%) with perennials	Mild (6–25%) with perennials	Moderate (26–50%) with perennials missing	Invasion dominated state
High resilience and resistance	• Monitor priority areas for new invaders, especially with disturbance or wildfire • Identify uninvaded areas and minimize disturbance to prevent new introductions • Manage livestock to maintain or increase perennial native grasses and forbs • Possibly no action post-disturbance (wildfire)	• Conduct EDRR monitoring every 3–5 years until detected • For new, small populations that are detected, herbicide use may be most efficient, but repeated application is required until control is achieved • Support natural recovery	• Manage for native perennials • Implement periodic grazing deferment • Prioritize treatment areas to maximize effectiveness • Use spot herbicide treatment for 3–5 years • Seed natives post-herbicide treatment	• Implement periodic grazing deferment • Evaluate site conditions for integrated weed management when grazing or fire management used • Use spot herbicide treatments, rather than broadcast treatments, for at least 5–10 years • Seed natives post-herbicide treatment	• Restoration success possible both pre- and post-fire • Avoid seeding introduced species to prevent spread • Use integrated herbicide application and seeding • Consider sagebrush transplants • Locate and maintain fuel breaks to prevent invasive plant introduction and spread and avoid intersecting uninvaded areas
Recovery potential		Very high	High	High to moderate	Moderate
Moderate resilience and resistance	*Management strategies for moderate R&R depend on soil temperature and moisture regimes. Treat relatively cool and moist areas similarly to high R&R areas. Treat relatively warm and dry areas similarly to low R&R areas.*				
Recovery potential		High	Moderate	Moderate	Moderate to low
Low resilience and resistance	• Identify uninvaded areas and prioritize prevention • Conduct EDRR annually • Monitor areas that are high priority or adjacent to high priority areas frequently • Monitor disturbed areas frequently • Avoid fuel break placements that connect invaded and uninvaded areas and avoid intersecting uninvaded areas	• Develop an EDRR network in high priority areas • Promote desirable native vegetation • Monitor herbicide treatments and continue treating as needed • Minimize disturbance and suppress wildfire to prevent new introductions • Locate and maintain fuel breaks to prevent invasive plant spread • Prioritize postfire monitoring for invasive plants	• Identify high fire risk areas and identify invasive plant populations in these areas and travel routes to minimize spreading invasives • Use significant and consistent treatments to prevent crossing threshold into heavy infestation and manage for native perennials • Minimize disturbance and suppress wildfire • Locate and maintain fuel breaks to prevent invasive plant introduction and spread • Prioritize postfire monitoring for invasive plants with subsequent treatment and revegetation	• Identify high fire risk areas and invasive plant populations in these areas • Use significant and consistent treatments for containment and suppression in high priority areas to prevent crossing threshold into heavy infestation and to protect adjacent high quality habitat • Suppress wildfire • Avoid fuel breaks that connect invaded and uninvaded areas and try not to intersect uninvaded areas	• Restoration not feasible for most areas. Restoration in high priority area will require significant and long-term funding • Consider targeted grazing to reduce invasive annual grass fuels to reduce fire risk to adjacent higher priority areas • Consider herbicide application and integrated biocontrol at perimeter to prevent spread • Consider fuel breaks around perimeter of invaded area to contain fine fuels • Restoration may require repeated interventions • Consider seeding nonnatives where natives fail to establish
Recovery potential		Moderate	Low	Low	Low to none

Early Detection and Rapid Response (EDRR) strategies survey for those new invasive plants most likely to increase in abundance (text box 5.1, Appendix 3) and pursue treatment as quickly as possible. An overview of the National Framework for Early Detection and Rapid Response to invasive plants is available on the USDA National Invasive Information Center website (https://www.invasivespeciesinfo.gov/toolkit/detection.shtml). An example of how EDRR can be incorporated into a monitoring strategy is in text box 5.1. Early detection and rapid response strategies are cost-effective and successful because they focus on eliminating new, small invasions, which are less costly to treat and easier to eliminate (Chippendale 1991 in Hobbs and Humphries 1995; Keller et al. 2007; Leung et al. 2002). The removal of small, separate populations of invasive plants (table 5.1) is a high priority because they often expand more rapidly and cover potentially greater areas than the edge of a large, single source population (Cousens and Mortimer 1995; Moody and Mack 1988). Most invasive plants have a long lag period before they spread following introduction, so they can usually be eradicated if treated as soon as possible after detection. Early detection can make the difference between employing feasible offensive strategies versus retreating to defensive strategies, which usually result in an infinite financial commitment (Rejmanek and Pitcairn 2004).

Extensive outreach and communication about new invaders, their identification, and life history characteristics and identifying the areas that are most at risk can help foster detection, reporting, and rapid response (see Appendix 3). Establishing a communication network among landowners, public land management agencies, recreation groups, conservation organizations, botanists, horticulturalists, and weed organizations to report new invasive plant infestations will help meet detection and monitoring objectives. Targeting species of known concern and high-risk invasion pathways, such as low resistance areas, roadsides, and areas disturbed by human development, can be a successful detection strategy (table 5.2).

Text Box 5.1—Monitoring for Early Detection of Invasive Species

Early Detection and Rapid Response (EDRR) provides an opportunity to control the spread of invasive species (USDOI 2016). Monitoring for early detection of invasive species requires the following:

1. Identify known high-risk invasive species and provide training for rapid species detection and identification.
2. Coordinate priority monitoring areas across land management jurisdictions.
3. Identify locations of existing invasions and likely invasion pathways to identify areas where invasive species may first establish (e.g., recreation sites, trails, and roadsides, and in areas with treatments, recent fires, energy development, and other types of disturbance).
4. Survey, report, and verify the presence of invasive species before the population becomes established or spreads so widely that eradication is no longer feasible.
5. Utilize early detection tools that can be readily accessed and allow data to be recorded and shared among networks of Federal, State, private and nongovernmental partners (e.g., Early Detection and Distribution Mapping System [EDDMapS]).
6. Use invasive plant species presence and abundance as monitoring indicators in other vegetation monitoring programs (e.g., the Bureau of Land Management's Assessment, Inventory, and Monitoring [AIM] and the Natural Resources Conservation Service's National Resources Inventory [NRI]).
7. Develop management triggers designed to address early invasions. Monitoring plans can be greatly improved when an invasive species list or georeferenced abundance data are available (Brooks and Klinger 2009).

Agency programs such as forestry, grazing, energy development, recreation, wildlife, and wildfire management have the responsibility to incorporate invasive species management strategies (Federal Noxious Weed Act, 7 U.S.C. §§ 2801-2814, January 3, 1975, as amended 1988 and 1994) and coordinate management actions with CWMAs. These management programs can identify geographic areas within their program jurisdictions that have either known populations of invasive plants or low resistance to certain species. They can also identify areas that serve as sources of invasive plants and conduits for their spread. Source areas for invasive plants include recent ecosystem disturbances, such as wildfire or die-offs due to drought, and anthropogenic developments, such as oil and gas wells or cropland conversion. Mapping overlays of resilience and resistance with known populations of invasive plants, disturbed areas, and road and trail networks can provide a broad-scale assessment of where to focus invasive plant prevention and control measures. For example, suppression and control of invasive plants along roads that link invaded areas to non-invaded areas can help to prevent or minimize movement along this vector. Similarly, the potential for spread of invasive plants can be considered when siting linear firebreak networks and determining follow-up actions. Monitoring programs that involve multiple management jurisdictions and program areas can be used to evaluate both the spread of invasive plants and the effectiveness of control measures.

Local Scale Considerations

Management Strategies

Management of invasive plants and restoration of native species require the capacity to address the full suite of management activities spanning inventory and mapping, prevention, EDRR, suppression/reduction and containment, collaboration and partnership development, data collection and sharing, and restoration and rehabilitation. General project priorities for invasive plant management exist (text box 5.2), but alignment of regional strategic goals for conservation and restoration of sagebrush ecosystems and the involvement of partnerships (e.g., CWMA, State and county governments) are needed. There also may be areas within the sagebrush biome that require immediate invasive plant management actions to reduce threats to other rare or unique plants. This kind of need can be highlighted with coordination and communication at the local scale.

Resilience and resistance concepts and decision matrices can be used in project selection and design for invasive species management. At the project scale, specific ecological site description information (e.g., precipitation and temperature regimes, soil characteristics, vegetation composition), state-and-transition models, and available invasive plant assessment data (inventory and monitoring data, risk assessments, predicted occurrence) help set priorities for management actions (see Miller et al. 2014, 2015). Because invasions can vary in distribution and abundance across project areas, a critical first step in diagnosing the level of threat is to complete inventories and assessments within the project boundary.

Once the size and impact of the invasion are determined, an evaluation of the recovery potential (resilience and resistance to the specific invader) will help determine and prioritize treatment activities with the highest chance of success for invasive plant eradication, suppression, reduction, or containment (table 5.1). New invasions, low density invasions, and invasions in areas of high to moderate resilience align well with the strategies of early detection, rapid response, and

Text Box 5.2—Invasive Plant Management Priorities and Limitations

Invasive plant management priorities and limitations need to be considered when developing broad-scale approaches.

Invasive Plant Management Priorities

1. Assess the extent of the invasion for spatial distribution and abundance.
2. Prevent new infestations and implement Early Detection and Rapid Response to maintain areas without invasive plant infestations that are ecologically intact.
3. Reduce densities and cover of invasive plants with invasive plant management while native plant species are available to respond and before the invaders dominate.
4. Consider containment of large, well-established infestations to prevent perimeter spread, rather than full-scale costly control efforts that may have a low chance of success.
5. Conduct revegetation efforts in high priority areas with a high probability of success based on ecological condition when sufficient resources are available.

Invasive Plant Management Limitations

1. Competing priorities among land managers that prevent common regional and local prioritization of project areas may create multiple, inconsistent efforts.
2. For many invasive species, detailed ecological knowledge on climatically suitable areas for their establishment and spread is lacking. Thus, it is difficult to characterize ecosystem resistance to these species, identify areas most at risk of invasion, or determine the most appropriate and effective management tools and methods.
3. Inconsistent and incompatible administrative procedures for operations, datasets, and databases among partners can slow or hinder effective communication and implementation (Ielmini et al. 2015).

suppression or reduction (table 5.2). Multi-year, consistent treatments in areas with high to moderate resilience and resistance to invasive plants may achieve eradication of new or small infestations (table 5.2). Larger, well-established infestations are likely to need long-term treatment measures for potential suppression or containment on the perimeter of large invaded patches.

If funding is available and it is a high priority conservation area, it may be feasible to try to restore areas that have large, well-established infestations using an integrated approach which includes invasive control measures and revegetation (tables 5.1, 5.2). Restoration to desired conditions may be feasible in areas with moderate to high resilience. However, in areas with low resilience, repeated interventions and greater levels of financial resources may be necessary. In areas dominated by invasive annual grasses, it may not be possible to establish perennial plants without significant and costly investments. In these cases, managers should consider the return on restoration investment carefully and work with scientists to test new methods for protecting restored areas that have low resilience to fire. The conservation value of a site and the associated cost:return ratio and likelihood of success are used to determine where to place resources for invasive species management (table 5.1). Identification of treatment options is then based on site-specific characteristics, the invasive species, the degree of invasion, potential for native plant recovery, and resources available (table 5.2).

Maintain Intact Native Communities. The most successful tool for maintaining resistance to plant invasions is generally to manage for sufficient density and cover of native perennial grasses and forbs and biological soil crusts to prevent the establishment or population growth of the invader (Chambers et al. 2014a,b). For example, research shows that about 20 percent cover of perennial native grasses and forbs is needed in Wyoming big sagebrush sites to prevent significant increases in cheatgrass and other exotic annuals after management treatments (sagebrush mowing and prescribed fire) (Chambers et al. 2014b). Similarly, about 18 percent cover of perennial native grasses and forbs or 10 perennial grasses per square meter (about 1 perennial grass per square foot) is needed to exclude medusahead rye from these sites (Davies 2008).

Decreases in perennial herbaceous species and biological soil crusts and reductions in resistance to invasive plants result from improper livestock grazing (Adler et al. 2005; Reisner et al. 2013, 2015), high severity wildfire, and juniper and piñon expansion into sagebrush ecosystems (Miller et al. 2013). Reductions in perennial native grasses and forbs are associated with increases in sagebrush density and cover (Chambers et al. 2017b; Cooper 1953), and juniper and piñon densities, canopy cover, or basal area (Guenther et al. 2004; Madany and West 1983; Shinneman and Baker 2009; Soulé et al. 2004). The increases in woody fuels can cause higher severity wildfires with the potential to increase mortality of perennial native species (Miller et al. 2013).

Carefully managed livestock grazing is crucial to maintain perennial herbaceous species, forbs, and biological soil crusts and thus resistance to invasive plants. The livestock grazing strategies identified in the Science Framework are broadly applicable to the sagebrush biome (table 1.4 and section 7). Implementing livestock grazing strategies that incorporate periodic deferment from use during the critical growth period, especially for cool season grasses, can help ensure maintenance of a mixture of native perennial grasses. Adjustments in timing, duration, and intensity of livestock grazing may be needed to reduce invasive species. Livestock grazing that creates patches of bare ground can result in avenues for invasion of species such as spotted and Russian knapweed and is associated with increases in cheatgrass (Reisner et al. 2013).

Other threats to maintaining intact native communities will require diligence in monitoring for new invasions in response to land use and land management practices. Oil and gas development, road maintenance, construction, and potentially even fuel breaks may create disturbances that foster colonization of invasive plants or bring in material contaminated with weed seed. The extent and placement of fuel breaks to reduce fire risk need to be considered and designed carefully to ensure that they do not inadvertently increase subsequent fire risk by creating disturbances conducive to new invasions, especially in uninvaded areas (table 5.1 and section 4). Other measures for preventing new invasions include sanitizing equipment and vehicles pre- and post-access; requiring certified weed-free seed, gravel, topsoil, and hay for construction or restoration; and education and outreach to public, staff, and partners in identification and management of invaders (Mealor et al. 2013; Pyke et al. 2016).

No Action Post-Disturbance. Areas characterized as having moderate to high resilience and resistance (table 1.3: cells 1B, 1C, 2B, 2C) with no current invasions may not require management intervention following disturbances such as wildfire (tables 5.1, 5.2). If these areas have sufficient perennial native grasses and forbs prior to disturbance, they are likely to maintain resistance to most invasive plant species, and invasive species management resources may be better spent in other areas. For example, in relatively cold and moist areas with high ecosystem resilience, allowing the area to recover after wildfire without intervention may be the most effective strategy for preventing increases in invasive plants. However, if invasive plants occur in the area or there are significant fire management activities including access roads and vehicles, then resources should be spent on a monitoring strategy to determine whether the invasive plants increase or colonize. Funding mechanisms should remain available for restoration activities if a no-action approach is not successful.

Invasive Plant Removal and Treatment. Control measures shown to be successful in reducing and removing invasive plants include biological, cultural, physical, and chemical treatments. The 2017 Weed Management Handbook (Peachey 2017) and Weed Control in Natural Areas in the West (DiTomaso et al. 2013) are comprehensive guides to invasive plant management that provide summaries of the requirements and advantages of different tools. Selection of the appropriate tool will vary based on the invasive plant species, extent of the invasion, and resilience of the site. The integration of different controls in treating invasive plants may offer more success over the long term at project scales. When using control methods, practitioners need to consider health, environmental, and economic risks. Selection of controls based on consensus building for common threat-reduction objectives, biology of invader, site conditions, environmental factors, and best available technology can achieve desired outcomes while minimizing effects to nontarget species and the environment. Individual controls that can be used at the project scale are summarized next.

(1) Biological control is the use of natural enemies—predators, parasites, pathogens, and competitors—to control invasive plants over multiple years. Invasive plants have many natural enemies including insects and plant pathogens. Biological control is often considered when the invasion is large and well established (table 5.1) because host plant density is a determinant of whether the biological control agent can become established (table 1.3: cells 1A, 1B, 2A, 2B, 3A, 3B; table 5.2). In practice, biological control options are best determined when the land manager and biological control practitioner coordinate closely to build a long-term biological control plan that includes a strong monitoring component for the targeted invasive plant and the respective biological control agent(s). Site conditions are important for selecting the appropriate biological control agent(s) for the targeted invasive species. Several resources exist for biological control information, including the reference compendium of information available online at https://www.ibiocontrol.org/catalog/.

Other types of control agents for invasive annual grasses, especially cheatgrass, may include fungal pathogens (Meyer et al. 2016) and bacterial agents (Kennedy et al. 2001). These are often mistaken for biocontrol, but they do not function in the same way as predation or feeding behavior, which is typical of classic biocontrol. Applying bacterial agents (e.g., weed suppressive bacteria) may be considered a biopesticide application and requires different application guidelines and policy compliance under State and Federal regulations than classic biocontrols. Multiple trials are underway to evaluate the effectiveness and application guidelines for the use of fungal pathogens and bacterial agents as biopesticides. However, there is currently very limited information demonstrating the effectiveness of either fungal pathogens or bacterial agents for cheatgrass control or the potential effects of these controls on native species. Fungal pathogens do result in large cheatgrass die-off areas that may provide restoration opportunities (Meyer et al. 2016).

Species such as knapweeds and leafy spurge have several biological control agents that may provide support for strategies of containment and suppression (Anderson et al. 2000). Integration of biocontrols with other control measures can have advantages and disadvantages. For example, herbicides could be used around the perimeter of large invaded patches with biocontrols released in the center of the patches to increase overall control. In contrast, release of the biocontrol with herbicide application at the time when biocontrols emerge may result in loss of the biocontrol.

(2) Cultural controls are management practices that reduce establishment, reproduction, dispersal, or survival of the invasive plant. For example, management actions that maintain or increase native perennial herbaceous species can help control many invasive plant species. Other cultural controls, such as prescribed fire or targeted grazing, can impact native communities and are best applied in areas dominated by the invasive plant. Typically, these are lower priority areas for sagebrush conservation and restoration (table 1.3: cells 2A and 3A; table 5.2), but they may be used to meet habitat objectives such as increasing habitat connectivity or establishing fuel breaks.

Prescribed fire may serve as a cultural control for cheatgrass dominated areas if applied during seed maturation in the spring; however, it is rarely an option due to narrow implementation requirements (Mealor et al. 2013). Prescribed fire may also be used as part of an integrated management strategy. Prescribed fire implemented when conditions are safe for burning can reduce standing litter and litter mats in cheatgrass dominated areas (Jones et al. 2015a,b). Reducing the litter in areas dominated by invasive plants can improve effectiveness of certain types of herbicide applications by allowing the herbicide to reach the soil surface (DiTomaso and Johnson 2006). It can also facilitate an integrated restoration approach that includes reducing litter through repeated burning (Jones et al. 2015b) or through prescribed grazing (Frost and Launchbaugh 2003); seeding with sterile cover crops such as common wheat (*Triticum aestivum*) to decrease cheatgrass reproduction and, thus, seedbanks; and then seeding the desired native perennial species (Jones et al. 2015a). If properly implemented, prescribed fire can provide some level of reduction for both invasive perennial and annual grasses and annual forbs. However, prescribed fire does not decrease, and may increase, perennial and biennial invasive forbs (DiTomaso and Johnson 2006).

The removal of cheatgrass by fire or livestock grazing may create conditions that allow release of perennial invasive plants, resulting in a bigger issue. Native species may take many years to increase from low densities following the removal of landscape disturbances such as grazing, perhaps due to seed limitation (Seabloom et al. 2003) or adverse interactions at seed and seedling stages (DiVittorio et al. 2007). In addition, prevention and early detection methods may be needed for recent prescribed fire (and wildfire) operations to ensure that suppression activities do not inadvertently increase risk for invasive plant colonization and spread.

Targeted grazing is the application of a specific kind of livestock at a determined season, duration, and intensity to accomplish defined vegetation or landscape goals (Launchbaugh and Walker 2006). Sheep and goats are effective tools for reducing invasive plants such as leafy spurge, spotted knapweed, and cheatgrass (Mosely 1996; Mosely et al. 2016). Intense sheep grazing of cheatgrass dominated sites can effectively suppress or even eliminate cheatgrass stands in as little as 2 years as was done in the urban interface above Carson City, Nevada (Mosley 1996). However, the effects of correctly applied targeted grazing are generally slow and cumulative (Launchbaugh and Walker 2006) and still need to be tested for applicability across broad areas.

Managed grazing may also reduce the risk and extent of wildfire in cheatgrass dominated areas (Diamond et al. 2009, 2012; Walker 2006). Because livestock grazing reduces herbaceous vegetation (fine fuels), grazing may reduce the extent of wildfire (Walker 2006) (table 5.2). Further, livestock tend to graze some areas more intensely than others, so grazing may create patchy vegetation that reduces the continuity of fuel loads and the fires that might burn those fuels (Walker 2006). In sagebrush ecosystems, exploratory high intensity targeted grazing to create fuel breaks can be tested by confining livestock to a strip of land with temporary fencing. In a fenced Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) ecosystem, cattle removed 80 to 90 percent of *B. tectorum* biomass in May during the boot phenological stage (Diamond et al. 2009). Grazing resulted in reductions in flame length and rate of spread compared to nongrazed plots in the first year; cheatgrass biomass and cover were reduced to the point that fires did not carry in the grazed plots in the second year (Diamond et al. 2009). However, grazing resulted in an increase of invasive annual forbs and Sandberg bluegrass (*Poa secunda*) (Diamond et al. 2012). This demonstrates there may be tradeoffs that will require secondary or additional management actions for other invasive species, such as the invasive annual forbs that responded to the grazing.

Effective grazing programs for invasive plant control require a clear statement of the kind of animal, timing, and rate of grazing necessary to suppress the invasive plant (Launchbaugh and Walker 2006). A successful grazing prescription should: (1) cause significant reduction in the target plant, (2) limit effects on the surrounding vegetation, and (3) be integrated with other control methods as part of an overall management strategy. Because targeted grazing by livestock is typically focused on heavily invaded areas, follow-up management such as seeding of the target area with the desired species may be needed. In big sagebrush areas with a cheatgrass understory where grazing is used to suppress cheatgrass, it may be possible to interseed the sagebrush with perennial grasses and forbs after treatment (Huber-Sannwald and Pyke 2005).

(3) Mechanical and physical controls such as hand pulling, mowing, or disking before seed production kill invasive plants directly, block establishment, or make the environment unsuitable for establishment. To date, these methods have not been widely applied to invasive annual grasses or perennial invasive plants in sagebrush ecosystems. There are potential tradeoffs of destroying biological soil crusts with some of these methods.

(4) Chemical control is the use of pesticides, which include herbicides, fungicides, or biopesticides (as mentioned in the discussion of biocontrols). Pesticides are typically used as an efficient and cost-effective approach to control invasive plant infestations, and, like other integrated pest management techniques, are best used in combination with other treatment approaches for more effective, long-term control. Ecological type or site descriptions and state-and-transition models that integrate information on resilience and resistance to invasive annual grasses (see Part 1, Appendices 5 and 6) can help determine whether herbicides are the best control method for larger invasions. Herbicides can be very useful for eradicating small patches of invasive plants or interrupting the spread of large patches along advancing fronts by containing the perimeter (Rinella et al. 2009) (tables 5.1, 5.2). In some situations, large-scale herbicide applications have been used to treat well-established plant invasions before implementing native plant restoration actions, in order to maximize effectiveness across large landscapes or along border areas. Evaluating the degree and extent of neighboring invasions can provide information on whether the invasive species can recolonize from a neighboring untreated area. Additionally, evaluating the existing seedbanks within a treated area can provide information to help determine whether repeated treatments are needed and, if so, for how long (e.g., 3–15 years).

Several basic elements should be included in all pesticide (herbicide) use proposals and application plans prior to implementing any herbicide application by trained and experienced personnel. These include proper selection of the appropriate herbicide product and adjuvants for the targeted invasive species, site condition, and the appropriate application technique and timing. Detailed knowledge of the soil and water conditions and other environmental concerns in the treatment area is also needed. Proper application of appropriate herbicidal products can be an effective solution for managing established invasive plant populations. Although there may be short-term collateral damage, proper herbicide application planning greatly reduces the chance of unintended negative impacts to nontarget native plants and associated fish and wildlife in the treatment area. For example, to minimize effects, herbicide applications may involve spot-spraying of localized invasive patches within the area by using a backpack sprayer, rather than aerial spraying the entire area, which may increase the risk of nontarget impacts. Further, while broadcast spray is a method for treating large, well-established invasions, the level of reduction in density or coverage accomplished and the effects on nontarget native plant communities, soils, or biological crusts, and costs of multi-year treatments needed should be carefully considered before implementation.

Conclusions

Sagebrush ecosystem conservation must recognize the need for greater investment in preventing additional plant invasions and limiting the spread of existing invasions across the entire sagebrush biome. This type of investment will support land owners and managers in a proactive management approach rather than the reactionary approach that is currently in place. Without prevention and a proactive approach, the ongoing expansion of invasive plants will continue to outpace restoration efforts and resources. Areas could be prioritized for proactive invasive plant management based on resources of concern, community needs, and opportunities for success according to resilience and resistance to invasion and current ecological site conditions such as the level of invasion. Uninvaded areas could be identified and monitored for new plant invasions and, if invasions occur, quickly treated and eradicated. Invasive annual grass control is the key to preventing and reducing uncharacteristic fuel and fire regimes. Partnerships are critical and must be developed to provide consistent invasive plant management to maintain weed-free areas and prevent mild invasions from spreading and crossing thresholds into heavy infestations. An all-hands-on-deck effort to leverage resources for restoration efforts is needed in high priority areas. Combating invasive plants pre- and postfire and addressing the technical, policy, communication, and operational challenges needs to be a priority. Addressing these challenges will help to prevent negative effects to ranching livelihoods and recreational opportunities and protect the sagebrush biome from overall ecosystem degradation.

References

Adler, P.B.; Milchunas, D.G.; Sala, O.E.; [et al.]. 2005. Plant traits and ecosystem grazing effects: Comparison of U.S. sagebrush steppe and Patagonian steppe. Ecological Applications. 15: 774–792.

Anderson, G.L.; Delfosse, E.S.; Spencer, N.R.; [et al.]. 2000. Biological control of leafy spurge: An emerging success story. In: Spencer, Neal R., ed. Proceedings, X International Biological Control Symposium; 1999 July 4–9; Bozeman, MT. Sidney, MT: U.S. Department of Agriculture, Agricultural Research Service, Northern Plains Agricultural Research Laboratory. http://www.team.ars.usda.gov/teampub/andersonpaper.html. [Accessed May 23, 2018].

Beier, P.; Hansen, L.J.; Helbrecht, L.; [et al.]. 2016. A how-to guide for coproduction of actionable science. Conservation Letters. 10: 288–296.

Boyte, S.P.; Wylie, B.K. 2015. Near-real-time cheatgrass percent cover in the northern Great Basin, USA—2015. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. https://www.sciencebase.gov/catalog/item/55ad3a16e4b066a2492409d5. [Accessed May 23, 2018].

Boyte, S.P.; Wylie, B.K. 2017. Near-real-time herbaceous annual cover in the sagebrush ecosystem. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. https://www.sciencebase.gov/catalog/item/595e6cc3e4b0d1f9f0570318. [Accessed May 23, 2018].

Boyte, S.P.; Wylie, B.K.; Major, D.J. 2016. Near-real-time cheatgrass percent cover in the northern Great Basin, USA—2016. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. https://www.sciencebase.gov/catalog/item/577fcb6ce4b0ef4d2f45fbf3. [Accessed May 23, 2018].

Brooks, M.L.; Brown, C.S.; Chambers, J.C.; [et al.]. 2016. Exotic annual *Bromus* invasions: Comparisons among species and ecoregions in the western United States. In: Germino, M.J.; Chambers, J.C.; Brown, C.S., eds. Exotic brome-grasses in arid and semiarid ecosystems of the western U.S. New York, NY: Springer: 11–60.

Brooks, M.L.; Klinger, R.C. 2009. Practical considerations for early detection monitoring of plant invasions. In: Inderjit, ed. Management of invasive weeds. Dordrecht: Springer: 9–33.

California Invasive Plant Council [Cal-IPC]. 2012. Preventing the spread of invasive plants: Best management practices for land managers. 3rd ed. Cal-IPC Publication 2012-03. Berkeley, CA: California Invasive Plant Council. www.cal-ipc.org. [Accessed October 26, 2018].

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017a. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p.

Chambers, J.C.; Board, D.I.; Roundy, B.A.; [et al.]. 2017b. Removal of perennial herbaceous species affects response of cold desert shrublands to fire. Journal of Vegetation Science. 28(5): 975-984.

Chambers, J.C.; Bradley, B.A.; Brown, C.A.; [et al.]. 2014a. Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. invasion in the cold desert shrublands of western North America. Ecosystems. 17: 360–375.

Chambers, J.C.; Miller, R.F.; Board, D.I.; [et al.]. 2014b. Resilience and resistance of sagebrush ecosystems: Implications for state-and-transition models and management treatments. Rangeland Ecology and Management. 67: 440–454.

Chambers, J.C.; Germino, M.J.; Belnap, J.; [et al.]. 2016. Plant community resistance to invasion by *Bromus* species—The role of community attributes, *Bromus* interactions with plant communities and *Bromus* traits. In: Germino, M.J.; Chambers, J.C.; Brown, C.S., eds. Exotic brome-grasses in arid and semiarid ecosystems of the western U.S. New York, NY: Springer: 275–306.

Chambers, J.C.; Roundy, B.A.; Blank, R.R.; [et al.]. 2007. What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? Ecological Monographs. 77: 117–145.

Chippendale, J.F. 1991. Potential returns to research on rubber vine (*Cryptostegia grandiflora*). Thesis. Brisbane, Australia: University of Queensland. 93 p. https://espace.library.uq.edu.au/view/UQ:260137. [Accessed May 23, 2018]

Collins, S.L.; Uno, G.E. 1985. Seed predation, seed dispersal, and disturbance in grasslands: A comment. American Naturalist. 125: 866–872.

Cooper, H.W. 1953. Amounts of big sagebrush in plant communities near Tensleep, Wyoming as affected by grazing treatment. Ecology. 34: 186–189.

Cousens, R.; Mortimer, M. 1995. Dynamics of weed populations. New York, NY: Cambridge University Press. 332 p.

Davies, K.W. 2008. Medusahead dispersal and establishment in sagebrush steppe plant communities. Rangeland Ecology and Management. 61: 110–115.

Diamond, J.M.; Call, C.A.; Devoe, N. 2009. Effects of targeted cattle grazing on fire behavior of cheatgrass-dominated rangeland in the northern Great Basin, USA. International Journal of Wildland Fire. 18: 944–950.

Diamond, J.M.; Call, C.A.; Devoe, N. 2012. Effects of targeted grazing and prescribed burning on community and seed dynamics of a downy brome (*Bromus tectorum*)-dominated landscape. Invasive Plant Science and Management. 5: 259–269.

DiTomaso, J.M.; Johnson, D.W., eds. 2006. The use of fire as a tool for controlling invasive plants. Cal-IPC Publication 2006-01. Berkeley, CA: California Invasive Plant Council. 56 p. http://www.cal-ipc.org/docs/ip/management/UseofFire.pdf. [Accessed May 23, 2018].

DiTomaso, J.M.; Kyser, G.B.; Oneto, S.R.; [et al.]. 2013. Weed control in natural areas in the western United States. Davis, CA: UC Davis Weed Research and Information Center. 544 p. http://www.cal-ipc.org/resources/library/publications/weedcontrol/. [Accessed May 23, 2018].

DiVittorio, C.T.; Corbin, J.D.; D'Antonio, C.M. 2007. Spatial and temporal patterns of seed dispersal: An important determinant of grassland invasion. Ecological Applications. 17: 311–316.

Doherty, K.E.; Evans, J.S.; Coates, P.S.; [et al.]. 2016. Importance of regional variation in conservation planning: A range-wide example of the Greater sage-grouse. Ecosphere. 7: e01462.

Federal Interagency Committee for the Management of Noxious and Exotic Weeds [FICMNEW], eds. 1998. Pulling together: National strategy for invasive plant management. Washington, DC: U.S. Government Printing Office. 14 p.

Frost, R.A.; Launchbaugh, K.L. 2003. Prescription grazing for rangeland weed management: A new look at an old tool. Rangelands. 25: 43–47.

Germino, M.J.; Belnap, J.; Stark, J.M.; [et al.]. 2016. Ecosystem impacts of the exotic annual invaders in the genus *Bromus*. In: Germino, M.J.; Chambers, J.C.; Brown, C.S., eds. Exotic brome-grasses in arid and semiarid ecosystems of the Western US: Causes, consequences and management implications. New York, NY: Springer: 61–98.

Guenther, D.; Stohlgren, T.J.; Evangelista, P. 2004. A comparison of a near-relict site and a grazed site in a pinyon-juniper community in the Grand Staircase-Escalante National Monument, Utah. In: Van Riper, C.; Cole, K.L., eds. The Colorado Plateau: Cultural, biological and physical research. Tucson, AZ: University of Arizona Press: 153–162.

Hill, J.P.; Germino, M.J.; Wraith, J.M.; [et al.]. 2006. Advantages in water relations contribute to greater photosynthesis in *Centarea maculosa* compared with established grasses. International Journal of Plant Sciences. 167: 269–277.

HilleRisLambers, J.; Yelenik, S.G.; Colman, B.P.; [et al.]. 2010. California annual grass invaders: The drivers or passengers of change? Journal of Ecology. 98: 1147–1156.

Hobbs, R.J.; Humphries, S.E. 1995. An integrated approach to the ecology and management of plant invasions. Conservation Biology. 9: 761–770.

Huber-Sannwald, E.; Pyke, D.A. 2005. Establishing native grasses in a big sagebrush-dominated site: An intermediate restoration step. Restoration Ecology. 13: 292–302.

Ielmini, M.R.; Hopkins, T.E.; Mayer, K.E.; [et al.]. 2015. Invasive plant management and greater sage-grouse conservation: A review and status report with strategic recommendations for improvement. Cheyenne, WY: Western Association of Fish and Wildlife Agencies. 47 p. https://www.wafwa.org/Documents%20and%20Settings/37/Site%20Documents/Initiatives/Sage%20Grouse/WAFWA%20Invasive%20Plant%20Management%20and%20Greater%20Sage-Grouse%20Report%20FINAL%203-28-15.pdf. [Accessed May 23, 2018].

Jones, R.O.; Chambers, J.C.; Board, D. I.; [et al.]. 2015a. The role of resource limitation in restoration of sagebrush ecosystems dominated by cheatgrass (*Bromus tectorum*) Ecosphere. 6: 107.

Jones, R.O.; Chambers, J.C.; Johnson, D.W.; [et al.]. 2015b. Effect of repeated burning on plant and soil carbon and nitrogen in cheatgrass (*Bromus tectorum*) dominated ecosystems. Plant and Soil. 386: 47–64.

Keeley, J.E.; Lubin, D.; Fotheringham, C.J. 2003. Fire and grazing impacts on plant diversity and alien plant invasions in the southern Sierra Nevada. Ecological Applications. 13: 1355–1374.

Keller, R.P.; Lodge, D.M.; Finnoff, D.C. 2007. Risk assessment for invasive species produces net bioeconomic benefits. Proceedings of the National Academy of Sciences. 104: 203–207.

Kennedy, A.C.; Johnson, B.N.; Stubbs, T.L. 2001. Host range of a deleterious rhizobacterium for biological control of downy brome. Weed Science. 49: 792–797.

Kesoju, S.R.; Shafii B.; Lass, L.W.; [et al.]. 2015. Predicting rush skeletonweed (*Chondrilla juncea*) dispersal by wind within the canyon grasslands of Central Idaho. International Journal of Plant Biology and Research. 3: 10262001. 1039.

Lass, L.W.; Prather, T.S.; Glenn, N.F.; [et al.]. 2005. A review of remote sensing of invasive weeds and example of the early detection of spotted knapweed (*Centaurea maculosa*) and babysbreath (*Gypsophila paniculata*) with a hyperspectral sensor. Weed Science. 53: 242–251.

Launchbaugh, K.; Walker, J. 2006. Chapter 1. Targeted grazing—A new paradigm for livestock management. In: Launchbaugh, K.; Walker, J., eds. Targeted grazing: A natural approach to vegetation management and landscape enhancement. Englewood, CO: American Sheep Industry Association: 2–8.

Leung, B.; Lodge, D.M.; Finnoff, D.; [et al.]. 2002. An ounce of prevention or a pound of cure: Bioeconomic risk analysis of invasive species. Proceedings of the Royal Society of London. 269: 2407–2413.

Madany, M.H.; West, N.E. 1983. Livestock grazing-fire regime interactions within montane forests of Zion National Park, Utah. Ecology. 64: 661–667.

Maestas, J.D.; Campbell, S.B.; Chambers, J.C.; [et al.]. 2016. Tapping soil survey information for rapid assessment of sagebrush ecosystem resilience and resistance. Rangelands. 38: 120–128.

Mealor, B.A.; Mealor, R.D.; Kelley, W.K.; [et al.]. 2013. Cheatgrass management handbook: Managing an invasive annual grass in the Rocky Mountain Region. Laramie, WY: University of Wyoming; Fort Collins, CO: Colorado State University. 131 p.

Meyer, S.E.; Beckstead, J.; Pearce, J. 2016. Community ecology of fungal pathogens on *Bromus tectorum.* In: Germino, M.J.; Chambers, J.C.; Brown, C.S.; eds. Exotic brome-grasses in arid and semiarid ecosystems of the western U.S. New York, NY: Springer: 193–226.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2014. A field guide to selecting the most appropriate treatments in sagebrush and pinyon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322-rev. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 68 p.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2015. A field guide for rapid assessment of post-wildfire recovery potential in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-338. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 70 p.

Miller, R.F.; Chambers, J.C.; Pyke, D.A.; [et al.]. 2013. A review of fire effects on vegetation and soils in the Great Basin Region: Response and ecological site characteristics. Gen. Tech. Rep. RMRS-GTR-308. Fort Collins, CO: Department of Agriculture, Forest Service, Rocky Mountain Research Station. 126 p.

Moody, M.E.; Mack, R.N. 1988. Controlling the spread of plant invasions: The importance of nascent foci. Journal of Applied Ecology. 25: 1009–1021.

Mosley, J.C. 1996. Prescribed sheep grazing to suppress cheatgrass: A review. Sheep and Goat Research Journal. 12: 74–80.

Mosley, J.C.; Frost, R.A.; Roeder, B.L.; [et al.]. 2016. Combined herbivory by targeted sheep grazing and biological control insects to suppress spotted knapweed (*Centaurea stoebe*). Invasive Plant Science and Management. 9: 22–32.

Peachey, E. 2018. 2018 PNW Weed Management Handbook. Corvallis, OR: Oregon State University Extension. https://catalog.extension.oregonstate.edu/weed. [Accessed May 23, 2015].

Pollnac, F.; Seipel, T.; Repath, C.; [et al.]. 2012. Plant invasion at landscape and local scales along roadways in the mountainous region of the Greater Yellowstone Ecosystem. Biological Invasions. 14: 1753–1763.

Prevey, J.S.; Germino, M.J.; Huntley, N.J. 2010. Loss of foundation species increases population growth of exotic forbs in sagebrush steppe. Ecological Applications. 20: 1890–1902.

Pyke, D.A.; Chambers, J.C.; Beck, J.L.; [et al.]. 2016. Land uses, fire and invasion: Exotic annual *Bromus* and human dimensions. In: Germino, M.J.; Chambers, J.C.; Brown, C.S., eds. Exotic brome-grasses in arid and semiarid ecosystems of the Western US: Causes, consequences and management implications. New York, NY: Springer: 307–337.

Reisner, M.D.; Doescher, P.S.; Pyke, D.A. 2015. Stress-gradient hypothesis explains susceptibility to *Bromus tectorum* invasion and community stability in North America's semi-arid *Artemisia tridentata wyomingensis* ecosystems. Journal of Vegetation Science. 26: 1212–1224.

Reisner, M.D.; Grace, J.B.; Pyke, D.A.; [et al.]. 2013. Conditions favouring *Bromus tectorum* dominance of endangered sagebrush steppe ecosystems. Journal of Applied Ecology. 50: 1039–1049.

Rejmanek, M.; Pitcairn, M.J. 2004. When is eradication of exotic pest plants a realistic goal? In: Veitch, C.R.; Clout, M.N., eds. Turning the tide: The eradication of invasive species. Gland, Switzerland and Cambridge, UK: IUCN SSC Invasive Species Specialist Group: 249–253.

Rew, L.J; Pokorny, M.L. 2006. Inventory and survey methods for nonindigenous plant species. Missoula, MT: Montana State University Extension. 78 p.

Rinella, M.J.; Maxwell, B.D.; Fay, P.K.; [et al.]. 2009. Control effort exacerbates invasive-species problem. Ecological Applications. 19: 155–162.

Romme, W.H.; Allen, C.D.; Bailey, J.D.; [et al.]. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in pinon-juniper vegetation in the western United States. Rangeland Ecology and Management. 62: 203–222.

Roundy, B.A.; Young, K.; Cline, N.; [et al.]. 2014. Piñon-juniper reduction increases soil water availability of the resource growth pool. Rangeland Ecology and Management. 67: 495–505.

Salo, L.F. 2005. Red brome (*Bromus rubens* subsp. *madritensis*) in North America: Possible modes for early introductions, subsequent spread. Biological Invasions. 7: 165–180.

Seabloom, E.W.; Harpole, W.S.; Reichman, O.J.; [et al.]. 2003. Invasion, competitive dominance, and resource use by exotic and native California grassland species. Proceedings of the National Academy of Sciences of the United States of America. 100: 13384–13389.

Shinneman, D.J.; Baker, W.J. 2009. Environmental and climatic variables as potential drivers of post-fire cover of cheatgrass (*Bromus tectorum*) in seeded and unseeded semiarid ecosystems. International Journal of Wildland Fire. 18: 191–202.

Soulé, P.T.; Knapp, P.A.; Grissino-Mayer, H.D. 2004. Human agency, environmental drivers, and western juniper establishment during the late Holocene. Ecological Applications. 14: 96–112.

Stromberg, M.R.; Griffin, J.R. 1996. Long-term patterns in coastal California grasslands in relation to cultivation, gophers, and grazing. Ecological Applications. 6: 1189–1211.

Stylinski, C.D.; Allen, E.B. 1999. Lack of native species recovery following severe exotic disturbance in southern Californian shrublands. Journal of Applied Ecology. 36: 544–554.

Trombulak, S.C.; Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology. 14: 18–30.

U.S. Department of the Interior [USDOI]. 2016. Safeguarding America's lands and waters from invasive species: A national framework for early detection and rapid response. Washington, DC: U.S. Department of the Interior. 55 p. https://www.doi.gov/sites/doi.gov/files/National%20EDRR%20Framework.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS] 2018. Data catalog. An invasive plant inventory and early detection prioritization tool. Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service. https://catalog.data.gov/dataset/an-invasive-plant-inventory-and-early-detection-prioritization-tool. [Accessed Sept. 15, 2018].

Walker, J. 2006. Chapter 12. Targeted grazing to manage fire risk. In: Launchbaugh, K.; Walker, J., eds. Targeted grazing: A natural approach to vegetation management and landscape enhancement. Englewood, CO: American Sheep Industry Association: 107–113.

Willard, E.E.; Bedunah, D.J.; Marcum, C.L.; [et al.]. 1988. Environmental factors affecting spotted knapweed. Biennial Report 1987–1988. Missoula, MT: University of Montana, School of Forestry, Montana Forest and Conservation Experiment Station. 21 p.


Lomatium dissectum

6. APPLICATION OF NATIONAL SEED STRATEGY CONCEPTS

Fred Edwards, Sarah M. Kulpa, and Francis F. Kilkenny

Introduction

Native plant species are the foundation of sagebrush ecosystems and provide essential habitat for wildlife species, such as Greater sage-grouse (*Centrocercus urophasianus*; hereafter, GRSG). The National Seed Strategy for Rehabilitation and Restoration (hereafter, Seed Strategy) (PCA 2015) strives to provide all land managers—Federal, tribal, State, county, private, and nongovernmental organization—the tools they need to address ecological restoration across the United States. The Seed Strategy provides a coordinated approach to improving the use of native seed, building Federal and private capacity, and increasing the supply of genetically appropriate native seed (PCA 2015). The Seed Strategy recognizes the value of existing native plants and soil seedbanks and acknowledges that not all disturbances or management treatments require active seeding to restore habitat. The Seed Strategy also recognizes that although many nonnative species have been seeded successfully and economically to provide forage and soil stabilization, their ability to support diversity and provide functioning ecosystems to meet multiple use and sustained yield mandates is limited (PCA 2015). Successful rehabilitation and restoration must always take into consideration compatibility of species in a seed mix, planting season, and appropriate seeding rates, techniques, technologies, and practices; that information is available elsewhere (e.g., Madsen et al. 2012, 2014; Monsen et al. 2004a,b,c; Ott et al. 2016; Pyke et al. 2015a,b, 2017).

Genetically appropriate native plant materials have been historically underdeveloped within the sagebrush biome. This section focuses on the logistics, challenges, opportunities, and considerations for procuring and using native seed in sagebrush ecosystems at broad (sagebrush biome), mid- (level III ecoregions), and local (project to site) scales. It also discusses local scale tradeoffs that should be considered when managers decide to use nonnative seeds within the sagebrush biome. It does not address restoration practices and techniques.

Top left: Bee on a native fiddleneck flower in Nevada (photo: USDOI Fish and Wildlife Service). Top center: Intern collecting Indian ricegrass seed (photo: Sophia Heston, USDOI Fish and Wildlife Service). Top right: Owen Baughman and Lauren Porensky preparing to fill the drill seeder (photo: Beth Leger, University of Nevada, Reno). Middle left: Native forbs in a seed increase field (photo by Anne Halford, USDOI Bureau of Land Management). Middle center: Sagebrush seedlings being grown for bare root stock at USDA Forest Service, Lucky Peak Nursery (photo: USDA Forest Service). Middle right: Western hawksbeard (photo: USDA Forest Service). Bottom left: Drill seeding in the snow (photo: Susan Fritts, USDOI Bureau of Land Management). Bottom right: Successful postfire seeding in a sagebrush ecosystem (photo: USDA Forest Service).

Conceptual Basis

Most gardeners and growers are familiar with the 2012 USDA Plant Hardiness Zone map (https://planthardiness.ars.usda.gov/PHZMWeb/) that is found on the back of almost every seed pack sold in the United States. This is the standard by which gardeners and growers can determine which plants are most likely to thrive at a location based on average annual minimum winter temperature, divided into 10-degree Fahrenheit zones. In this context, seed transfer guidelines, which include mapped seed zones (fig. 6.1), are just a more sophisticated and accurate way to understand what seeds and plants thrive best at a location. Seed transfer guidelines are management tools that define acceptable distances seed can be moved from the point of origin, while considering genetic adaptation (Bower et al. 2014; Kilkenny 2015; St. Clair et al. 2013). For more detail, see Part 1, Appendix 11 of the Science Framework (Chambers et al. 2017; hereafter, Part 1).

Variations in biotic and abiotic factors cause plants to experience natural selection across their range. When adaptive evolution occurs in response to local selective pressures, populations are considered to be locally adapted (Leimu and Fischer 2008; McKay et al. 2005). Common garden studies and reciprocal transplant studies have shown that plant populations are often adapted to local environmental conditions (e.g., Clausen et al. 1941; Hiesey et al. 1942; Joshi et al. 2001; Turesson 1922). For restoration projects, this means locally adapted plants can generally outperform nonlocal plants (e.g., Bischoff et al. 2006; Humphrey and Schupp 2002; Leimu and Fischer 2008; Rice and Knapp 2008; Rowe and Leger 2012).

Ecosystem resilience to disturbance and resistance to invasive annual grasses can be increased by considering both seed source and genetic diversity, in combination with other factors, when selecting seeds and plant materials. Besides project failure, poor seed mix choices may have long-term consequences including genetic degradation of the surrounding plant population, loss of fitness, and loss of evolutionary potential and, consequently, reduction of future

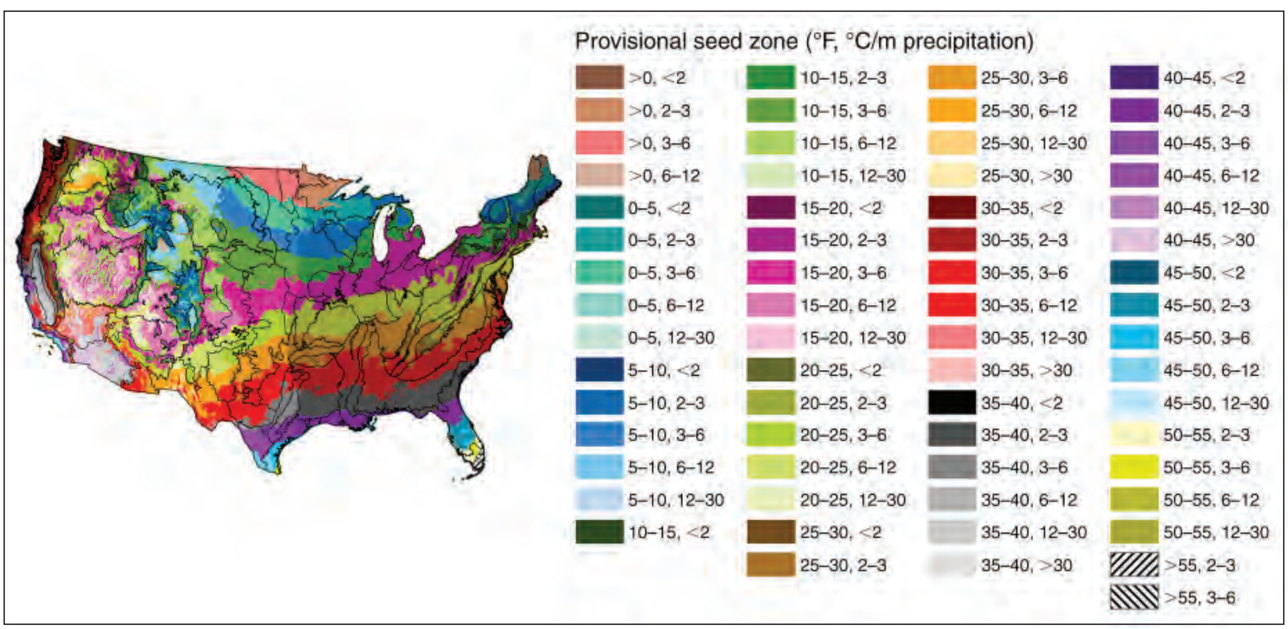

Figure 6.1—Provisional seed zones for native plants (color polygons) overlain with Omernik's (1987) level III ecoregion boundaries (black lines). Provisional seed zones are the first step in defining seed transfer guidelines. Level III ecoregions can be used to refine seed movement within a provisional seed zone. In the legend, the first range of numbers is the temperature class band (°F) and the second range of numbers is the annual heat:moisture (AH:M) index class bands (°C/m precipitation; from Bower et al. 2014) (Chambers et al. 2017, Appendix 11 fig. A.11.2).

plant community resilience and resistance to invasive annual grasses (Crémieux et al. 2010; McKay et al. 2005; Mijnsbruggea et al. 2010; Schröder and Prasse 2013). The Seed Strategy provides a path forward for developing and procuring genetically appropriate native seed sources that have the best genetic fit for individual restoration and vegetation management projects by identifying the research, technology, and monitoring needs for integrating and managing genetic diversity across the sagebrush biome.

Considerations for Enhancing Resilience and Resistance Using Seed Strategy Concepts

Broad- to Mid-Scale Considerations

Prioritizing Native Seed Development

The geospatial data layers and analyses described in Part 1, sections 8.1 and 8.2 of the Science Framework can help prioritize sagebrush ecosystems for native plant materials development, postfire rehabilitation, and restoration. Analyses are conducted at the ecoregion scale because similarities in ecoregional climate, soil properties, resilience to disturbance, and resistance to invasive annual grasses can provide economies of scale compatible with seed development. Collectively, the sagebrush biome includes most of 14 different Omernik (1987) level III ecoregions: Eastern Cascades Slopes and Foothills, Columbia Plateau, Blue Mountains, Idaho Batholith, Snake River Plain, Northern Basin and Range, Central Basin and Range, Wasatch and Uinta Mountains, Middle Rockies, Wyoming Basin, Colorado Plateaus, Southern Rockies, Northwestern Great Plains, and Northwestern Glaciated Plains. Omerick's level III ecoregions served as the basis for the U.S. Environmental Protection Agency (EPA) level III ecoregions described in Part 1 and are synonymous with EPA level III ecoregions (fig. 1.1). For example, warmer and drier areas with low resilience and resistance to invasive annual grasses might require additional seeding after a disturbance to supplement natural recovery. Therefore, ecoregions with predominantly warm and dry soil temperature and moisture regimes, such as the Columbia Plateau, Northern Basin and Range, Central Basin and Range, Snake River Plain, and Colorado Plateaus, may be a higher priority for the development of native plant materials.

Key data layers for prioritizing areas for native plant materials development include: (1) resilience and resistance to invasive annual grasses as indicated by soil temperature and moisture regimes, (2) GRSG breeding habitat probabilities and densities or habitats of other sagebrush obligate habitats, (3) the primary threats for the ecoregion (see Part 1, section 8), and (4) generalized or provisional seed zones (fig. 6.1) (Bower et al. 2014; Part 1, Appendix 11). For example, in the Great Basin, Jensen and Stettler (2012) reported that over the last 30 years, 90 percent of fire rehabilitation projects on Federal land occurred in three major generalized or provisional seed zones. In the eastern range 78 percent of oil and gas development occurs in six major generalized or provisional seed zones (see Part 1, Appendix 8 for data sources). Thus, initial seed development efforts should focus on developing native plant materials for the most appropriate species (most likely native perennial grasses) for these provisional seed zones.

Primary considerations in prioritizing areas for native plant materials development based on resilience and resistance to invasive annual grasses follow (see tables 1.3 and 1.4, especially the sections on postfire rehabilitation and climate change).

- In general, areas with moderate and, especially, high resilience and resistance often recover without seeding following wildfire and vegetation management. Shrubs, particularly sagebrush, may or may not require seeding or transplanting. These areas are relatively low priority for development of native plant materials (table 1.3: cells 1B, 1C, 2B, 2C).
- Priority increases as resilience and resistance decrease and habitat probability for GRSG increases. High priorities include ecological types with low to moderate resilience and resistance that (1) may lack sufficient native perennial grasses and forbs to recover on their own, but (2) have nearby areas still supporting GRSG habitat (table 1.3: cells 2B, 2C, 3B, 3C).
- Areas of low habitat probability for GRSG (table 1.3: cells 1A, 2A, 3A) are generally lower priority, but may become higher priority if they support other species or resources at risk or can be used to increase connectivity among areas with intact sagebrush.
- Areas may be considered for prioritization regardless of resilience and resistance if repeated large fires or other habitat disturbances are causing habitat fragmentation and seeding or transplanting of sagebrush is needed to maintain habitat connectivity.

Because resilience and resistance to invasive annual grasses increase along soil temperature and moisture gradients, an understanding of the relationship of major sagebrush taxa to soil temperature and moisture regimes can help in prioritizing sagebrush and their associated species for seed development by using seed zones and seed transfer guidelines. Within the big sagebrush complex in the western portion of the range, mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) occurs on cold to cool moist sites, while in the eastern portion of the range it occurs on cold and cool wet, summer moist, or winter moist sites. In the western portion of the range, Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis*) and basin big sagebrush (*A. tridentata* ssp. *tridentata*) typically occur on relatively warm and dry sites, whereas in the eastern portion of the range, these species occur on a spectrum of sites, ranging from cool and summer moist to warm and dry. Thus, Wyoming big sagebrush and basin big sagebrush may be considered a higher priority for native plant materials development in the western portion of the range based on low resilience and resistance to invasive annual grasses on the sites where they grow.

Some dwarf sagebrush species, such as warm springs low sagebrush (*Artemisia arbuscula* ssp. *thermopola*), alkali sagebrush (*A. longiloba*), and Wyoming threetip sagebrush (*A. tripartita* ssp. *rupicola*) occur on relatively cold to cool sites with high resistance and resilience to invasive annual grasses (Miller et al. 2014) and, therefore, are a lower priority for native plant materials development and restoration. However, other Dwarf sagebrush species—black sagebrush (*A. nova*), pygmy sagebrush (*A. pygmaea*), low sagebrush (*A. arbuscula* ssp. a*rbuscula*), and alkali sagebrush (*A. arbuscula* ssp. *longiloba*)—grow on relatively warm and dry sites (Miller et al. 2014). Although this appears to indicate that the ecosystems where these species are most abundant have low resilience and resistance to invasive annual grasses, soil and vegetation community characteristics need to be taken into account. For example, black sagebrush grows on shallow, stony, calcareous soils which are sparsely vegetated, and thus has a low fuel load and low likelihood of needing restoration. Therefore, black sagebrush is typically a lower priority for native plant materials development and restoration. However, monitoring of all sagebrush ecological types is needed to determine whether declines are occurring due to climate, wildfire, improper grazing, disease, or other perturbations.

Developing the Mechanism for Seed Increase

Vegetation community lists from the available ecological site descriptions for Natural Resources Conservation Service (NRCS) Major Land Resource Areas can be used to identify the native shrub, grass, and forb species needed to restore ecosystem function. Development of lists can be prioritized based on resilience and resistance concepts and the considerations just described. Vegetation community lists can also be used to prioritize species for native plant materials development and regional procurement objectives. One caveat is that ecological site descriptions tend to be dominated by later successional species. In some cases earlier successional species may need to be included in a seed mix to help establish initial site resistance to invasive annual grasses. To achieve this, local expertise and herbarium records coupled with ecological site descriptions should be used to develop the most comprehensive vegetation community lists.

Intact sagebrush communities with low and moderate resilience and resistance to invasive annual grasses can be identified for wildland seed collection or the establishment of commercial seed collection areas. These sagebrush communities can provide reliable, source-identified sagebrush seed for restoration projects. Alternatively, where local seed sources have been depleted or are not available for seed collection (such as Wilderness areas), the development of seed orchards based on seed transfer guidelines and seed zones may be useful.

Potential Tradeoffs and Management Challenges at the Broad and Mid-Scale

Changes in precipitation and temperature regimes are projected to have large consequences for species distributions across the sagebrush biome (see Part 1, section 4.2). This is a challenge for management because the vegetation communities we currently manage may or may not be the same in the future. Developing native plant materials that include the genetic diversity of a species by seed zones can help species seeded onto a site adapt to future changes in climate. Predictive models of changes in climate can be used to assess threats to important restoration species and identify opportunities for targeting, prioritizing, and implementing restoration projects that consider potential changes in species distribution and plant community composition. Modeling changes in species distributions and seed zone boundaries will help identify potential refugia areas and bottlenecks to species' movement and select appropriate plant populations for inclusion in restoration projects to reduce the risk of future maladaptation.

At the broad scale, prioritizing ecoregions and sagebrush ecological types within them (for example, Wyoming big sagebrush ecological types in the Columbia Plateau), may mean that seed needed for restoration within areas that have high and moderate resilience and resistance to invasive annual grasses may not always be as readily available as seed for areas with low resilience and resistance to invasive annual grasses. Therefore, when making seeding decisions, it is important not to waste seed, and seed only when necessary. In areas with high resilience and resistance to invasive annual grasses, not seeding or other passive restoration treatments may be more practical (Pyke et al. 2015a). In areas with lower resilience and resistance to invasive annual grasses that require seeding, individual project planning can help mitigate the need for seed. By building reasonable timelines within individual projects, local seed collection and seed increase can be conducted to ensure that sufficient genetically appropriate native seed is available.

Land managers may want to rehabilitate and restore rangelands that have low GRSG habitat value or other resource management value, but are currently dominated by cheatgrass (*Bromus tectorum*), crested wheatgrass (*Agropyron cristatum*), or some other undesirable plant species, because these rangelands are prevalent at the mid-scale. Under these circumstances, where other range management objectives have a higher priority than GRSG management objectives, the financial costs to procure genetically appropriate native seed, the size and scale of the project, or adverse impacts to remaining local native seed sources (e.g., improper grazing) may preclude the use of native seed. Nonnative species and native cultivars that originate from sites with similar temperature and precipitation regimes may provide an acceptable management tradeoff. However, if native ecosystem restoration is the goal, seed of genetically appropriate native grasses is relatively inexpensive and can be the first step of a "staged planting" approach, whereby grasses and forbs are planted in successive years or forbs are added to a limited number of favorable areas (i.e., forb islands) (Benson et al. 2011) (see section on local-scale tradeoff).

Local Scale Considerations

In this section, local scale refers to individually funded vegetation management activities within a district or field office. At this scale, managers need to carefully consider seed mixes and seed sources because of the critical role they play in managing for resilience and resistance to invasive annual grasses. The importance of deciding when seeding is or is not needed cannot be overstated. Such decisions should be tied to site-specific assessments of current conditions, past management, and the potential for a site to recover without management intervention. Local monitoring data can be used to provide information on seed mixes and seed sources, as well as the need to seed (text box 6.1). Monitoring treatment effectiveness can provide the necessary information on species performance to adjust seed sources over time.

Planning for and initiating collection, seed increase, and long-term storage of native seed are important components of the management and development of native plant materials. Forward planning for the use of genetically appropriate

Text Box 6.1—Monitoring to Inform Selection of Species and Seed Sources and to Evaluate Seed Source Performance

Monitoring data play an important role in selecting species and seed sources and evaluating species performance. First, monitoring data from a project area or site prior to treatment can provide the necessary information on the species composition to select the most appropriate restoration species. Such data can also provide information on suitable areas for seed collection. Second, information about the seed source is essential for selecting plant materials that are genetically adapted to the site conditions. Selecting appropriate seed sources can ensure that the desired species establish and persist and is necessary for achieving successful and effective restoration projects.

Information on the seed sources used in a restoration project should be recorded and tracked in a systematic manner. Relating data on seed sources to seedling establishment as a part of effectiveness monitoring provides critical information on species and seed source performance that can be used to inform future restoration efforts. When only anecdotal data are available, project managers can draw or perpetuate erroneous conclusions regarding the effectiveness of seeding outcomes. Data on seed sources, along with other treatment information, could be recorded in the Land Treatment Digital Library, a catalog of information about land treatments on Federal lands in the western United States (https://ltdl.wr.usgs.gov/). This need is identified and described in the Seed Strategy under Action Item 2.4.1, "Analyze new and existing monitoring methodologies to evaluate restoration outcomes" (PCA 2015).

native seed based on quantities requested annually, number of acres seeded annually, fire projections, or some other metric is critical. Forward planning when seeding with cultivars or nonnatives is generally not crucial because of their widespread availability.

For the western range, Miller et al. (2014, 2015) provide a framework for evaluating postwildfire resilience and resistance to invasive annual grasses, potential successional pathways, and the need to seed at the local scale. A similar framework can be developed for the eastern range. Additionally, the Seedlot Selection Tool (https://seedlotselectiontool.org/sst/) can help with seed source decisionmaking based on climate information. General seeding strategies by resilience and resistance category are:

- **High Resilience and Resistance**. The potential for native shrubs, grasses, and forbs to recover after disturbance without seeding is typically high. Shrubs, particularly sagebrush, may or may not require seeding or transplanting. If sites require seeding, the use of locally sourced or source-identified seed from the same seed zone will improve project success while maintaining genetic adaptation and diversity.
- **Moderate Resilience and Resistance**. The potential for native shrubs, grasses, and forbs to recover after disturbance is usually moderately high, especially on cooler and moister sites. Seeding following disturbance or treatment may be needed in areas with depleted native perennial grasses and forbs. Including perennial grasses in seed mixes that can compete with and provide resistance to invasive annual grasses is recommended. Including locally sourced or source-identified forbs from the same seed zone may be necessary to meet habitat management objectives, but their seeding depends on the degree of site preparation, capabilities of the seeding equipment, and expectation of weed invasion.
- **Low Resilience and Resistance**. Recovery potential after overlapping disturbances (e.g., wildfire, improper grazing) is usually low and seeding is needed in areas with depleted native shrubs, grasses, and forbs. The use of perennial grasses in seed mixes is recommended to provide competition with invasive annual grasses. Decisions on the use of native (locally sourced or source identified from the same seed zone), grasses, native cultivars, or nonnative grasses depends on the availability of seed sources and degree of invasion by nonnative annual grasses. On degraded sites, forbs may be absent. Including locally sourced or source-identified forbs from the same seed zones may be necessary to meet habitat management objectives. However, to successfully seed forbs it is necessary to consider the degree of site preparation, capabilities of the seeding equipment, and expectation of weed invasion.

Good species selections and seed source choices can strengthen community resilience and resistance to invasive annual grasses, whereas poor species selections and seed source decisions can erode long-term community resilience and resistance. Management considerations for resilience and resistance at the local scale include:

- **Incorporating native perennial grasses in all seed mixes used on sites with moderate and low resilience and resistance to invasive annual grasses**. Native perennial grasses compete directly with cheatgrass and other introduced annual grasses for space, water, and nutrients (Blank and Morgan 2012; Chambers et al. 2007; Leger 2008). Including genetically appropriate native perennial grasses adapted to site-specific temperature and precipitation regimes increases resilience and resistance as well as site diversity. Empirical seed zones are available for many of the common native perennial grasses used in rehabilitation and restoration of sagebrush ecosystems.

- **Designing a diverse seed mix of native shrubs, grasses, and forbs for all project seed mixes.** Species diversity is the hallmark of a healthy ecosystem; diverse seed mixes of native shrubs, grasses, and forbs can increase site resistance by filling ecological niches and competing with nonnative invasive annual grasses. Seed mixes should integrate information about ecological site conditions and successional stage for best success. For example, if forbs are included in a seed mix, site preparation and management should prevent cheatgrass invasion, such as through the "staged planting" approach (Benson et al. 2011). Temperature and precipitation conditions that favor seed germination and seedling establishment vary from year to year, so seeding a diverse mix of early and late successional stage native shrubs, grasses, and forbs may increase resilience by providing a range of species capable of germinating and establishing in response to a variety of environmental conditions.
- **Using the right sagebrush in the right place.** With 27 sagebrush species and subspecies across the sagebrush biome, using the correct sagebrush species or subspecies source identified to the same seed zone in restoration projects is essential to creating sagebrush communities that are resilient and resistant to invasive annual grasses. Variations in biotic and abiotic factors cause plants to undergo natural selection and adaptive evolution; thus, individual sagebrush species and subspecies have evolved to grow best under different soil environments, temperature, and precipitation regimes (Dumroese et al. 2015; Miller et al. 2011). Consequently, sagebrush species and subspecies are not interchangeable in a restoration seed mix. For example, Richardson et al. (2015) found that Wyoming big sagebrush has a significantly greater seed weight than basin big sagebrush and determined that 83 percent of certified seed lots used in 2013 and 2014 were labeled as Wyoming big sagebrush but were actually basin big sagebrush. Furthermore, data indicate that local adaptation in sagebrush plays an important role in long-term survivorship. In an Idaho Department of Fish and Game study, Sands and Moser (2012) found locally sourced Wyoming sagebrush seed had 100 percent survivorship after 20 years, while non-locally sourced seed had less than 50 percent survivorship.
- **Including native forbs to create healthier food webs.** Complex and diverse food webs are a hallmark of intact ecosystems with high resilience and resistance to invasive annual grasses. Native forbs are a major component of sage-grouse chick diets (Dumroese et al. 2015), are critical to native pollinators (Pollinator Health Task Force 2015), and can be abundant in sagebrush communities (Humphrey and Schupp 2001; James et al. 2014). In healthy sagebrush ecosystems, native forbs have continuous and overlapping flowering and seed production throughout the growing season—meaning that a variety of ecological niches are filled by a diversity of species. On degraded sites, land managers can attempt to create or repair flowering phenology and reproduction through carefully planned seed mixes. Restoring the native plant community, especially the native forb component, is likely to result in a cascading response in which other native species increase. Thus, native forbs are an important component of sagebrush ecosystem restoration and should be included in seed mixes.

- **Considering use of ruderal or annual native forbs in project seed mixes to increase resistance to cheatgrass where they are naturally abundant and seed sources have been developed.** Some native annual species (such as bristly fiddleneck [*Amsinckia tesselata*]) have been shown to compete well and suppress nonnative invasive annual species due to phenological similarities (Leger et al. 2014; Uselman et al. 2014). Developing competitive, native annual species for use in future seed mixes may improve seeding outcomes in some disturbed rangeland ecosystems. However, the potential amount of seed required, availability, and costs of including native annuals should be carefully considered during project planning.
- **Considering long-term planning at the local scale to preserve seed sources from low resilience and resistance sites that are at high risk of cheatgrass invasion or wildfire.** In these cases, long-term planning can provide seed sources adapted at the seed zone level which will be adapted to site conditions within a seed zone.

Potential Tradeoffs and Management Challenges at the Local Scale

If a decision is made to seed, there are five major tradeoffs related to resilience and resistance concepts and implementation of Seed Strategy concepts. Tradeoffs should not be considered individually, but rather in the context of meeting project objectives while maintaining site resilience and resistance to invasive annual grasses. Local tradeoffs in the context of seed source choices (fig. 6.2) are discussed briefly.

The Tradeoff between Seed Source and the Need for Follow-up Management to Meet GRSG Habitat Objectives. Nonnative species, such as crested wheatgrass and forage kochia (*Bassia substrata*), are widely seeded for rangeland revegetation, postfire rehabilitation, invasive plant control, and green stripping, because they germinate and establish quickly, are readily available for purchase, are cheaper than native species, provide good livestock forage, and compete with nonnative invasive species (Brooks and Pyke 2001; Harrison et al. 2000; Monaco et al. 2003; Pellant 1994; Richards et al. 1998). These nonnative species are used as placeholder or bridge species to convert annual invasive grass-dominated rangelands into native perennial-dominated plant communities (Monaco et al. 2003); however, follow-up restoration rarely happens. Putting this concept into practice has not been widely realized and some of the positively perceived attributes of these species, such as competitive ability, can negatively impact native plant community structure and function.

The wide use of nonnative species in some circumstances represents a tradeoff for achieving diverse ecosystem and habitat management objectives for GRSG, pollinators, and other sagebrush dependent species. For example, crested wheatgrass can be highly competitive with native sagebrush and perennial grass species (Asay et al. 2001; Bakker and Wilson 2001; Hull and Klomp 1967; Marlette and Anderson 1986). Crested wheatgrass can dominate the soil seedbank (Marlette and Anderson 1986) and limit the growth and establishment of native plants (Gunnell et al. 2010; Heidinga and Wilson 2002; Hendersen and Naeth 2005). Attempts to reintroduce native species into crested wheatgrass monocultures suggest that costly and time-intensive repeated treatments are required because this species recovers rapidly from mechanical and chemical control treatments (Davies et al. 2013; Fansler and Mangold 2011; Hulet et al. 2010; McAdoo et al. 2016). Short and long-term (13 years) studies suggest that even if seeded at low rates in a seed mix, crested wheatgrass may ultimately

Project Seed Options

	Will follow-up management be needed to meet sage-grouse habitat objectives?	Could there be negative affects to the adjacent plant community?	Will established plants reproduce?	Will seed establish?	Can seed be procured quickly?
Locally sourced seed	High certainty additional management not needed	High certainty there will be no impacts	High certainty will reproduce	Moderate certainty will establish	Requires advance planning and dedicated funding
Source identified to the same seed zone	High certainty additional management not needed	Moderate certainty there will be no impacts	Moderate certainty will reproduce	Moderate certainty will establish	Seed availability unknown; depends on species and project seed zone
Native cultivated commercial variety	Moderate certainty additional management not needed	Low or unknown certainty due to potential genetic dilution or hybridization of local populations	Reproduction depends on germplasm origin and climactic similarities to target site	Establishment depends on germplasm origin and climactic similarities to target site	High certainty seed available
Persistent nonnative species like crested wheatgrass or forage kochia	Additional management needed to diversify the plant community and meet habitat objectives	Low or unknown certainty because can potentially spread beyond the project area	High certainty will reproduce but has potential to form a monoculture	High certainty will establish	High certainty seed available
Non-persistent, nonnative place holder species (such as sterile wheatgrass)	Additional management needed to diversify the plant community and meet habitat objectives	Moderate certainty there will be no impacts	Not expected to reproduce	Moderate certainty will establish	High certainty seed available

High certainty
Moderate certainty
Low or unknown certainty
Not applicable

Local Level Implementation Considerations

Figure 6.2—Seed source and local-level considerations for selecting seed sources and types.

become the most abundant grass in a mixed bunchgrass community (Bakker and Wilson 2004; Nafus et al. 2015).

The Tradeoff between Seed Source and Potential Impacts to the Adjacent Plant Community. Plants established as part of a seeding project interact with the surrounding environment and interbreed with native, resident (local) plant populations. Local seeds or seed sources identified by seed zone are advantageous because they are unlikely to be invasive or overly competitive with other native plants. Local seeds or seed sources identified by seed zone should be most genetically similar to the existing native plant populations and have the lowest potential for adverse genetic impacts.

Seeding with nonnatives may represent an ecological tradeoff because they have the potential to invade and spread beyond a project boundary. For example, Gray and Muir (2013) found that on sites seeded 3 to 24 years earlier, forage kochia spread as much as 710 meters (2,330 feet) into both intact and disturbed plant communities for an estimated rate of 25 meters (82 feet) per year.

Just as individual plants may spread, genes are also capable of spreading into adjacent, resident plant populations. Seeding with native cultivars may represent a genetic tradeoff because of potential adverse impacts to local population genetics through hybridization, potentially affecting overall species fitness

(Hereford 2009; Leimu and Fischer 2008). Seed source is often not a criterion for developing native cultivars. Native cultivars have been developed over many years in an agronomic setting and are often selected for specific traits (see next paragraph), which may or may not align with restoration success (Johnson et al. 2010; Jones and Larson 2005; Leger and Baughman 2015). Introduced seed has the potential to hybridize with native populations and result in maladaptation or negative long-term impacts that could affect a plant community's ability to adapt to changing environmental conditions.

The Tradeoff between Seed Sources and Seed Germination, Establishment, and Reproduction. Traits selected for and often prioritized in native cultivars are: forage quality and yield, seed yield, seedling vigor, ability to establish and persist, and drought tolerance across a range of environmental conditions (Leger and Baughman 2015). Nonnative species are selected for traits similar to those selected in native cultivars. For example, the crested wheatgrass germplasm 'Ephraim' was selected for forage quality and yield, ability to establish, and a rhizomatous growth form for site stabilization (USDA NRCS 2012). In contrast, locally sourced native seeds and seed sources are more likely to be adapted to the environmental conditions in the seed zones where they are collected.

Locally sourced, native seed may need one or more growing seasons to germinate and establish on a site due to seed dormancy or other physiologic mechanisms. Seed of nonnatives and native cultivars typically germinate and establish quickly because they are selected for little or no seed dormancy. However, this represents a tradeoff because nonnatives and native cultivars may not meet long-term habitat objectives for sage-grouse, pollinators, other wildlife species, or special status plant species. Additionally, using a nonnative species like crested wheatgrass will support site resistance to invasive annual grasses because it is a good competitor with cheatgrass. However, it is less likely to support long-term site resilience because of the low species diversity it maintains (see preceding discussion). Treatment effectiveness monitoring that tracks native seed sources and their performance in the field can be used to inform both native species and seed source selection (text box 6.1).

The Tradeoff between Seed Sources and Procurement. Until the seed market can be fully developed, there is a tradeoff between the species desired for a seed mix and their availability. Anticipating and planning for native species needed to develop a seed mix is an important aspect of project management because seed of desired native plant species and seed sources usually are not immediately available. At the local scale, it is possible to plan and collect local seed that can be sent to a grower to increase it to the desired quantities. Advance planning (such as performing project-specific seed collections and seed increase with a commercial grower) will make species more available, but this represents a tradeoff in how quickly a project can be implemented. Purchasing and using native cultivars or nonnative species is a tradeoff that saves time and money, allowing a project to move forward quickly. Native cultivars (such as 'Sherman' Sandberg bluegrass *[Poa secunda*] or 'Magnar' basin wildrye [*Leymus cinereus*]) or nonnative species (such as crested wheatgrass and forage kochia) are often immediately available and can be bought from the commercial market in large quantities. However, using native cultivars or nonnative species results in tradeoffs regarding potential adverse impacts to future resilience and resistance to invasive annual grasses and a need for follow-up management (see earlier discussion).

Conclusions

Balancing locally adapted seed sources, cultivars, and nonnative species against the realities of implementing a project in the field is a series of tradeoffs. Every project is unique and a one-size-fits-all approach will not work. Sometimes seeding is used as a way to mitigate management risk or simply as insurance. Regardless of why and what is being seeded, the judicious use of seed will not only save money, but also minimize the risk of unintended ecological consequences to naturally recovering native plant communities. As part of any decision to seed, potential tradeoffs should be carefully weighed against the potential future economic and ecosystem costs. Seeding should not always be the first choice. For example, where prescriptive treatments are desired to minimize erosion risks to infrastructure, one-time physical barriers (such as straw wattles and straw mulch) may be more desirable and cost-effective where sufficient native perennial plants exist to promote recovery (e.g., Robichaud et al. 2010).

References

Asay, K.H.; Hoeron, W.H.; Jensen, K.B.; [et al.]. 2001. Merits of native and introduced Triticeae grasses on semiarid rangelands. Canadian Journal of Plant Science. 81: 45–52.

Bakker, J.D.; Wilson, S.D. 2001. Competitive abilities of introduced and native grasses. Plant Ecology. 157: 117–125.

Bakker, J.D.; Wilson, S.D. 2004. Using ecological restoration to constrain biological invasion. Journal of Applied Ecology. 41: 1058–1064.

Benson, J.E.; Tveten, R.T.; Asher, M.G.; [et al.]. 2011. Shrub-steppe and grassland restoration manual for the Columbia River Basin. Olympia, WA: Washington Department of Fish and Wildlife. https://wdfw.wa.gov/publications/01330/. [Accessed May 23, 2018].

Bischoff, A.; Vonlanthen, B.; Steinger, T.; [et al.]. 2006. Seed provenance matters—Effects on germination of four plant species used for ecological restoration. Basic and Applied Ecology. 7: 347–359.

Blank, R.S.; Morgan, T. 2012. Suppression of *Bromus tectorum* L. by established perennial grasses: Potential mechanisms—Part One. Applied Environmental Soil Science. 2012: Article 632172.

Bower, A.D.; St. Clair, J.B.; Erickson, V. 2014. Generalized provisional seed zones for native plants. Ecological Applications. 24: 913–919.

Brooks, M.L.; Pyke, D.A. 2001. Invasive plants and fire in the deserts of North America. In: Galley, K.E.M; Wilson, T.P., eds. Proceedings of the invasive species workshop: The role of fire in the control and spread of invasive species. Miscell. Publ. No. 11. Tallahassee, FL: Tall Timbers Research Station: 1–14.

Chambers, J.C.; Roundy, B.A.; Blank, R.R.; [et al.]. 2007. What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? Ecological Monographs. 77: 117–145.

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications.

Gen. Tech. Rep. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p.

Clausen, J.; Keck, D.D.; Hiesey, W.M. 1941. Regional differentiation in plant species. American Naturalist. 75: 231–250.

Crémieux, L.; Bischoff, A.; Müller-Schärer, H.; [et al.]. 2010. Gene flow from foreign provenances into local plant populations: Fitness consequences and implications for biodiversity restoration. American Journal of Botany. 97: 94–100.

Davies, K.W.; Boyd, C.S.; Nafus, A.M. 2013. Restoring the sagebrush component in crested wheatgrass-dominated communities. Rangeland Ecology and Management. 66: 472–478.

Dumroese, R.K.; Luna T.; Richardson, B.A.; [et al.]. 2015. Conserving and restoring habitat for Greater sage-grouse and other sagebrush obligate wildlife: The crucial link of forbs and sagebrush diversity. Native Plants Journal. 16: 276–299.

Fansler, V.A.; Mangold, J.M. 2011. Restoring native plants to crested wheatgrass stands. Restoration Ecology. 19: 16–23.

Gray, E.C.; Muir, P.S. 2013. Does *Kochia prostrata* spread from seeded sites? An evaluation from southwestern Idaho, USA. Rangeland Ecology and Management. 66: 191–203.

Gunnell, K.T.; Monaco, T.A.; Call, C.A.; [et al.]. 2010. Seedling interference and niche differentiation between crested wheatgrass and contrasting native Great Basin species. Rangeland Ecology and Management. 63: 443–449.

Harrison, R.D.; Chatterton, N.J.; Waldron, B.L.; [et al.]. 2000. Forage kochia: Its compatibility and potential aggressiveness on intermountain rangelands. Utah Agricultural Experiment Station Research Report 162. Logan, UT: Utah State University. 66 p.

Heidinga, L.; Wilson, S.D. 2002. The impact of invading alien grass (*Agropyron cristatum*) on species turnover in native prairie. Diversity and Distributions. 8: 249–258.

Henderson, D.C.; Naeth, M.A. 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. Biological Invasions. 7: 639–650.

Hereford, J. 2009. A quantitative survey of local adaptation and fitness trade-offs. American Naturalist. 173: 579–588.

Hiesey, W.M.; Clausen, J.; Keck, D.D. 1942. Relations between climate and intraspecific variation in plants. American Naturalist. 76: 5–22.

Hulet, A.; Roundy, B.A.; Jessop, B. 2010. Crested wheatgrass control and native plant establishment in Utah. Rangeland Ecology and Management. 63: 450–460.

Hull, A.C.; Klomp, G.J. 1967. Thickening and spread of crested wheatgrass stands on southern Idaho ranges. Journal of Range Management. 20: 222–227.

Humphrey, L.D.; Schupp, E.W. 2001. Seed banks of *Bromus tectorum*-dominated communities in the Great Basin. Western North American Naturalist. 61: 85–92.

Humphrey, L.D.; Schupp, E.W. 2002. Seedling survival from locally and commercially obtained seeds on two semiarid sites. Restoration Ecology. 10: 88–95.

James, D.G.; Seymour, L.; Lauby, G.; [et al.]. 2014. Beneficial insects attracted to native flowering buckwheats (*Eriogonum* Michx) in central Washington. Environmental Entomology. 43: 942–948.

Jensen, S.; Stettler, J. 2012. Applying provisional seed zones to plant materials development in the Great Basin and cultural practice notes. Presentation to the Great Basin Native Plant Project. http://www.fs.fed.us/rm/boise/research/shrub/projects/PowerPoint_Presentations/2012/Jensen.pdf. [Accessed May 23, 2018].

Johnson, R.; Stritch, L.; Olwell, P.; [et al.]. 2010. What are the best seed sources for ecosystem restoration on BLM and USFS lands? Native Plants Journal. 11: 117–131.

Jones, T.A.; Larson, S.R. 2005. Status and use of important native grasses adapted to sagebrush communities. In: Shaw, N.L.; Pellant, M.; Monsen, S.B., eds. Sage-grouse habitat restoration symposium proceedings; 2001 June 4–7; Boise, ID. Proceedings RMRS-P-38. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 49–55.

Joshi, J.; Schmid, B.; Caldeira, M.; [et al.]. 2001. Local adaptation enhances performance of common plant species. Ecology Letters. 4: 536–544.

Kilkenny, F.F. 2015. Genecological approaches to predicting the effects of climate change on plant populations. Natural Areas Journal. 35: 152–164.

Leger, E.A. 2008. The adaptive value of remnant native plants in invaded communities: An example from the Great Basin. Ecological Applications. 18: 1226–1235.

Leger, E.A.; Baughman, O.W. 2015. What seeds to plant in the Great Basin? Comparing traits prioritized in native plant cultivars and releases with those that promote survival in the field. Natural Areas Journal. 35: 54–68.

Leger, E.A.; Goergen, E.M.; Forbis de Queiroz, T. 2014. Can native annual forbs reduce *Bromus tectorum* biomass and indirectly facilitate establishment of a native perennial grass? Journal of Arid Environments. 102: 9–16.

Leimu, R.; Fischer, M. 2008. A meta-analysis of local adaptation in plants. PLoS ONE. 3: 1–8.

Madsen, M.D.; Davies, K.W.; Williams, C.J.; [et al.]. 2012. Agglomerating seeds to enhance native seedling emergence and growth. Journal of Applied Ecology. 49: 431–438.

Madsen, M.D.; Davies, K.W.; Mummey, D.L.; [et al.]. 2014. Improving restoration of exotic annual grass-invaded rangelands through activated carbon seed enhancement technologies. Rangeland Ecology and Management. 67: 61–67.

Marlette, G.M.; Anderson J.E. 1986. Seed banks and propagule dispersal in crested-wheatgrass stands. Journal of Applied Ecology. 23: 161–175.

McAdoo, J.K.; Swanson, J.C.; Murphy, P.J.; [et al.]. 2016. Evaluating strategies for facilitating native plant establishment in northern Nevada crested wheatgrass seedings. Restoration Ecology. 25: 53–62.

McKay, J.K.; Christian, C.E.; Harrison, S.; [et al.]. 2005. "How local is local?" A review of practical and conceptual issues in the genetics of restoration. Restoration Ecology. 13: 432–440.

Mijnsbruggea, K.V; Bischoff, A.; Smith, B. 2010. A question of origin: Where and how to collect seed for ecological restoration. Basic and Applied Ecology. 11: 300–311.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2014. A field guide for selecting the most appropriate treatment in sagebrush and pinon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses, and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322-rev. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 68 p.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2015. A field guide for rapid assessment of post-wildfire recovery potential in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-338. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 70 p.

Miller, R.F.; Knick, S.T.; Pyke, D.A.; [et al.]. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 145–185.

Monaco, T.A.; Waldron, B.L.; Newhall, R.L.; [et al.]. 2003. Re-establishing perennial vegetation in cheatgrass monocultures-planting prostrate kochia in 'greenstrips' may be a viable option to decrease cheatgrass dominance. Rangelands. 25: 26–29.

Monsen, S.B.; Stevens, R; Shaw, N.L., comps. 2004a. Restoring western ranges and wildlands. Gen. Tech. Rep. RMRS-GTR-136-vol-1. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Pages 1–294 plus index.

Monsen, S.B.; Stevens, R; Shaw, N.L., comps. 2004b. Restoring western ranges and wildlands. Gen. Tech. Rep. RMRS-GTR-136-vol-2. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Pages 295–698 plus index.

Monsen, S.B.; Stevens, R; Shaw, N.L., comps. 2004c. Restoring western ranges and wildlands. Gen. Tech. Rep. RMRS-GTR-136-vol-3. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Pages 699–884 plus appendices and index.

Nafus, A.M.; Svejcar, T.J.; Ganskopp, D.C.; [et al.]. 2015. Abundances of coplanted native bunchgrasses and crested wheatgrass after 13 years. Rangeland Ecology and Management. 68: 211–214.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers. 77: 118–125.

Ott, J.E.; Cox, R.D.; Shaw, N.L.; [et al.]. 2016. Postfire drill-seeding of Great Basin plants: Effects of contrasting drills on seeding and nonseeded species. Rangeland Ecology and Management. 69: 373–385.

Pellant, M. 1994. History and applications of the Intermountain greenstripping program. In: Monsen, S.B.; Kitchen, S.G., eds. Proceedings of the Ecology and Management of Annual Rangelands. Gen. Tech. Rep. GTR-INT-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 63–68.

Plant Conservation Alliance [PCA]. 2015. National seed strategy for rehabilitation and restoration 2015–2020. Washington, DC: U.S. Department of the Interior, Bureau of Land Management. 50 p. https://www.fs.fed.us/wildflowers/Native_Plant_Materials/documents/SeedStrategy081215.pdf. [Accessed May 23, 2018].

Pollinator Health Task Force. 2015. National strategy to promote the health of honey bees and other pollinators. Washington, DC: The White House. 58 p. https://obamawhitehouse.archives.gov/sites/default/files/microsites/ostp/Pollinator%20Health%20Strategy%202015.pdf. [Accessed May 23, 2018].

Pyke, D.A.; Chambers, J.C.; Pellant, M.; [et al.]. 2015a. Restoration handbook for sagebrush steppe ecosystems with emphasis on greater sage-grouse habitat—Part 1. Concepts for understanding and applying restoration. Circular 1416. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 44 p. https://doi.org/10.3133/cir1416

Pyke, D.A.; Chambers, J.C.; Pellant, M.; [et al.]. 2017. Restoration handbook for sagebrush steppe ecosystems with emphasis on greater sage-grouse habitat—Part 3. Site Level Restoration Decisions. Circular 1426. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 62 p. https://www.treesearch.fs.fed.us/pubs/53743. [Accessed May 23, 2018].

Pyke, D.A.; Knick, S.T.; Chambers, J.C.; [et al.]. 2015b. Restoration handbook for sagebrush steppe ecosystems with emphasis on greater sage-grouse habitat—Part 2. Landscape Level Restoration Decisions. Circular 1418. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 19 p. https://doi.org/10.3133/cir1418.

Rice, K.J.; Knapp, E.E. 2008. Effects of competition and life history stage on the expression of local adaptation in two native bunchgrasses. Restoration Ecology. 16: 12–23.

Richards, R.T.; Chambers, J.C.; Ross, C. 1998. Use of native plants on federal lands: Policy and practice. Journal of Range Management. 51: 625–632.

Richardson, B.A.; Ortiz, H.G.; Carlson, S.L.; [et al.]. 2015. Genetic and environmental effects on seed weight in subspecies on big sagebrush: Applications for restoration. Ecosphere 6: 1–13.

Robichaud, P.R.; Ashmun, L.E.; Sims, B.D. 2010. Post-fire treatment effectiveness for hillslope stabilization. Gen. Tech. Rep. RMRS-GTR-240. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 62 p.

Rowe, C.L.J.; Leger, E.A. 2012. Seed source affects establishment of *Elymus multisetus* in postfire revegetation in the Great Basin. Western North American Naturalist. 74: 543–553.

Sands, A.; Moser, A. 2012. Sagebrush restoration final report. March 24, 2012. Boise, ID: Idaho Department of Fish and Game. 19 p.

Schröder, R.; Prasse, R. 2013. From nursery into nature: A study on performance of cultivated varieties of native plants used in re-vegetation, their wild relatives and evolving wild × cultivar hybrids. Ecological Engineering. 60: 428–437.

St. Clair, J.B.; Kilkenny, F.F.; Johnson, R.C.; [et al.]. 2013. Genetic variation in adaptive traits and seed transfer zones for *Pseudoroegneria spicata* (bluebunch wheatgrass) in the northwestern United States. Evolutionary Applications. 6: 933–938.

Turesson, G. 1922. The genotypical response of the plant species to the habitat. Hereditas. 14: 99–152.

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2012. 'Epharim' crested wheatgrass (*Agropyron cristatum*). A conservation plant release. Aberdeen, ID: U.S. Department of Agriculture, Natural Resources Conservation Service, Aberdeen Plant Materials Center. https://www.nrcs.usda.gov/Internet/FSE_PLANTMATERIALS/publications/idpmcrb11478.pdf. [Accessed May 23, 2018].

Uselman, S.M.; Snyder, K.A.; Leger, E.A.; [et al.]. 2014. First-year establishment, biomass, and seed production of early vs. late seral natives in two medusahead (*Taeniatherum caput-medusae*) invaded soils. Invasive Plant Science and Management. 7: 291–302.

7. LIVESTOCK GRAZING MANAGEMENT

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Introduction

Part 1 of the Science Framework identifies livestock grazing as the most widespread land use in the sagebrush biome (Chambers et al. 2017a; hereafter, Part 1). In the Conservation Objectives Team Report (USDOI FWS 2013) improper livestock grazing is considered a present and widespread threat to Greater sage-grouse (*Centrocercus urophasianus*; hereafter, GRSG) for most GRSG populations. Livestock grazing affects the composition and structure of plant communities across the sagebrush biome and, consequently, the habitats of GRSG, other species at risk, and high value resources (Boyd et al. 2014). Livestock grazing can also affect habitat restoration efforts and thus the capacity to achieve broad-scale conservation and restoration goals.

The effects of livestock grazing on ecosystem composition, pattern, and function are well recognized (Beck and Mitchell 2000; Boyd et al. 2014; Cagney et al. 2010; Freilich et al. 2003; Fuhlendorf and Engle 2001; Knick et al. 2011). Major differences in plant responses to livestock grazing exist among ecoregions due to evolutionary adaptations to grazing and browsing, plant phenology relative to the timing of grazing, and selectivity of grazers for different plant species within the community (see Part 1, section 5.3.7). The effects of livestock grazing are strongly influenced by season of grazing relative to plant tolerance to grazing and the availability of water for plant regrowth after grazing. In the Cold Deserts water storage and plant growth depend on winter precipitation, and cool season plants (see definitions in Appendix 1) dominate plant communities (Part 1, sections 4.2 and 4.3). In the Cold Deserts both stocking rates (Briske et al. 2011) and grazing season affect plant responses to grazing (Briske and Richards 1995). Grazing of perennial grasses during inflorescence development (late spring) when moisture is becoming limited can negatively affect plant regrowth and recovery (Briske and Richards 1995). In contrast, in the West-Central Semiarid Prairies more moisture is available during summer and a mixture of cool season plants and warm season grasses, which have greater water use efficiency, dominate plant communities (Part 1, section 4.1). In both the West-Central Semiarid Prairies and Western Cordillera, precipitation during the growing season may increase tolerance to grazing, but cool season grasses can be eliminated by seasonal grazing that impacts them but not warm season plants.

Livestock grazing has the greatest potential to affect GRSG habitat by changing the composition, structure, and productivity of the herbaceous plants used by GRSG for nesting and early brood-rearing (Part 1, section 5.3.7; Beck and Mitchell 2000; Boyd et al. 2014; Cagney et al. 2010; Hockett 2002). The available research indicates that GRSG nest and early brood microhabitat selection and brood-rearing success are closely tied to areas with greater

Left: Utah rancher Bill Kennedy (photo: Jesse Bussard. USDA Forest Service). Top right: Livestock grazing in a sagebrush ecosystem (photo: Joe Smith, Sage Grouse Initiative/ University of Montana). Middle right: Placing fence markers to prevent sage-grouse strikes (photo: USDA Forest Service). Bottom right: Cattle and Greater sage-grouse in a sagebrush ecosystem (photo: USDA Forest Service).

sagebrush and grass canopy cover and height than are randomly available in sagebrush landscapes (Dinkins et al. 2016; Doherty et al. 2011, 2014; Hagen et al. 2007; Kirol et al. 2012; Thompson et al. 2006). However, the reported effects of grass-related variables on nest site selection and nest survival have been less consistent in the literature (Part 1, section 5.3.7; Coates et al. 2017; Smith 2016). Thus, it has been suggested that management prescriptions for livestock grazing within nesting habitats consider the potential regional variation in grass-related variables and the effects associated with plant phenology. Current vegetation habitat objectives for breeding and nesting seasonal habitat, and brood-rearing and summer seasonal habitat, consider key plant community indicators such as sagebrush cover, sagebrush height, sagebrush shape, and perennial grass and perennial forb cover and height (Stiver et al. 2015). These vegetation habitat objectives also consider how plant community indicators vary between wetter and drier ecological sites (Stiver et al. 2015). Livestock grazing management is key to either maintaining or attaining these habitat objectives.

Livestock, primarily cattle and sheep, are grazed across the sagebrush biome on Federal, State, tribal, and private lands. Grazing practices and flexibility in those practices can vary according to the land manager or owner. Because many livestock grazing operations span multiple management jurisdictions, it is necessary to consider management opportunities and restrictions on each parcel that the operator uses.

Federal and State agencies are working together with private landowners to maintain or improve sagebrush habitat on rangelands in a manner appropriate for the site conditions and landowner interests. The Federal Land Policy and Management Act of 1976 stated that Federal land management agencies must ‘‘manage the public lands under principles of multiple use and sustained yield’’ (Public Law 94–579, Sec. 302). The Public Rangelands Improvement Act of 1978 (Public Law 95–514) further commits Federal land management agencies to providing regular updates on the condition and trend of rangelands. These legislative actions typically translate into management of livestock use in ways that sustain other land uses (e.g., wildlife conservation) and involve monitoring of livestock grazing effects.

This section begins by discussing the administration of livestock grazing on public and private lands and the ongoing review of grazing authorization (permits and leases) and processing in GRSG habitat. Then information is provided on the use of resilience and resistance concepts and the Science Framework to inform livestock grazing management. Considerations for the use of this information are presented for both the mid-scale (ecoregion or Management Zone) and local scale (field office or district), with an emphasis on grazing management practices to improve habitats of GRSG and other species and values at risk. Finally, select ecological types and state-and-transition models (STMs) (see Appendix 1 for definitions) are used as the basis for identifying livestock grazing management practices within the GRSG range that can be implemented to maintain or improve the resilience and resistance of sagebrush plant communities and the quality of GRSG nesting and early brood-rearing habitat.

Livestock Grazing Management on Public and Private Lands

The Bureau of Land Management (BLM) manages livestock grazing on 155 million acres (73 million hectares) of public land and administers nearly 18,000 permits and leases held by ranchers who graze their livestock at least part of the year on more than 21,000 allotments. A grazing permit is a document authorizing grazing use of the public lands within an established grazing district. A grazing lease is a document authorizing grazing use outside of an established grazing district. A grazing allotment is an area of land designated and managed for the grazing of livestock. Allotments may consist of BLM-administered lands as well as other Federally managed, State-owned, and private lands. Livestock numbers and periods of use are specified for each allotment. Permits and leases specify all authorized livestock grazing use including the total number of animal unit months (AUMS) and the area (allotment) authorized for grazing use.

Permits and leases generally cover a 10-year period and are renewable if the BLM determines that the terms and conditions of a permit or lease are being met. The terms and conditions for grazing on BLM-managed lands (such as stipulations on forage use and season of use) are set forth in the permits and leases issued by the BLM to public land ranchers. The amount of grazing that takes place each year on BLM-managed public lands can be affected by such factors as drought, wildfire, and market conditions.

The Forest Service manages livestock grazing on over 95 million acres (38 million hectares) of National Forest System lands on 7,275 allotments spread across 29 States. Grazing use is administered through a grazing permit system similar to that used by the BLM. Permits are issued for a 10-year period with the current permittee having the preference to reapply for the permit upon expiration provided that he or she has complied with the terms and conditions of the current permit. The Forest Service administers about 6,400 permits for 5,897 permittees. The majority (90 percent) of those permits are for cattle and sheep. The remaining 10 percent include bison, goat, donkey, burro, horse, and mule.

Potential livestock grazing management practices designed to improve sagebrush habitats can be incorporated into livestock grazing management alternatives during the grazing authorization (grazing permits and grazing leases) renewal process. When vegetation habitat objectives for GRSG and land health standards are not met because of current livestock grazing management, changes in livestock grazing management are needed to ensure significant progress toward achieving the vegetation habitat objectives for GRSG and land health standards. Current BLM livestock grazing regulations require that monitoring data or field observations, or both, be used to support decisions about stocking rates on allotments (43 CFR 4110.3) (text box 7.1).

Setting priorities for review and processing of grazing authorizations (permits and leases) is ongoing within the BLM (USDOI BLM 2017a) and other agencies. Priorities for review and processing of grazing authorizations are (1) areas where rangeland health standards have not been evaluated, and (2) areas that are not achieving rangeland health standards. In areas with GRSG habitat, BLM and its partners have developed specific vegetation habitat objectives for breeding and nesting seasonal habitat, and brood-rearing and summer seasonal habitat, for GRSG in Montana (USDOI BLM 2015a, table 2.3-2; USDOI BLM 2015c, table 2-6; USDOI BLM 2015d, table 2-2), North Dakota (USDOI BLM 2015f, table 2-2), South Dakota (USDOI BLM 2015h, table 2-6), the Wyoming Basin Ecoregion and northeast Wyoming (USDOI BLM 2015j, tables 2-2 and 2-3), Oregon and Washington (USDOI BLM 2015g, table 2-2), Utah (USDOI BLM 2015i, table 2-2), Nevada and northeastern California (USDOI BLM 2015e, table 2-2), and Idaho and southwestern Montana (USDOI BLM 2015b, table 2-2). In areas with GRSG habitat, managers will need to evaluate vegetation habitat objectives for GRSG when conducting an evaluation of rangeland

Text Box 7.1—Monitoring Livestock Grazing

In 1995, through regulation in 43 CFR 4180, livestock grazing on BLM-administered lands was required to ensure the attainment of Fundamentals of Rangeland Health. The Fundamentals of Rangeland Health address minimum standards for ecosystem functioning including: (1) properly functioning watersheds; (2) ecological processes of the hydrologic cycle, nutrient cycle, and energy flow; (3) water quality; and (4) wildlife habitat quality (43 CFR 4180.1). The BLM was required to develop rangeland health standards that would conform to the Fundamentals of Rangeland Health within individual regions in consultation with local Resource Advisory Councils (43 CFR 4180.2). To evaluate land health, BLM field office personnel are required to perform individual, on-the-ground evaluations of these rangeland health standards in all grazing allotments. Current livestock grazing use is monitored within grazing allotments to ascertain whether current livestock grazing use is allowing for achievement of rangeland health standards. Collection of monitoring data on the effects of current livestock grazing use constitutes a major priority for livestock grazing management on BLM-administered lands.

health standards. If the BLM finds that vegetation habitat objectives for GRSG are not being achieved because of current livestock grazing, then the agency modifies the livestock grazing management practices to ensure that progress will be made toward achieving the vegetation habitat objectives for GRSG. It may be necessary to modify and update the vegetation habitat objectives over time as additional information on GRSG habitat requirements and ecological site potentials to support GRSG habitat become available and additional policy direction is provided (USDOI BLM 2017b).

Private landowners generally use range management principles and tools provided by entities such as the Agricultural Research Service, Natural Resources Conservation Service, and State and university extension programs. Use of proven range management principles and tools can ensure that private lands are managed in a manner that maintains or improves rangeland resilience and resistance to invasive annual grasses and provides the necessary resources for GRSG and other wildlife species. Tools for private lands include range management plans that are based on local ecological site information and rangeland plant inventories. It is recommended that range management plans incorporate flexibility in season of use and stocking rates to allow for implementing adaptive management of GRSG habitat. It is generally recognized that by promoting diverse and productive native perennial plant communities, private landowners can ensure that rangelands remain resilient to disturbance and resistant to invasive plants. As a result, drought, annual grass invasions, and wildfires are less likely to impact GRSG and other sagebrush dependent species.

Using Resilience and Resistance Concepts and the Science Framework Approach to Inform Livestock Grazing Management

Designing livestock grazing management practices to improve habitats of GRSG and other species and values at risk requires a consistent approach that can be applied across jurisdictions. In Part 1 of the Science Framework, an approach is identified for determining the suitability of an area for a management action and the most appropriate management action that can be applied to livestock grazing management. At the mid-scale, geospatial analyses can be used to evaluate: (1) the likely response of an area to disturbance or management actions (i.e., resilience to disturbance and resistance to invasion by annual grasses), (2) the capacity of an area to support target species or resources, and (3) the predominant threats. Many of the data layers used in the mid-scale geospatial analyses for the Science Framework (see Part 1, sections 8.1 and 8.2) can be used to help inform livestock grazing administration and identify appropriate livestock grazing management practices. Key data layers include resilience and resistance to invasive annual grasses as indicated by soil temperature and moisture regimes (Maestas et al. 2016), GRSG breeding habitat probabilities (Doherty et al. 2016), and the primary threats within the assessment area.

At the local scale the Science Framework approach includes: (1) identifying the different ecological types or ecological sites that exist within the management area and determining their relative resilience to disturbance and resistance to invasive annual grasses; (2) evaluating the current ecological dynamics of the ecological types or ecological sites and, where possible, their restoration pathways; and (3) selecting livestock grazing management practices that have the potential to increase overall ecosystem functioning and habitat conditions. Ecological types or ecological site descriptions and STMs that explicitly consider ecosystem resilience to disturbance and resistance to invasive annual grasses provide the basis for selecting appropriate livestock grazing management practices (see Part 1, section 9). Consideration of habitat objectives for GRSG and other species and values at risk is used to assess whether the management area (e.g., grazing allotment) has the potential to attain the habitat objectives and, if so, the specific livestock grazing management practices needed to achieve the objectives (Beck and Mitchell 2000; Boyd et al. 2014; Hockett 2002).

In general, areas that support GRSG habitat or other important species or resources are high priorities for livestock grazing management that maintains or improves GRSG habitat values (tables 1.3, 1.4). Areas with moderate to high resilience and resistance to invasive annual grasses often have the potential to recover from disturbances through successional processes. These areas represent significant opportunities to use livestock grazing management and other management activities to direct plant succession to improve habitat. Areas with low resilience and resistance often lack the potential to recover from improper livestock grazing without significant intervention, and are among the highest priorities for improved livestock grazing management.

To step down to the local scale, ecological types or ecological site descriptions and their associated STMs can be used to evaluate current ecological dynamics and determine appropriate livestock grazing management practices (text box 7.2). In the Science Framework, generalized ecological types and STMs have been described for the range of environmental conditions in the eastern and western portions of the sagebrush biome. These ecological types and STMs are characterized according to their resilience to disturbance and resistance to invasive annual grasses based on soil temperature and moisture regimes and other biophysical characteristics (Part 1, Appendices 5 and 6). They provide information on the alternative states, ranges of variability within states, and processes that cause plant community shifts within states as well as transitions among states. Examples of how to use these resilience-based ecological types and STMs for managing ecosystem threats across the sagebrush biome are in Part 1, section 9.2. Information on using the ecological types and STMs in sagebrush and juniper (*Juniperus* spp.) and piñon (*Pinus* spp.) ecosystems of the Great Basin for selecting appropriate treatments is in Miller et al. (2014). Information on assessing postwildfire recovery potential and making restoration decisions is in Miller et al. (2015) and Pyke et al. (2017).

Text Box 7.2—Using Ecological Site Descriptions and State-and-Transition Models

Ecological site descriptions and their associated state-and-transition models (STMs) provide essential information for determining treatment feasibility and type of treatment. Ecological site descriptions are part of a land classification system that describes the potential of a set of climate, topographic, and soil characteristics and natural disturbances to support a dynamic set of plant communities (Bestelmeyer et al. 2009; Stringham et al. 2003). Ecological site descriptions have been developed by the Natural Resources Conservation Service and its partners to assist land management agencies and private landowners with making resource decisions. For a detailed description of ecological site descriptions and access to available ecological site descriptions see: http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/ecoscience/desc/.

STMs are a central component of ecological site descriptions that are widely used by managers to illustrate changes in plant communities and associated soil properties, causes of change, and effects of management interventions (Briske et al. 2005; USDA NRCS 2015; Stringham et al. 2003). STMs use the concepts of **states** (a relatively stable set of plant communities that are resilient to disturbance) and **transitions** (change among alternative states caused by disturbances or other drivers) to describe the range in composition and function of plant communities within ecological site descriptions (Stringham et al. 2003) (see Appendix 1 for definitions). The reference state is based on the natural range of conditions associated with the historical range of variation and often includes several plant communities (**phases**) that differ in dominant plant species relative to type and time since disturbance (Caudle et al. 2013). Alternative states describe new sets of communities that result from factors such as improper livestock use, invasion by nonnative species, or changes in fire regimes. Changes or transitions among states often are characterized by **thresholds** or conditions that may persist over time without active intervention, potentially causing irreversible changes in community composition, structure, and function. **Restoration pathways** are used to identify the environmental conditions and management actions that will facilitate return to a previous state.

Examples of Using Resilience-Based State-and-Transition Models to Identify Potential Livestock Grazing Management Practices

The dominant ecological types and STMs provide the basis for identifying livestock grazing management practices that can be implemented to maintain or improve the resilience and resistance of sagebrush plant communities and the quality of GRSG nesting and early brood-rearing habitat. Here, examples of ecological types and STMs are provided for different ecoregions and sage-grouse management zones (fig. 1.1). The examples were chosen to illustrate the differences in potential management strategies for ecological types that support GRSG populations and can often benefit from improved livestock grazing management. Some states within the STMs, and plant community phases within the states, do not provide the vegetation necessary for nesting and early brood-rearing habitat for GRSG as identified in vegetation habitat objectives for breeding and nesting seasonal habitat and brood-rearing and summer seasonal habitat (e.g., USDOI BLM 2015a, table 2.3-2; 2015b, table 2-2; 2015c, table 2-6; 2015d, table 2-2; 2015e, table 2-2; 2015f, table 2-2; 2015g, table 2-2; 2015h, table 2-6; 2015i, table 2-2; 2015j, tables 2-2 and 2-3). Potential livestock grazing management practices are presented that can be implemented to help improve ecological conditions and achieve the vegetation habitat objectives for nesting and early brood-rearing habitat for GRSG.

West Central Semi-Arid Prairies—Frigid Bordering on Cryic/ Ustic Bordering on Aridic, Grass Dominated with Silver Sagebrush (Management Zone I)

Potential Livestock Grazing Management Practices for the Reference State

There are two primary goals for livestock grazing management practices in the reference state of the silver sagebrush (*Artemisia cana*), 10–14 inch (25–36 centimeter) precipitation zone ecological type (fig. 7.1). The first is to maintain the reference state and prevent a transition to the unsustainable grazing state. The second is to facilitate achievement of vegetation habitat objectives for breeding and nesting seasonal habitat, and brood-rearing and summer seasonal habitat, for GRSG in Montana (USDOI BLM 2015a, table 2.3-2; 2015c, table 2-6; 2015d, table 2-2), North Dakota (USDOI BLM 2015f, table 2-2), and South Dakota (USDOI BLM 2015h, table 2-6).

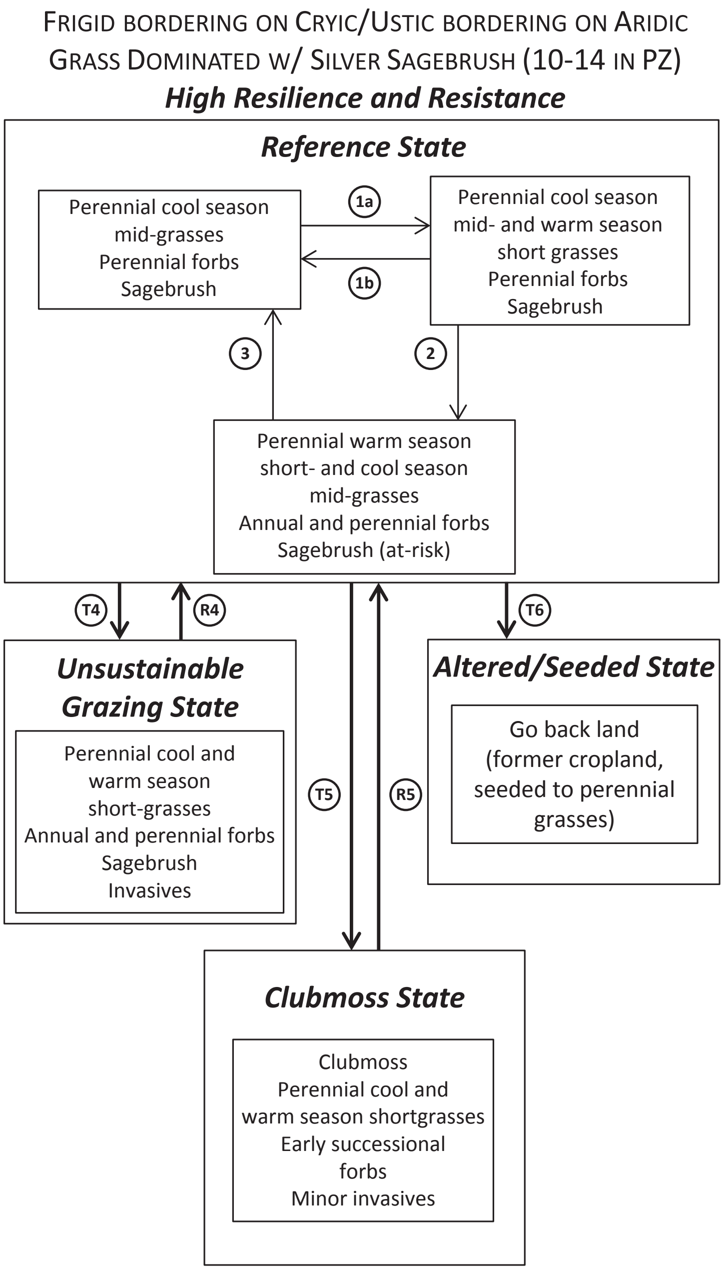
Plant communities in the reference state provide nesting and early brood-rearing habitat for GRSG. Plant communities in the reference state are dominated by perennial cool-season mid-grasses, with less abundance of perennial warm-season short grasses and silver sagebrush. Silver sagebrush is present within a matrix of perennial cool-season mid-grasses and perennial warm-season short grasses.

Consistent year to year, early spring use by livestock will reduce the abundance of perennial cool-season mid-grasses (Adams et al. 2004) and cause a transition to the unsustainable grazing state. Livestock grazing that is deferred to a late spring onset of grazing can improve plant vigor and productivity of the perennial cool-season mid-grasses and provide increased plant cover, reducing the potential conflict between livestock and GRSG during breeding and nesting (Adams et al. 2004). Managing for light grazing intensity of no more than about 25 to 40 percent annual utilization of the perennial grasses can maintain the productivity of the perennial grasses, provide cover to conceal GRSG nesting sites, and improve breeding and brood-rearing habitat (Adams et al. 2004).

Deferred rotation grazing systems can reduce the impacts of livestock to GRSG nesting sites by resting pastures from livestock grazing in the nesting and brood-rearing seasons and rotating early-season grazing among pastures (Adams et al. 2004). Rest-rotation grazing systems can increase perennial grass height in these plant communities compared with season-long grazing (Smith 2016).

In central Montana GRSG nesting habitat comprising mixed stands of silver sagebrush/Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) with perennial cool-season mid-grasses, the cover of silver sagebrush and Wyoming big sagebrush was comparatively more important than the cover and height of herbaceous vegetation, for GRSG nest site selection and nest survival (Smith 2016). Maintaining or increasing the cover of sagebrush in these plant communities is important to maintain breeding habitat for GRSG (Smith 2016). Grazing by livestock does not have direct effects on the cover of silver sagebrush. However, silver sagebrush is often low in stature and can be vulnerable to trampling by livestock, particularly if livestock congregate within silver sagebrush stands in winter (Adams et al. 2004).

To improve early brood-rearing habitat, large flood plain and overflow sites composed of western wheatgrass (*Pascopyrum smithii*)/silver sagebrush plant communities can be fenced off and managed separately as riparian pastures. Forb production can be stimulated with periodic light grazing in spring, at light stocking rates for a short duration, and then grazed again in late summer or fall after the brood-rearing season (Adams et al. 2004).



1a Sagebrush increases and proportion of cool season mid-grass Functional/Structural Group decreases due to disturbances such as drought (3-5 years) and spring grazing.

1b Normal precipitation patterns favor herbaceous understory. Grazing intensity and/or duration is reduced to allow for herb recovery.

2 Sagebrush increases and proportion of cool and warm season mid-and short-grass Functional/Structural Groups increases due to prolonged drought (5-7 years), increased grazing intensity and duration, and lack of fire. Plant community is at-risk of leaving reference state with extended drought and continued grazing pressure.

3 With favorable precipitation, disturbance such as fire, and a grazing system that provides rest and recovery of preferred species, cool season mid-grass Functional/Structural Groups increase.

T4 Extended drought (>7 years) along with high intensity and long duration grazing result in transition to a state resistant to grazing that is dominated by cool and warm season short-grass Functional/Structural Groups. Silver sagebrush cover is at its highest, and early seral forbs are present. There is potential for invasive species such as field brome in high moisture years and/or due to removal of grazing, lack of fire, and other conditions causing accumulation of excessive litter.

R4 Normal precipitation patterns, fire or fire surrogates (herbicides and/or mechanical treatments), and a grazing regime with proper timing and intensity that varies season of use can return the site to the reference state.

T5 Extended drought (>7 years) may result in dense stands of clubmoss. However, no grazing, light grazing, and rotational grazing combined with drought can result in more rapid increase in clubmoss than drought alone. Lack of fire may contribute to this transition as well. Potential for invasives such as field brome is minor, and this transition occurs more often on older, more developed soils with an argillic horizon.

R5 Extended periods of normal and above average precipitation, mechanical renovation, chemical treatment, fertilizer/manure application, seeding (if an adequate seedbank does not exist), fire, and/or periods of rest or light grazing can return the site to the reference state.

T6 Former cropland seeded to introduced and/or native perennial grasses, largely funded by government programs. In the 1960-1970s seedings were primarily introduced species such as crested wheatgrass, intermediate wheatgrass, and smooth brome. From 1985 to present both introduced and native species were used, mainly under the Conservation Reserve Program. Sagebrush is largely absent from this state. There is potential for invasive species such as field brome in high moisture years and/or due to removal of grazing, lack of fire, and other conditions that would result in an accumulation of excessive litter.

Figure 7.1—State-and-transition model for a silver sagebrush, 10–14 inch precipitation zone ecological type applicable to the West Central Semiarid Prairies in the eastern part of the sagebrush biome and GRSG range in Montana, North Dakota, and South Dakota (Management Zone I). Large boxes illustrate states that are made up of community phases (smaller boxes). Transitions among states are shown with arrows starting with T; restoration pathways are shown with arrows starting with R. The "at risk" community phase is most vulnerable to transition to an alternative state (figure source: Chambers et al. 2017a, Appendix 5).

Potential Livestock Grazing Management Practices for the Unsustainable Grazing State

Livestock grazing management practices in the unsustainable grazing state (fig. 7.1) have the goal of stimulating a transition of the unsustainable grazing state to a reference state. Plant communities in the reference state provide improved nesting and early brood-rearing habitat for GRSG. Livestock grazing management practices should facilitate achievement of vegetation habitat objectives for breeding and nesting seasonal habitat, and brood-rearing and summer seasonal habitat, for GRSG in Montana (USDOI BLM 2015a, table 2.3-2; 2015c, table 2-6; 2015d, table 2-2), North Dakota (USDOI BLM 2015f, table 2-2), and South Dakota (USDOI BLM 2015h, table 2-6).

Grazing management practices that increase the amount of rest in a pasture can be useful in providing more cover for GRSG (Adams et al. 2004). Adams et al. (2004) recommend rest-rotation grazing systems to improve grass and silver sagebrush plant communities that are depauperate in perennial cool-season mid-grasses and aid regeneration of silver sagebrush plants if moisture is available to support resprouting.

Cold Deserts—Frigid/Ustic Bordering on Aridic Wyoming Big Sagebrush (Management Zones II and VII)

Potential Livestock Grazing Management Practices for the Reference State

Livestock grazing management practices in the reference state in the Wyoming big sagebrush, 10–14 inch precipitation zone ecological type (figs. 7.2, 7.3) have two primary goals. The first goal is to maintain the reference state and prevent a transition to the grazing resistant state. The grazing-resistant state results from continuous spring grazing with cattle during the critical growth period for cool season grasses and eventual dominance of grazing-tolerant species: perennial cool-season rhizomatous grasses, short or sod-forming warm-season grasses, and mat-forming forbs. The second goal is to facilitate achievement of vegetation habitat objectives for breeding and nesting seasonal habitat, and brood-rearing and summer seasonal habitat, for GRSG in the Wyoming Basin ecoregion (USDOI BLM 2015j, tables 2-2 and 2-3). Plant communities in the reference state provide nesting and early brood-rearing habitat for GRSG.

A livestock grazing strategy that prevents grazing of the perennial cool-season bunchgrasses during the critical growing season (mid-May through mid-June) in at least two out of every three consecutive years is likely to maintain the reference state and prevent a transition to a grazing resistant state (Cagney et al. 2010).

Late season and winter grazing of the reference state may help promote the long-term persistence of perennial cool-season bunchgrasses, but can cause a reduction in the residual herbaceous material of these bunchgrasses that is needed for nesting cover for GRSG the next spring. Residual grasses remaining from the previous year provide the initial herbaceous cover available for nesting GRSG. Thus, late season and winter grazing is not always a grazing management practice that would allow for achieving nesting habitat objectives for GRSG (Cagney et al. 2010).

Potential Livestock Grazing Management Practices for the Grazing Resistant State

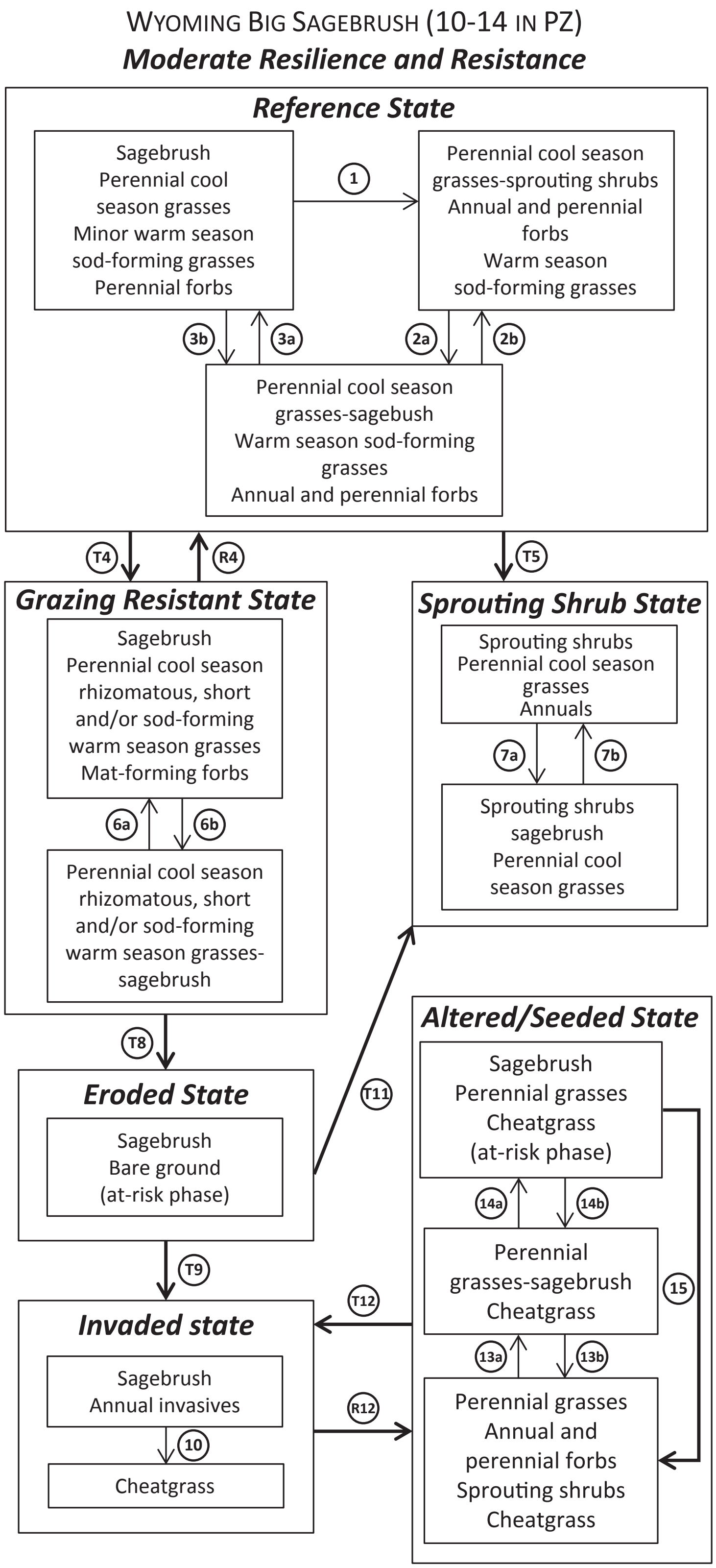
Livestock grazing management practices in the grazing resistant state (figs. 7.2, 7.4) have the goal of stimulating a transition of the grazing resistant state to a reference state. Plant communities in the reference state provide improved nesting and early brood-rearing habitat for GRSG. Livestock grazing management practices should help to achieve the vegetation habitat objectives for breeding and nesting seasonal habitat, and brood-rearing and summer seasonal habitat, for GRSG in the Wyoming Basin ecoregion (USDOI BLM 2015j, tables 2-2 and 2-3).

Grazing resistant grasses, specifically rhizomatous grasses and bluegrasses, are unlikely to decrease in abundance with changes in livestock grazing management alone (Cagney et al. 2010). Further, changing livestock grazing management, or eliminating grazing, is likely to have a limited effect on increasing the abundance of large bunchgrasses (Cagney et al. 2010). However, light to moderate grazing with periodic rest during critical growth periods along with fire, herbicides, mechanical treatments, or a combination thereof, may result in return to the reference state. If the grazing resistant state is burned or is treated with herbicides, causing a decrease in the canopy cover of sagebrush, it is advisable to defer livestock grazing during at least the first two growing seasons after fire or herbicide disturbance on these sites. Grazing deferment for two or more growing seasons will allow the remaining perennial, cool season bunchgrasses in this grazing resistant state to increase in abundance (Cagney et al. 2010). Heavy, continuous livestock grazing can cause a decrease in the herbaceous species and a more rapid increase in sagebrush, which will cause the site to progress back to the grazing resistant state (Cagney et al. 2010).

Targeted livestock grazing by domestic sheep in the grazing resistant state can cause browsing of sagebrush that decreases the canopy cover of sagebrush. It also opens up niches for establishment and increases in abundance of the grazing resistant rhizomatous grasses and bluegrasses as well as any remaining cool-season perennial bunchgrasses (Cagney et al. 2010). This treatment is applied in fall or winter when perennial cool-season bunchgrasses are not actively growing. Supplemental feeding of livestock in the winter on this grazing resistant state may be necessary to effectively implement this strategy. However, if these systems are grazed too intensely or too often, they can convert to a sprouting shrub state.

Potential Livestock Grazing Management Practices for the Eroded State

Changes in livestock grazing management alone are unlikely to cause an increase in perennial grasses on the eroded state (figs. 7.2, 7.5) (Cagney et al. 2010). Moreover, livestock grazing management practices alone cannot be used to achieve the vegetation habitat objectives for breeding and nesting seasonal habitat, and brood-rearing and summer seasonal habitat, for GRSG on the eroded state in the Wyoming Basin ecoregion (USDOI BLM 2015j, tables 2-2 and 2-3). Interseeding with native perennial grasses and forbs may be needed to meet habitat objectives (Huber-Sannwald and Pyke 2005). Grazing deferment for two or more grazing seasons is recommended for seedling establishment.



(1) Perennial grass, sprouting shrubs, and forbs become dominant due to disturbances that decrease sagebrush like prolonged or severe drought, freezing, flooding, wildfire, insects, disease, and pathogens.

(2a) Sagebrush increases with time until co-dominant with the herbaceous understory.

(2b) Perennial grass, sprouting shrubs, and forbs become dominant due to disturbances that decrease sagebrush.

(3a) Sagebrush increases with time until dominant.

(3b) Perennial grass and forbs increase due to disturbances that decrease sagebrush.

(T4) Continuous spring grazing with cattle during the critical growth period of cool season grasses results in dominance of grazing tolerant species that may include warm season grasses (e.g., blue grama). As bare ground increases, surface erosion (e.g., rills, sheet erosion) and pedestalled plants (especially bunchgrasses) may result.

(R4) Light to moderate grazing with periodic rest during critical growth periods along with fire, herbicides, and/or mechanical treatments can result in return to reference state.

(T5) An increase in the disturbance cycle by fire, fire surrogates, mechanical types of disturbance, and/or high density/frequency grazing will favor sprouting shrubs such as rabbitbrush. Annual invasives can occur.

(6a) Sagebrush increases with time. Cheatgrass and other weeds can be present, but do not dominate.

(6b) Perennial cool season grasses increase due to disturbances that decrease sagebrush. A temporary flush of annual invaders is expected.

(7a) Sagebrush increases with time and removal of disturbances until co-dominant with herbaceous understory.

(7b) Perennial cool season grasses and sprouting shrubs increase due to disturbances that decrease sagebrush.

(T8) Perennial grasses and forbs are eliminated and sagebrush increases with high density/frequency grazing by cattle, resulting in altered biotic, hydrologic, and soil function. This state is at-risk to invasion by annuals such as cheatgrass, especially after a stand-replacing, sagebrush killing event.

(T9) If a cheatgrass seed source is introduced, and weather conditions are conducive to establishment (warm wet spring), it will invade, especially after a stand-replacing event that eliminates sagebrush.

(10) Fire and fire surrogates that kill sagebrush will dramatically increase cheatgrass.

(T11) Multiple chemical and/or mechanical treatments or biological disturbances that reduce sagebrush will result in a shift toward sprouting shrub dominance with potential for cheatgrass to invade.

(T12) Catastrophic climatic events and/or fire can result in cheatgrass dominance, especially when in the sagebrush dominant phase of the altered state.

(R12) A restoration treatment, including chemical treatment for cheatgrass and seeding can restore a perennial grass community and eventually support an altered sagebrush community with invaders.

(13a) Sagebrush increases with time and no disturbances until co-dominant with the herbaceous understory, but cheatgrass will be present.

(13b) Perennial grass and forbs become dominant due to disturbances that decrease sagebrush.

(14a) Sagebrush increases with time and no disturbances until dominant, but cheatgrass may be present.

(14b) Perennial grass and forbs become dominant due to minor disturbances that decrease sagebrush.

(15) Perennial grass and annual/perennial forbs become dominant due to disturbances that decrease sagebrush.

Figure 7.2—State-and-transition model for a Wyoming big sagebrush, 10–14 inch precipitation zone ecological type applicable to the Cold Deserts in the eastern part of the sagebrush biome and GRSG range in the Wyoming Basin in the western and central portions of Wyoming (Management Zones II and VII). Large boxes illustrate states that are made up of community phases (smaller boxes). Transitions among states are shown with arrows starting with T; restoration pathways are shown with arrows starting with R. The “at risk” community phase is most vulnerable to transition to an alternative state (figure source: Chambers et al. 2017a, Appendix 5).

Figure 7.3—Example of a plant community phase in the reference state in the Wyoming big sagebrush, 10–14 inch precipitation zone ecological type (fig. 7.2) in Wyoming. The site is dominated by Wyoming big sagebrush with an herbaceous understory dominated by cool-season perennial bunchgrasses. This plant community phase provides nesting and early brood-rearing habitat for GRSG (photo by Jim Cagney, used with permission).

Cold Deserts—Mesic/Aridic Bordering on Xeric Wyoming Big Sagebrush (Management Zones III, IV, and V)

Potential Livestock Grazing Management Practices for the Invaded State

Livestock grazing management practices in the invaded state (figs. 7.6, 7.7) can be used to promote an increase of perennial grasses to increase resistance to invasive annual grasses. Livestock grazing management practices can also help achieve the vegetation habitat objectives for nesting and brood-rearing seasonal habitat for GRSG in Oregon and Washington (USDOI BLM 2015g, table 2-2), Utah (USDOI BLM 2015i, table 2-2), Nevada and northeastern California (USDOI BLM 2015e, table 2-2), and Idaho and southwestern Montana (USDOI BLM 2015b, table 2-2).

Effects of grazing on the abundance of annual grasses such as cheatgrass (*Bromus tectorum*) depend on multiple factors including: (1) the relative resilience of the site as indicated by soil temperature and moisture regimes, (2) the relative resistance of the site as indicated by its climatic suitability for cheatgrass (fig. 7.8) (Strand et al. 2014), and (3) the relative abundance of competitive perennial grasses and forbs (Chambers et al. 2014a,b). If sufficient

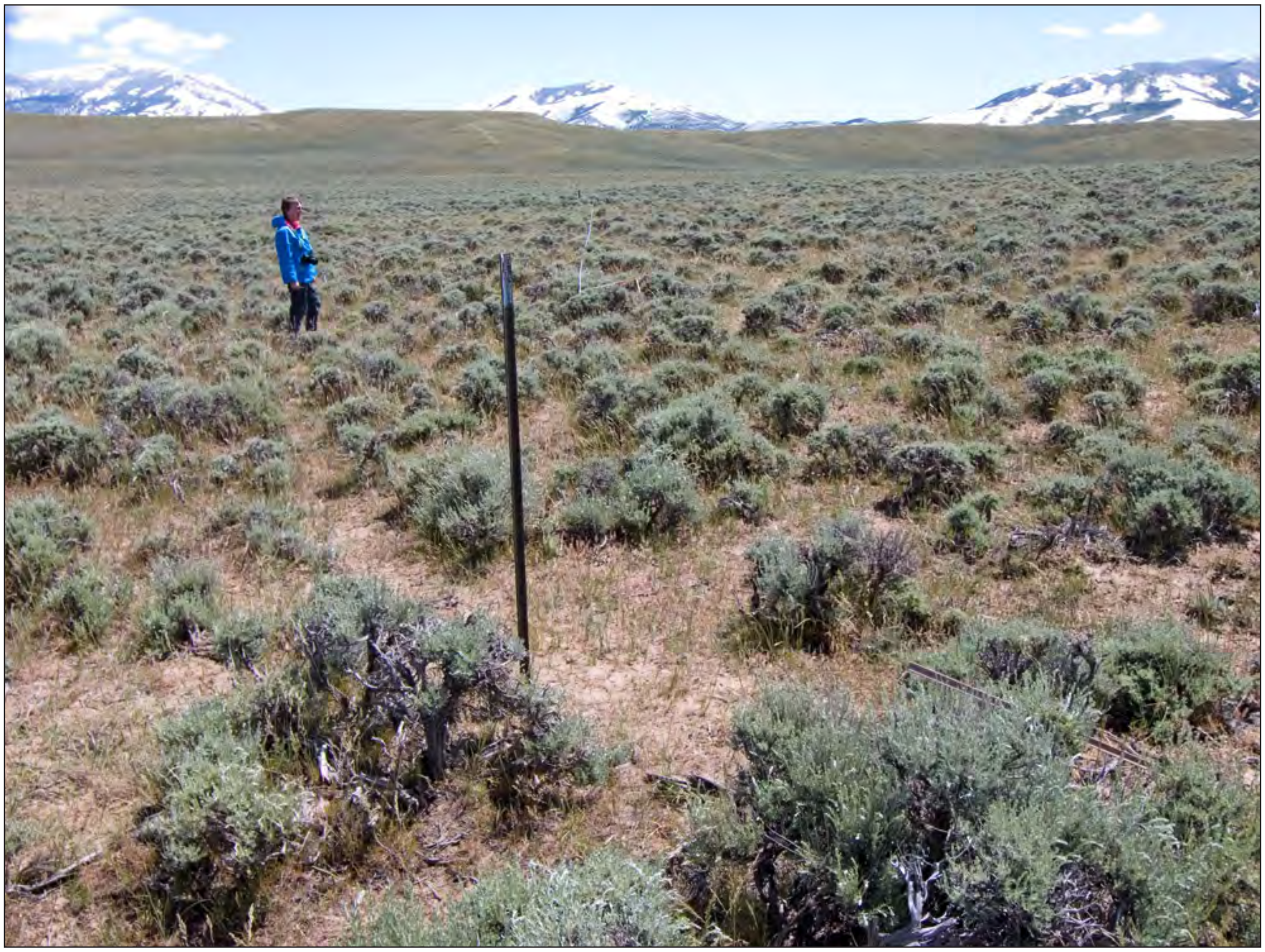


Figure 7.4—Example of a plant community phase in the grazing resistant state in the Wyoming big sagebrush, 10–14 inch precipitation zone ecological type (fig. 7.2) in Wyoming. The site is dominated by Wyoming big sagebrush with an herbaceous understory dominated by rhizomatous grasses and bluegrasses. If the herbaceous understory is not depleted, this plant community phase can provide nesting habitat for GRSG. With a depleted herbaceous understory, this plant community phase does not provide nesting habitat for GRSG (photo by Jim Cagney, used with permission).

perennial native grasses remain on the site, managed livestock grazing may result in an increase in perennial grasses and forbs and a decrease in invasive annual grasses, especially on relatively cool and moist sites. Grazing when perennial grasses are beginning to flower is likely to cause a decline in perennial grasses and an increase in cheatgrass (fig. 7.8) (Strand et al. 2014). Early spring grazing may suppress the abundance of cheatgrass and promote an increase of perennial grasses if grazing is applied when the annual grasses are starting to produce seeds but before the perennial grasses begin to bolt (fig. 7.8) (Strand et al. 2014). Livestock grazing persisting into the time when perennial grasses are beginning active growth can be detrimental to the perennial grasses (fig. 7.8) (Strand et al. 2014). Early spring grazing of cheatgrass can be difficult to plan for year after year and can be challenging to implement in a livestock grazing permit or lease on Federal land. This is because the amount of cheatgrass forage available in the early spring depends on the amount and timing of precipitation and varies considerably from year to year (Chambers et al. 2014b; West and Yorks 2002). Thus, the length of time that cheatgrass forage is available to be grazed in the early spring will vary from year to year, and permittees and lessees will have a difficult time planning ahead for how many animals will be required to consume the cheatgrass (Schmelzer et al. 2014).

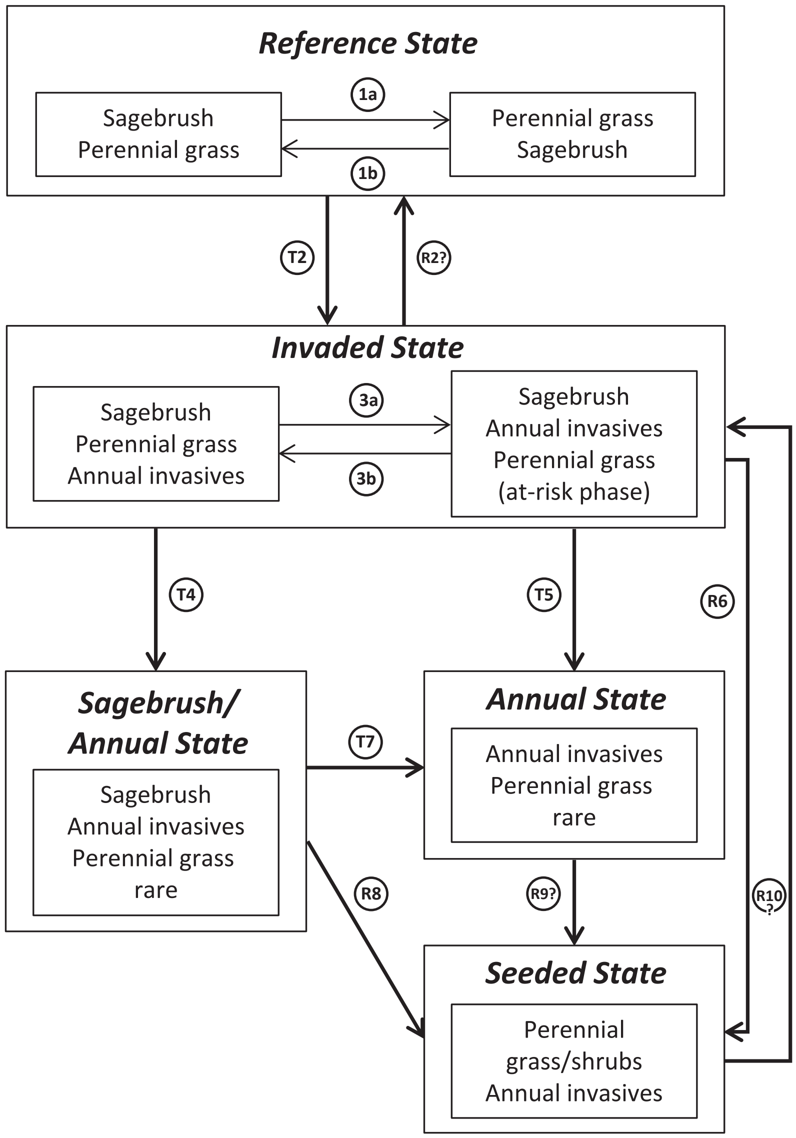
Figure 7.5—Example of a plant community phase in the eroded state in the Wyoming big sagebrush, 10–14 inch precipitation zone ecological type (fig. 7.2) in Wyoming. The site is dominated by Wyoming big sagebrush and bare ground. Herbaceous vegetation is located primarily beneath shrubs or cactus. This plant community phase is not providing nesting or early brood-rearing habitat for GRSG (photo by Jim Cagney, used with permission).

Grazing with cattle during the fall at appropriate levels repeatedly over time may reduce the abundance of cheatgrass and will probably not decrease the abundance of the perennial grasses. But few longer-term data exist (Schmelzer et al. 2014; Strand et al. 2014) (see fig. 7.8).

Once the perennial native herbaceous species have been depleted, recovery of perennial native grasses is likely to be a slow process in this ecological type even with long-term rest from livestock grazing (e.g., West et al. 1984). Further, once the perennial native herbaceous species have been depleted, sagebrush and other shrubs may continue to increase in abundance for a decade or more even with removal of livestock (Chambers et al. 2017b; West et al. 1984). Thus, for areas within the invaded state with moderate cover of perennial native grasses, grazing practices to maintain or increase the cover of these species is a priority.

The effects of livestock grazing on wildfire potential in the invaded and other states depend on the relative proportion of sagebrush to herbaceous fuels combined with weather conditions. The potential for grazing to be effective in reducing the risk of fire initiation and spread is greatest when sagebrush cover is low and fire weather severity is low to moderate (fig. 7.9) (Strand et al. 2014). Long-term removal of grazing may increase the likelihood of wildfire-induced mortality of perennial bunchgrasses in some ecological sites because of fuel

Cold Deserts – Mesic/aridic bordering on Xeric
Wyoming big sagebrush (8-12 in PZ)
Low to Moderate Resilience and Low Resistance



1a Perennial grass increases due to disturbances that decrease sagebrush like wildfire, insects, disease, and pathogens.

1b Sagebrush increases with time.

T2 An invasive seed source and/or improper grazing trigger an invaded state.

R2 Proper grazing, fire, herbicides and/or mechanical treatments are unlikely to result in return to the reference state on all but the coolest and wettest sites.

3a Perennial grass decreases and both sagebrush and invasives increase with improper grazing resulting in an at-risk phase. Decreases in sagebrush due to insects, disease or pathogens can further increase invasives.

3b Proper grazing and herbicides or mechanical treatments that reduce sagebrush may restore perennial grass and decrease invaders on wetter sites (10-12”). Outcomes are less certain on drier sites (8-10”) and/or low abundance of perennial grass.

T4 Improper grazing triggers a largely irreversible threshold to a sagebrush/annual state.

T5 Fire or other disturbances that remove sagebrush result in an annual state.
T7 Perennial grass is rare and recovery potential is low due to low precipitation and competition from annual invasives. Repeated fire can cause further degradation.

R6 Seeding following fire and/or invasive species control results in a seeded state.
R8 R9 Sagebrush may recolonize depending on patch size, but annual invasives are still present.

R10 Seeding effectiveness and return to the invaded state are related to site conditions, seeding mix, and post-treatment weather.

Figure 7.6—State-and-transition model for a Wyoming big sagebrush, 8–12 inch precipitation zone ecological type applicable in the Cold Deserts in the western part of the sagebrush biome and GRSG range in the Snake River Plain, Northern Basin and Range, and Central Basin and Range ecoregions (Management Zones III, IV, and V). Large boxes illustrate states that are made up of community phases (smaller boxes). Transitions among states are shown with arrows starting with T; restoration pathways are shown with arrows starting with R. The “at risk” community phase is most vulnerable to transition to an alternative state (figure source: Chambers et al. 2017a, Appendix 6).

Figure 7.7—Example of a plant community phase in the invaded state in the Wyoming big sagebrush, 8–12 inch precipitation zone ecological type (fig. 7.6) in Nevada. The plant community phase is dominated by Wyoming big sagebrush and cheatgrass with some perennial grasses. This site is not providing optimum nesting or early brood-rearing habitat for GRSG (photo by BLM).

buildup on the root crown of perennial bunchgrasses (Davies et al. 2009, 2010). While grazing may decrease fuels and reduce wildfire severity or extent in some cases (fig. 7.9), as weather conditions become extreme, the potential role of grazing in wildfire behavior decreases and may become meaningless (Strand et al. 2014).

Potential Livestock Grazing Management Practices for the Annual State

Shifts in plant communities in sagebrush ecosystems toward invasive annual grass dominance were caused in part by historical improper livestock grazing (Davies et al. 2014). However, changes in grazing practices in the annual state (figs. 7.6, 7.10) are not likely to aid conversion of annual grass-dominated plant communities back to native species-dominated communities (Davies et al. 2014; Strand et al. 2014). Similarly, changes in grazing practices in the annual state cannot be used to achieve vegetation habitat objectives for nesting and brood-rearing seasonal habitat for GRSG in Oregon and Washington (USDOI BLM 2015g, table 2-2), Utah (USDOI BLM 2015i, table 2-2), Nevada and northeastern California (USDOI BLM 2015e, table 2-2), and Idaho and southwestern Montana (USDOI BLM 2015b, table 2-2).

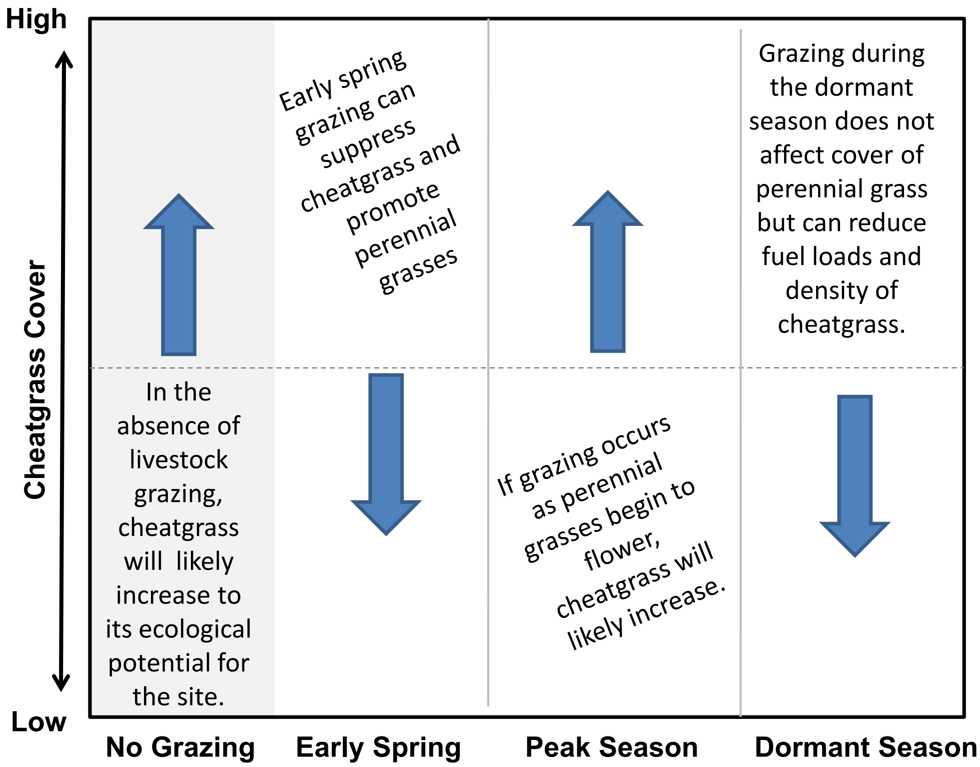


Figure 7.8—Conceptual depiction of how livestock grazing can influence cheatgrass abundance in sagebrush dominated ecosystems with a significant component of perennial grasses. Grazing can suppress or promote cheatgrass depending primarily on the season of grazing. Grazing suppresses cheatgrass when applied (1) in early spring when annuals begin to produce seeds and before native perennial grasses initiate bolting, and (2) during the dormant season (figure source: Strand et al. 2014, used with permission).

Grazing of the annual state can be effective in reducing the risk of fire initiation and spread (fig. 7.9). Targeted grazing, or the application of a specific kind of livestock at a determined season, duration, and intensity, can be used to achieve defined vegetation or broad-scale goals within annual states (Launchbaugh and Walker 2006; Mosley and Roselle 2006). For example, intense sheep grazing of cheatgrass dominated sites can effectively suppress or even eliminate cheatgrass stands in as little as 2 years, as was done in the urban interface above Carson City, Nevada (Mosley 1994). Managed grazing may also reduce the risk and extent of wildfire in cheatgrass dominated areas (Diamond et al. 2009, 2012; Walker 2006).

In sagebrush ecosystems, high intensity targeted grazing may best be used to create firebreaks by confining livestock to a strip of land with temporary fencing. This type of grazing may reduce the spread of wildfire by reducing herbaceous vegetation (fine fuels that carry fire) (Walker 2006). Further, because livestock tend to graze some areas more intensely than others, grazing may create patchy vegetation that reduces the continuity of fuel loads and the fires (Walker 2006). However, this reduction in fuel continuity is influenced strongly by multi-year precipitation patterns (Pilliod et al. 2017) and timing of grazing.

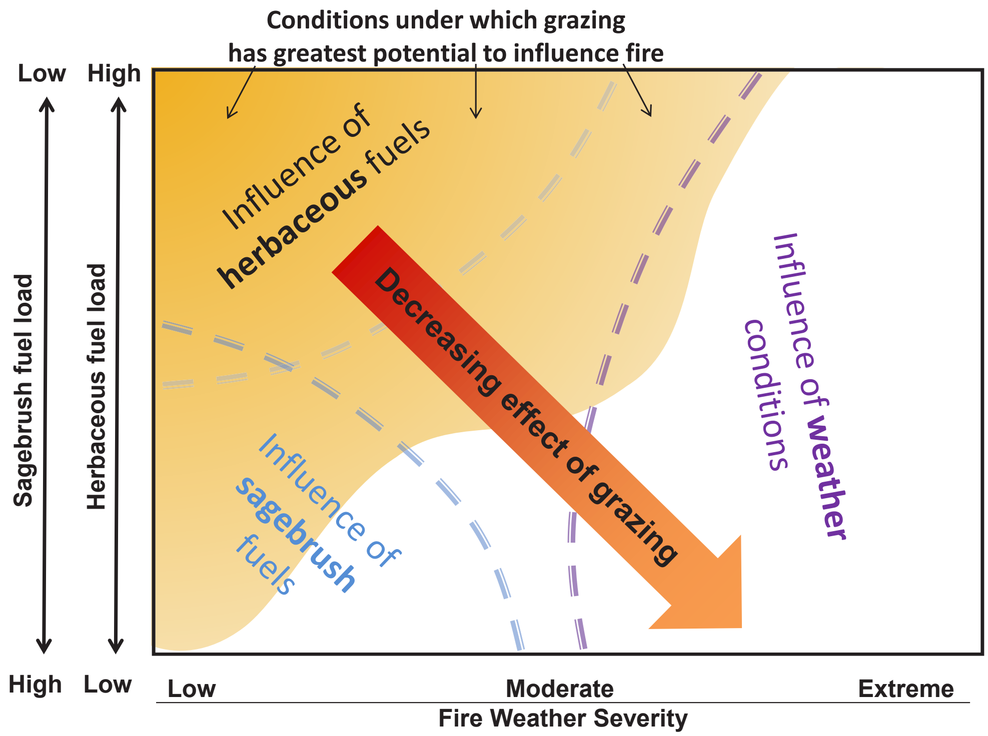


Figure 7.9—Conceptual model illustrating how the potential for grazing to influence fire behavior occurs along continuums of fuel and weather conditions. Fuel composition is displayed on the y-axis and fire weather condition is displayed on the x-axis. Low fire weather severity is characterized by high fuel moistures, high relative humidity, low temperature, and low wind speeds, whereas extreme fire weather is characterized by the opposite conditions. The potential for grazing to be effective in reducing the risk of fire initiation and spread is greatest when the sagebrush cover is low and the fire weather severity is low to moderate (figure source: Strand et al. 2014, used with permission).

Effective grazing programs for invasive plant control require a clear statement of the kind of animal and timing and rate of grazing necessary to suppress the invasive plant (Launchbaugh and Walker 2006). A successful targeted grazing prescription should: (1) cause significant reductions in the target plant(s), (2) limit effects to the surrounding vegetation, and (3) be integrated with other control methods as part of an overall management strategy. Because targeted grazing by livestock is typically focused on heavily invaded areas, follow-up management, such as seeding the target area with the desired perennial species, may be needed.

Potential Livestock Grazing Management Practices for the Seeded State

After wildfire, areas within the Wyoming big sagebrush, 8–12 inch (20–30 centimeter) precipitation zone that support GRSG are often a priority for seeding because residual perennial native grasses are typically insufficient to promote recovery (fig. 7.11). Seeding with a diverse mix of native shrubs, grasses, and forbs can increase resilience to disturbance as well as resistance to invasive annual grasses through increased competition with the invaders over the long term (see section 6).

Grazing rest and deferment schedules are needed to ensure establishment of the seeded species and recovery of the site after postwildfire rehabilitation (Pyke et al. 2017). Newly seeded and surviving plants are at risk of repeated defoliation due to animal preference for foraging in burned areas (Veblen et al. 2015). Thus,

Figure 7.10—Example of a plant community phase in the annual state in the Wyoming big sagebrush, 8–12 inch precipitation zone ecological type (fig. 7.6). The plant community phase is dominated by exotic annual grasses and forbs such as cheatgrass, medusahead (*Taeniatherum caput-medusae*), and tumblemustard (*Thelypodiopsis* spp.). The site is located in the Jackies Butte allotment in the Jordan Resource Area of the BLM's Vale District in Oregon. This site is not providing nesting or early brood-rearing habitat for GRSG (photo by Jon Sadowski, used with permission).

grazing should be resumed only after perennial grasses have established and are producing viable seed at levels equal to grasses on unburned sites. Failure to implement a program of grazing rest or deferment may slow or prevent site recovery (Kerns et al. 2011) and promote invasive annual grasses and other undesirable plants.

Once postfire grazing resumes on a site, use should be deferred until after seed maturity or shatter to promote bunchgrass recovery (Bates et al. 2009; Bruce et al. 2007). In addition, postfire grazing after rest or during deferment periods will probably need to be lighter than grazing recommendations for unburned areas, which are no more than 50 percent utilization during active growth, and no more than 60 percent during dormancy (Guinn and Rouse 2009). Under certain conditions (e.g., in warm or dry areas, after high severity fires, or during low precipitation years), even lower utilization may be required to allow seeded species to establish and soils to recover. Options for mitigating livestock distribution problems in large grazing units include fencing, herding, and strategic placement of water, salt, and supplements.

Careful monitoring and assessment is an integral part of a grazing program to determine when grazing may be resumed, whether postfire grazing management has been effective, and whether changes in grazing management are needed.

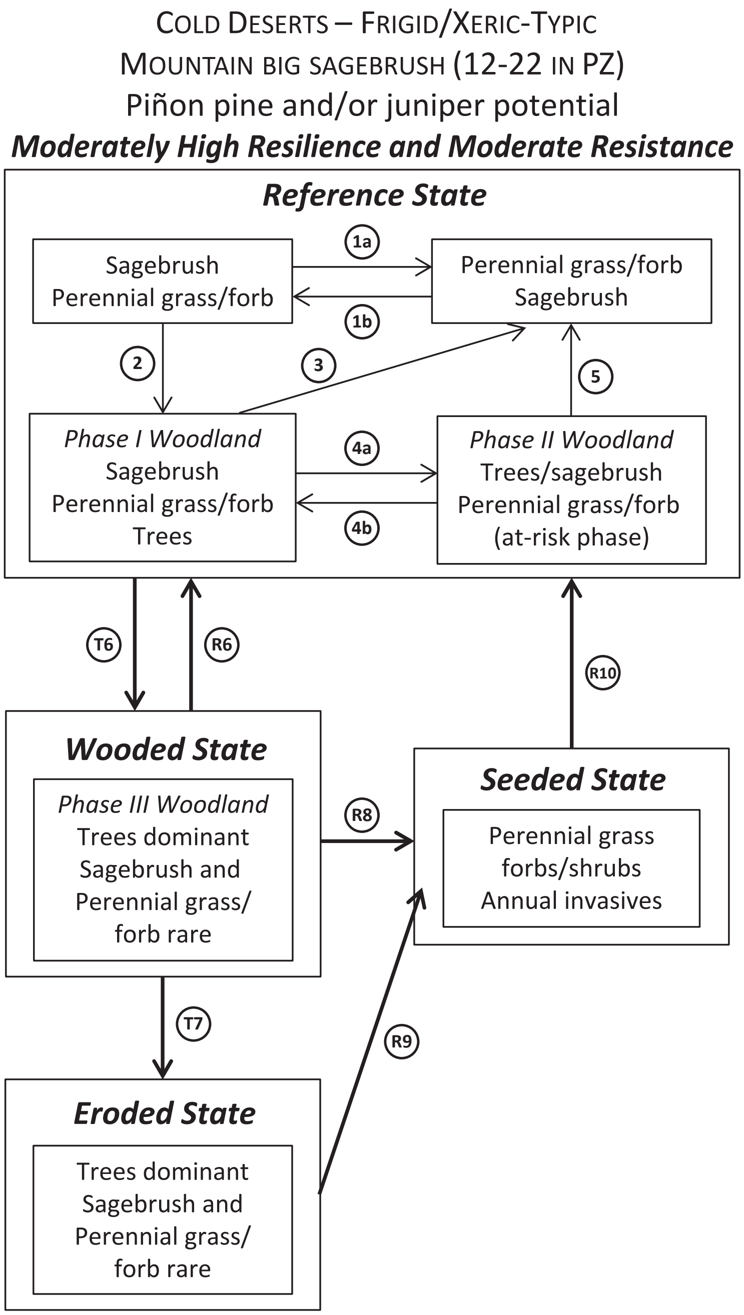
Figure 7.11—Example of a plant community phase in the seeded state in the Wyoming big sagebrush, 8–12 inch precipitation zone ecological type (fig. 7.6). Plant community phase is a seeding dominated by Fairway crested wheatgrass. The site is located in the Jackies Butte allotment in the Jordan Resource Area of the BLM's Vale District in Oregon. This site is not providing nesting or early brood-rearing habitat for GRSG (photo by Jon Sadowski, used with permission).

Cold Deserts—Frigid/Xeric-Typic Mountain Big Sagebrush with Piñon Pine and/or Juniper Potential (Management Zones III, IV, and V)

Potential Livestock Grazing Management Practices for the Reference State—Phase I and II Woodland

Managing livestock grazing in plant communities with phase I and II juniper and piñon in the reference state (figs. 7.12, 7.13) to maintain perennial grasses can decrease the rates of juniper and piñon expansion and infill into adjacent sagebrush ecosystems (Guenther et al. 2004; Madany and West 1983; Shinneman and Baker 2009; Soulé et al. 2004). Grazing management to maintain perennial grasses can increase the resilience of these plant communities and their capacity to recover after wildfire (Chambers et al. 2014a). It can also increase resistance to invasive annual grasses on warmer and drier sites (Chambers et al. 2014a,b).

In studies that compared adjacent grazed and historically ungrazed areas, juniper and piñon densities, canopy cover, or basal area were greater in the grazed than ungrazed pastures (Guenther et al. 2004; Madany and West 1983; Shinneman and Baker 2009; Soulé et al. 2004). Further, shrubs often act as nurse plants for juniper and piñon by modifying temperatures and increasing resource availability (Chambers 2001; Johnsen 1962; Miller and Rose 1995; Soulé and Knapp 2000, Soulé et al. 2004). Shrub abundance can increase after fire in



1a Disturbances such as wildfire, insects, disease, and pathogens result in less sagebrush and more perennial grass/forb.

1b Sagebrush increases with time.

2 Time combined with seed sources for piñon and/or juniper trigger a Phase I Woodland.

3, 5 Fire and or fire surrogates (herbicides and/or mechanical treatments) that remove trees may restore perennial grass/forb and sagebrush dominance.

4a Increasing tree abundance results in a Phase II woodland with depleted perennial grass/forb and shrubs and an at-risk phase.

4b Fire surrogates (herbicides and/or mechanical treatments) that remove trees may restore perennial grass/forb and sagebrush dominance.

T6 Infilling of trees and/or improper grazing can result in a biotic threshold crossing to a wooded state with increased risk of high severity crown fires.

R6 Fire, herbicides and/or mechanical treatments that remove trees may restore perennial grass/forb and sagebrush dominance.

T7 An irreversible abiotic threshold crossing to an eroded state can occur depending on soils, slope, and understory species.

R8, R9 Seeding after treatments or fire may be required on sites with depleted perennial grass/forb, but seeding with aggressive introduced species can decrease native perennial grass/forb. Annual invasives are typically rare. Seeded eroded states may have lower productivity.

R10 Depending on seed mix and grazing, return to the reference state may be possible if an irreversible threshold has not been crossed.

Figure 7.12—State-and-transition model for a mountain big sagebrush, 12–22 inch precipitation zone ecological type applicable in the Cold Deserts in the western part of the sagebrush biome and GRSG range in the Snake River Plain, Northern Basin and Range, and Central Basin and Range ecoregions (Management Zones III, IV, and V). Large boxes illustrate states that are made up of community phases (smaller boxes). Transitions among states are shown with arrows starting with T; restoration pathways are shown with arrows starting with R. The “at risk” community phase is most vulnerable to transition to an alternative state (figure source: Chambers et al. 2017a, Appendix 6).

Figure 7.13—Example of a phase II woodland plant community in the reference state of the mountain big sagebrush, 12–22 inch precipitation zone ecological type (fig. 7.12) in Nevada. This woodland is dominated by piñon pine. Piñon pine is continuing to expand and increase in density and canopy cover, and mountain big sagebrush and bluebunch wheatgrass (*Pseudoroegneria spicata*) are declining in canopy cover. This plant community phase is not providing nesting or early brood-rearing habitat for GRSG (photo by Jeanne Chambers).

response to grazing that removes perennial grasses in mountain big sagebrush ecological types (Chambers et al. 2017b). A recent simulation model that evaluated woodland expansion across the Intermountain West identified grazing as the key factor leading to juniper expansion through reduction of perennial grass and shrub cover as well as decreases in fire occurrence (Caracciolo et al. 2017).

Areas with more than 2 percent conifer cover severely compromise GRSG habitat use and can result in greater bird mortality (Coates et al. 2017; Severson et al. 2016). Thus, changes in grazing management alone in phase I or phase II plant communities in the reference state (figs. 7.12, 7.13) cannot be used to achieve vegetation habitat objectives for nesting and brood-rearing seasonal habitat for GRSG in Oregon and Washington (USDOI BLM 2015g, table 2-2), Utah (USDOI BLM 2015i, table 2-2), Nevada and northeastern California (USDOI BLM 2015e, table 2-2), and Idaho and southwestern Montana (USDOI BLM 2015b, table 2-2). However, phase I and phase II expansion woodlands are often targeted for conifer removal treatments to improve GRSG habitat. Treatments may include cutting and leaving the trees, shredding or masticating the trees, and in some cases, prescribed fire. Bunchgrasses and other perennial vegetation may exhibit increases in cover, but may take several years to fully

Figure 7.14—Example of a plant community phase in the wooded state in the mountain big sagebrush, 12–22 inch precipitation zone ecological type (fig. 7.12) in Nevada. The site is a phase III woodland dominated by piñon pine that was dominated in the past by sagebrush and Thurber's needlegrass (*Achnatherum thurberianum*). This plant community phase is not providing nesting or early brood-rearing habitat for GRSG (photo by Jeanne Chambers).

recover, especially on warmer and drier sites and following prescribed fire (Williams et al. 2017). During the recovery period, many of the same livestock grazing management practices as used after fire and rehabilitation seeding may be used, including rest and deferment, decreased levels of utilization, changes in the timing of livestock grazing, and increased emphasis on livestock distribution.

Potential Livestock Grazing Management Practices for the Wooded State—Phase III Woodland

Because GRSG do not use phase III woodland (fig. 7.14) (Severson et al. 2017), changes in grazing management alone cannot be used to achieve vegetation habitat objectives for nesting and brood-rearing seasonal habitat for GRSG in the wooded state in Oregon and Washington (USDOI BLM 2015g, table 2-2), Utah (USDOI BLM 2015i, table 2-2), Nevada and northeastern California (USDOI BLM 2015e, table 2-2), and Idaho and southwestern Montana (USDOI BLM 2015b, table 2-2). However, following wildfire and postfire rehabilitation seeding or tree removal in these areas to increase connectivity of sagebrush habitat, many of the same livestock grazing management practices as used after wildfire and postfire rehabilitation seeding may be used.

Conclusions

Livestock grazing management is a critical aspect of maintaining and improving resilience to disturbance and resistance to invasive annual grasses in sagebrush ecosystems and GRSG habitat. Livestock grazing has well-recognized effects on ecosystem structure and function that vary among ecoregions and GRSG Management Zones. Consideration of the potential regional variation in grass-related variables and the effects associated with plant phenology can help in the development of management prescriptions for livestock grazing to attain habitat objectives within nesting habitats. Potential livestock grazing management practices designed to improve sagebrush habitats can be incorporated into livestock grazing management alternatives during the grazing authorization (grazing permits and grazing leases) renewal process, which is ongoing within the BLM (USDOI BLM 2017a) and other agencies. Specific vegetation habitat objectives for breeding and nesting seasonal habitat, and brood-rearing and summer seasonal habitat, have been developed by BLM and its partners. But it may be necessary to modify and update these as additional information on GRSG habitat requirements and ecological site potentials to support GRSG habitat become available and additional policy direction is provided (USDOI BLM 2017b).

The Science Framework provides an approach for managing sagebrush ecosystems based on their relative resilience and resistance to invasive annual grasses. This approach can be used to evaluate the likely response of an area to disturbance or management actions and the capacity of an area to support target species or resources at the mid-scale. At the local scale, ecological types or ecological site descriptions and their associated STMs can be used to evaluate current ecological dynamics and determine appropriate livestock grazing management practices. In this section, examples of ecological types and STMs illustrate the use of these tools for identifying livestock grazing management practices that can be implemented to maintain or improve the resilience and resistance of sagebrush plant communities and the quality of GRSG nesting and early brood-rearing habitat.

References

Adams, B.W.; Carlson, J.; Milner, D.; [et al.]. 2004. Beneficial grazing management practices for sage-grouse (*Centrocercus urophasianus*) and ecology of silver sagebrush (*Artemisia cana*) in southeastern Alberta. Tech. Rep. Publication No. T/049. Lethbridge, Alberta, Canada: Public Lands and Forests Division, Alberta Sustainable Resource Development. 60 p. https://albertawilderness.ca/wordpress/wp-content/uploads/20040101_rp_goa_BeneficialGrazingManagement-SageGrouse-SilverSagebrush.pdf. [Accessed May 23, 2018].

Bates, J.D.; Rhodes, E.C.; Davies, K.W.; [et al.]. 2009. Postfire succession in big sagebrush steppe with livestock grazing. Rangeland Ecology and Management. 62: 98–110.

Beck, J.L.; Mitchell, D.L. 2000. Influences of livestock grazing on sage grouse habitat. Wildlife Society Bulletin. 28: 993–1002.

Bestelmeyer, B.T.; Tugel, A.J.; Peacock, G.L., Jr.; [et al.]. 2009. State-and-transition models for heterogeneous landscapes: A strategy for development and application. Rangeland Ecology and Management. 62: 1–15.

Boyd, C.S.; Beck, J.L.; Tanaka, J.A. 2014. Livestock grazing and sage-grouse habitat: Impacts and opportunities. Journal of Rangeland Applications. 1: 58–77.

Briske, D.D.; Fuhlendorf, S.D.; Smeins, F.E. 2005. State-and-transition models, thresholds, and rangeland health: A synthesis of ecological concepts and perspectives. Rangeland Ecology and Management. 58: 1–10.

Briske, D.D.; Richards, J.H. 1995. Plant responses to defoliation: A physiological, morphological, and demographic evaluation of individual plants to grazing: Current status and ecological significance. In: Bedunah, D.J.; Sosebee, R.E., eds. Wildland plants: Physiological ecology and developmental morphology. Denver, CO: Society for Range Management: 635–710.

Briske, D.D.; Sayre, N.F.; Huntsinger, L.; [et al.]. 2011. Origin, persistence, and resolution of the rotational grazing debate: Integrating human dimensions into rangeland research. Rangeland Ecology and Management. 64: 325–334.

Bruce, L.B.; Perryman, B.; Conley, K.; [et al.]. 2007. Grazing management on seeded and unseeded post-fire public rangelands. Professional Animal Scientist. 23: 285–290.

Cagney, J.; Bainter, E.; Budd, B.; [et al.]. 2010. Grazing influence, objective development, and management in Wyoming's Greater sage-grouse habitat, with emphasis on nesting and early brood rearing. Extension Bulletin B-1203. Laramie, WY: University of Wyoming, Cooperative Extension Service. 60 p.

Caracciolo, D.; Istanbulluoglu, E.; Noto, L.V. 2017. An ecohydrological cellular automata model investigation of juniper tree encroachment in a western North America landscape. Ecosystems. 20: 1104–1123.

Caudle, D.; DiBenedetto, J.; Karl, M.; [et al.]. 2013. Interagency ecological site handbook for rangelands. Washington, DC: U.S. Department of the Interior, Bureau of Land Management; U.S. Department of Agriculture, Forest Service and Natural Resources Conservation Service. http://directives.sc.egov.usda.gov/viewerFS.aspx?hid=33150. [Accessed May 23, 2018].

Chambers, J.C. 2001. *Pinus monophylla* establishment in an expanding pinon-juniper woodland: Environmental conditions, facilitation and interacting factors. Journal of Vegetation Science. 12: 27–40.

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017a. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p.

Chambers, J.C.; Board, D.I.; Roundy, B.A.; [et al.]. 2017b. Removal of perennial herbaceous species affects response of cold desert shrublands to fire. Journal of Vegetation Science. 28(5): 975–984.

Chambers, J.C.; Bradley, B.A.; Brown, C.A.; [et al.]. 2014a. Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. invasion in the cold desert shrublands of western North America. Ecosystems. 17: 360–375.

Chambers, J.C.; Miller, R.F.; Board, D.I.; [et al.]. 2014b. Resilience and resistance of sagebrush ecosystems: Implications for state and transition models and management treatments. Rangeland Ecology and Management. 67: 440–454.

Coates, P.S.; Prochazka, B.G.; Ricca, M.A.; [et al.]. 2017. Pinyon and juniper encroachment into sagebrush ecosystems impacts distribution and survival of Greater sage-grouse. Rangeland Ecology and Management. 70: 25–38.

Davies, K.W.; Bates, J.D.; Svejcar, T.J.; [et al.]. 2010. Effects of long-term livestock grazing on fuel characteristics in rangelands: An example from the sagebrush steppe. Rangeland Ecology and Management. 63: 662–669.

Davies, K.W.; Svejcar, T.J.; Bates, J.D. 2009. Interaction of historical and nonhistorical disturbances maintains native plant communities. Ecological Applications. 19: 1536–1545.

Davies, K.W.; Vavra, M.; Schultz, B.W.; [et al.]. 2014. Implications of longer term rest from grazing in the sagebrush steppe. Journal of Rangeland Applications. 1: 14–34.

Diamond, J.M.; Call, C.A.; Devoe, N. 2009. Effects of targeted cattle grazing on fire behavior of cheatgrass-dominated rangeland in the northern Great Basin, USA. International Journal of Wildland Fire. 18: 944–950.

Diamond, J.M.; Call, C.A.; Devoe, N. 2012. Effects of targeted grazing and prescribed burning on community and seed dynamics of a downy brome (*Bromus tectorum*)-dominated landscape. Invasive Plant Science and Management. 5: 259–269.

Dinkins, J.B.; Smith, K.T.; Beck, J.L.; [et al.]. 2016. Microhabitat conditions in Wyoming's sage-grouse core areas: Effects on nest site selection and success. PLoS ONE 11: e0150798.

Doherty, K.E.; Beck, J.L.; Naugle, D.E. 2011. Comparing ecological site descriptions to habitat characteristics influencing greater sage-grouse nest site occurrence and success. Rangeland Ecology and Management. 64: 344–351.

Doherty, K.E.; Evans, J.S.; Coates, P.S.; [et al.]. 2016. Importance of regional variation in conservation planning: A range-wide example of the Greater sage-grouse. Ecosphere. 7: e01462.

Doherty, K.E.; Naugle, D.E.; Tack, J.D.; [et al.]. 2014. Linking conservation actions to demography: Grass height explains variation in greater sage-grouse nest survival. Wildlife Biology. 20: 320–325.

Federal Register [FR]. 2005. 43 CFR 4180.2. Standards and guidelines for grazing administration [68 FR 33804. June 5, 2003.]. https://www.gpo.gov/fdsys/granule/CFR-2005-title43-vol2/pdf/CFR/2005-title43-vol2-sec4180-2.pdf. [Accessed Oct. 30, 2018].

Federal Register [FR]. 2012. 43 CFR 4110.3. Implementing changes in active use. https://www.gpo.gov/fdsys/granule/CFR-2012-title43-vol2/pdf/CFR/2012-title43-vol2-sec4110-3-2.pdf. [Accessed Oct. 25, 2018].

Freilich, J.E.; Emlen, J.M.; Duda, J.J.; [et al.]. 2003. Ecological effects of ranching: A six-point critique. BioScience. 53(8): 759–765.

Fuhlendorf, S.D.; Engle, D.M. 2001. Restoring heterogeneity on rangelands: Ecosystem management based on evolutionary grazing patterns. BioScience. 51(8): 625–632.

Guenther, D.; Stohlgren, T.J.; Evangelista, P. 2004. A comparison of a near-relict site and a grazed site in a pinyon-juniper community in the Grand Staircase-Escalante National Monument, Utah. In: Van Riper, C.; Cole, K.L., eds. The Colorado Plateau: Cultural, biological and physical research. Tucson, AZ: University of Arizona Press: 153–162.

Guinn, K.; Rouse, G. 2009. Grazing management guidelines. Range Tech. Note-34, revised. Spokane, WA: U.S. Department of Agriculture, Natural Resources Conservation Service. 10 p.

Hagen, C.A.; Connelly, J.W.; Schroeder, M.A. 2007. Meta-analysis of Greater sage-grouse *Centrocercus urophasianus* nesting and brood-rearing habitats. Wildlife Biology. 13: 42–50.

Hockett, G.A. 2002. Livestock impacts on the herbaceous components of sage grouse habitat: A review. Intermountain Journal of Sciences. 8: 105–114.

Huber-Sannwald, E.; Pyke, D.A. 2005. Establishing native grasses in a big sagebrush-dominated site: An intermediate restoration step. Restoration Ecology. 13: 292–301.

Johnsen, T.N., Jr. 1962. One-seed juniper invasion of northern Arizona grasslands. Ecological Monographs. 32: 187–207.

Kerns, B.K.; Buonopane, M.; Thies, W.G.; [et al.]. 2011. Reintroducing fire into a ponderosa pine forest with and without cattle grazing: Understory vegetation response. Ecosphere. 2: 59.

Kirol, C.P.; Beck, J.L.; Dinkins, J.B.; [et al.]. 2012. Microhabitat selection for nesting and brood-rearing by the Greater sage-grouse in xeric big sagebrush. The Condor. 114: 75–89.

Knick, S.T.; Hanser, S.E.; Miller, R.F.; [et al.]. 2011. Ecological influence and pathways of land use in sagebrush. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse—Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 203-251.

Launchbaugh, K.; Walker, J. 2006. Chapter 1. Targeted grazing—A new paradigm for livestock management. In: Launchbaugh, K.; Walker, J., eds. Targeted grazing: A natural approach to vegetation management and landscape enhancement. Englewood, CO: American Sheep Industry Association: 2–8.

Madany, M.H.; West, N.E. 1983. Livestock grazing-fire regime interactions within montane forests of Zion National Park, Utah. Ecology. 64: 661–667.

Maestas, J.D.; Campbell, S.B.; Chambers, J.C.; [et al.]. 2016. Tapping soil survey information for rapid assessment of sagebrush ecosystem resilience and resistance. Rangelands. 38: 120–128.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2014. A field guide to selecting the most appropriate treatments in sagebrush and pinyon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322-rev. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 68 p.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2015. A field guide for rapid assessment of post-wildfire recovery potential in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-338. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 70 p.

Miller, R.F.; Rose, J.A. 1995. Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. Great Basin Naturalist. 55: 37–45.

Mosley, J.C. 1994. Prescribed sheep grazing to suppress cheatgrass: A review. Sheep Research Journal. 12: 79-91.

Mosley, J.C.; Roselle, L. 2006. Chapter 8. Targeted livestock grazing to suppress invasive annual grasses. In: Launchbaugh, K.; Walker, J., ed. Targeted grazing: A natural approach to vegetation management and landscape enhancement. Denver, CO: American Sheep Association: 67–72.

Pilliod, D.S.; Welty, J.L.; Arkle, R.S. 2017. Refining the cheatgrass-fire cycle in the Great Basin: Precipitation timing and fine fuel composition predict wildfire trends. Ecology and Evolution. 7: 8126–8151.

Pyke, D.A.; Chambers, J.C.; Pellant, M.; [et al.]. 2017. Restoration handbook for sagebrush steppe ecosystems with emphasis on greater sage-grouse habitat—Part 3. Site level restoration decisions. Circular 1426. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 62 p. https://www.treesearch.fs.fed.us/pubs/53743. [Accessed Sept. 20, 2017].

Schmelzer, L.; Perryman, B.; Bruce, B.; [et al.]. 2014. Case study: Reducing cheatgrass (*Bromus tectorum* L.) fuel loads using fall cattle grazing. The Professional Animal Scientist. 30: 270–278.

Severson, J.P.; Hagen, C.A.; Maestas, J.D.; [et al.]. 2016. Effects of conifer expansion on Greater sage-grouse nesting habitat selection. Journal of Wildlife Management. 81: 86–95.

Severson, J.P.; Hagen, C.A.; Maestas, J.D.; [et al.]. 2017. Short-term response of sage-grouse nesting to conifer removal in the northern Great Basin. Rangeland Ecology and Management. 70: 50–58.

Shinneman, D.J.; Baker, W.J. 2009. Environmental and climatic variables as potential drivers of post-fire cover of cheatgrass (*Bromus tectorum*) in seeded and unseeded semiarid ecosystems. International Journal of Wildland Fire. 18: 191–202.

Smith, J.T. 2016. Landscape to local: A multi-scale evaluation of voluntary efforts to reduce fragmentation and enhance management of rangelands for sage-grouse. Dissertation. Missoula, MT: University of Montana. 140 p. http://scholarworks.umt.edu/etd/10902. [Accessed May 23, 2018].

Soulé, P.T.; Knapp, A.K. 2000. *Juniperus occidentalis* (western juniper) establishment history on two minimally disturbed research natural areas in central Oregon. Western North American Naturalist. 60: 26–33.

Soulé, P.T.; Knapp, P.A.; Grissino-Mayer, H.D. 2004. Human agency, environmental drivers, and western juniper establishment during the late Holocene. Ecological Applications. 14: 96–112.

Stiver, S.J.; Rinkes, E.T.; Naugle, D.E.; [et al.], eds. 2015. Sage-grouse habitat assessment framework: A multiscale assessment tool. Technical Reference 6710-1. Denver, CO: U.S. Department of the Interior, Bureau of Land Management and Western Association of Fish and Wildlife Agencies.

Strand, E.K.; Launchbaugh, K.L.; Limb, R.; [et al.]. 2014. Livestock grazing effects on fuel loads for wildland fire in sagebrush dominated ecosystems. Journal of Rangeland Applications. 1: 35–57.

Stringham, T.K.; Krueger, W.C.; Shaver, P.L. 2003. State and transition modeling: An ecological process approach. Journal of Range Management. 56: 106–113.

Thompson, K.M.; Holloran, M.J.; Slater, S.J.; [et al.]. 2006. Early brood-rearing habitat use and productivity of Greater sage-grouse in Wyoming. Western North American Naturalist. 66: 332–342.

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2015. Ecological site descriptions. Washington, DC: U.S. Department of Agriculture, Natural Resources Conservation Service. https://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/ecoscience/desc/. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2015a. HiLine District Office Greater sage-grouse approved resource management plan. BLM/MT/PL-15/012+1610. Washington, DC: U.S. Department of the Interior, Bureau of Land Management, HiLine District Office. https://eplanning.blm.gov/epl-front-office/projects/lup/68346/88890/106398/HiLine_ARMP_2015.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2015b. Idaho and southwestern Montana greater sage-grouse approved resource management plan amendment. BLM/ID/SG/EIS-15+1610. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office. https://eplanning.blm.gov/epl-front-office/projects/lup/31652/63337/68679/IDMT_ARMPA_print.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2015c. Miles City Field Office approved resource management plan. BLM/MT/PL-15/010+1610. Miles City, MT: U.S. Department of the Interior, Bureau of Land Management, Miles City Field Office. https://eplanning.blm.gov/epl-front-office/projects/lup/59042/86804/104007/Miles_City_Field_Office_Approved_Resource_Management_Plan_(2015).pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2015d. Lewistown Field Office greater sage-grouse approved resource management plan amendment. BLM/MT/PL-15/009+1610. Billings, MT: U.S. Department of the Interior, Bureau of Land Management, Montana State Office. https://eplanning.blm.gov/epl-front-office/projects/lup/36877/63212/68461/LFO_ARMPA_camera-508.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI, BLM]. 2015e. Nevada and northeastern California greater sage-grouse approved resource management plan amendment. BLM/NV/NV/PL/15-14+1600. Reno, NV: U.S. Department of the Interior, Bureau of Land Management, Nevada State Office. https://eplanning.blm.gov/epl-front-office/projects/lup/21152/63235/68484/NVCA_Approved_RMP_Amendment.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2015f. North Dakota greater sage-grouse approved resource management plan amendment. BLM/MT/PL-15/008+1610. Dickinson, ND: U.S. Department of the Interior, Bureau of Land Management, North Dakota Field Office. https://eplanning.blm.gov/epl-front-office/projects/lup/36811/63113/68356/NDFO_ARMPA_camera_508.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2015g. Oregon greater sage-grouse approved resource management plan amendment. BLM/OR/WA/PL-15/051+1792. Portland, OR: U.S. Department of the Interior, Bureau of Land Management, Oregon/Washington State Office. https://www.blm.gov/or/energy/opportunity/files/or_armpa.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2015h. South Dakota approved resource management plan. BLM/MT/PL-15/013+1610. Belle Fourche, SD: U.S. Department of the Interior, Bureau of Land Management, South Dakota Field Office. https://www.blm.gov/sites/blm.gov/files/South%20Dakota%20Approved%20RMP.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2015i. Utah greater sage-grouse approved resource management plan amendment. DOI-BLM-UT-9100-2013-0002-EIS. Salt Lake City, UT: U.S. Department of the Interior, Bureau of Land Management, Utah State Office. https://eplanning.blm.gov/epl-front-office/projects/lup/68351/87600/104856/Utah_ARMPA.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2015j. Bureau of Land Management Casper, Kemmerer, Newcastle, Pinedale, Rawlins, and Rock Springs Field Offices approved resource management plan amendment for greater sage-grouse. Cheyenne, WY: U.S. Department of the Interior, Bureau of Land Management, Wyoming State Office. https://eplanning.blm.gov/epl-front-office/projects/lup/9153/63189/68431/002_Wyoming_ARMPA_Main-Body.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management, Washington Office [USDOI BLM]. 2017a. Instruction Memorandum No. 2018-024. Setting priorities for review and processing of grazing authorizations and related livestock grazing monitoring. Washington, DC: U.S. Department of the Interior, Bureau of Land Management. https://www.blm.gov/policy/im-2018-024. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2017b. Instruction Memorandum No. 2018-025. Implementation of the habitat objectives table from the 2015 Greater Sage-Grouse Approved Resource Management Plans and Amendments. Washington, DC: U.S. Department of the Interior, Bureau of Land Management. https://www.blm.gov/policy/im-2018-025. [Accessed May 23, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2013. Greater sage-grouse (*Centrocercus urophasianus*) Conservation Objectives: Final Report. Denver, CO: U.S. Department of the Interior, Fish and Wildlife Service. 91 p. https://www.fws.gov/greatersagegrouse/documents/COT-Report-with-Dear-Interested-Reader-Letter.pdf. [Accessed May 23, 2018].

Veblen, K.E.; Newingham, B.A.; Bates, J.; [et al.]. 2015. Post-fire grazing management in the Great Basin. Great Basin Factsheet Series. Number 7. Reno, NV: Great Basin Fire Science Exchange.

Walker, J. 2006. Chapter 12. Targeted grazing to manage fire risk. In: Launchbaugh, K.; Walker, J., eds. Targeted grazing: A natural approach to vegetation management and landscape enhancement. Englewood, CO: American Sheep Industry Association: 107–113.

West, N.E.; Provenza, F.D.; Johnson, P.S.; [et al.]. 1984. Vegetation change after 13 years of livestock grazing exclusion on sagebrush semidesert in west central Utah. Journal of Range Management. 37: 262–264.

West, N.E.; Yorks, T.P. 2002. Vegetation responses following wildfire on grazed and ungrazed sagebrush semi-desert. Rangeland Ecology and Management. 55: 171–181.

Williams, R.E.; Roundy, B.; Hulet, A.; [et al.]. 2017. Pretreatment tree dominance and conifer removal treatments affect plant succession in sagebrush communities. Rangeland Ecology and Management. 70: 759–773.

8. WILD HORSE AND BURRO CONSIDERATIONS

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Introduction

Wild horses (*Equus caballus*) and wild burros (*E. asinus*), like domestic livestock, can alter sagebrush ecosystem structure and composition and affect habitat quality for sagebrush dependent species (Beever and Aldridge 2011). The presence of Federally protected wild horses and wild burros can also have substantial effects on the capacity for habitat restoration efforts to achieve conservation and restoration goals. In the Conservation Objectives Team Report (USDOI FWS 2013), the presence of wild horses and burros was considered a threat to Greater sage-grouse (*Centrocercus urophasianus*; hereafter, GRSG) habitat quality, particularly in the sage-grouse's western range (USDOI FWS 2013). Four years after the Conservation Objectives Team Report was published, wild horse population sizes on Bureau of Land Management (BLM) and Forest Service lands have almost doubled (USDOI BLM 2017).

Lands with Federally protected wild horses and burros are managed for multiple uses, so it can be difficult to separate their ecological effects. However, scientific studies designed to isolate the effects of various land uses lead to the conclusion that landscapes with greater wild horse and burro abundance tend to have lower resilience to disturbance and resistance to invasive plants than similar landscapes with herds at or below target levels (Beever and Aldridge 2011; Chambers et al. 2017 [hereafter, Part 1], section 5.3.8). Many studies corroborate the general understanding that wild horses can lead to biologically significant changes in sagebrush ecosystems, particularly when their populations are overabundant relative to forage and water resources. In the Great Basin, areas without wild horses had higher shrub cover, plant cover, species richness, native plant cover, and overall plant biomass, and lower cover of grazing-tolerant, unpalatable, and invasive plant species such as cheatgrass (*Bromus tectorum*), when compared to areas with horses (Beever et al. 2008; Boyd et al. 2017; Davies et al. 2014; Smith 1986; Zeigenfuss et al. 2014). There were also measurable increases in soil penetration resistance and erosion, decreases in ant mound and granivorous small mammal densities, and changes in reptile communities (Beever et al. 2003; Beever and Brussard 2004; Beever and Herrick 2006; Ostermann-Kelm et al. 2009).

Top: Wild horses at Cherry Spring in the Maverick-Medicine Herd Management Area, Nevada (photo: USDOI Bureau of Land Management). Middle left: Wild horses in Divide Basin Horse Management Area, Wyoming (photo: USDOI Bureau of Land Management). Middle right: Wild burros at Wood Hills spring in the Elko, Nevada, BLM District (photo: USDOI Bureau of Land Management). Bottom left: Wild Horses at Victoria spring in the Antelope Triple B complex (photo: USDOI Bureau of Land Management). Bottom right: Wild horse gather by the BLM (photo: USDOI Bureau of Land Management).

Wild horses can have severe impacts on water source quality, aquatic ecosystems, and riparian communities (Barnett 2002; Beever and Brussard 2000; Earnst et al. 2012; Kaweck 2016; Nordquist 2011; USDOI FWS 2008, 2012) and can sometimes exclude native ungulates from water sources (Gooch et al. 2017; Hall et al. 2016; Ostermann-Kelm et al. 2008; Perry et al. 2015; USDOI FWS 2008). Bird nest survival may be lower in areas with wild horses (Zalba and Cozzani 2004), and bird populations have recovered substantially after livestock or wild horses, or both, have been removed (Batchelor et al. 2015; Earnst et al. 2005, 2012). Wild horses can spread nonnative plant species, including cheatgrass, and may limit the effectiveness of reseeding projects (Beever et al. 2003; Couvreur et al. 2004; Jessop and Anderson 2007; Loydi and Zalba 2009). Even after domestic livestock are removed, continued wild horse use above appropriate management levels can cause ongoing detrimental ecosystem effects (Davies et al. 2014; USDOI FWS 2008), which may require several decades for recovery (e.g., Anderson and Inouye 2001).

Wild burros can have grazing and trampling impacts that are similar to wild horses (Carothers et al. 1976; Douglas and Hurst 1983; Hanley and Brady 1977) and can substantially affect riparian habitats (e.g., Tiller 1997) and native wildlife (e.g., Seegmiller and Ohmart 1981). Where wild burros and GRSG co-occur, year-round use by burros in low elevation habitats may lead to a high degree of overlap between burros and GRSG (Beever and Aldridge 2011).

In contrast to managed domestic livestock grazing (see section 7), neither the seasonal timing nor the intensity of grazing by Federally protected wild horses and burros can be managed, except through efforts to manage their numbers and distribution. Wild horses roam freely on the range year-round, and wild horse populations have the potential to grow 15 to 20 percent or more per year (Dawson 2005; Eberhardt et al. 1982; Garrott et al. 1991; Roelle et al. 2010; Scorolli and Cazorla 2010; Wolfe 1980). Although annual growth rates may be marginally lower in some areas where mountain lions (*Puma concolor*) can take foals (Turner 2015; Turner and Morrison 2001), horses tend to favor use of more open habitats (Schoenecker et al. 2016) that are dominated by grasses and shrubs and where ambush is less likely. For the majority of wild horse herds, there is little evidence that population growth is significantly affected by predation. As a result of the potential for wild horse populations to grow rapidly, impacts of wild horses on water, soil, vegetation, and native wildlife resources can increase exponentially unless there is active management to limit their population sizes.

On lands administered by the BLM, there were an estimated 72,674 BLM-administered, Federally protected wild horses and burros as of March 1, 2017, not including foals born in 2017 (USDOI BLM 2017). Approximately 60 percent of those are present within 13 million acres (5 million hectares) of GRSG habitat. Federal protections exist for an estimated 7,100 wild horses and 900 wild burros that occupy approximately 2 million acres (800,000 hectares) of Forest Service-administered lands. Approximately 446,065 acres (180,523 hectares) of active Territories administered by the Forest Service contain GRSG habitat, which is occupied by an estimated 3,400 wild horses and burros. Some wild horses also inhabit other Federal lands in the sagebrush biome, including lands administered by the National Park Service, U.S. Fish and Wildlife Service, or Department of Defense, and Native American reservations and tribal lands.

Although wild horses and burros can present challenges to achieving desired habitat conditions, wild horse management is a necessary requirement of planning for long-term sagebrush ecosystem and GRSG conservation. This section relates to management of Federal lands and the terms “wild horses” and “wild burros” are used throughout. However, the specific legal status for any given wild horse or burro population has a large influence on management objectives and the ability to manage wild horse and burro impacts.

In the biological sense, all free-roaming horses and burros in North America are feral, meaning that they are descendants of domesticated animals brought to the Americas by European colonists. Horses went extinct in the Americas by the end of the Pleistocene, about 10,000 years ago (MacFadden 2005; Webb 1984). Burros evolved in Eurasia (Geigl et al. 2016). The published literature refers to free-roaming horses and burros as either feral or wild. In the ecological context the terms are interchangeable, but the term “wild” horse is associated with a specific legal status. Wild and free-roaming horses and burros under the jurisdiction of the BLM and Forest Service are designated “wild” as legally defined by the Wild Free Roaming Horses and Burros Act of 1971 (WFRHBA) as amended (Public Law 92–195), and are under the protection, management, and control of the BLM and Forest Service. Only those horses whose unbranded and unclaimed ancestors were present on BLM and Forest Service lands at the time of the passage of the WFRHBA are managed in accordance with the WFRHBA, and only those lands where wild horses and burros were found when the WFRHBA was passed can be managed to maintain Federally recognized wild horse and burro populations.

Other populations of feral horses and burros on Federal lands (i.e., those on lands administered by the U.S. Fish and Wildlife Service, National Park Service, or Department of Defense; and Native American reservations and tribal trust lands) are generally subject to other Federal regulations and relevant State laws, but are not subject to provisions of the WFRHBA. This section draws on scientific studies of feral horses and burros, some of which also have wild horse or wild burro legal status. Clarification of which horses and burros are considered Federally protected is provided in the BLM regulation (43 CFR 4700 [FR 2011]), BLM wild horse and burro management handbook and manuals (USDOI BLM 2010a,b,c,d), Forest Service manual (FSM 2260.5), and Forest Service regulation (36 CFR 222.20(b)(13), 36 CFR 222.63 [FR 2012]). The legal designation of a particular herd is not expected to change the animals’ ecological effects, but it will influence management options. Discussions about management in this section reflect constraints for Federally designated wild horses and burros.

This section begins with information on wild horse and burro management structure, population estimates and spatial distribution, and management actions to maintain wild horses and burros at appropriate management levels. Then it discusses using resilience and resistance concepts to inform management of wild horses and burros. It concludes with management considerations at the project scale. This section refers mainly to wild horses because wild burros are not nearly as numerous as wild horses in most areas of the sagebrush biome.

Wild Horse and Burro Management Structure

For lands administered by the BLM, Herd Areas (HAs) are defined as areas where wild horses and burros existed at the time of passage of the WFRHBA. Herd Management Areas (HMAs)—the subset of lands designated for active management of wild horses and burros as part of multiple use management—can be designated only within HAs during land use planning activities. In most cases, each HMA is intended to support only wild horses or wild burros, but there are some HMAs that contain both. For HAs that do not have an HMA designation, it generally has been determined that resources are limiting and that wild horse and burro populations cannot be maintained for the long term. The Forest Service-administered Wild Horse Territories (WHTs), Wild Burro Territories (WBTs), and Wild Horse and Burro Territories (WHBTs) are designated according to the species that occupy the Territory. There are some Territories without any wild horses or burros that are considered “inactive,” where it has been determined that there are not sufficient resources to maintain wild horses and burros, or where wild horses and burros no longer exist. The numbers of wild horses and burros in HMAs, WHTs, WBTs, or WHBTs and the overlap with GRSG habitat are in text box 8.1.

When two or more HMAs, WHTs, WBTs, or WHBTs are located close to one another, with the potential for wild horses and burros to move freely among them, those areas may be managed collectively as a “complex” (or “joint management area”). Complexes sometimes cross administrative boundaries between BLM field or district offices and Forest Service districts.

The spatial scales of wild horse management are the entire population at the West-wide scale; complexes or groups of HMAs or WHTs, WBTs, and WHBTs with interchange for the regional scale; and individual herds for the local scale. A National Academies of Science report (National Research Council 2013) suggested that wild horse management should be focused more broadly on meta-populations, in which HMAs, WHTs, WBTs, and WHBTs are grouped where interchange occurs, regardless of administrative boundaries. Thus, relative to the spatial scales presented in section 1 of this report, the BLM and Forest Service manage wild horses between the regional and local project levels. The actual spatial scale for any given wild horse population should be determined in consultation with the local staff that manages those populations (i.e., BLM wild horse and burro specialist; Forest Service rangeland management specialist).

Importantly, each HMA, WHT, WBT, and WHBT has an established target population size range for wild horses (and a separate target for wild burros, if they are present), known as the appropriate management level (AML). The BLM and Forest Service view AML as a target population size range which, if maintained, should allow for a thriving ecological balance and multiple use relationship (43 CFR 4710.3-1 [USDOI BLM 2010b]; 43 CFR 4770.3(c) [USDOI 2012]; 36 CFR 222.60(b)(3), 36 CFR 222.61(a)(1), 36 CFR 222.69(a)

Text Box 8.1—Wild Horse and Burro Population Sizes

The BLM manages wild horses and burros within a total of 177 Herd Management Areas (HMAs), which range in size from 3.0 square miles (777 hectares) to 2,033.8 square miles (526,754.2 hectares). As of March 1, 2017, the estimated number of wild horses and burros managed by BLM was 72,674. A total of 105 HMAs overlap with approximately 13 million acres (5 million hectares) of GRSG habitat.

The Forest Service manages 34 active and 19 inactive wild horse and burro administrative units that include: Wild Horse Territories (WHTs; 27 active, 16 inactive), Wild Burro Territories (WBTs; 4 active, 3 inactive), and Wild Horse and Burro Territories (WHBTs; 3 active). These range in size from 5.4 square miles (1,398.6 hectares) to 530.4 square miles (137,373.6 hectares). The Forest Service manages approximately 8,000 wild horses and burros. Thirteen active Territories overlap with approximately 446,000 acres (180,000 hectares) of GRSG habitat.

One thousand or more wild horses on three WHTs and five HMAs live on or near Bi-State GRSG habitat (about 70,000 Forest Service acres [28,000 hectares] and 82,403 BLM acres [33,348 hectares]) (Bi-State Technical Advisory Committee 2012). The Bi-State population has been identified as a Distinct Population Segment of GRSG and is managed under a separate conservation Action Plan.

[FR 2012]). This view reflects an assumption that wild horse and burro populations at AML should allow for land health standards to be met (USDOI BLM 2010a). The AML generally is a range between a low and high value, to allow for some variability in population size across years (USDOI BLM 2010a). The AML is typically determined at the activity planning level through site-specific analysis or, in some cases, through the land use planning process. Monitoring information that couples data on wild horse and burro populations and rangeland status and trends is used to establish or adjust AMLs (text box 8.2). Progress toward attainment of site-specific and landscape-level management objectives or multiple use objectives is also considered. Future studies at local scales could test the assumption that wild horse and burro populations at AML allow for land health standards to be met.

Text Box 8.2—Monitoring Considerations for Wild Horses and Burros

Reliable estimates of population sizes and habitat data provide the basis for management decisions regarding wild horses and wild burros. Understanding the annual growth rates of wild horse and burro populations and the status and trends of rangelands occupied by wild horses and burros is essential for making informed management decisions.

Inventory (monitoring) data for wild horse and burro populations include information on the numbers of animals, their use patterns, and spatial distribution. Habitat data include grazing utilization, range ecological condition and trend, actual use, and climate (weather) data. Habitat monitoring data collection should be coordinated with other resource programs (e.g., range, watershed, wildlife) to maximize efficiency and minimize duplication.

Data and analyses of populations and habitats are used in concert to:

- Establish or adjust Appropriate Management Levels (AMLs);
- Make a determination of excess wild horses or burros (i.e., establish the need to gather and remove excess animals in order to reach and stay at AML);
- Develop or revise Herd Management Area (HMA) boundaries; and
- Evaluate conformance with Land Health Standards, Land Use Plan goals and objectives, or other site-specific or landscape-level objectives.

Data and methods used to inform decisions should be scientifically defensible. The public should be able to understand the methods used and how they are implemented and also to access the data used to make decisions.

Data on Population Estimates and Spatial Distribution of Wild Horses and Burros

Population estimates for each HA and HMA are reported annually in the Public Land Statistics (http://www.blm.gov/public_land_statistics/); spatial data are available via the BLM GeoCortex, which is available to managers for analyses and planning and is useful in determining the number of excess animals present on the range (https://webmaps.blm.gov/Geocortex/Html5Viewer/Index.html?viewer=whb). The Forest Service reports population estimates for each territory on the Forest Service wild horse and burro program website (https://www.fs.fed.us/wild-horse-burro/territories/index.shtml). The BLM and Forest Service have recently adopted a statistically valid, standardized methodology for estimating wild horse population sizes (Lubow and Ransom 2009, 2016; Ransom 2012) that accounts for animals that were present, but not seen by observers. In most cases, reported population estimates are based on the statistical analysis of aerial survey data; BLM policy calls for each HMA (and complexes that include both BLM lands and Forest Service WHTs, WBTs, or WHBTs) to be surveyed at least once every 2 years (USDOI BLM 2010e). For both agencies, population size estimates are projected for intervening years based on the best available information about expected population growth rates for each area. As previously discussed, wild horse growth rates can typically be assumed to be about 15 percent to 20 percent per year (National Research Council 2013) unless there is a contraceptive project to limit reproduction. However, in some places the annual growth rate may be greater than 20 percent. The range-wide population estimates are used to develop BLM geospatial data (accessible at the BLM GeoCortex site) and the status of a population relative to high AML within a particular HMA.

Although it is the intended management goal that wild horses remain only on HMAs, WHTs, or WHBTs, the current reality is that Federally protected wild horses are also present on many HAs and on other Federal, State, tribal, and private lands outside of these administrative boundaries. As a result, the user must be cautiously aware that the data representing boundaries of and populations within HMAs, HAs, WHTs, WBTs, and WHBTs may not portray the actual spatial distribution of all wild horse and burro populations. Continued increases in wild horse and burro populations, relative to AML, will result in a more widespread distribution of herds, including into areas outside designated boundaries. In areas where road or trail access allows for observations and on-the-ground documentation of horse sign (e.g., trailing, scat piles, evidence of horse grazing and browsing), the local designated staff is likely to have a broad understanding of where the animals tend to go in different seasons, which water sources they rely on, and the general pattern of their movements.

Management Actions to Maintain Wild Horses and Burros at Appropriate Management Levels

The 1971 WFRHBA directs the BLM and Forest Service to remove excess animals from the range (43 CFR 4720.1 and 36 CFR 222.69, respectively) to maintain a thriving natural balance. The number of wild horses or burros greater than a designated high AML for a HMA, WHT, WBT, or WHBT is considered to be the number of "excess" animals in the area. In order to take management action, the agencies must make two determinations: (1) that an overpopulation exists, and (2) whether or not it will be necessary to remove excess animals.

Historically, the BLM and Forest Service reduced herd population sizes to the low value of AML. This was accomplished by removing excess animals from the range. The population would then typically grow to reach the high value of the AML range within 3 to 4 years, unless some form of contraception was used to limit population growth rates. Natural regulation via starvation or dehydration is generally not acceptable to many members of the public (National Research Council 2013).

After removal, animals were placed in holding facilities, offered to the public for adoption, and then kept in holding facilities indefinitely if there was no adoption demand. However, removing all excess wild horses and holding them in off-range facilities for the remainder of their lives would be prohibitively expensive (Garrott and Oli 2013). In many recent years, the BLM has not had the budgetary capacity to remove more than approximately 3,500 animals per year from the range. Further, the more than 45,000 BLM-administered, captive wild horses currently in long-term holding (of which about 850 are horses from Forest Service Territories) require over $50 million per year to maintain. As a result, populations of wild horses and burros across all BLM-administered lands (and on some Forest Service Territories) have not been gathered so frequently. Average population sizes are now more than three times greater than the high end of the total AML and these populations are growing.

In 2015, the BLM, the Forest Service, and other agencies identified certain areas as the most important habitats for GRSG and other sagebrush obligates. None of those areas overlapped with Forest Service-administered wild horse or wild burro populations. The BLM developed a 5-year gather schedule to achieve AML by 2020 in 22 HMAs that overlapped areas identified as the most important habitats for GRSG and other sagebrush obligates. However, under budget projections made in FY2017, the BLM will not have the fiscal capacity to conduct gathers within GRSG Priority Habitat Management Areas until 2020 or later, and has no capacity to manage wild horse populations that overlap with GRSG General Habitat Management Areas. Unless there are Congressionally directed changes to the BLM program, it is expected that the number of wild horses within GRSG habitat could surpass 65,000 horses in 2019. Furthermore, maintaining any wild horse population at or below AML will require an active and ongoing program of population growth suppression or scheduled removals (or both) of excess animals. Without such a program, habitat restoration will quickly be at risk as wild horse populations again grow to exceed AML.

Currently used population growth suppression methods include gelding and the immunocontraceptives porcine zona pellucida (PZP) and GonaCon (National Research Council 2013). Both vaccines may be effective for only 1 year, unless booster doses are given (National Research Council 2013). Repeated PZP boosters require annual darting or recapture of the vast majority of wild horses under BLM or Forest Service management, which is infeasible on many HMAs and Territories, would be prohibitively expensive to apply across the range of wild horses and burros, and may lead to more stress for wild horses as a result of frequent capture. The BLM is supporting ongoing research initiatives to develop and test longer-term contraception for wild horses and burros and to improve contraceptive efficacy and production (USDOI BLM 2015). However, planning decisions that propose to remove excess horses or utilize population growth suppression on any BLM lands are often appealed and litigated by interested members of the public. This results in a high degree of uncertainty about the ability of designated Federal agencies to maintain wild horse populations within AML.

Using Resilience and Resistance Concepts and the Science Framework to Inform Management of Wild Horses and Burros

Information on relative ecosystem resilience to disturbance and resistance to invasive annual grasses can be used to help understand the responses of sagebrush ecosystems, species at risk, and other resources to wild horse and burro use and to the interactions of wild horse and burro use with other potential disturbance factors such as wildfire and invasive plants. Information on resilience and resistance to invasive annual grasses, coupled with information on current and projected wild horse and burro population sizes relative to AML and other predominant threats and disturbance factors, can be used to inform conservation and restoration strategies in sagebrush ecosystems across scales.

Part 1 of the Science Framework provides an approach based on an understanding of ecosystem resilience to disturbance and resistance to invasive annual grasses that uses assessments at the mid-scale (ecoregional or GRSG Management Zone) (fig. 1.1) to help prioritize areas for management and determine effective management strategies (Chambers et al. 2017). The approach is based on: (1) the likely response of an area to disturbance or stress due to threats or management actions (i.e., resilience to disturbance and resistance to invasive annual grasses), (2) the capacity of an area to support target species or resources, and (3) the predominant threats. The geospatial data layers and analyses used in the approach are described in Part 1, sections 8.1 and 8.2. The process involves overlaying key data layers including resilience and resistance to invasive annual grasses as indicated by soil temperature and moisture regimes (Maestas et al. 2016), sage-grouse breeding habitat probabilities (Doherty et al. 2016), the densities or distributions of other sagebrush dependent species, and the primary threats for the ecoregions or Management Zones in the assessment. The maps and analyses that managers derive from this process are an essential component of prioritizing areas for management actions and developing management strategies.

Wild horse and burro densities and AMLs can be used similarly to other threats and disturbance factors in the analyses. Managers can devise categories to evaluate the degree to which wild horse and burro populations are within or exceed AMLs for HMAs, WHTs, WBTs, and WHBTs. Here, three abundance categories relative to AML were developed based on available abundance estimates for BLM lands and Forest Service lands: within AML, more than 100 percent to 200 percent of AML, and more than 200 percent of AML. The wild horse HMAs were overlaid with these three abundance categories (fig. 8.1). Note that this figure also depicts HAs where the target population for wild horses is zero, but where wild horses are present.

The three abundance categories were overlaid with: (1) the three resilience and resistance categories derived from soil temperature and moisture regime information, and (2) GRSG breeding habitat probabilities (see Part 1, sections 8.1 and 8.2). This analysis does not include areas outside the boundaries of HMAs, HAs, WHTs, WBTs, and WHBTs where horses and burros have expanded their use. The data used in the analyses can be found at: https://www.sciencebase.gov/catalog/item/576bf69ce4b07657d1a26ea2.

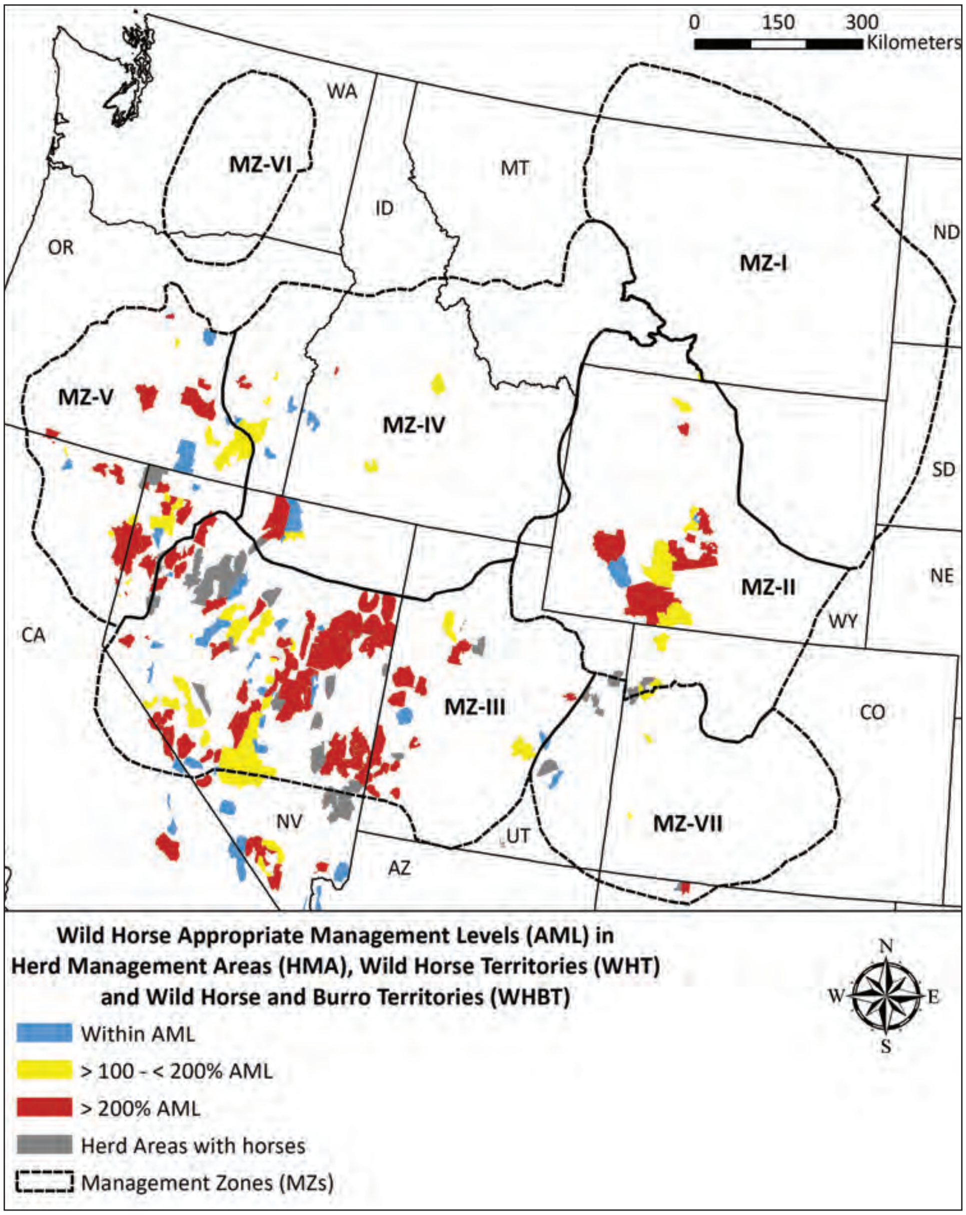

Figure 8.1—Categories of estimated wild horse abundance as of March 1, 2017 relative to Appropriate Management Level (AML) for wild horse Herd Management Areas (HMAs) on BLM lands and Wild Horse Territories (WHTs) and Wild Horse and Burro Territories (WHBTs) on Forest Service lands. Gray polygons indicate Herd Areas where the target population for wild horses is zero, but where wild horses are present. Estimated wild horse abundance exceeds AML in most HMAs, WHTs, and WHBTs.

Analyses of Appropriate Management Levels, Ecosystem Resilience and Resistance, and Breeding Bird Habitat Probabilities

Sixty percent of HMAs, WHTs, and WHBTs managed by the BLM and Forest Service are in areas categorized as having low resilience and resistance (fig. 8.2, table 8.1). In contrast, 33 percent have moderate resilience and resistance and only 7 percent have high resilience and resistance. In the area with low resilience and resistance, 60 percent has wild horse abundance that exceeds 200 percent of the horse AML.

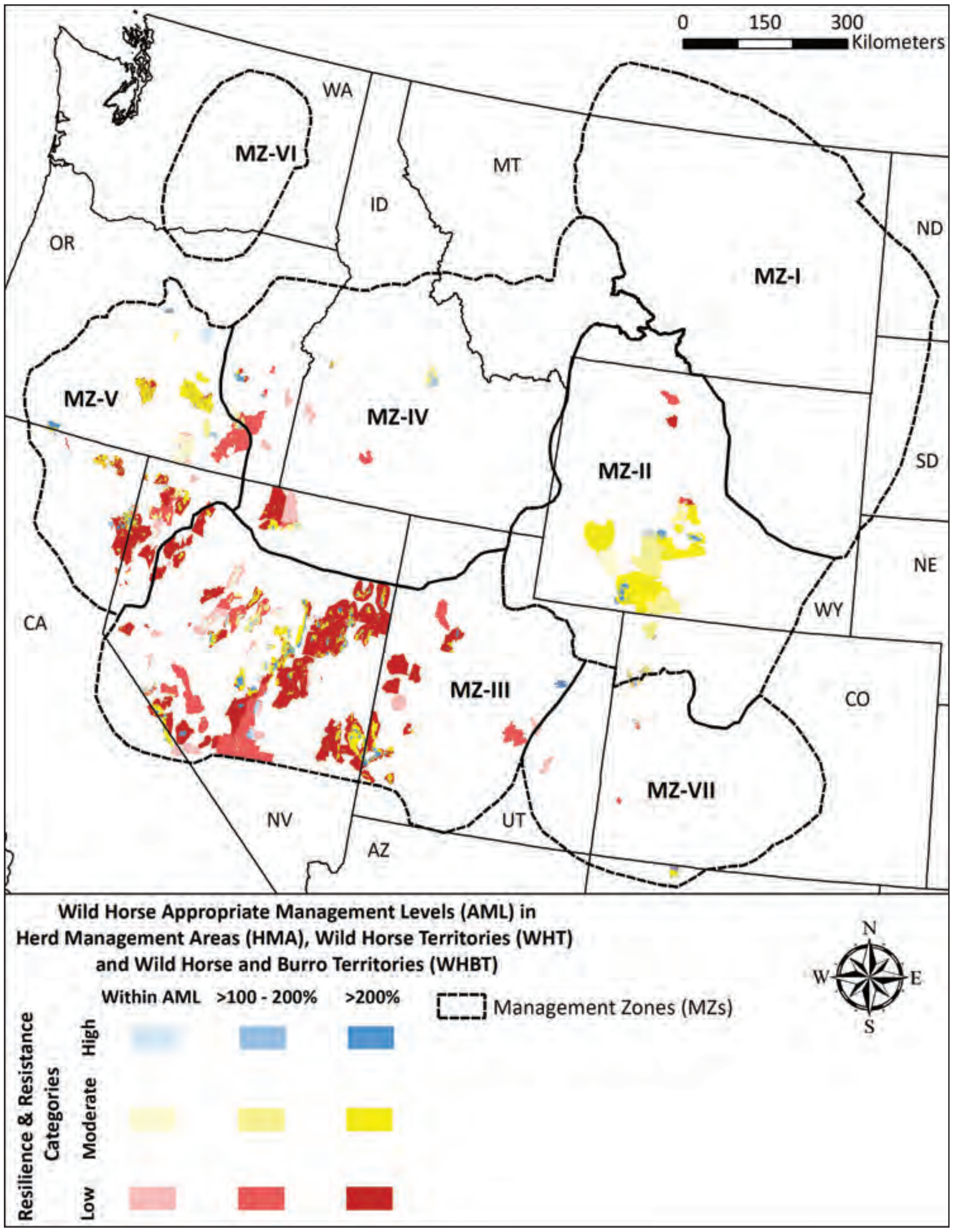

Figure 8.2—Categories of estimated wild horse abundance as of March 1, 2017 relative to Appropriate Management Level (AML), overlaid with the resilience and resistance classes within wild horse Herd Management Areas (HMAs) on BLM lands and Wild Horse Territories (WHTs) and Wild Horse and Burro Territories (WHBTs) on Forest Service lands. Most HMAs, WHTs, and WHBTs are in low to moderate resilience and resistance categories and exceed AML.

Table 8.1—The area and percentage of Herd Management Areas, Wild Horse Territories, and Wild Horse and Burro Territories for the Bureau of Land Management and Forest Service by wild horse Appropriate Management Level (AML) class and resilience and resistance class. Percentages within a Management Zone (MZ) add to 100.

	Resilience and resistance					
Percent Horse	Low		Moderate		High	
AML class	Acres	%	Acres	%	Acres	%
MZ I						
<100	0	0	0	0	0	0
>100–200	0	0	4,326	57	3,200	43
>200	0	0	0	0	0	0
Total	0	0	4,326	57	3,200	43
MZ II						
<100	0	0	414,831	8	2,204	1
>100–200	182,045	4	1,578,883	31	68,236	1
>200	108,086	2	2,548,764	50	166,862	3
Total	290,131	6	4,542,478	89	237,302	5
MZ III						
<100	1,161,465	8	233,713	2	146,235	1
>100–200	2,965,677	19	368,132	2	168,363	1
>200	7,916,216	52	1,743,470	11	618,498	4
Total	12,043,358	79	2,345,315	15	933,096	6
MZ IV						
<100	560,601	27	67,981	3	19,771	1
>100–200	490,895	23	198,977	9	89,076	4
>200	560,706	27	90,401	4	49,144	2
Total	1,612,201	77	357,359	16	157,991	7
MZ V						
<100	193,058	4	426,958	8	186,252	4
>100–200	942,681	18	336,100	6	85,331	2
>200	1,618,840	31	1,119,312	22	276,522	5
Total	2,754,579		1,882,370		548,105	
MZ VII						
<100	130,987	38	0	0	0	0
>100–200	47,132	13	64,758	19	29,502	8
>200	8,427	2	40,236	12	27,286	8
Total	186,546	53	104,994	31	56,788	16
All MZs						
<100	2,046,111	7	1,143,483	4	354,462	1
>100–200	4,628,430	17	2,551,176	9	443,708	2
>200	10,212,274	36	5,542,187	20	1,138,311	4
Total	16,886,815	60	9,236,846	33	1,936,481	7

Differences in both resilience and resistance and the abundance categories exist among Management Zones for wild horses (fig. 8.2, table 8.1). In Management Zone III, where the majority of wild horses are found, lands managed for wild horses are primarily within low resilience and resistance areas (79%). In the area with low resilience and resistance, 52 percent has wild horse abundance in excess of 200 percent of the horse AML. In Management Zones IV and V, lands managed for wild horses also are primarily within low resilience and resistance areas: 77 percent and 53 percent, respectively. In both of these areas, most lands managed for wild horses have horse abundance greater than 100 to 200 percent of the horse AML.

For wild burro populations, most of the land area in HMAs, WBTs, and WHBTs included in this analysis is in low resilience and resistance areas (80 percent), followed by moderate resilience and resistance areas (18 percent) (fig. 8.3, table 8.2). Moreover, 73 percent of the lands managed for wild burros in this analysis have wild burro abundance in excess of 200 percent of the burro AML.

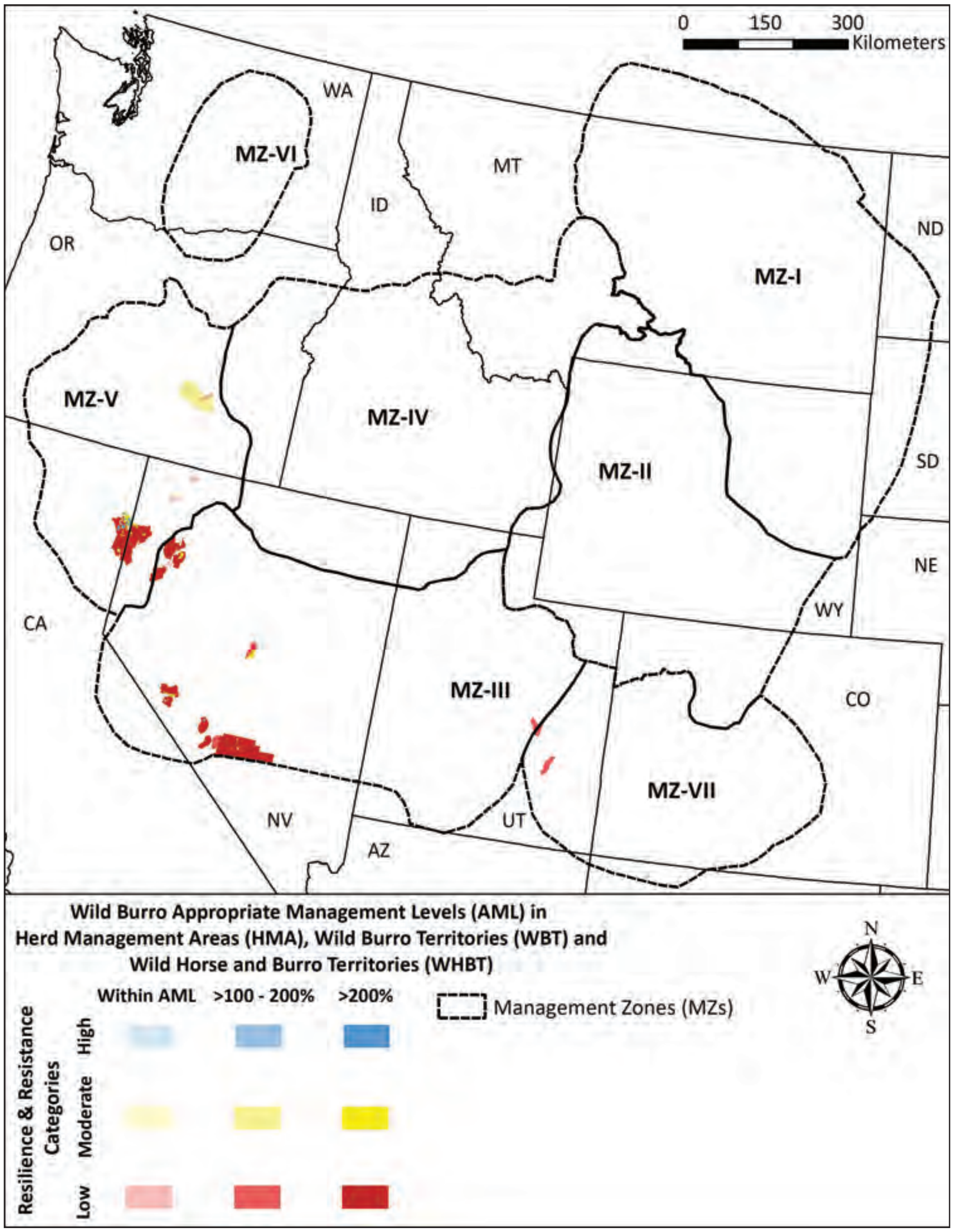

Figure 8.3—Categories of estimated wild burro abundance as of March 1, 2017 relative to Appropriate Management Level (AML), overlaid with the resilience and resistance classes within wild burro Herd Management Areas (HMAs) on BLM lands and Wild Burro Territories (WBTs) and Wild Horse and Burro Territories (WHBTs) on Forest Service lands. Estimated wild burro abundance exceeds AML in most HMAs, WHTs, and WHBTs.

Table 8.2—The area and percentage of Herd Management Areas, Wild Burro Territories, and Wild Horse and Burro Territories for the Bureau of Land Management and Forest Service by wild burro Appropriate Management Level (AML) class and resilience and resistance class. Percentages within a Management Zone (MZ) add to 100.

Percent Burro AML class	Resilience and resistance					
	Low		Moderate		High	
	Acres	%	Acres	%	Acres	%
MZ III						
<100	18,063	1	0	0	0	0
>100–200	162,160	8	9,563	1	0	0
>200	1,655,499	87	59,095	3	4,076	0
Total	1,835,722	96	68,658	4	4,076	0
MZ V						
<100	77,478	5	44,492	3	0	0
>100–200	30,008	2	442,165	29	20,651	1
>200	795,307	52	80,589	5	51,215	3
Total	902,793	59	567,246	37	71,865	4
MZ VII						
<100	0	0	0	0	0	0
>100–200	130,987	100	0	0	0	0
>200	0	0	0	0	0	0
Total	130,987	100	0	0	0	0
All MZs						
<100	95,541	3	44,492	1	0	0
>100–200	323,155	9	451,728	13	20,651	1
>200	2,450,806	68	108,351	4	55,290	1
Total	2,869,502	80	635,940	18	75,941	2

In Management Zones III and V the highest percentage of land is in low resilience and resistance areas with wild burro abundance more than 200 percent of the burro AML. Most of the burros managed by the BLM are located in Arizona and southern Nevada (USDOI 2017), which is outside of the sagebrush biome and the area of this analysis.

Overlaying the categories of wild horse abundance relative to AMLs with the sage-grouse breeding habitat probabilities shows that 42 percent of the lands managed for wild horses occur in the low, 40 percent in the moderate, and 18 percent in the high GRSG breeding habitat probability (fig. 8.4, table 8.3). In the high breeding habitat probability areas, which are the highest priority for protection, and in the moderate breeding habitat probability areas, which often provide opportunities for conservation actions, about two-thirds of the lands managed for wild horses have horse abundance in excess of 200 percent of the horse AML.

Analysis of the sage-grouse breeding habitat probabilities overlaid on categories of wild burro abundance relative to AML shows that 46 percent, 46 percent, and 8 percent of those GRSG breeding habitats managed for wild burros and included in this analysis occur in the low, moderate, and high breeding habitat probability areas, respectively (table 8.4). Within low, moderate, and high GRSG breeding habitat probability areas, 69 percent, 72 percent, and 38 percent, respectively, of the lands managed for wild burros have burro abundance greater than 200 percent of the burro AML. Management Zone V has a higher land area managed for wild burros with GRSG breeding habitat than Management Zone III, and a higher percentage of the wild burro population is in moderate and high GRSG breeding habitat probability areas.

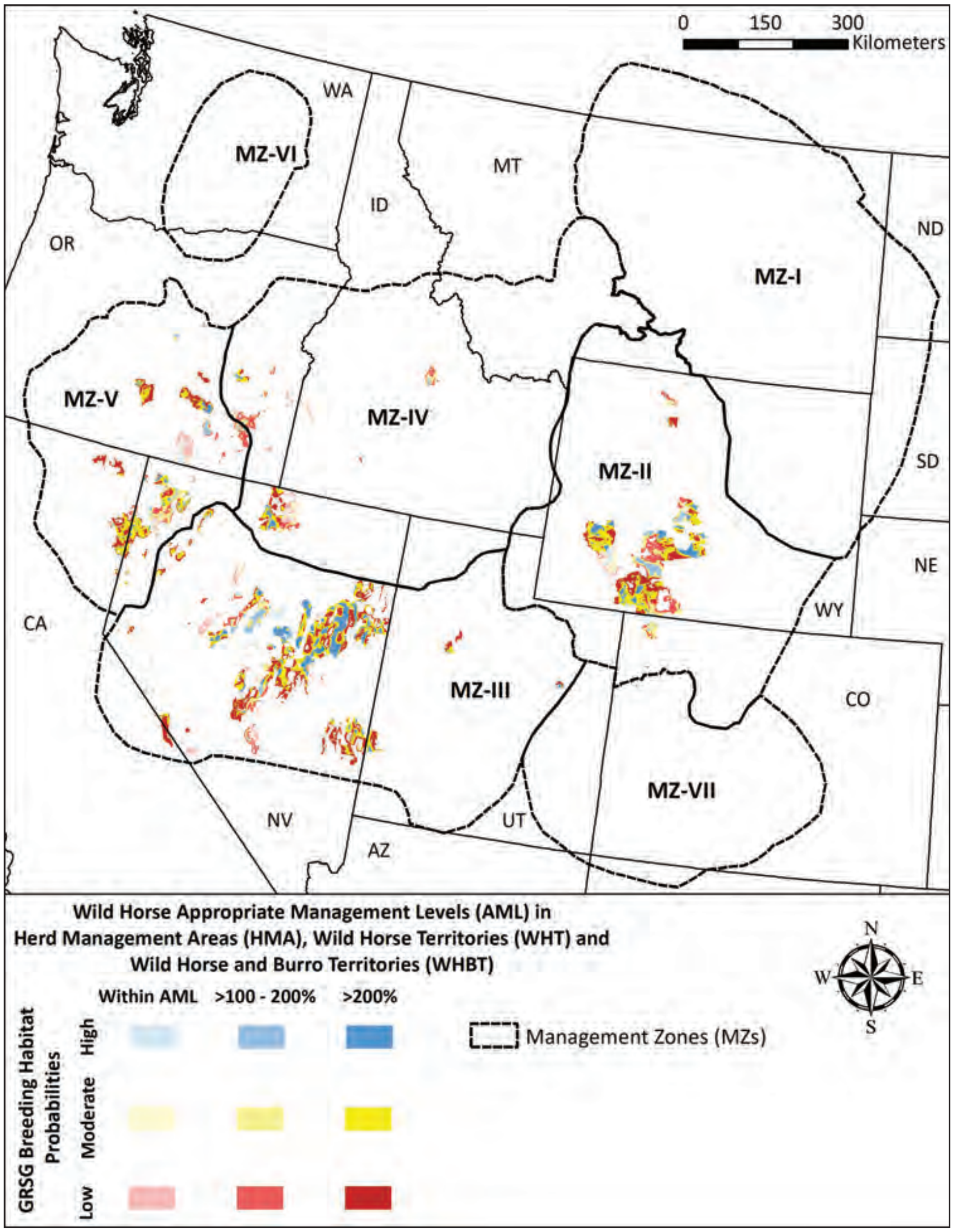

Figure 8.4—Categories of estimated wild horse abundance as of March 1, 2017 relative to Appropriate Management Level (AML), overlaid with the GRSG breeding habitat probabilities within wild horse Herd Management Areas (HMAs) on BLM lands and Wild Horse Territories (WHT) and Wild Horse and Burro Territories (WHBTs) on Forest Service lands. Estimated wild horse abundance exceeds AML in many areas with moderate to high GRSG breeding habitat probabilities.

Table 8.3—The area and percentage of Herd Management Areas, Wild Horse Territories, and Wild Horse and Burro Territories for the Bureau of Land Management and Forest Service by wild horse Appropriate Management Level (AML) class and Greater sage-grouse (GRSG) breeding habitat probability class. Percentages within a Management Zone (MZ) add to 100.

Percent Horse AML class	GRSG breeding habitat probability					
	Low		Moderate		High	
	Acres	%	Acres	%	Acres	%
MZ II						
<100	92,230	2	198,329	4	77,042	2
>100–200	573,836	13	557,183	13	255,275	6
>200	924,545	21	1,298,137	29	462,370	10
Total	1,590,610	36	2,053,649	46	794,686	18
MZ III						
<100	353,147	5	148,052	2	85,319	1
>100–200	312,594	4	319,359	5	273,905	4
>200	2,319,075	33	2,028,561	29	1,185,258	17
Total	2,984,816	42	2,495,972	36	1,544,482	22
MZ IV						
<100	234,091	16	208,371	14	10,955	1
>100–200	293,756	20	160,647	11	33,053	2
>200	212,954	14	224,679	15	95,330	7
Total	740,802	50	593,697	40	139,338	10
MZ V						
<100	281,312	9	161,838	5	94,638	3
>100–200	334,833	10	320,755	10	142,127	4
>200	867,460	27	832,302	26	178,115	6
Total	1,483,605	46	1,314,895	41	414,880	13
MZ VII						
<100	0	0	0	0	0	0
>100–200	252	3	2,494	29	5,748	68
>200	0	0	0	0	0	0
Total	252	3	2,494	29	5,748	68
All MZs						
<100	960,780	6	716,590	4	267,954	2
>100–200	1,515,271	9	1,360,438	8	710,108	4
>200	4,324,034	27	4,383,679	27	1,921,073	12
Total	6,800,085	42	6,460,707	40	2,899,135	18

Using the Science Framework to Inform Management Decisions

Primary considerations for wild horse and burro management from the Science Framework approach are presented next (see tables 1.3, 1.4).

- In general, areas that support medium to high sage-grouse breeding habitat probabilities or other important resources are high priorities for management (table 1.3: cells 2A, 2B, 2C, 3A, 3B, 3C), especially low resilience and resistance categories that lack the potential to recover from disturbances such as excessive wild horse and burro use without significant intervention (table 1.3: cells 2C, 3C). These areas could be considered priorities for wild horse and burro gathers and fertility control where horse and burro abundance exceeds target AMLs and the area is not highly degraded.
- Areas with moderate and, especially, high resilience and resistance often have the potential to recover through successional processes (table 1.3: cells 1B, 1C, 2B, 2C).

Table 8.4—The area and percentage of Herd Management Areas, Wild Burro Territories, and Wild Horse and Burro Territories for the Bureau of Land Management and Forest Service by wild burro Appropriate Management Level (AML) class and Greater sage-grouse (GRSG) breeding habitat probability class. Percentages within a Management Zone (MZ) add to 100.

Percent Burro AML class	GRSG breeding habitat probability					
	Low		Moderate		High	
	Acres	%	Acres	%	Acres	%
MZ III						
<100	107	1	0	0	0	0
>100–200	9,882	3	12,082	4	8,717	3
>200	168,963	58	86,373	30	2,943	1
Total	178,952	62	98,455	34	11,660	4
MZ V						
<100	23,217	2	68,662	7	18,022	2
>100–200	147,908	14	91,557	8	50,412	5
>200	263,516	24	364,745	34	44,423	4
Total	434,640	40	524,964	49	112,857	11
All MZs						
<100	23,217	2	68,662	5	18,022	1
>100–200	157,790	12	103,638	8	59,130	4
>200	432,479	32	451,118	33	47,366	3
Total	613,486	46	623,418	46	124,518	8

    - These areas represent significant opportunities to improve habitat and could also be considered priorities for wild horse and burro gathers and fertility control where horse and burro abundance exceeds target AMLs and removals are likely to result in habitat improvement.
    - In areas where wild horses and burros exceed target AMLs (including occupied areas outside of HMAs, HAs, WHTs, WBTs, and WHBTs), managers should carefully consider the current spatial extent and growth potential of any nearby wild horse herds and their potential effects on management actions to improve habitat.
- New postfire rehabilitation areas and areas that provide sagebrush habitat connectivity for GRSG and other species at risk are conservation priorities and, thus, could be priorities for wild horse and burro gathers, where abundance exceeds AMLs.

Ecological type or ecological site descriptions and their associated state-and-transition models (STMs) can be used to help evaluate potential effects of wild horse and burro use and the likely success of conservation and restoration actions. In the Science Framework, generalized ecological types and STMs have been developed for the range of environmental conditions in the eastern and western portions of the sagebrush biome (see Part 1, Appendices 5 and 6). The ecological types and STMs are characterized according to their resilience to disturbance and resistance to invasive annual grasses based on soil temperature and moisture regimes and other biophysical characteristics such as plant community composition. They provide information on the alternative states, ranges of variability within states, and processes that cause plant community shifts within states as well as transitions among states. These ecological types and STMs can be used to: (1) identify the different ecological types that exist within the HMA or Territory and determine their relative resilience to disturbance and resistance to invasive annual grasses; (2) evaluate the current ecological dynamics of the ecological types or ecological sites and, where possible, their restoration pathways; (3) increase understanding of the potential effects of wild horse and burro use; and (4) determine the likelihood of conservation and restoration actions succeeding given ongoing wild horse and burro use (Part 1, section 9).

Section 7 uses these STMs to illustrate potential livestock management strategies for ecological types that support GRSG populations and that may benefit from improved livestock grazing management. Information on how to use these resilience-based ecological types and STMs for managing ecosystem threats across the sagebrush biome is in Part 1, section 9.2. Information on how to use resilience-based ecological types and STMs for selecting appropriate treatments for assessing postwildfire recovery and restoration decisions in sagebrush and juniper-piñon ecosystems in the Great Basin is in Miller et al. (2014, 2015) and Pyke et al. (2017), respectively.

Management Considerations at the Project Scale

An assessment of the ecological sites in the project area and their relative resilience to disturbance and resistance to invasive annual grasses can help determine the potential for conservation and restoration treatments to succeed. More detailed information can be obtained from ecological site descriptions for those areas where they have been developed (see http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/ecoscience/desc/). Ecological type and ecological site descriptions provide basic information on the climate and soil characteristics of an area and the potential of the area to support a dynamic set of plant communities. The associated STMs provide information on the current states and the potential transitions among them due to disturbances and other drivers such as wild horse and burro use as well as management treatments. Assessing the states and the plant communities within the states based on STMs provides information on both the disturbances and the drivers that have led to the current state and the potential restoration pathways. For example, plant communities within the reference state or within states that have feasible restoration pathways may respond favorably to conservation and restoration actions if the wild horse population can be managed at or below AML. However, plant communities in other states, such as an invaded state or annual state (see figs. 7.2, 7.6) may not respond favorably to conservation and restoration actions if the wild horse population cannot be managed at or below AML. Ecological types or ecological sites with relatively low resilience and resistance to invasive annual grasses often require more than one intervention for restoration efforts to succeed and wild horse and burro use can have significant effects on project success.

Effects of wild horses and burros on project success depend on the number of wild horses and burros that can reach the site. If the project site is located within an HMA, WHT, WBT, or WHBT, then grazing and trampling pressure from wild horses should be expected in most cases. Even if the project area is outside any HMA, WHT, WBT, and WHBT, managers should carefully consider the current spatial extent, and growth potential, of any nearby wild horse population. Higher population sizes tend to lead to an expanded spatial area used by the wild horse population. If the number of wild horses is at AML, and there are measures in place to limit the population's growth rate, then wild horse use across the landscape may be distributed enough that a conservation or restoration project could achieve habitat quality goals. Thus, managers should carefully evaluate the likelihood of success of planned conservation and restoration activities if a local or adjacent wild horse population cannot be kept at AML.

Project success is also likely to be influenced by distance to the nearest drinking water source for wild horses. The greater the distance, the lower the grazing pressure that can be expected. Horses require access to large amounts of water; an individual can drink an average of 7.4 gallons [28.0 liters] of water per day (Groenendyk et al. 1988). Despite a general preference for habitats near water (e.g., Crane et al. 1997), wild horses will routinely commute long distances (e.g., 10+ miles [16 kilometers] per day) between water sources and palatable vegetation (Hampson et al. 2010). Managers should expect that any restoration project less than 5 miles [8 kilometers] from water will be subject to use by wild horses in the area. Riparian and wildlife habitat improvement projects that intend to increase the availability of grasses, forbs, riparian habitats, and water are likely to attract and be subject to heavy grazing and trampling by wild horses that live near the project.

Managers need to understand and consider the potential effects of wild horses and burros on conservation and restoration projects and plan accordingly. For certain habitat restoration projects, managers may want to consider installing fencing to discourage use by wild horses, particularly around riparian areas. On BLM and Forest Service lands, temporary fencing for habitat rehabilitation is generally acceptable on HMAs, HAs, WHTs, WBTs, and WHBTs. But permanent fencing often requires a more in-depth environmental assessment or land use plan revision, and should be designed in a way that allows for wild horse and burro movement throughout the rest of the HMA or Territory. The Forest Service also requires National Environmental Policy Act analysis for fence installation. Fencing that excludes wild horses and burros from riparian areas or water development projects that are designed to disperse both riparian and upland use by wild horses and burros are important management tools to protect riparian habitat. Fencing riparian areas to exclude wild horses and burros is generally acceptable as long as water from the area continues to be available to them, and solid pipe fencing is used that can withstand pressure from wild horses and burros. Continued monitoring to assess changes in plant communities and wild horse and burro abundance should be part of any conservation or restoration project where these animals are found.

If AML cannot be achieved, it may be more reasonable to forego a habitat restoration project entirely instead of spending time and resources on projects with a low probability of success. Managers deciding about any project that is near a wild horse or burro population should consider population sizes of wild horses and burros relative to the AML, including explicit schedules for wild horse and burro removals or population growth suppression treatments that are adequate to limit population growth. Unfortunately, high populations of wild horses or burros can substantially affect the ability of land managers to implement conservation measures in some areas. A potential project area with high current wild horse or burro population sizes may become suitable for restoration if the manager can influence priorities and policies such that wild horse and burro populations in the project area are reduced to and maintained at or below high AML.

References

Anderson, J.E.; Inouye, R.S. 2001. Landscape-scale changes in plant species abundance and biodiversity of a sagebrush steppe over 45 years. Ecological Monographs. 71: 531–556.

Barnett, J. 2002. Monitoring feral horse and burro impacts on habitat, Sheldon National Wildlife Refuge. Unpublished report on file with: Sheldon National Wildlife Refuge, Lakeview, OR.

Batchelor, J.L.; Ripple, W.J.; Wilson, T.M.; [et al.]. 2015. Restoration of riparian areas following the removal of cattle in the northwestern Great Basin. Environmental Management. 55: 930–942.

Beever, E.A.; Aldridge, C.L. 2011. Influences of free-roaming equids on sagebrush ecosystems, with focus on Greater sage-grouse. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 273–290.

Beever, E.A.; Brussard, P.F. 2000. Examining ecological consequences of feral horse grazing using exclosures. Western North American Naturalist. 63: 236–254.

Beever, E.A.; Brussard, P.F. 2004. Community- and landscape-level responses of reptiles and small mammals to feral-horse grazing in the Great Basin. Journal of Arid Environments. 59: 271–297.

Beever, E.A.; Herrick, J.E. 2006. Effects of feral horses in Great Basin landscapes on soils and ants: Direct and indirect mechanisms. Journal of Arid Environments. 66: 96–112.

Beever, E.A.; Tausch, R.J.; Brussard, P.F. 2003. Characterizing grazing disturbance in semiarid ecosystems across broad scales, using diverse indices. Ecological Applications. 13: 119–136.

Beever, E.A.; Tausch, R.J.; Thogmartin, W.E. 2008. Multi-scale responses of vegetation to removal of horse grazing from Great Basin (USA) mountain ranges. Plant Ecology. 196: 163–184.

Bi-State Technical Advisory Committee. 2012. Bi-state action plan for the conservation of the Greater sage-grouse bi-state distinct population segment. Prepared for the Bi-State Executive Oversight Committee for conservation of Greater sage-grouse. Nevada and California. March 2012. Carson City, NV: Resource Concepts, Inc. https://www.fws.gov/greatersagegrouse/Bi-State/Bi-State%20Action%20Plan.pdf. [Accessed May 23, 2018].

Boyd, C.S.; Davies, K.W.; Collins, G.H. 2017. Impacts of feral horse use on herbaceous riparian vegetation within a sagebrush steppe ecosystem. Rangeland Ecology and Management. 70: 411–417.

Carothers, S.W.; Stitt, M.E.; Johnson, R.R. 1976. Feral asses on public lands: An analysis of biotic impact, legal considerations and management alternatives. North American Wildlife Conference. 41: 396–405.

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p.

Couvreur, M.; Christian, B.; Verheyen, K.; [et al.]. 2004. Large herbivores as mobile links between isolated nature reserves through adhesive seed dispersal. Applied Vegetation Science. 7: 229–236.

Crane, K.K.; Smith, M.A.; Reynolds, D. 1997. Habitat selection patterns of feral horses in south central Wyoming. Journal of Range Management. 50: 374–380.

Davies, K.W.; Collins, G.; Boyd, C.S. 2014. Effects of free-roaming horses on semi-arid rangeland ecosystems: An example from the sagebrush steppe. Ecosphere. 5: 1–14.

Dawson, M. 2005. The population ecology of feral horses in the Australian Alps, management summary. Unpublished report on file with: Australian Alps Liaison Committee, Canberra, Australia.

Doherty, K.E.; Evans, J.S.; Coates, P.S.; [et al.]. 2016. Importance of regional variation in conservation planning: a range-wide example of the Greater sage-grouse. Ecosphere. 7: e01462.

Douglas, C.L.; Hurst, T.L. 1993. Review and annotated bibliography of feral burro literature. CPSU/UNLV 044/02. Las Vegas, NV: University of Nevada, Cooperative National Park Resources Studies Unit. 132 p.

Earnst, S.L.; Ballard, J.A.; Dobkin, D.S. 2005. Riparian songbird abundance a decade after cattle removal on Hart Mountain and Sheldon National Wildlife Refuges. Gen. Tech. Rep. PSW-GTR-191. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 550–558.

Earnst, S.L.; Dobkin, D.S.; Ballard, J.A. 2012. Changes in avian and plant communities of aspen woodlands over 12 years after livestock removal in the northwest Great Basin. Conservation Biology. 26: 862–872.

Eberhardt, L.L.; Majorowicz, A.K.; Wilcox, J.A. 1982. Apparent rates of increase for two feral horse herds. Journal of Wildlife Management. 46: 367–374.

Federal Register [FR]. 2011. 43 CFR 4700. Protection, management, and control of wild free-roaming horses and burros. https://www.gpo.gov/fdsys/granule/CFR-2011-title43-vol2/pdf/CFR/2011-title43-vol2-part4700. [Accessed Oct. 25, 2018].

Federal Register [FR]. 2012. 36 CFR 222. Range management. https://www.gpo.gov/fdsys/granule/CFR-2012-title36-vol2/CFR-2012-title36-vol2-part222. [Accessed Oct. 25, 2018].

Garrott, R.A.; Oli, M.K. 2013. A critical crossroad for BLM's wild horse program. Science. 341: 847–848.

Garrott, R.A.; Siniff, D.B.; Eberhardt, L.L. 1991. Growth rates of feral horse populations. Journal of Wildlife Management. 55: 641–648.

Geigl, E.M.; Bar-David, S.; Beja-Pereira, A.; [et al.]. 2016. Genetics and paleogenetics of equids. In: Ransom, J.I.; Kaczensky, P., eds. Wild equids: Ecology, management, and conservation. Baltimore, MD: Johns Hopkins University Press: 87–104.

Gooch, A.M.; Petersen, S.L.; Collins, G.H.; [et al.]. 2017. The impacts of feral horses on the use of water by pronghorn in the Great Basin. Journal of Arid Environments. 168: 38–43.

Groenendyk, P.; English, B.; Abetz, I. 1988. External balance of water and electrolytes in the horse. Equine Veterinary Journal. 20: 189–193.

Hall, L.K.; Larsen, R.T.; Westover, M.D.; [et al.]. 2016. Influence of exotic horses on the use of water by communities of native wildlife in a semi-arid environment. Journal of Arid Environments. 127: 100–105.

Hampson, B.A.; de Laat, M.A.; Mills, P.C.; [et al.]. 2010. Distances travelled by feral horses in 'outback' Australia. Equine Veterinary Journal. 42: 582–586.

Hanley, T.A.; Brady, W.W. 1977. Feral burro impact on a Sonoran Desert range. Journal of Range Management. 30: 374–377.

Jessop, B.D.; Anderson, V.J. 2007. Cheatgrass invasion in salt desert shrublands: Benefits of postfire reclamation. Rangeland Ecology and Management. 60: 235–243.

Kaweck, M.M. 2016. Impacts of wild horses and grazing ungulates on riparian areas in Idaho. Thesis. Moscow, ID: University of Idaho. 111 p.

Loydi, A.; Zalba, S.M. 2009. Feral horses dung piles as potential invasion windows for alien plant species in natural grasslands. Plant Ecology. 201: 471–480.

Lubow, B.; Ransom, J.I. 2009. Validating aerial photographic mark-recapture for naturally marked feral horses. Journal of Wildlife Management. 73: 1420–1429.

Lubow, B.C.; Ransom, J.I. 2016. Practical bias correction in aerial surveys of large mammals: Validation of hybrid double-observer with sightability method against known abundance of feral horse (*Equus caballus*) populations. PLoS ONE. 11: e0154902.

MacFadden, B.J. 2005. Fossil horses—Evidence of evolution. Science. 307: 1728–1730.

Maestas, J.D.; Campbell, S.B.; Chambers, J.C.; [et al.]. 2016. Tapping soil survey information for rapid assessment of sagebrush ecosystem resilience and resistance. Rangelands. 38: 120–128.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2014. A field guide to selecting the most appropriate treatments in sagebrush and pinyon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322-rev. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 68 p.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2015. A field guide for rapid assessment of post-wildfire recovery potential in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-338. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 70 p.

National Research Council [NRC]. 2013. Using science to improve the BLM wild horse and burro program: A way forward. Washington, DC: The National Academies Press. https://doi.org/10.17226/13511

Nordquist, M.K. 2011. Stable isotope diet reconstruction of feral horses (*Equus caballus*) on the Sheldon National Wildlife Refuge, Nevada, USA. Thesis. Provo, UT: Brigham Young University. 77 p. https://scholarsarchive.byu.edu/etd/3183/. [Accessed May 23, 2018].

Ostermann-Kelm, S.; Atwill, E.R.; Rubin, E.S.; [et al.]. 2008. Interactions between feral horses and desert bighorn sheep at water. Journal of Mammalogy. 89: 459–466.

Ostermann-Kelm, S.D.; Atwill, E.A.; Rubin, E.S.; [et al.]. 2009. Impacts of feral horses on a desert environment. BMC Ecology. 9: 1–10.

Perry, N.D.; Morey, P.; Miguel, G.S. 2015. Dominance of a natural water source by feral horses. Southwestern Naturalist. 60: 390–393.

Public Law 92–195. 1971. Wild Free Roaming Horses and Burros Act. Authenticated U.S. Government Information. Washington, DC: U.S. Government Printing Office. https://www.gpo.gov/fdsys/pkg/STATUTE-85/pdf/STATUTE-85-Pg649.pdf. [Accessed Oct. 25, 2018].

Public Law 94–579. 1976. Federal Land Policy and Management Act of 1976. Authenticated U.S. Government Information. Washington, DC: U.S. Government Printing Office. https://www.gpo.gov/fdsys/pkg/STATUTE-90/pdf/STATUTE-90-Pg2743.pdf. [Accessed Oct. 25, 2018].

Public Law 95–514. 1978. Public Rangelands Improvement Act of 1978. Authenticated U.S. Government Information. Washington, DC: U.S. Government Printing Office. https://www.gpo.gov/fdsys/pkg/STATUTE-92/pdf/STATUTE-92-Pg1803.pdf. [Accessed Oct. 25, 2018].

Pyke, D.A.; Chambers, J.C.; Pellant, M.; [et al.]. 2017. Restoration handbook for sagebrush steppe ecosystems with emphasis on greater sage-grouse habitat—Part 3. Site level restoration decisions. Circular 1426. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 62 p. https://www.treesearch.fs.fed.us/pubs/53743. [Accessed Sept. 20, 2017].

Ransom, J.I. 2012. Detection probability in aerial surveys of feral horses. Journal of Wildlife Management. 76: 299–307.

Roelle, J.E.; Singer, F.J.; Zeigenfuss, L.C.; [et al.]. 2010. Demography of the Pryor Mountain wild horses 1993–2007. Scientific Investigations Report 2010–5125. Washington, DC: U.S. Department of the Interior, U.S. Geological Survey. 31 p.

Schoenecker, K.A.; King, S.R.B.; Nordquist, M.K.; [et al.]. 2016. Habitat and diet of equids. In: Ransom, J.I.; Kaczensky, P., eds. Wild equids: Ecology, management, and conservation. Baltimore, MD: Johns Hopkins University Press: 41–57.

Scorolli, A.L.; Cazorla, A.C.L. 2010. Demography of feral horses (*Equus caballus*): A long-term study in Tornquist Park, Argentina. Wildlife Research. 37: 207–214.

Seegmiller, R.F.; Ohmart, R.D. 1981. Ecological relationships of feral burros and desert bighorn sheep. Wildlife Monographs. 78: 3–58.

Smith, M.A. 1986. Impacts of feral horse grazing on rangelands: An overview. Journal of Equine Science. 6: 236–238.

Tiller, B.L. 1997. Feral burro populations: Distribution and damage assessment. Pacific Northwest National Laboratory Report PNNL-11879. Fort Irwin, CA: U.S. Army, Department of Public Works. https://www.osti.gov/servlets/purl/663550. [Accessed May 23, 2018].

Turner, J.W. 2015. Environmental influences on movements and distribution of a wild horse (*Equus caballus*) population in western Nevada, USA: A 25-year study. Journal of Natural History. 49: 2437–2464.

Turner, J.W.; Morrison, M.L. 2001. Influence of predation by mountain lions on numbers and survivorship of a feral horse population. Southwestern Naturalist. 46: 183–190.

U.S. Department of Agriculture, Forest Service [USDA FS]. Forest Service manual. FSM 2260.5. Washington, DC: U.S. Department of Agriculture, Forest Service.

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2010a. Wild horses and burros management handbook H-4700-1. Washington, DC: U.S. Department of the Interior, Bureau of Land Management.

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2010b. Wild horse and burro management manual 4710: Management Considerations. Washington, DC: U.S. Department of the Interior, Bureau of Land Management.

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2010c. Wild horse and burro management manual 4720: Removals. Washington, DC: U.S. Department of the Interior, Bureau of Land Management.

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2010d. Wild horse and burro management manual 4740: Motor vehicles and aircraft. Washington, DC: U.S. Department of the Interior, Bureau of Land Management.

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2010e. Wild horse and burro population inventory and estimation: Bureau of Land Management Instructional Memorandum No. 2010-057. Washington, DC: U.S. Department of the Interior, Bureau of Land Management. 4 p.

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2012. Administrative remedies. Washington, DC: U.S. Department of the Interior, Bureau of Land Management. https://www.gpo.gov/fdsys/pkg/CFR-2012-title43-vol2/pdf/CFR-2012-title43-vol2-sec4770-3.pdf. [Accessed Nov. 5, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2015. Research with universities to improve fertility control tools and methods. Washington, DC: U.S. Department of the Interior, Bureau of Land Management. https://www.blm.gov/programs/wild-horse-and-burro/herd-management/science-and-research. [Accessed May 23, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 2017. Herd Area and Herd Management Area statistics. Washington, DC: U.S. Department of the Interior, Bureau of Land Management. https://www.blm.gov/sites/blm.gov/files/wildhorse_programdata_2017hmastats.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2008. Revised, final environmental assessment for horse and burro management at Sheldon National Wildlife Refuge. April 2008. Lake County, OR: U.S. Department of the Interior, Fish and Wildlife Service. https://www.fws.gov/sheldonhartmtn/pdf/Sheldon%20Horse%20and%20Burro%20EA%20Master%20Copy.draft%20final.7Apr08.pdf. [Accessed May 23, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2012. Sheldon National Wildlife Refuge comprehensive conservation plan. Lakeview, OR: U.S. Department of the Interior, Fish and Wildlife Service. https://ecos.fws.gov/ServCat/DownloadFile/7829?Reference=8078. [Accessed May 23, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2013. Greater Sage-grouse conservation objectives: Final report. Denver, CO: U.S. Department of the Interior, Fish and Wildlife Service. February 2013. https://www.fws.gov/greatersagegrouse/documents/COT-Report-with-Dear-Interested-Reader-Letter.pdf. [Accessed May 23, 2018].

Webb, S.D. 1984. Ten million years of mammal extinction in North America. In: Martin, P.S.; Klein, R.G., eds. Quaternary extinctions: A prehistoric revolution. Tucson, AZ: University of Arizona Press: 189-210.

Wolfe, M.L. 1980. Feral horse demography: A preliminary report. Journal of Range Management. 33: 354–360.

Zalba, S.M.; Cozzani, N.C. 2004. The impact of feral horses on grassland bird communities in Argentina. Animal Conservation. 7: 35–44.

Zeigenfuss, L.C.; Schoenecker, K.A.; Ransom, J.I.; [et al.]. 2014. Influence of nonnative and native ungulate biomass and seasonal precipitation on vegetation production in a Great Basin ecosystem. Western North American Naturalist. 74: 286–298.








9. INTEGRATION AND TRADEOFFS

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Introduction

Managing for sagebrush ecosystems that are resilient to disturbance and resistant to invasive plants often requires managers to make tough decisions in the face of considerable complexity and uncertainty. The decisionmaking environment is often characterized by multiple management objectives, limited management authority and capabilities, dynamic ecosystems and plant communities, and uncertain responses to management actions. Resource decisionmakers must be able to determine appropriate objectives based on desired management outcomes and sort through the different management considerations involved in obtaining those desired outcomes. Decisionmakers must also be able to evaluate the tradeoffs associated with diverse and often competing management considerations and determine the long-term positive or negative effects of particular management actions on the resource.

Management decisions are most effective when developed and implemented in an adaptive management framework. Adaptive management promotes flexible decisionmaking and allows adjustments in management as part of an iterative learning process (fig. 2.1) (Goldstein et al. 2013; USDOI 2009). This "decisionmaking process" emphasizes: (1) using the best available information to inform decisions, (2) learning from the results of management decisions and actions, and (3) adjusting management as outcomes from management actions and prior uncertainties become better understood. Adaptive management recognizes the importance of changing ecological and socioeconomic conditions in contributing to ecological resilience to disturbance and resistance to nonnative invasive plants. Rigorous monitoring of management outcomes related to clearly defined objectives provides the scientific basis for adjusting policies or management actions in response to dynamic conditions. Adaptive management is a means for making more effective decisions over time that when properly implemented can help to meet ecological, social, and economic goals, increase scientific knowledge, and reduce tensions among stakeholders.

Decisionmaking in an adaptive management context requires a collaborative process where tradeoffs among resources and management objectives are carefully considered. A structured approach to decisionmaking in natural resources can increase both accountability and specificity (Goldstein et al. 2013; USDOI 2009). Greater attention to key elements (text box 9.1) in the decisionmaking process can help decisionmakers focus on what, why, where, and how actions will be taken.

Top left: Sagebrush ecosystem (photo: Tom Koerner, USDOI Fish and Wildlife Service). Top right: Fire suppression in a cheatgrass dominated site (photo: USDOI Bureau of Land Management). 2nd left: Hand removal of piñon pine with a chainsaw (photo: SageSTEP.org). 2nd right: Pinyon jay, a juniper and piñon obligate species (photo: Richard Crossley from Wikimedia Commons). 3rd left: Deep gas drill rig outside of Pinedale, Wyoming (photo: Tomas J. Christensen, retired, Wyoming Game and Fish Department). 3rd right: Conversion of a sagebrush ecosystem in the West-Central Prairies to agricultural land (photo: John Carlson, USDOI, Bureau of Land Management). Bottom left: Greater sage-grouse at a lek site (photo: USDOI Fish and Wildlife Service). Bottom right: Santa Rosa Mountains and cattle (photo by Nolan Preece, used with permission).

Managers need to take into account many different factors when developing management objectives and deciding on alternative actions aimed at maintaining or increasing resilience to disturbance and resistance to nonnative invasive plants.

Spatial and Temporal Scale. In the Science Framework a multi-scale approach is used to inform different aspects of planning and implementation: (1) the sagebrush biome scale, where consistent data for the range of sagebrush and Greater sage-grouse (*Centrocercus urophasianus*; hereafter, GRSG) can inform budget prioritization; (2) the mid-scale (ecoregion or Management Zone), where assessments are typically conducted to inform budget prioritization and develop priority planning areas; and (3) the local scale, where local data and expertise are used to select project sites and determine appropriate management strategies and treatments within priority planning areas (table 1.2, fig. 1.1). In the decisionmaking process it is necessary to ask whether decisions made at one scale will affect the ability to obtain objectives at other scales. For example, will management decisions at the local scale regarding the locations of fuel treatments or restoration activities have net positive, negative, or neutral effects on landscape connectivity, GRSG, and other species at risk at larger scales? It also is important to ask what the effects of decisions made today will be in 10 or 20 years. For example, will seeding an introduced species in an area that may recover on its own or where restoration of native species may eventually be needed have a net positive, negative, or neutral effect on agency budgets and ecological conditions?

Nontarget Resources. Another important question to ask in the decisionmaking process is: How will decisions to either leave current management practices in place or change management practices affect the resource being managed and nontarget resources over time? For example, will maintaining current grazing practices have net positive, negative, or neutral effects on forage production and habitat quality for GRSG and other species at risk? What will be the longer-term consequences on rangeland health of failure to manage wild horses and burros at Appropriate Management Levels (AMLs)?

Data Availability and Quality. Resource management increasingly involves the use of geospatial data, models, and maps to identify optimal management strategies. The quality and availability of data affect the information available

Text Box 9.1—Activities in a Structured Decisionmaking Approach (Based on USDOI 2009)

- Engage the relevant experts and stakeholders in the decision making process;
- Identify the problem to be addressed;
- Specify objectives and tradeoffs that capture the effects on the ecosystem and the values of stakeholders;
- Obtain the best available information on potential management outcomes and identify the range of decision alternatives from which actions are to be selected;
- Specify assumptions about resource structures and functions and the effects of management outcomes;
- Project the consequences of alternative actions;
- Identify key uncertainties;
- Evaluate risk tolerance for potential consequences of decisions;
- Account for future impacts of present decisions; and
- Account for legal guidelines and constraints.

for making decisions, the management actions that are implemented, and the outcomes of those actions. Consequently, it is necessary to stay informed about new data layers and decision-support systems and their relative strengths and weaknesses (text box 9.2). It also is important to consider both the source and quality of the science that is being used and ensure that it has been published in the peer-reviewed literature (text box 9.2).

Text Box 9.2—Data Considerations for the Science Framework

The models, maps, and data layers used throughout the Science Framework (Chambers et al. 2017; Crist et al. this volume) represent the best scientific information available at the time these documents were written. This information is the result of cutting-edge techniques in remote-sensing of plant communities (e.g., Boyte and Wylie 2017; Xian et al. 2013), combination of data from different spatial scales (Maestas et al. 2016), and new analytical techniques for combining complex datasets (Doherty et al. 2016). This information may be updated as we advance our understanding of these complex ecosystems and develop new and improved data layers and decision-support tools. In addition, new data may arise from interpretation of existing information or application of improved techniques for measuring and modeling dynamic and variable systems across space and time. Updates on the models, maps, and data layers used in the Science Framework are intended to be provided as new science information and geospatial data become available.

When selecting information to inform a decision or updating data layers, practitioners need data that are appropriate to the scale of interest. Technological advances in remote sensing and analysis are providing data with increasingly finer temporal, thematic, and spatial resolution. Although this provides tremendous opportunities for understanding and targeting actions, users must ensure that they have selected the best data to meet project objectives or answer the management questions. For example, most of the species distribution modeling literature uses landscape cover metrics derived from remotely sensed land cover maps that characterize ecological communities (i.e., LANDFIRE). Recently, remote-sensing products have been developed that provide continuous vegetation component values that are more equivalent to ground-based vegetation surveys (Xian et al. 2013). These two types of data are not directly interchangeable and it will be necessary to evaluate which data type is better for the intended application.

Users should critically evaluate uncertainty, measurement error, and model assumptions to understand potential limits to application and inference whenever selecting data for analyses. The original scientific publications should be consulted for information on the types of error, degree of uncertainty, and underlying assumptions. This is particularly important for modeling that integrates multiple spatial datasets, because the degree of error and uncertainty can vary across different datasets and can be compounded when data are combined to create new models or decision-support tools. However, integrative models and spatial products still offer very useful ways of understanding and visualizing complex information when the potential errors and uncertainties are understood and specified. These models and spatial products can guide practitioners to places on the landscape that can be verified by field surveys and local knowledge.

Finally, practitioners should consider the source and quality of the science they are using, because new geospatial layers, tools, and applications are being developed rapidly. They should use data that have been published in the peer-reviewed literature. The rigors of the peer review process necessary for publication in respected sources result in quality control and assurance that nonpeer-reviewed literature may or may not have acquired. Although new maps, data, or tools may appear to provide exciting new opportunities for analysis and decisionmaking, caution should be used in applying this information before adequate documentation is available and peer-review of methods and assumptions has been completed.

Dealing with uncertainty is one of the greatest challenges in decisionmaking. Changes in administrative priorities, policies, and economic resources can all cause uncertainty in the types of decisions that should be made as well as the outcomes of those decisions. In addition, there are several well-recognized sources of uncertainty specific to making natural resource decisions (Conroy et al. 2011; USDOI 2009; Williams et al. 2002). **First**, environmental uncertainty, or uncertainty in ecosystem and species responses to factors such as disturbances, weather events, climate change, and management actions, is a well-known source of uncertainty that characterizes all natural systems and requires little explanation. **Second**, partial observability, or the need to estimate and model the relevant "quantities" that characterize natural systems because of our inability to directly observe nature, often limits our ability to accurately determine the resource "quantities" that are the targets of management. For example, the amount of forage production on an allotment is often estimated from sampling a small number of plots and estimating values; the acres of habitat to support a particular species is often estimated from limited research on habitat requirements, often in a different location. **Third**, partial controllability is the frequent inability to apply management actions directly and with high precision. An example is aerial seeding of postfire reclamation species. **Fourth**, structural uncertainty is the uncertainty in the models that predict system responses to specific management actions. Structural uncertainty is often represented by alternative models of system dynamics, each with associated measures of relative credibility. Reducing this type of uncertainty is a key objective of adaptive management (Walters 1986; Williams et al. 2002). Dealing with uncertainty in decisionmaking requires recognizing its existence, establishing rules whereby an optimal decision can be made in the face of uncertainty, and reducing uncertainty where possible (Conroy et al. 2011).

Application to Management

This section is intended to facilitate the decisionmaking process by integrating the management considerations for each of the management topics addressed in this volume and identifying the tradeoffs involved in managing for the different objectives and resources associated with each management topic. On October 17–19, 2017, management and science experts from different agencies and organizations met in Boise, Idaho, to evaluate the management considerations and tradeoffs for the different topics. Specific objectives were to: (1) identify and discuss how to integrate project objectives and evaluate the tradeoffs that need to be considered across scales in decisions about land management activities in sagebrush ecosystems, and (2) develop scenarios that identify and discuss how tradeoffs influence priorities for managing dominant threats in the western and eastern portions of the sagebrush biome. As a result of this meeting and subsequent work by the editorial team, "management scenarios" were developed that focus on the management considerations and tradeoffs involved in managing (1) invasive annual grasses and uncharacteristic wildland fire, (2) juniper (*Juniperus* spp.) and piñon pine (*Pinus* spp.) expansion, and (3) land use and development (e.g., cropland conversion and associated invasion of nonnative species). In addition, an "integration table" was developed that includes all paired combinations of the topics addressed in this volume and identifies the desired outcome, management considerations, and tradeoffs for each paired combination. The integration table also includes any critical information needs and policy needs that were identified at the meeting.

The scenarios and integration table were not developed for a particular management agency and thus do not consider the different policies of individual agencies. Instead, collective management considerations are provided for all managing entities. Managers can incorporate other management considerations and tradeoffs important for their particular agency, geographic region, or program.

Management Scenarios

The management scenarios illustrate how different management considerations and tradeoffs (table 9.1) are taken into account when developing management actions and making management decisions about potential actions. Supporting information is found in Part 1 of the Science Framework (Chambers et al. 2017a; hereafter, Part 1). An overview of persistent ecosystem threats is in Part 1, section 5. These threats include nonnative invasive plant species, altered fire regimes, conifer expansion, and climate change, as well as land use and development threats including cropland conversion, energy development, mining, roads and other infrastructure, urban and exurban development, recreation, wild horse and burro use, and improper livestock grazing. Geospatial analyses with overlays of key data layers can help (1) evaluate the type, presence, and level of threat to ecological types and vegetation communities; (2) target areas for adaptive management; and (3) determine the most appropriate types of management actions. Part 1, section 8 presents data and analytical methods for identifying priority areas for management within ecoregions or Management Zones and evaluating both persistent ecosystem and land use and development threats. The use of higher resolution spatial data, combined with local information and knowledge, helps managers and stakeholders refine project areas and determine the most appropriate management strategies and is detailed in Part 1, section 9. Management strategies for persistent ecosystem threats, climate change, and land use and development threats are identified in table 1.4 (this volume), and recommendations for prioritizing and targeting strategies are in table 1.3 (this volume).

Invasive Annual Grasses and Uncharacteristic Wildfire

This scenario addresses the ongoing spread of invasive annual grasses and resulting uncharacteristic wildfires. The desired outcome is to reduce the occurrence and spread of invasive annual grasses in these landscapes and the loss of sagebrush habitat due to uncharacteristic wildland fire. The emphasis is on landscapes with low to moderate resilience and resistance, where these issues are most problematic and additional management focus is needed. Although the scenario was developed largely for the northern and central Great Basin and Columbia Plateau in the Cold Deserts, it also is applicable to the Western Cordillera (see fig. 1.1), where invasive annual grasses are spreading and uncharacteristic wildfires are occurring.

Three management approaches are provided to help address the threats of invasive annual grasses and wildfire in low to moderate resilience and resistance landscapes. These approaches are intended to work in tandem with the management considerations and tradeoffs described in the integration table (table 9.1) and build on the information provided in tables 5.1 and 5.2. These approaches are: (1) **prevention** of invasion of existing intact sagebrush habitat by nonnative invasive annual grasses, (2) **intervention** to help restore areas at risk of becoming dominated by invasive annual grasses and higher fire frequencies, and (3) **containment** of invasive annual grasses to decrease the effects and spread of the fire/invasive annual grass cycle in low to moderate resilience and resistance areas.

The use of the different management approaches depends on the extent and relative abundance of invasive annual grasses and associated wildfire occurrences. Multi-scale assessments that include geospatial datasets, monitoring data, and field surveys can help identify the most appropriate scale for applying the management approaches within a region. Geospatial datasets and methods are provided in text box 9.3 to help identify areas on the landscape where these management approaches apply. Areas managed for **prevention** are those where sagebrush communities are ecologically intact and have little to no cover of invasive annual grasses. Areas managed for **intervention** typically have lower cover of sagebrush or shrubs and perennial grasses and forbs, but a relatively low cover of invasive annual grasses. These areas may be at risk of invasive annual grass dominance and intervention may help them return to a more native species-dominated state. Areas managed for **containment** have moderate to high cover of invasive annual grasses and very low cover of shrubs and native grass and forbs. These areas are difficult to restore to a native species-dominated state due to invasive dominance. The three management approaches align with the five invasion states in tables 5.1 and 5.2. **Prevention** areas can be defined as "invasion free" and "trace"; **intervention** areas as "mild" to "moderate"; and **containment** areas as "invasion dominated."

Text Box 9.3—Mapping Prevention, Intervention, and Containment Areas for Managing Invasive Annual Grasses and Uncharacteristic Wildfire

A multi-scale spatial assessment can be used to identify and delineate where to apply **prevention, intervention,** and **containment** management approaches in landscapes with low to moderate resilience and resistance to invasive annual grasses. Geospatial data layers and a mapping framework for prioritizing areas for management at regional scales are in Part 1, section 8 and Appendix 8. The highest resolution data available for the assessment area should be used (text box 9.2). The categorization of an area for **prevention, intervention,** or **containment** should include characteristics such as: (1) the cover of native, intact sagebrush ecosystems; (2) the degree of connectivity among sagebrush habitats; and (3) priority resource values such as Greater sage-grouse (GRSG) habitat. Fire risk assessments should be used to identify areas with low to moderate resilience and resistance that have a higher probability of experiencing fire (Part 1, Appendix 10). Relevant data layers include ecological site types or vegetation cover types, resilience and resistance categories, and surface land management. Other information may include the potential of an area to provide native seed sources and reserves.

The proportion of the landscape dominated by sagebrush land cover provides information on the landscape context and potential habitat suitability for GRSG and for other sagebrush dependent species at risk (Chambers et al. 2017a,c; Knick et al. 2013). For example, sagebrush cover categories are based on the proportion of the landscape dominated by sagebrush (5-kilometer [3-mile] rolling window; low = 1–25 percent; moderate = 26–65 percent; high = >65 percent land cover). Data on topography, postfire recovery sites, rare species habitats, migratory pathways, and GRSG lek locations or population indices can refine the identification of these areas.

The use of Assessment, Inventory, and Monitoring (AIM) data, field survey data, and local expertise can be used for refining distinctions between the different areas. Tables 5.1 and 5.2 provide five invasion states that can further refine the delineation of **prevention**, **intervention**, and **containment** areas. **Prevention** areas can be defined as "invasion free" and "trace"; **intervention** areas as "mild" to "moderate"; and **containment** areas as "invasion dominated."

To implement the approaches, land management objectives of "**prevention**," "**intervention and restoration**," and "**containment and long-term rehabilitation**" are developed and assigned based on coordination among the science and resource specialists across a management jurisdiction. Strong partnerships and collaboration between State and Federal invasive programs are needed for targeted prevention, control, and eradication of invasive plants. In **prevention** areas, managers should minimize management activities known to spread invasive plants and implement a strong monitoring and eradication program, such as an Early Detection and Rapid Response program (EDRR) (USDOI 2016). Other prevention measures are in table 5.1. In **intervention** areas (previously burned or unburned), managers should emphasize restoring and maintaining resilience to wildfire and resistance to nonnative annual grass invasions. Primary intervention objectives include increasing the extent, connectivity, and ecological functioning of sagebrush ecosystems. These objectives are requisite to meeting other landscape objectives such as increasing the sustainability and resilience of habitat for different species of wildlife, forage for livestock, and other resources such as native seed reserves. Intervention measures include eradication, suppression, containment, and active restoration. Their use should be aligned with local environmental conditions to optimize success.

In **containment** areas, the management focus is on removal and containment of invasive plants to protect adjacent or nearby areas from invasion and address the higher frequency and larger extent of wildfire in these areas. The effectiveness of different herbicide treatments and seeding strategies can be tested through carefully designed treatments and long-term monitoring. This information can be shared among land management agencies. Monitoring of **containment** areas can provide information on changes in invasive annual grasses and other invasive plants and help identify new invaders. Monitoring along the interfaces of highly invaded sites and intact sagebrush communities can help provide information on where containment strategies have been successful and where adaptive management is needed. Once an area is designated as having a management objective of "containment," loss of ecological function may occur due to containment strategies. However, there may be opportunities for rehabilitation when methods become available in the future. In addition, surveys of potential **containment** areas for endangered, threatened, or sensitive species, species of concern, and known rare species can be used to determine whether these areas or portions of these areas should be reclassified as **intervention** areas to protect these resources. The development of evaluation criteria for restoration potential, along with an understanding of associated tradeoffs, will help inform the classification of these areas.

Climate Variability and Adaptation

Climate and climate variability have a strong influence on management considerations and tradeoffs and, thus, management approaches for low to moderate resilience and resistance areas. Identification of which invasive plants are likely to spread and of the areas susceptible to invasion coupled with EDRR monitoring can help managers decide where to implement **prevention, intervention,** and **containment** strategies to facilitate climate adaptation. Scenario planning also can assist with balancing the tradeoffs of different management approaches (e.g., assisted migration).

To help maintain or enhance the resilience and resistance of areas managed for **prevention** and **intervention**, native plant species distributions should be allowed to transition and adapt to changing climatic and environmental conditions. In areas managed as **intervention** and **containment**, resilience and resistance may be maintained or facilitated through vegetation treatments that help communities transition to new states or site types where appropriate. The use of carefully designed treatments and monitoring can help identify successful methods for assisted migration of native plant species (Bucharova 2017). Monitoring for appearance of novel invaders, changes in biodiversity and native species populations, and movement of key species can be used to evaluate how changing landscapes are responding to treatments in all three management approaches.

Land Uses, Development, and Rehabilitation

Anthropogenic land uses and developments that are known to serve as invasive and noxious weed vectors, such as roads, pipelines, fuel breaks, utility corridors, juxtaposed agricultural practices, grazing, and mining, should be addressed in all three management approaches. Land uses and developments that serve as vectors for invasive plants should be redirected around **prevention** areas or reduced in number, frequency, and extent to reduce impacts; minimized and monitored in **intervention** areas; and where resource values are not at risk, focused in **containment** areas. Management activities should use defined best management practices (BMPs) for preventing the spread of invasive plants. See tables 5.1 and 5.2 for other "on the ground" **prevention, intervention,** and **containment** strategies.

Grazing should be minimized in **protection** areas and potentially refocused to other areas that are more resilient to grazing to maintain no to low levels of invasive plants. For **intervention** areas, use of alternative grazing strategies (e.g., shifting the season of use, using outcome-based grazing, creating grass banks) can help contain spread of invasive plants. Where alternative grazing strategies may increase risk to the operator or permittee, outcome-based grazing and evaluating the degree of risk can help provide effective solutions. Identifying **containment** areas that may be used as grass banks or to extend grazing seasons may also address these tradeoffs. Grazing permits should include the season, duration, and amount of grazing that can sustain native grasses and forbs based on state-and-transition models for low and moderate resilience and resistance sites (section 7). They also should include plans for drought conditions and changing weather and climate patterns. Alternative grazing strategies such as changing season of use, targeted grazing, and grass banks could be focused in containment areas to reduce contiguous fuels throughout these areas. Grazing strategies developed for the three approaches will need to be adaptive and responsive to climate and weather patterns that result in changes in forage availability.

Control and removal of invasive plants through the use of adaptive management, EDRR strategies (USDOI 2016), and focused invasive plant removal treatments should become a primary management goal for all Federal and State management agencies. At the field office and district scale, spatial mapping, field surveys, and use of monitoring data can augment geospatial data to refine **prevention, intervention,** and **containment** areas (text box 9.3). The primary factors to consider are site conditions, relative abundance of residual grasses and forbs, relative abundance of the invader, and proximity and juxtaposition to invasive plant dispersal vectors. See tables 5.1 and 5.2 for "on the ground" **prevention, intervention,** and **containment** strategies.

In **prevention** areas EDRR is used to quickly remove new invasive plants. In **intervention** and **containment** areas strategies depend on the magnitude of the invasion, but can include a variety of treatments such as herbicides, seeding, and transplants, to reduce the cover and spread of invasive plants (section 5). In **intervention** areas, invasive plant control treatments should minimize soil surface disturbance and disturbance of biological soil crusts. Restoration should focus on seeding in areas that lack perennial grasses and forbs. Spatial mapping can be used to target restoration efforts between intact sagebrush patches to increase sagebrush habitat connectivity over the long term. Use of herbicides followed by seeding should be prioritized to control spread from **containment** areas, especially those located adjacent to **intervention** or **prevention** areas. Treatment success may be challenging and multiple interventions may be required, especially in **containment** areas. In general, long-term monitoring and adaptive management practices should be used to evaluate treatment successes, test other invasive plant removal strategies, identify challenging areas, and determine when **intervention** areas may need to be considered **containment** areas.

Following wildfire or other disturbances, tradeoffs to consider for invasive plant management in **prevention** areas include the potential negative effects of using herbicides and seeding on native species recovery versus allowing natural recovery. In **intervention** areas, herbicide application, seeding treatments, and other postfire or disturbance recovery efforts should be targeted. Management objectives for seeding in **prevention** and **intervention** areas should focus on reestablishing native species and ecological diversity rather than seeding specifically for livestock grazing benefits. Establishing restoration islands of diverse native forbs, bunchgrasses, and other shrubs can mimic natural recovery and succession after wildfire in sagebrush communities with depleted native herbaceous species.

The use of specific livestock grazing regimes for low to moderate resilience and resistance areas is essential for all restoration and postfire recovery efforts because grazing or use of seeded areas may inhibit recovery. Managers should consider structuring grazing regimes depending on the designated management approach—**prevention, intervention,** or **containment**. For example, spring and early summer grazing could be prioritized in **containment** areas before **intervention** and **prevention** areas. Focused monitoring and management of cattle grazing are needed to adapt grazing strategies where recovery goals are not met. Prioritizing management of wild horse and burro populations for population reductions where these populations exceed AML and are affecting ecological conditions will help protect treated and seeded areas (section 8). These types of strategies are applicable to other restoration activities for invasive plant control after disturbance.

Wildland Fire Management

Fire risk assessments are useful in determining priorities for wildfire management objectives for **prevention, intervention,** and **containment** areas. **Prevention** and **intervention** areas should receive higher priority for fire suppression efforts, especially if located next to a containment area. This juxtaposition increases the risk of wildfire and conversion to annual invasive grasses in **prevention** and **intervention** areas. Fuel treatments should be focused in **intervention** and **containment** areas. Tables 5.1 and 5.2 offer more specific management strategies. The following approaches, when integrated, can help reduce the occurrence of fire disturbances in lower resilience and resistance areas and mitigate potential natural resource tradeoffs in fuel treatments and wildfire management decisions.

First, a strong emphasis on wildland fire prevention strategies in wildland-human interface areas that focus on common causes of human ignitions such as powerlines, fireworks, campfires, target shooting, and vehicles parking on roadsides is needed to help reduce wildland fires in **prevention, intervention,** and **containment** areas. Across the western states, human-caused fires accounted for 31 percent to 97 percent of all wildfires (Balch et al. 2017). Strong partnerships and collaboration are needed between State and Federal wildfire prevention and mitigation programs to help reduce human-caused fires. Industries, land users, and recreationists need to be included in these partnerships.

Second, siting of fire suppression activities (e.g., firelines, burnouts) and equipment in containment areas where they occur adjacent to intervention and prevention areas can be used to minimize disturbance in intervention and prevention areas. Other strategies include training on invasive plant awareness, and incorporating invasive plant information and management into Fire Incident Action Plans.

Third, strategically placed and consistently maintained fuel treatments such as fuel breaks alongside roads within **intervention** and **containment** areas may help reduce substantial losses of sagebrush communities due to wildfire by aiding wildfire suppression efforts and reducing fire spread. The use of fuel breaks should be prioritized for areas of higher fire frequency to help protect wildland-urban interface areas, **prevention** areas, and **intervention** areas. The effectiveness of fuel breaks across large landscapes is unknown, and fuel breaks alone may not reduce the extent of uncharacteristic fire in sagebrush communities (Shinneman et al. 2018). However, different lengths and widths of fuel break networks can be tested using fire simulation modeling to identify strategic placement and design. Design and placement should take into account the fuels in the landscape, fire response, and operational efficiency. Monitoring and adaptive management will further inform their best use and placement over time.

Fuel breaks are for the sole purpose of wildland fire management and should not be used to achieve other management goals. Plant materials used in fuel breaks should have traits such as low stature to reduce flame lengths or resistance to invasive plant species. Native and nonnative species selected for seeding fuel breaks should not be managed as forage for wildlife or have traits that rely on grazing regimes to retain low biomass.

To help avoid unintended management consequences and ecological impacts, the design, placement, and long-term management of fuel breaks should be carefully evaluated before construction. Fuel breaks can become dominated by invasive annual grasses and serve as fire ignition points, especially when located next to wildland-urban interface areas or popular recreation sites. Therefore, consistent fuel break maintenance in perpetuity needs to be a high management priority to maintain their effectiveness for fire suppression efforts over time. To help mitigate unintended ecological impacts, managers should assess effects on wildlife habitat and adjacent ecosystems before deciding to construct a fuel break network (see Shinneman et al. 2018). Tradeoffs, such as habitat loss, fragmentation, and impeding wildlife species movements, may be mitigated by using wildlife habitat fragmentation thresholds and varying fuel break width, length, and placement across the landscape.

Several sections in this volume will be useful in evaluating management considerations and tradeoffs associated with fuel breaks (sections 4 through 6). Also see table 1.4.

In conclusion, this scenario provides a spatially integrated management approach that builds on many of the strategies in tables 5.1 and 5.2. There are many other factors to consider for applying **prevention, intervention,** and **containment** management approaches in low to moderate resilience and resistance areas, including:

- Special status wildlife and plant species
- Availability of seed
- Land use plan flexibility
- Stakeholders' willingness to engage and collaborate
- Unforeseen or unplanned disturbance
- Staff turnover—key personnel
- Topography and terrain access
- Availability of grass banks and grazing options
- Availability of useful monitoring data in and adjacent to site
- Emerging invasive species that pose a risk to these sites (early watch species)

Juniper and Piñon Pine Expansion

This scenario addresses the expansion of juniper and piñon pine trees into sagebrush ecosystems and the associated decline in sagebrush dependent species and resource values. The desired outcome is to reduce the loss of sagebrush resulting from juniper and piñon expansion, while maintaining a mosaic of sagebrush and juniper and piñon habitats needed for species dependent on these ecosystems. The focus is on moderate to high resilience and resistance areas at mid- to high elevations where juniper and piñon expansion is causing sagebrush habitat loss. This integrated management scenario discusses identifying juniper and piñon areas for targeted removals, addressing the threat of increasing invasive plants during site selection and treatment implementation, and using treatment methods that mimic natural disturbances which may help mitigate the negative effects on the species that depend on these expansion areas.

Identification of areas where juniper and piñon are expanding into currently occupied GRSG habitats or other threatened or at-risk species habitats is needed to locate the highest priority sites for tree removal treatments to maintain or restore sagebrush communities. The framework and geospatial datasets provided in Part 1, section 8 can be used to help select potential treatment sites. After identifying potential treatment sites, managers should coordinate with other science and resource specialists (State and Federal) to evaluate potential conflicts with other species' conservation needs and other resources to determine appropriate treatments. An approach for evaluating a site's relative resilience to disturbance and resistance to nonnative invasive plants and selecting appropriate treatment methods is in Miller et al. (2014).

Management objectives for juniper and piñon removals should incorporate potential changes in native juniper and piñon species distributions, fluctuations in populations, and adaptations to changing climatic and environmental conditions. Considering this information in site and treatment selection can help in managing for longer-term ecosystem resilience and multiple uses. When identifying juniper and piñon removal sites, practitioners should consider the presettlement distribution and history in relation to the number of acres (hectares) of juniper and piñon lost to disturbances, such as wildland fire, insects, and drought (see Board et al. 2018), as well as past removals over a specified period of time (past one to two decades), to help determine the appropriate number of acres for targeted removal. Continued monitoring of juniper and piñon as well as sagebrush habitats that are lost to disturbances such as wildland fire and drought over time can be used to identify where adjustments are needed in proposed removals and help adapt management strategies for local and regional areas. Recent increases in loss of juniper and piñon woodlands through natural disturbances may be contributing to removal goals, or these goals may have even been met in some areas. This type of information will improve understanding of how much targeted removal should occur across a geographic area and help to plan removals in the context of natural disturbances and climate change.

Areas should be prioritized for treatment where removals will not result in increases or dominance of invasive plants because of the disturbance caused by the removal treatment. Field-based surveys are needed to identify areas for removals that have sufficient cover of sagebrush and native grasses and forbs in the understory for site recovery (Miller et al. 2014). If expansion sites are relatively warm and dry, invasive annual grasses are present, and sagebrush or perennial grasses have low abundance, there is a strong possibility that the site will convert to invasive plant dominance after tree removal. Managers can consider treating the site with pre-emergent herbicides after tree removal and monitoring for recovery of perennial grasses and forbs (but see Pyke et al. 2014). However, seeding perennial native grasses and forbs may be required to facilitate recovery of these types of sites, and investments in tree removal will produce higher returns in areas that have the potential to recover without additional treatments.

Thresholds of native perennial grasses and forbs needed to ensure recovery of sagebrush ecosystems can be found in Davies (2008), Chambers et al. (2014d), and Miller et al. (2014). Recent research related to juniper and piñon treatments is in sections 4 and 5.

Removal of juniper and piñon in expansion areas may have negative consequences for species dependent on the different habitat conditions these areas provide (e.g., seed caching areas for pinyon jay [*Gymnorhinus cyanocephalus*] and winter habitat for mule deer [*Odocoileus hemionus*]). Expansion areas include edge and open transitional habitats important to a variety of species including some that are in sharp decline. Designing removals that mimic the patterns of natural disturbance such as wildland fire and drought will help ensure that the habitat needs of these species are taken into account and that objectives in land management plans for maintaining a mosaic of sagebrush and juniper and piñon habitats are achieved. To meet these needs, removal treatments can be designed to incorporate the following:

- Creation of transitional (feathered) and more convoluted-shaped edge habitats between sagebrush and juniper and piñon to avoid sharply contrasting and straight edges (e.g., dense juniper and piñon woodland adjacent to sagebrush)
- Creation of openings within juniper and piñon stands with high density and cover
- Leaving older piñon pine trees that produce pine nuts

During and after removals and associated treatments, there may be a need to temporarily change grazing management regimes. Shifting seasons of grazing use depending on climate and weather patterns can help encourage recovery of sagebrush habitats and deter invasive plants from spreading into treated sites. However, this can have economic effects on the grazing operator or permittee. Planning for the use of alternative grazing areas for the time needed to allow recovery after removal will help mitigate effects on the grazing operator. Where wild horse and burro management areas overlap or are adjacent to areas for targeted juniper and piñon removal, it may be necessary to reduce wild horse and burro populations to AML if the juniper and piñon removal treatments are to succeed (section 8).

Land Use and Development Threats

This scenario addresses two closely related issues. The first is type conversions such as those resulting from agricultural uses that degrade habitat quality or remove habitat through conversion to other land uses. The desired outcome is to prevent loss of sagebrush habitats and reduce fragmentation while maintaining or improving connectivity at multiple scales. The second issue is land uses that facilitate increases in invasive annual grasses and forbs. Here the desired outcome is to prevent new invasions and reduce expansion and spread of existing invasive plant threats that may be increased with surface-disturbing activities, such as energy development and conversion of sagebrush communities to cropland. The emphasis is on the eastern portion of the sagebrush biome, including the Northwestern Plains, Wyoming Basin, and Colorado Plateau, and Southern Rockies (see fig. 1.1), but management strategies are broadly applicable.

In the eastern portion of the sagebrush biome, land use impacts often represent a more immediate risk to high quality, intact, and connected GRSG habitat than wildland fire, invasive plant species, or the effects of a changing climate. For example, cropland conversion can pose a more immediate and lasting risk to GRSG habitat quantity or connectivity than is posed by invasive plant species or wildland fire. In the Conservation Objectives Team Report (USDOI FWS 2013), cropland conversion was ranked a widespread and persistent threat on more productive soils for 6 of 15 GRSG populations in the eastern range. The West-Central Semiarid Prairies (Management Zone I) has the highest percentage of private lands and highest amounts of filled cropland of the Management Zones (Doherty et al. 2016; Knick et al. 2011, table 12.1). GRSG extirpations have occurred in areas where cultivated crops exceeded 25 percent of landscape cover (Aldridge et al. 2008) and recent studies show that 96 percent of active leks are surrounded by less than 15 percent cropland in Management Zone I (SGI 2015; Smith et al. 2016). Loss of landscape cover of sagebrush associated with energy development has been well documented in recent analyses, especially for oil and gas. Oil and gas development affects 8 percent of sagebrush habitats, with the highest intensities occurring in Management Zone I and Management Zone II (Part 1, section 5.3.2). Mining is considered a persistent and widespread threat to 8 of 15 GRSG populations in the eastern range (USDOI FWS 2013) (Part 1, section 5.3.2).

Numerous studies have found invasive plant species associated with soils disturbed by development activities and have noted that restoration becomes much more difficult once these species are established (see Part 1, section 5.3.6). The cumulative effects of anthropogenic development and persistent ecosystem threats may be most evident for sites with relatively warm or dry soil temperature and moisture regimes that have relatively low resilience and resistance; these effects may intensify as the climate warms (Part 1, section 5.3.6). The most successful tool for maintaining sagebrush ecosystem resistance to nonnative plant invasions is generally to manage for sufficient density and cover of native perennial grasses and forbs and biological soil crusts to prevent the establishment or population growth of the invader (Chambers et al. 2014b,d).

Best management practices can reduce or prevent introductions of invasive plant species to new areas and can help maintain the resistance of the ecosystem to invasion. Monitoring (including EDRR) can be used to identify areas where preventive action can decrease the risk of reaching the levels of invasive annual grasses currently found in parts of the Great Basin. Monitoring can also provide the necessary information to quickly respond to reports of new sightings of invasive plant species. Although invasive annual grasses are arguably the most widespread ecosystem disrupters across the sagebrush biome, other plant life forms are also responsible for impacts to the sagebrush uplands and the riparian and wet meadow habitats. These invasive plant species should be included in EDRR efforts as well (see section 5 and Appendix 3). EDRR for these species can be enhanced through the use of standardized vegetation monitoring programs such as the Bureau of Land Management's Assessment Inventory and Monitoring (AIM) and Natural Resources Conservation Service's National Resources Inventory (NRI) efforts which, when combined with enhanced data tracking systems, can be used to locate and treat identified areas in the same year that they are discovered.

In much of the eastern part of the sagebrush biome, the culture, customs, and practices of landscape management have formed within a relatively resilient ecosystem. Failure to consider how land uses and impacts can degrade habitat and increase the likelihood for invasive plants may give a false sense of resilience and resistance. It is important to ask how decisions to either leave current management practices in place or change management practices will affect the resource being managed and nontarget resources over time. To fully address this question, it will be necessary to reexamine current assumptions about the effects of weather and climate on environmental responses and underlying assumptions about the expected results of management actions. Use of appropriate BMPs can help adapt management over time.

Type Conversions

Several management strategies can be used to prevent habitat loss from land uses that degrade habitat and conversion of sagebrush to cropland. These include conservation agreements (easements and Federal and private lands programs, such as the Conservation Reserve Program, Agricultural Conservation Easement Program, Environmental Quality Incentives Program, and Candidate Conservation Agreements with Assurances), land use regulations, and land acquisitions. Factors to consider are:

- Willingness of private landowners to utilize conservation programs
- Wildlife and habitat resource values
- Subsidies for conversion
- Benefits in terms of larger scale connectivity
- International agreements
- Cost of managing the land after acquisition or agreement
- Spatial strategy for acquiring lands or conservation easements (or both) to improve connectivity
- Positioning of existing conservation easements
- Subsurface mineral ownership issues potentially impacting durability and benefit of conservation actions used to address other primary threats
- Existing regulations that may limit the amount of disturbance allowed

There are tradeoffs to consider when easements or land purchases are used to meet conservation objectives. Easements may limit or restrict other land uses and result in a potential long-term economic loss to farmers or to the community. Acquisition of lands results in both short-term and long-term costs associated with managing the land to achieve desired conditions or management goals. Additionally, acquiring easements opportunistically based on the willingness of landowners may not be the most strategic approach to reaching desired outcomes, such as habitat connectivity, or may not occur in areas with the most important resource conditions (i.e., low versus high resilience and resistance and wildlife habitat values, such as GRSG population densities and seasonal habitats).

Land Uses that Facilitate Increases in Invasive Annual Grasses and Forbs

Preventing new nonnative plant invasions and reducing the expansion or spread of existing invasive plants begins by identifying uninvaded areas and areas at increased risk of invasion and prioritizing management responses. Once the size and impact of an invasion are determined, the recovery potential of the area is evaluated. Uninvaded areas, especially those with lower resilience and resistance, are often at risk and should be identified for prevention strategies to keep "clean areas clean."

Tables 5.1 and 5.2 provide many management strategies for prevention of invasive grasses and forbs. Integrated pest management techniques are used to prevent introductions and reduce or control invasive plant spread into sagebrush habitat. Increased EDRR monitoring for invasive annual grasses and forbs, such as cheatgrass (*Bromus tectorum*), medusahead (*Taeniatherum caput-medusae*), ventenata grass (*Ventenata dubia*), leafy spurge (*Euphorbia esula*), and Russian knapweed (*Acroptilon repens*), is used in high priority areas (i.e., high GRSG population density and GRSG breeding habitat) near areas with development potential (cropland conversion or oil and gas potential). Strong working partnerships with landowners and local governments are developed to treat invasive plant species across ownership boundaries. Where development will occur, Conditions of Approval are employed for regulated activities to reduce the invasion and spread of unwanted nonnative invasive plants. Examples are reducing or controlling invasive plants in an area before disturbance and during active development and production; power-washing construction equipment before transporting to the project area; reclaiming the site to meet objectives for resistance to invasive plants and other objectives, such as value to wildlife; and educating vehicle operators about the dangers of fire ignition resulting from sparks caused by drag chains, cigarettes, and other ignition sources.

Factors to consider are:

- Willingness of private landowners to treat invasive plants
- Adequacy of post-disturbance reclamation requirements, implementation, and outcomes
- Coordination of treatments across ownership boundaries
- Use of methods other than chemical treatment, such as targeted livestock grazing, to control invasive plants
- Durability of treatment efforts to ensure that treatments are maintained long enough to avoid reestablishment of invasive plants and the potential for other land uses (development, infrastructure, grazing [livestock, wild horse and burro, wildlife]) to undo the efforts being implemented

Several tradeoffs need to be considered when implementing these strategies, including: (1) costs of conducting monitoring and potential treatments necessary to control invasive plants versus not having influence on how sagebrush communities are managed, (2) fewer resources for monitoring elsewhere or for other resources, (3) possible increased use of herbicides (which may have unintended impacts to nontarget species), and (4) herbicide application without emphasis on increasing desirable native species (herbicide treatments may create voids in which new invasive plants may occur).

Integration Table

The integration table is a tool that can be used to help develop management objectives and make management decisions regarding potential actions (table 9.1). The table is designed to help identify the relevant management considerations and tradeoffs involved for the different management topics addressed in this volume. It can be used to cross-check the relevant topics for a particular objective or desired outcome to ensure that all of the relevant management considerations and tradeoffs have been taken into account.

Table 9.1—The desired outcomes, management considerations, and tradeoffs, as well as any critical information needs and policy needs, for each combination of the topics included in this volume. The information provided for the integrated topics can be used to help managers determine whether all of the relevant management considerations and tradeoffs have been taken into account when making decisions regarding potential management actions. The length of the table and the inclusion of some repetition reflects the need to ensure that the relevant management considerations and tradeoffs were included for each integrated topic. It is anticipated that only a subset of the integrated topics will need to be reviewed for any particular action.

MONITORING and CLIMATE ADAPTATION

Desired Management Outcome:

Resilience and resistance are maintained and transitions to desirable new states or site types are facilitated by collecting monitoring data that can be used to understand where and how ecosystems are changing and to inform adaptive management.

Management Considerations:

(1) Identify monitoring questions, ecosystem attributes, and indicators needed to evaluate effects of climate change and incorporate them into monitoring programs.

> **Tradeoff**: Durability of conservation and restoration efforts may be impacted if projects do not incorporate climate change or transition zone information due to changes in resilience and resistance, soils, and other resource conditions.

(2) Incorporate climate change information into project planning and use it to prioritize monitoring efforts among resources and treatments. Then adapt management based on results.

> **Tradeoff:** Increased monitoring requires greater investment and other areas or resources may be monitored less intensively. If Assessment Inventory and Monitoring (AIM) sampling is increased, it may be difficult to maintain sampling rigor.

(3) Monitor areas projected to change rapidly and areas with strong environmental gradients (transitions). Focus on resources and species within these areas.

> **Tradeoff:** Additional climate and weather monitoring stations and downscaled climate projections will be needed for areas projected to undergo changes or transitions.

(4) Use vegetation metrics to evaluate relative changes and impacts on different resources and wildlife species if possible.

> **Tradeoff:** Interactions among climate variables, the metrics for evaluating change, and species will need to be evaluated carefully.

(5) Use local climate data and climate projections to help indicate possible wildfire activity, potential for reclamation, grazing impacts, limits to recreational activities, and impacts to habitats.

> **Tradeoffs:** (a) Some activities may be limited or prohibited due to climatic conditions for short (summer) or long (years) terms. (b) The current spatial mismatch between the location and coverage of climate monitoring and the location and scales where we are making management decisions makes it difficult to incorporate climate impacts into assessments of other impacts, and to understand the effectiveness of management actions. (c) A temporal mismatch between the climate information collected and weather data, especially drought and seasonal weather which can influence management decisions, limits our ability to predict how seasonal weather has impacted things like wildfire, drought, and seeding effectiveness.

(Continued)

Table 9.1—(Continued).

Critical Information Needs:

(1) Create a systematic approach to monitoring weather and climate, building on existing monitoring networks that provide compatible data across the environmental gradients in the sagebrush biome. Without an expanded weather network, weather and climate data for mid- and upper elevations will have larger error rates because of spatial mismatches between weather stations and areas where management decisions are being made.

(2) Assess the relationships among various land changes, management outcomes, and climate to determine potential longer-term effects of climate change and to inform monitoring and adaptive management.

MONITORING and WILDLAND FIRE AND VEGETATION MANAGEMENT

Desired Management Outcome:

The effectiveness of wildfire suppression and vegetation management on current uncharacteristic wildfire regimes in sagebrush systems and the capacity to maintain resilience and resistance are positively related to current policies and practices on the ground across scales and over time.

Management Considerations:

(1) Identify and refine monitoring metrics for successful wildfire suppression, vegetation management, and invasive plant control with a focus on effectiveness and outcomes (e.g., acres with invasive plants reduced) rather than outputs (e.g., acres treated) to facilitate adaptive management. Utilize project "failure" information from monitoring results and focus on what we can learn from challenging postfire restoration or reclamation projects.

> **Tradeoff:** Reporting to Congress on short-term actions versus long-term outcomes creates too much focus on implementation rather than the effectiveness of treatments and other management actions.

(2) Change current monitoring (e.g., Fuel Treatment Effectiveness Monitoring) from a binary yes and no response to focus on more meaningful and quantifiable information for adaptive management.

> **Tradeoff:** It may be difficult to obtain the resources needed to monitor adequately.

(3) Use existing monitoring protocols to track long-term dynamics in grass/fire cycles and grass/shrub ecosystems. Base monitoring on timeframes beyond those specified in current protocols that require short-time measurement intervals at small scales (e.g., seasonal versus annual data over multiple years).

> **Tradeoff:** Results will need to be analyzed in a consistent and timely manner so that the results are meaningful at multiple scales, and land management decisions and actions can be adapted quickly.

(4) Monitor the spread of annual invasive grasses and their effects on fire processes.

> **Tradeoff:** If annual invasive grasses are shown to have widespread effects on fire spread, changes in firefighting strategies may be needed.

(5) Monitor the rates of recovery of sagebrush ecosystems in terms of the effects on different wildlife species with varying habitat requirements.

> **Tradeoff:** Failure to consider and plan for a variety of wildlife species and resources in management decisions can have undesired outcomes. For example, postfire recovery efforts may have negative effects on certain wildlife species by changing the composition of plant communities.

(6) Monitor fuel breaks to determine the effectiveness for wildfire suppression activities and the consequences for ecosystems. Monitoring should include quantifying vegetation loss due to fuel break construction and maintenance.

> **Tradeoff:** Monitoring may show that fuel breaks may be installed and maintained that either do not fully meet project objectives to aid wildfire suppression efforts or provide protection for fire suppression personnel. Monitoring may also show that extensive implementation of fuel breaks may increase both fragmentation and the chance of nonnative plant invasions into sagebrush ecosystems as a result of increased disturbance or intentionally seeding potentially invasive introduced species such as forage kochia (see section 6).

(7) Allocate both staff time and funding to conduct effectiveness monitoring to increase the return on investment. Embed costs of monitoring within estimated project costs up front and indicate the monetary tradeoff for monitoring to document effectiveness (outcomes) compared to only implementation (outputs).

> **Tradeoff:** Resources are limited for conservation actions. Although funding for this activity will divert resources from action implementation (outputs), it will provide critical information on success of those actions (outcomes).

(8) Provide the necessary training for conducting monitoring and evaluating the data across scales.

> **Tradeoff:** Without training, the data collected may be less accurate and fail to provide the desired information.

(9) Monitor current exposure to threats. Use that information to evaluate potential future exposure to the threat and to plan conservation and restoration efforts.

(Continued)

Table 9.1—(Continued).

Tradeoff: It will be necessary to determine whether resources will be used to protect those areas most at risk due to threats such as wildfire and plant invasions, or to protect those areas at least risk to maintain current values.

MONITORING and INVASIVE PLANTS

Desired Management Outcome:

Information on resilience and resistance and the current distribution and abundance, vectors, pathways, and impacts of invasive plants is used to inform prioritization of treatment areas, target monitoring efforts, and evaluate treatment effectiveness across scales.

Management Considerations:

(1) Monitor for high priority invasive plants with Early Detection and Rapid Response (EDRR) (USDOI 2016) protocols to prevent additional management burden due to new invasions and to detect spread from existing invasions.

Tradeoff: Without adequate monitoring to locate new invasions, invasive plants may spread and increase in abundance, degrading sagebrush habitat and understory and increasing the risk of catastrophic wildfires. Existing invasions may require long-term efforts and monitoring to achieve and identify success. EDRR monitoring can reduce the management burden and costs through eradication of the invasive plant that can be measured with monitoring within a shorter timeframe.

(2) Link prevention and EDRR strategies to agencies' implementation responses to invasive plant species in the sagebrush biome.

Tradeoff: An agency needs funds and capacity to be able to respond quickly, validate new reports, and have decision rules for level of response.

(3) Use resilience and resistance classes to stratify areas to monitor for invasive plants, focusing on areas of lower resistance and areas of high resource value.

Tradeoff: Monitoring in low resilience and resistance areas can help prevent spread and reduce current risk. Monitoring in high resilience and resistance areas is necessary to prevent new invasions and reduce future risk.

(4) Monitor the effectiveness of treatment strategies for invasive plants across ecological site types to provide more local and regional information on treatments or other management actions that have higher likelihoods of controlling invasive plants and thus will save time and resources.

Tradeoff: Monitoring may take resources from short-term actions (outputs), but having longer-term information on success (outcomes) will improve overall cost-effectiveness of future actions.

(5) Conduct posttreatment effectiveness monitoring following Emergency Stabilization and Rehabilitation (ES&R) to determine invasive plant response and report results to common agency databases.

Tradeoff: ES&R efforts for invasive plant control often have limited monitoring timeframes and can identify short-term reductions in invasive plants. However, additional resources for longer-term monitoring are needed to identify invasive plant treatment needs for effective restoration. Forgoing this monitoring may appease sociopolitical needs or concerns, or partners' concerns if resources are instead used for actions; however, efforts to control or reduce invasive plants in areas that are important for GRSG or other sagebrush dependent species may fail.

(6) Incorporate data or information on invasive plant presence into project planning to better assess the risk of invasive plant spread from existing invasions and in response to disturbance, development, vectors, and pathways.

Tradeoff: Federal land management agencies have mandates for multiple land use, yet authorized uses may increase the spread of invasive plants. Without incorporating information on the existing distributions and abundances of invasive plants into planning efforts, the risk of invasion from disturbance, development, vectors, and pathways may be underrepresented.

(7) Use Citizen Science opportunities to assist with EDRR monitoring for presence of new invasive plants.

Tradeoff: Citizen Science may not collect all of the information needed to confirm or evaluate the presence or abundance of an invasive plant and may be opportunistic and inconsistent. However, it is an opportunity to engage the public and can help identify new invasions.

(8) Identify opportunities to participate in collaborative efforts that are evaluating which tools (e.g., managing for perennial native grasses, selective use of herbicides and targeted grazing) can effectively control annual grasses over large enough areas to reduce risks associated with invasive plant spread and wildfire.

Tradeoff: Unless these efforts are focused and well-conceived, time and resources may be lost for reducing the population while waiting for results.

(Continued)

Table 9.1—(Continued).

Critical Information Needs:

(1) Develop better spatial information related to presence and cover of invasive plants to better target monitoring.

(2) Determine the climatic suitability and risk of future invasion for priority invasive plants. Use this information to determine the relationship between invasive plants and the resilience and resistance categories.

(3) Conduct long-term monitoring across a variety of ecological and geographical areas on native vegetation response to invasive species management tools: cultural (grazing, fire), mechanical (cutting, mowing), pesticides, and biological (pests, pathogens, bacteria, fungi).

MONITORING and SEED STRATEGY

Desired Management Outcome:

Implementation and effectiveness monitoring is used to ensure that projects and seeding strategies increase resilience and resistance by remaining flexible and adaptive and by tracking seed sources, species performance, and the outcomes of different seeding methods.

Management Considerations:

(1) Use monitoring information to determine whether seeding is necessary based on factors such as disturbance history, relative abundances of native perennial plant species, proximity to intact habitat, potential for invasive plant species competition with seeded species, and likely seed sources. If seeding is necessary, select appropriate species based on management objectives and ecological site characteristics, such as precipitation and soil type.

> **Tradeoff:** Although additional investments are necessary, much of the information required for determining the need to seed and selecting the species to seed could be determined by coupling prior monitoring data with resilience and resistance information and local knowledge about past fires/treatment success (Miller et al. 2015). For example, the response of postfire treatments in loamy, 8- to 12-inch [20–30 centimeter] ecological site types with Wyoming big sagebrush and bluebunch wheatgrass can be determined largely based on vegetation composition and cover prior to the wildfire, intensities of past burns, and past and current site-disturbance legacies, such as spring versus fall livestock grazing, or multiple livestock classes using the same allotment.

(2) Use effectiveness monitoring to assess the need for follow-up seeding, the addition of other species, and other management actions due to the effects of disturbances such as improper livestock grazing.

> **Tradeoff:** Monitoring the appropriate information for a sufficient period of time to determine the need for follow-up actions requires additional resources, but can help ensure longer-term treatment success.

(3) Record seed sources, pure live seed (PLS), and seeding methods. Monitor the germination and establishment of the different seed sources in a consistent manner.

> **Tradeoff:** With only anecdotal data, project managers can draw or perpetuate erroneous conclusions about the effectiveness of seeding outcomes. They may not be able to identify the cause of a seeding failure and prevent the failure from being repeated in the future.

(4) Develop monitoring protocols for managers and practitioners that are simple and infer results quickly in order to adaptively manage seeding strategies (e.g., Wirth and Pyke 2009).

> **Tradeoff:** More simplistic monitoring protocols may not capture long-term successes and failures. Implementation of nonstandardized protocols does not allow for comparisons of results among sites or the ability to analyze data at broad scales to identify trends that may affect seeding strategies across large areas.

Critical Information Needs:

(1) Better understand environmental cues that trigger germination in species we predominantly use or want to use in restoration, such as forbs, to determine why species perform poorly or seedings fail.

(2) Further develop climate tools to time seeding treatments to the most appropriate climate window(s). Effective use of these tools would require a new way to get and keep restoration funding to use when those windows are open (Hardegree et al. 2017).

(3) Develop equipment that ensures that native species seed is placed at the right depth in the seedbed.

(Continued)

Table 9.1—(Continued).

MONITORING and LIVESTOCK GRAZING MANAGEMENT

Desired Management Outcome:

Resilience and resistance of lands grazed by livestock are maintained or improved by using monitoring information to evaluate how and to what extent livestock grazing is influencing an area's rangeland health, effects on wildlife habitat, and forage production and to adaptively manage the timing, intensity, and frequency of livestock use.

Management Considerations:

(1) Collect monitoring data and analyze the results to evaluate the effectiveness of grazing strategies. Revise grazing permits and leases where rangeland health standards are not being achieved because of current livestock grazing management.

Tradeoff: Monitoring of grazing effects is at the local level and is the primary monitoring activity for most field offices. Although data collection is generally occurring, failure to analyze the data and revise permits and leases as needed can result in declines in rangeland health and forage production.

(2) Identify expectations should monitoring data show that a grazing management change is warranted. Communicate these expectations to grazing permittees and lessees.

Tradeoff: Monitoring data can indicate improper grazing of public lands, which can strain relationships with grazing permittees and lessees. These may be the same grazing permittees and lessees with whom managers would like to work to implement GRSG habitat improvements.

(3) Assess grazing utilization earlier than at the end of the grazing season to have the opportunity to make management changes (e.g., move livestock) before reaching utilization levels that can cause negative vegetation impacts.

Tradeoff: Without this type of monitoring information and proactive management, rangeland health may decline over time.

(4) Use monitoring to determine how long to defer the onset of grazing after restoration or postfire rehabilitation to allow seeded species to establish and gain the vigor needed to withstand grazing pressures.

Tradeoffs: Native grass species have not been selected to produce large amounts of aboveground biomass, are more susceptible to spring grazing, and are generally more palatable than nonnative species, leading to preferential grazing by livestock. (a) It may be necessary to defer the onset of grazing longer in areas where local native seed is used for restoration. (b) Producers may need other grazing options during the deferment in order to provide the treated or seeded area with the necessary time for recovery. Expected outcomes and estimated yields or treatment effectiveness may help achieve buy-in on deferments.

MONITORING and WILD HORSE AND BURRO CONSIDERATIONS

Desired Management Outcome:

Resilience and resistance are maintained by determining the effects of wild horses and burros (WHBs) on sagebrush ecosystems and whether Appropriate Management Levels (AMLs) for WHBs are appropriately set into the future.

Management Considerations:

(1) Continue aerial surveys using defensible methods to evaluate WHB distribution and abundance.

Tradeoff: Increased conflict regarding WHB management could arise without rigorous measures of WHB distribution and abundance.

(2) Conduct utilization monitoring, keeping livestock grazing numbers and WHB abundance measures as covariates in the analyses. An assessment of range condition before livestock grazing and after grazing has ended in a particular year may help identify which impacts are from livestock and which are from WHBs.

Tradeoff: Determining the effects of livestock versus WHB grazing is challenging, and this approach may not accurately portray WHB effects. However, by not monitoring WHB utilization and managing to AML, certain allotments may not be able to withstand the grazing pressure from both livestock and WHBs.

(3) Implement a monitoring program that includes measures of WHB impacts at or near water sources because WHBs are known to impair soil penetration, water quality, and flow at spring sites, especially when WHBs are at high densities.

Tradeoff: Other areas may need to be less intensively monitored due to budget constraints.

(4) Include measures of WHB herd size (i.e., densities relative to AML) in the analysis of status and trends monitoring datasets that can be aggregated over the landscape based on data from multiple monitoring sites.

Tradeoff: The spatial scale of project sites and vegetation monitoring may be very small compared to the scale of a local wild horse herd.

(Continued)

Table 9.1—(Continued).

(5) Consider including specific levels of WHB population, relative to AML as soft or hard triggers requiring a WHB gather in herd management area plans.

Tradeoff: These adaptive management triggers and responses have a high likelihood of ending up in litigation, which is also a management consideration.

(6) Consider distance to water as an important covariate in monitoring program design (site selection) in areas with high populations of WHBs.

Tradeoff: By not incorporating this information, monitoring could underestimate population densities and ecosystem impacts.

(7) Use adaptive management with WHBs and vegetation monitoring (validation monitoring) to answer the question: "Will habitats recover if WHBs are kept at AML?"

Tradeoff: If monitoring data show that WHBs are causing damage or negative impacts, policy changes may be needed to address management needs and actions. These adaptive management triggers and responses have a high likelihood of ending up in litigation, which is also a management consideration.

CLIMATE ADAPTATION and WILDLAND FIRE AND VEGETATION MANAGEMENT

Desired Management Outcome:

Resilience and resistance are maintained and transitions to desirable new states or site types are facilitated through effective prioritization and implementation of vegetation management treatments and other wildland fire management activities as wildfire regimes continue to change and additional conservation priorities arise.

Management Considerations:

(1) Use regional climate information to better predict high fire years.

Tradeoff: This requires additional investment but can assist with fire preparedness.

(2) Expect that increases in fire potential will lead to increases in fire staff and the need for greater coordination of emergency services at the local level.

Tradeoffs: Project implementation may be postponed until conditions improve, and budget priorities may shift to emergency services. Fire restrictions could impact recreational and other land uses.

(3) Clearly identify objectives when prioritizing habitats or species for protection and determining vegetation management strategies.

Tradeoff: Managing for connectivity will facilitate dispersal and adaptation of species. However, assisted migration of native plant species may introduce species into new environments where they are not adapted or alter ecosystem processes (Bucharova 2017). Resources may be wasted if low priority habitats are selected for protection and management.

(4) Consider the climate vulnerability of species when prioritizing habitats or species for protection.

Tradeoff: Protecting habitats or species in their current location that is not expected to support them in the future may preclude protecting another location that may be viable for them in the future.

Critical Information Needs:

(1) Determine how climate change is likely to alter vegetation across the landscape to guide management decisions.

(2) Evaluate how climate change will influence wildfire frequency and size across the sagebrush biome to allow for repositioning suppressive resources (e.g., local fire personnel and equipment) and potentially for locating fuel breaks or green strips.

(3) Research how climate change will affect landscape scale connectivity, species' vulnerability to climate change, and their projected distributions.

CLIMATE ADAPTATION and INVASIVE PLANTS

Desired Management Outcome:

Resilience and resistance are maintained and transitions to desirable new states or site types are facilitated by identifying new plant invasions; effectively treating, suppressing, containing, and where possible eradicating existing invasions; and identifying die-offs and restoration opportunities.

(Continued)

Table 9.1—(Continued).

Management Considerations:

(1) Increase EDDR efforts to detect new invasive plants and monitor for die-offs with a focus along climatic transition zones.

Tradeoff: This may result in other areas being monitored less intensively.

(2) Use permanent monitoring plots in Areas of Critical Environmental Concern and Research Natural Areas that are generally not grazed by livestock and WHBs, or in ungrazed national wildlife refuges, to detect emerging invasive plant species.

Tradeoff: Emerging invasive plants may be detected, but not necessarily in systems where new invasions are most likely.

(3) Use all permanent plots (e.g., AIM, possibly National Resources Inventory) to track changes in invasive plants over time.

Tradeoff: Taking advantage of existing systems is cost-effective.

(4) Identify refugia for climate change that include redundancy and a range of values for stepping stones (linkages) for native species movements.

Tradeoff: Identification of refugia that maintain representative native ecosystems and prevent extinctions will require substantial investment. Refugia would need to be intensively monitored for invasive plant species.

(5) Use resilience and resistance (soil temperature and moisture regimes) to help evaluate potential nonnative plant invasions.

Tradeoff: This provides a good first filter, especially for invasive annual grasses, but additional information and investment are required to relate soil temperature and moisture regimes to the distributions of many other invasive plants. Changes in climate may modify the distribution of soil temperature and moisture regimes on the landscape (i.e., change the distribution of resilience and resistance on the landscape).

(6) Use information about resilience and resistance to determine the types of actions for addressing plant invasions. In areas with high resilience and resistance, the priority may be to maintain intact, uninvaded ecosystems. In areas with low resilience and resistance, the priority may be to prevent degradation due to soil erosion, protect groundwater, and manage fire risk.

Tradeoff: Caution is needed to prevent areas with low resilience and resistance from being managed solely for livestock forage and wildfire prevention. Intact areas with low resilience and resistance need to be identified and protected.

(7) Determine whether programmatic environmental assessments or environmental impact statements are needed to address invasive plant impacts that affect all programs.

Tradeoff: Budgets for inventory of invasive plants and control treatments are expensive and long-term costs usually fall to one program (e.g., range in the Bureau of Land Management [BLM]).

Critical Information Needs:

(1) Improve capacity to map the extent of all major invasive annual grasses, not just cheatgrass.

(2) Obtain information on the climate suitability of all major invasive plants (including biennial and perennial forbs) that can be used to understand and map the probability of invasion of these species.

(3) Increase understanding of how changes in climate are likely to influence the resilience and resistance of sagebrush and juniper and piñon ecosystems.

Policy Need:

(1) State laws for reclamation and restoration standards are needed to address invasive plant species. If no standards are set (or met), then an increase in spread is likely to be followed by a failure to meet habitat needs. Private companies doing business on public land may push back if stricter reclamation standards are applied. However, if the companies are not responsible or not held accountable, then the land management agency must pay for the long-term invasive control or the problem of invasion will continue to spread.

CLIMATE ADAPTATION and SEED STRATEGY

Desired Management Outcome:

Resilience and resistance are maintained and transitions to desirable new states or site types are facilitated by selecting adapted seed sources, using effective restoration methods, monitoring outcomes, and adapting management. Seeding creates plant communities that are adapted to current climate conditions and can adapt to future conditions. Species should be able to move, adapt, and establish in their future climate zones.

(Continued)

Table 9.1—(Continued).

Management Considerations:

(1) Prioritize where to invest in restoration and seed based on resilience and resistance considerations—what to collect, what to produce, and what to put on the ground.

> **Tradeoff:** It may be necessary to choose between doing nothing, using native species with the best available information and seed sources, and using introduced species (mid- or local scale).

(2) Use seed sources that are adapted to site conditions and that maintain genetic diversity.

> **Tradeoff:** Broad- and mid-scale shifts in vegetation species will directly impact local seed collections and needs. Areas exhibiting climate change may no longer support certain native species, including sagebrush (see Chambers et al. 2017a, section 5.2). Information to facilitate transitions is just now being developed and assisted migration is controversial.

(3) Develop maps that pre-specify seed mixes and treatments before wildfires based on ecological types and ecosystem conditions.

> **Tradeoff:** This requires additional upfront resources, but may substantially increase success.

(4) Use a continuum in restoration—seed sources, implementation, monitoring, and adaptive management—and recognize differences among stabilization, rehabilitation, and restoration. Also consider incorporating concepts and tools from the Society for Ecological Restoration's International Standards for the Practice of Ecological Restoration (McDonald et al. 2016).

> **Tradeoff:** Funding additional education of staff is likely to be well worth the investment.

(5) Use adaptive management and monitoring to identify changes with climate in considering the best places for assisted migration. Accidental assisted migration is already occurring but may not have the desired outcome where the environmental requirements of the cultivated species used in restoration do not match the environmental conditions in which they are planted (Bucharova 2017).

> **Tradeoff:** Without information on species adaptations to the new site or how the new species will affect the communities where they are introduced, the results may not be as desired.

(6) Consider species' current and future distributions and seed zone boundaries to select populations for inclusion in restoration projects that will reduce the risk of future maladaptation and to identify potential bottlenecks to species movement.

> **Tradeoff:** Development of climate shift models is time consuming and will require active planning and coordination to target species populations for collection and growth in order to increase availability in the market (5+ years per seed collection). It is difficult to respond quickly to new information on shifting climates.

Critical Information Needs:

(1) Continue to develop seed zones for more local restoration species—forbs, grasses, and shrubs.

(2) Set aside areas to be used for common garden studies across Management Zones.

(3) Develop and evaluate models of how seed zones may shift as climate changes.

(4) Develop seeding and monitoring strategies that incorporate and test assisted migration.

(5) Identify genotypes for focal restoration species that are widely adapted and will lend themselves to facilitated migration as the climate changes.

(6) Ensure that seed zone development captures seed sources across a species range. Evaluate and develop models on how seed zones may shift as climate changes.

CLIMATE ADAPTATION and LIVESTOCK GRAZING MANAGEMENT

Desired Management Outcome:

Resilience and resistance are maintained and transitions to desirable new states or site types are facilitated by adjusting grazing permits and leases as rangeland ecological condition, forage production, and the level of animal stress change.

Management Considerations:

(1) Revise ecological site descriptions and grazing management to permit adaptation to changing climate conditions.

> **Tradeoff:** This will require information on projected changes in plant species composition and productivity. Changes in long-term habitat objectives, allotment management plans, and grazing permits and leases may be needed.

(2) Change both the locations and timing of livestock use.

(Continued)

Table 9.1—(Continued).

Tradeoff: Analysis of permittee and lessee flexibility will be needed; some will have capacity to move and some will not. Land use plan amendments may be needed.

(3) Create regional networks of grass banks to increase flexibility.

Tradeoff: This may require adjusting other land uses such as WHB AMLs and may have unintended effects on species at risk.

(4) Allow managers to manage for performance (i.e., maintaining or improving resilience and resistance).

Tradeoff: This may increase capacity to manage for resilience and resistance, but would require developing the correct metrics for monitoring.

(5) Develop the capacity to support outcome-based grazing management under a changing climate by adjusting livestock grazing based on current conditions to allow for corrections to occur as climate gradually changes.

Tradeoff: The method for determining animal unit months (AUMs) may need to be modified so that future projections of site productivity and site capacity for livestock grazing take into account the influence of climate change.

(6) Develop drought plans that identify thresholds and list responses. Ideally such plans would be coordinated with drought planning for the permittee's base property.

Tradeoff: Additional management investment and proactive coordination that considers impacts to economies and way of life as well as ecological damage or desertification will be required.

(7) Evaluate changes in wildfire risk due to a warming environment and increases in invasive annual grasses in the context of allotments and the potential mitigation of wildfire effects by grazing, including fuels and the probability of ignition.

Tradeoff: Identifying short-term objectives and the correct metrics will be required. Prioritizing protection of habitats over other resources may be a hard sell at local, mid-, and broad scales.

(8) Evaluate potential changes in native ungulate distributions attributable to changing climate and their interaction with livestock grazing.

Tradeoff: This requires an understanding of potential changes in native ungulate populations and distributions and likely impacts on vegetation communities, soil erosion, and disease transmission.

Critical Information Need:

(1) Identify how and where vegetation composition and productivity and thus AUMs will change in response to climate change.

Policy Need:

(1) Evaluate the policy changes needed to allow grazing management to adapt to climate change.

CLIMATE ADAPTATION and WILD HORSE AND BURRO CONSIDERATIONS

Desired Management Outcome:

Resilience and resistance are maintained and transitions to desirable new states or site types are facilitated by managing WHB populations at AMLs that will sustain ecosystems in the face of reduced water and forage availability and increased competition for these resources by livestock and native ungulates.

Management Considerations:

(1) Reevaluate AML to account for warming and drying conditions. This will require reevaluating site productivity and capacity to support WHBs during drought.

Tradeoff: Failure to adjust AML as climate changes will decrease water and forage for livestock and native ungulates, and place other plant and animal species at greater risk. It may also increase stress on individual WHBs in overpopulated areas. Evaluating and monitoring WHB populations and their use of the landscape will require additional resources that could be spent elsewhere.

(2) Increase understanding of how WHBs use the landscape. This will provide information on how natural water resources may be altered, which in turn can inform management decisions relative to livestock and native ungulate grazing.

Tradeoff: Failure to understand how WHBs use water sources (seeps, springs, riparian systems) will accelerate degradation. Evaluating and monitoring WHB populations and their use of the landscape will require additional resources that could be spent elsewhere.

(3) Adjust public expectations.

Tradeoff: Failure to effectively educate the public will result in increased conflict when and if AMLs are adjusted and gathers are increased.

(Continued)

Table 9.1—(Continued).

WILDLAND FIRE AND VEGETATION MANAGEMENT and INVASIVE PLANTS

Desired Management Outcome:

Allocations for fuel treatments and postfire rehabilitation in agency budgets are prioritized for invasive plant management to decrease the invasive grass/fire cycle that causes large losses of sagebrush habitats. Agency staffs and the public are knowledgeable about the negative effects of the spread of invasive plants on public lands and are supportive of rapid response and eradication efforts.

Management Considerations:

(1) Curtail or change management practices (e.g., some grazing practices) that promote spread of annual invasive grasses and in turn increase fire occurrence and spread.

Tradeoff: Such practices require proactive management by local staff and may not always be agreeable to permittees and lessees.

(2) Change vegetation management priorities and budget allocations to protect postfire recovery efforts and address invasive plants adjacent to postfire recovery areas so that they do not spread into rehabilitated areas.

Tradeoff: Allocation of funds to invasive plant management may decrease funds for other management activities.

(3) Use integrated modeling of resilience and resistance, fire risk, and resource values to determine configuration and placement of fuel treatments in conjunction with district-wide, programmatic National Environmental Planning Act (NEPA) analyses to address invasive annual grass/wildfire concerns.

Tradeoff: Certain assumptions may be required regarding effects of fuel treatments on fire risk. Additional resources will be required to complete the necessary models and NEPA documents.

(4) Design and locate fuel treatments and fuel breaks based on ignition sources and accessibility for firefighters and maintenance activities.

Tradeoff: Fuel breaks may increase wildlife habitat fragmentation and loss, and function as a vector for invasive plants into high quality sagebrush habitats.

(5) Monitor and remove invasive plants in vegetation or fuel treatments and fuel breaks and remove any nonnatives planted in fuel breaks that have spread outside of fuel breaks to ensure that they do not act as a vector for invasion.

Tradeoff: It will be necessary to recognize that although fuel breaks may have a single management objective and result in an ecological type conversion, they should still be managed to prevent plant invasions.

(6) Consider designing prescribed burns that result in a mosaic of burned and unburned patches to maintain seed sources and habitat connectivity rather than designing larger, more extensive burns.

Tradeoff: Additional planning and careful execution is needed to create mosaics that will enhance connectivity.

(7) Use resilience and resistance classes to prioritize areas for postwildfire recovery efforts to increase cost:benefit ratios.

Tradeoff: This approach requires additional staff training to implement and monitoring to evaluate effectiveness.

(8) Continue partnerships, such as the multi-jurisdictional Cooperative Weed Management Area (CWMA) partnership, for invasive plant management.

Tradeoff: Prioritizing for the largest invasion or for protection of more intact uninvaded sagebrush systems, especially at low resilience and resistance, will require partner engagement. Determining which agency programs should cover the cost of treatment is challenging.

(9) Focus eradications and rapid response efforts on areas that act as invasive plant vectors (e.g., along roadsides).

Tradeoff: This requires proactive collaboration with and education of State or county agencies responsible for road maintenance and of grazing lessees who may not treat invasive plants on private lands because of the cost.

(10) Keep annual invasive patches small and focus efforts on proactively treating these before they expand.

Tradeoff: Budgets are limited and treating invasive plants, which includes initial and follow-up treatments and monitoring, is expensive.

(11) Train field specialists, staff, and the public (including permittees) to recognize local weeds and invasive plants and their negative effects on public lands.

Tradeoff: This takes additional resources initially, but can yield large benefits.

(12) Incorporate monitoring of any new “invasions” into existing vegetation monitoring efforts.

Tradeoff: Funding and staffing will be needed, as will time to develop collaborative partnerships across jurisdictional and private property boundaries.

(Continued)

Table 9.1—(Continued).

Critical Information Needs:

(1) Determine how to best address invasive plants in low resilience and resistance areas at a large scale.

(2) Evaluate the use of a variety of plant species, including native species, for fuel breaks.

(3) Develop an understanding of how many plants per square foot or how much cover of perennial grasses is needed following wildfires and prescribed fires to promote recovery and effectively keep annual grasses under control. (This is likely to vary by ecological site type.)

(4) Develop better metrics for measuring perennial grass mortality following both wildfires and prescribed burns and for determining the need to seed.

WILDLAND FIRE AND VEGETATION MANAGEMENT and SEED STRATEGY

Desired Management Outcome:

Resilience and resistance of sagebrush ecosystems are maintained and transitions to desirable new states or site types are facilitated through stabilization, rehabilitation, and restoration treatments following wildfire.

Management Considerations:

(1) Capitalize on natural recovery following wildfires by evaluating the burned areas' environmental conditions and identifying where native plant species will recover on their own and where native plant species should be planted, seeded, or both.

> **Tradeoff:** Additional effort is required to assess postfire areas to determine the ecological site types and their resilience and resistance after wildfire (see Miller et al. 2015). If bunchgrasses are not adequate for natural succession and site recovery, seeding is likely to be necessary.

(2) Use genetically appropriate seed sources identified by seed transfer zones, rather than nonnative species or native cultivars, to avoid introducing species that are invasive or overly competitive with native species.

> **Tradeoffs:** Seeding with nonnatives represents an ecological tradeoff because they have the potential to invade, compete with native species, or spread beyond a project boundary. Seeding with native cultivars represents a genetic tradeoff because of potential adverse impacts to local population genetics through hybridization that may affect overall species fitness. However, seed choices may be limited until more source-identified germplasm is developed by seed zone for native forbs, grasses, and shrubs.

(3) Better match local site conditions with seeded species (right seed, right place, right time) to minimize ecological impacts and increase treatment success (e.g., avoid seeding low sagebrush sites with big sagebrush species).

> **Tradeoff:** More effort and resources are needed to adequately assess sites, determine the appropriate species, and obtain the needed seed sources. Many native species are not readily available and require time for cultivation practices to be developed and for larger-scale seed increase to occur.

(4) Increase sources of sagebrush by developing seed orchards through the private sector for the different ecoregions in the sagebrush biome.

> **Tradeoff:** Seed sources must be carefully chosen and trusted contractors located.

(5) Evaluate several approaches for seeding on harsh sites, such as encapsulating seed.

> **Tradeoff:** Successfully implementing more effective seeding approaches may increase expense and will necessitate monitoring outcomes.

(6) Follow seedings over time using effectiveness monitoring to determine whether and when retreatment is needed or whether the treatment was successful.

> **Tradeoff:** Monitoring resources must be allocated to determine treatment effectiveness.

(7) Carefully evaluate whether and when herbicide application is needed for postfire reclamation of areas with invasive plants.

> **Tradeoff:** Application of pre-emergent herbicides with active ingredients like Imazapic prior to seeding may be appropriate for burned areas with high risk of invasive annual grass or sites where release of native species would be enhanced by reducing annual grass invasion risk. However, depending on application rates, surviving native species and seedbanks may be affected for several years post-application.

(8) Carefully evaluate the use of drill seeding and aerial seeding treatments.

> **Tradeoff:** Aerial application of seed after wildfires has been shown to be largely ineffective, except on moister sites (Knutson et al. 2014). However, drill seeding may not be possible in some areas due to terrain conditions. Seeded species may interfere with native species recovery (section 6) and before deciding whether a site even requires seeding, it is necessary to first determine whether there are sufficient native species for recovery. On sites where seeding would be beneficial, but aerial seeding is unlikely to be successful and drill access is limited, it may be necessary to allow recovery without seeding and manage some risk of an invasive plant species component.

(Continued)

Table 9.1—(Continued).

(9) Test species known to be tolerant of fire and to increase resistance to invasion in fuel breaks.

Tradeoff: Seeding of native species that are not preferred by cattle in fuel breaks could help reduce the spread of cheatgrass in fuel breaks. However, managers and practitioners are not always comfortable using species that they are unfamiliar with or have not used previously.

WILDLAND FIRE AND VEGETATION MANAGEMENT and LIVESTOCK GRAZING MANAGEMENT

Desired Management Outcome:

Grazing management is flexible enough to allow livestock to be moved as needed to maintain the resilience and resistance of sagebrush ecosystems and to provide for grazing deferment following postfire restoration.

Management Considerations:

(1) Train field personnel in how to manage grazing pre-fire to minimize fire risk in fire susceptible areas and post-fire to promote site recovery.

Tradeoff: This type of training needs to balance the needs to reduce fuels, while maintaining or increasing perennial native grasses to promote postfire recovery. If grazing is not carefully managed, it can decrease resistance to invasive annual grasses and increase fire risk.

(2) Consider all available options for managing grazing (e.g., season of use, number of animals, type of livestock), and determine whether those options are sufficient to achieve objectives or whether new options need to be explored.

Tradeoff: The grazing permit states the number of livestock (AUMs and season of use) and it is legally binding for grazing on public lands. Permits may need to be adjusted to maintain resilience and resistance and provide for grazing deferment following postfire restoration.

(3) Minimize grazing use, or adjust the timing or levels of grazing use that are currently promoting spread of annual invasive grasses, which in turn increase fire occurrence and spread.

Tradeoff: Permittees or lessees may not have sufficient flexibility or be receptive to these types of changes even though failure to change may increase fire risk.

(4) Manage for threatened and endangered (T&E) species' habitats, riparian areas, and restoration and postfire rehabilitation areas that may need a reduction in livestock grazing impacts.

Tradeoff: Managers may be pressured to allow livestock grazing to take precedence over other resources.

(5) Work with permittees or lessees in an adaptive management setting to defer the onset of grazing to allow for successful postfire restoration projects.

Tradeoff: Grazing is addressed at the local level with each ranch being its own unit. Postfire grazing deferments may depend on the size of the fire, the resources at risk, and impacts to the grazing permittee or lessee. Permittee or lessee willingness to move livestock in relation to seeding and grazing tolerance may vary by geographic area.

(6) Strategically place targeted grazing in areas where it will be the most effective for fuel reduction and managing fuel breaks.

Tradeoffs: Targeted grazing practices may not always work for permittees or lessees because of the time and management practices required to implement it effectively (e.g., it is expensive for permittees or lessees, or permittees or lessees may not want to participate). If not properly executed, targeted grazing may increase invasion by nonnative annual grasses and fire risk.

Critical Information Needs:

(1) Determine the effectiveness of grazing to maintain fuel breaks along roadsides or other linear features at operational scales.

(2) Evaluate the effects of targeted grazing to control invasive annual grasses on establishing and maintaining native grasses.

WILDLAND FIRE AND VEGETATION MANAGEMENT and WILD HORSE AND BURRO CONSIDERATIONS

Desired Management Outcome:

Wild horses and burros are maintained at AML, which are intended to be population levels that provide for resilience and resistance of rangeland ecosystems and are consistent with other land uses and resources. WHBs are limited to designated management areas: Herd Management Areas (HMAs) and Herd Areas (HAs) on BLM lands; and Wild Horse Territories (WHTs), Wild Burro Territories (WBTs), and Wild Horse and Burro Territories (WHBTs) on Forest Service lands.

(Continued)

Table 9.1—(Continued).

Management Considerations:

(1) Monitor vegetation and fuel loads to determine the effects of WHBs on wildfire and the fire/invasive annual grass cycle and ecosystem resilience and resistance.

Tradeoff: WHBs may decrease fuel loads and the potential for wildfire, but may also reduce perennial grasses and forbs, decrease forage for livestock, and compete with wildlife.

(2) When WHB management areas experience large fires and large-scale WHB removals are not possible, plan for lands to be grazed or browsed by WHBs.

Tradeoff: During wildland or prescribed fires, burned fences can lead to WHB movement outside of established pastures. If WHBs are above AML, they may decrease postfire recovery and increase the risk of nonnative invasive plant spread.

(3) Explore and fund options for effective exclusion of WHBs in areas of postfire vegetation recovery.

Tradeoffs: Given that horses can routinely move 10 miles (16 kilometers) between water and available forage (Hampson et al. 2010), any seeding area, as well as newly revegetated areas after burns, can be attractive forage to WHBs if the areas have palatable forage. WHB presence in postfire recovery areas is likely to decrease seeding success, especially if WHBs are above AML.

(4) For prescribed fires, consult with the local WHB specialist or other appropriate agency staff about which gates should be left open to allow WHBs to escape burn areas.

Tradeoff: WHBs have the potential to impact adjacent areas.

(5) Temporarily remove most WHBs from a landscape (with an emergency gather, holding in BLM facility) to facilitate postfire rehabilitation.

Tradeoff: The efficacy of such options should be weighed against expense and effects on livestock grazing movements. Emergency gathers require agency approval, and may require NEPA analysis.

Critical Information Needs:

(1) Determine the conditions under which WHBs spread invasive annual grasses and affect invasive plant species distributions, which in turn influence fire processes.

(2) Determine the effects of WHBs on fuels and wildfire probabilities and evaluate the tradeoffs between reducing fuels and ecological resilience and resistance.

INVASIVE PLANTS and SEED STRATEGY

Desired Management Outcome:

Management practices are modified to maintain or increase resilience and resistance by protecting native seed sources, providing sufficient native seed for restoration or rehabilitation projects, and establishing mixes of species that can compete effectively with invasive plant species.

Management Considerations:

(1) Ensure that permitting for native seed collection is not resulting in overcollection of native populations by not allowing seed collection in the same areas every year.

Tradeoff: Native seed collections may require additional oversight to ensure permit compliance and cost more.

(2) Diversify seed mixes to include a variety of life forms (shrubs, grasses, and forbs) that increase ecosystem function and provide the range of plant phenologies and rooting depths necessary for long-term resilience to disturbance and resistance to invasive annual grasses.

Tradeoff: Until the availability of genetically appropriate native plant material increases, it may be difficult to develop more diverse seed mixes.

(3) Use restoration and rehabilitation practices that will help ensure establishment and persistence of diverse mixtures of seeded species.

Tradeoff: Diverse seed mixes may require adjusting seeding methods, such as seeding depth, based on seed size and germination requirements of the individual species.

(4) Evaluate site conditions on low resilience and resistance areas to determine whether ecological thresholds have been crossed that may influence the choice of seeded species.

Tradeoffs: Use of nonnative species and native cultivars on highly disturbed or invaded sites that have crossed ecological thresholds may meet objectives for site stabilization or fuel breaks. However, it is necessary to acknowledge that these types of seedings are not designed to meet wildlife habitat objectives.

(Continued)

Table 9.1—(Continued).

(5) Use postfire vegetation monitoring and reporting to evaluate the competitive ability of both native plant species and mixtures, including forbs, with invasive annual grasses.

> **Tradeoff:** Seed mixes need to match site conditions well in order to effectively evaluate their competitive ability.

Policy Need:

(1) Change current seed laws to increase consistency in not allowing cheatgrass seed in commercial seed sources, because it is difficult and expensive to remove from purchased seed and seeded sites. This requires evaluation. If seed law required cheatgrass-free seed, then there could be economic impacts and less native seed availability if it is cost-prohibitive or operationally impossible to provide cheatgrass-free seed.

INVASIVE PLANTS and LIVESTOCK GRAZING MANAGEMENT

Desired Management Outcome:

Grazing management maintains or increases resilience and resistance by decreasing or minimizing dispersal and growth of invasive plant populations and does not increase invasive plants when used as a tool for reducing fuels.

Management Considerations:

(1) Evaluate the different vectors (dispersers) of nonnative invasive plants, including livestock grazing, WHBs, and wildlife, to determine the relative effects of the different vectors.

> **Tradeoff:** Vehicle and livestock movement among parcels can transport and assist dispersal of invasive plant seed, increasing invasive plant species spread and necessitating early detection and treatment based on vector management. If movement among parcels is prevented, then additional areas may be needed for grazing. If invasive plant species spread is not addressed through vector management and hence restriction of the invasion to the original location, a much larger invasive plant species management problem may develop.

(2) Consider both the state of invasion and resilience and resistance when developing or modifying grazing management practices in areas with invasive annual grasses.

> **Tradeoff:** There are general management strategies for cheatgrass and other nonnative invasive annual grasses based on resilience and resistance and the invasion state (tables 5.1, 5.2) that can be used to help evaluate whether grazing management is appropriate for the site conditions and degree of invasion. Monitoring to ensure that grazing management decreases the degree of invasion or at a minimum does not increase it can be used to develop more effective grazing strategies, but may require additional investment.

(3) Consider the state and condition of the areas being evaluated for targeted grazing, including relative resilience and resistance, the degree of invasion by nonnative annual grasses, and proximity to invaded areas.

> **Tradeoff:** Targeted grazing may help reduce the biomass of nonnative invasive annual grasses and thus fuels once these grasses are dominant, but in uninvaded or low invasion areas improper grazing may increase invasive plant species. Appropriate use will depend on the degree of invasion.

(4) Conduct coordinated research and management trials to evaluate the effectiveness of targeted grazing for setting up fuel breaks or fuel reduction. This effort should be limited. Managers should evaluate the amount of time and infrastructure required and strategize as to where to try targeted grazing.

> **Tradeoff:** Targeted grazing to establish effective fuel breaks requires intense livestock management during a short time period. It may be difficult or expensive for permittees or lessees to implement and require close monitoring of contractors. Annual maintenance would be required; species other than cattle, such as sheep and goats, may have less impact, but carry disease risk if bighorn sheep (Ovis canadensis) are in the area. Targeted grazing may increase invasive annual plants, facilitate new invasions attributable to livestock movement, or reduce vigor of extant native plants.

(5) Evaluate the need to move livestock grazing operations outside of the allotment or into different pastures within an allotment after a treatment or disturbance until the desired outcomes are obtained.

> **Tradeoff:** The producer has to keep livestock off the allotment or off certain pastures within an allotment for a set number of years depending on resilience and resistance and current level of invasion by nonnative annual grasses. But policy or landowner agreements limit the flexibility to change implementation guidelines. Returning livestock to the allotment earlier than guidelines suggest may decrease overall sustainability of ecological conditions and forage sources.

(6) Require the use of weed-free hay for supplemental feeding of livestock following wildfire.

> **Tradeoff:** Requiring weed-free hay is expensive in the short term, but can reduce long-term costs of managing invasive plants.

(Continued)

Table 9.1—(Continued).

(7) Consider creating grass banks where livestock can be moved during the period required for areas to recover after restoration or rehabilitation activities.

Tradeoff: Nonnative plant species could be seeded to provide for grazing in certain areas, such as those with low resilience and resistance, rather than seeding with native plant species, but this may have negative ecological effects in the long term.

(8) Use a holistic approach when evaluating effects of livestock grazing on invasive plants that considers: (a) the management objectives; (b) current ecological state, resilience and resistance, and geographic area; (c) wildlife resources; (d) distance to water to prevent concentration of impacts from grazers; (e) different management needs for managing different kinds of livestock (cattle, sheep, goat, horse); and (f) control of livestock for utilization and ability for timing and frequency of movement of the herd.

Tradeoff: Clear information on appropriate grazing management (timing of grazing, number of livestock) based on the ecological site type and kind of livestock is needed for this type of approach but is often lacking.

INVASIVE PLANTS and WILD HORSE AND BURRO CONSIDERATIONS

Desired Management Outcome:

Wild horses and burros are maintained at AML, which is intended to be population levels that allow for the resilience and resistance of rangeland ecosystems and are consistent with other land uses and resources. WHBs are limited to designated management areas: HMAs on BLM lands; and WHTs, WBTs, and WHBTs on Forest Service lands.

Management Considerations:

(1) Evaluate the degree to which WHBs versus livestock are acting as vectors (dispersers) of invasive plants.

Tradeoff: If movement of WHBs among parcels is prevented to manage weed invasions, then additional areas for grazing, or gathers, may be needed. If invasive plant species spread is not addressed through this type of vector management and thus restriction of the invasion to the original location, a much larger invasive plant species management problem may develop.

(2) Consider both the state of invasion of invasive annual grasses and resilience and resistance of the area when evaluating the effects of WHBs and the need for gathers.

Tradeoff: There are general management strategies for cheatgrass and other invasive plants based on resilience and resistance and the invasion state (tables 5.1, 5.2) that can be used to help evaluate site conditions and the degree of invasion within management areas. Monitoring to ensure that WHBs grazing does not increase the state of invasion by nonnative annual grasses can be used to evaluate the need for gathers, but may require additional investment.

(3) Identify areas without WHBs present that may be higher priority for conservation and restoration. Consult with local WHB specialists or agency staff to identify areas beyond HMA or WHT, WBT, or WHBT boundaries that WHBs occupy.

Tradeoff: Areas with valuable resources that have WHBs above AML may fail to receive restoration or conservation actions.

(4) Consider how water sources influence WHB movement patterns when developing invasive plant management plans. (WHBs will congregate around water sources and move up to 10 miles each way from forage to water [Hampson et al. 2010], increasing the likelihood of spreading invasive plants.)

Tradeoff: This requires an extra step in developing invasive plant management plans but may have large benefits.

SEED STRATEGY and GRAZING MANAGEMENT

Desired Management Outcome:

Livestock grazing is managed to maintain or increase the resilience and resistance of restored or rehabilitated native plant communities.

Management Considerations:

(1) Consider creating grass banks where livestock can be moved during the period required for areas to recover after restoration or rehabilitation activities.

Tradeoff: Areas already seeded with nonnative plant species could be used as grass banks. Nonnative plant species could also be seeded to provide for grazing in certain areas, such as those with low resilience and resistance, rather than seeding with native plant species, but this may have negative ecological effects in the long term.

(Continued)

Table 9.1—(Continued).

(2) Consider using ecological site descriptions and state-and-transition models within the project area to evaluate the relative resilience and resistance of the area to be seeded.

Tradeoff: Ecological types and ecological sites with relatively low resilience and resistance often require more than one intervention for restoration efforts to succeed. Livestock use can have negative effects on project success.

(3) Evaluate the distance to the nearest drinking water source for livestock during project planning.

Tradeoff: The shorter the distance, the greater the grazing pressure that can be expected, potentially decreasing the likelihood of success.

(4) Consider installing fencing to prevent use by livestock on certain habitat restoration projects, particularly those associated with riparian areas.

Tradeoff: Temporary fencing for habitat rehabilitation is generally acceptable, but permanent fencing often requires a more in-depth environmental assessment or land use plan revision, and should be designed in a way that allows livestock to reach drinking water and move throughout the rest of the allotment.

(5) Consider forgoing a habitat restoration project entirely instead of spending time and resources on projects where spring, summer, and fall season of use occurs or where permittees do not have the flexibility or desire to change grazing system or season of use.

Tradeoff: Areas in need of active restoration may not be treated unless grazing permits are revised.

SEED STRATEGY and WILD HORSE AND BURRO CONSIDERATIONS

Desired Management Outcome:

Wild horse and burro populations are managed at AML to protect sagebrush ecosystems from overgrazing and maintain resilience and resistance in areas where native seedings have been conducted.

Management Considerations:

(1) Consider using ecological site descriptions and state-and-transition models within the HMA to evaluate the relative resilience to disturbance and resistance to invasive annual grasses of the area to be seeded.

Tradeoff: Ecological types or ecological sites with relatively low resilience and resistance often require more than one intervention for restoration efforts to succeed. WHBs use can have negative effects on project success.

(2) Assess the current spatial extent and population size of any nearby WHB population during project planning.

Tradeoff: Effects of WHBs on seedings depend on the number of WHBs that can enter the site, and high numbers can limit project success.

(3) Evaluate the distance to the nearest drinking water source for wild horses during project planning.

Tradeoff: The shorter the distance, the greater the grazing pressure that can be expected, potentially decreasing the likelihood of success. Horses can travel long distances (10 or more miles per day) from water to forage in arid to semi-arid environments (Hampson et al. 2010).

(4) Consider installing fencing to discourage use by WHBs on certain habitat restoration projects, particularly those associated with riparian areas.

Tradeoff: Temporary fencing for habitat rehabilitation is generally acceptable, but permanent fencing often requires a more in-depth environmental assessment or land use plan revision. Permanent fencing should be designed in a way that lets WHBs reach drinking water, and allows their movement throughout the rest of the HMA.

(5) Consider forgoing a habitat restoration project entirely instead of spending time and resources on projects in areas with wild horse populations above AMLs.

Tradeoff: Areas in need of active restoration may not be treated until WHB populations have been reduced.

GRAZING MANAGEMENT and WILD HORSE AND BURRO CONSIDERATIONS

Desired Management Outcome:

Wild horses and burros are maintained at AML, which is intended to be population levels that provide for resilience and resistance and allow for other land uses and resources (including livestock grazing). WHBs are limited to designated management areas: HMAs on BLM lands; and WHTs, WBTs, and WHBTs on Forest Service lands.

Management Considerations:

(1) Maintain WHBs at AML because overpopulated WHB numbers along with management actions for grazing may have effects that are counter to rangeland health objectives.

(Continued)

Table 9.1—(Continued).

Tradeoff: Local economies may be impacted in the form of reduced grazing opportunities, reduced wildlife populations, and other multiple uses if WHBs are not maintained at AML.

(2) Reduce WHBs to AML in order to stabilize rangeland conditions.

Tradeoffs: (a) Periodic deferment or movement of domestic livestock and adjustments of livestock allotments do not reduce WHB grazing impacts, especially where WHB densities are well above AML. (b) Livestock producers may not have the flexibility to move and adjust in response to WHBs and their impacts.

(3) Once WHB populations are maintained at AML, monitor to determine whether land health standards are being achieved.

Tradeoff: If land health standards are not being achieved, and WHBs are causing the non-achievement, it is necessary to consider a reduction in AML in land use plans.

(4) Where WHB numbers are above AML, consider using other available livestock management tools to minimize overlap with areas that have high WHB densities.

Tradeoff: Use of such tools presents challenges and costs to livestock producers. Livestock numbers may need to be decreased, but livestock grazing must be considered as a legally protected multiple use of the rangeland resource.

References

Aldridge, C.L.; Nielsen, S.E.; Beyer, H.L.; [et al.]. 2008. Range-wide patterns of greater sage-grouse persistence. Diversity and Distributions. 14: 983–994.

Balch, J.K.; Bradley, B.A.; Abatzoglou, J.T.; [et al.]. 2017. Human-started wildfires expand the fire niche across the United States. Proceedings of the National Academy of Science, USA. 114: 2946–2951.

Boyte, S.P.; Wylie, B.K. 2017. Near-real-time herbaceous annual cover in the sagebrush ecosystem. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. https://www.sciencebase.gov/catalog/item/595e6cc3e4b0d1f9f0570318. [Accessed May 23, 2018].

Board, D.I.; Chambers, J.C.; Miller, R.F.; [et al.]. 2018. Fire patterns in piñon and juniper land cover types in the Semiarid Western United States from 1984 through 2013. RMRS-GTR-372. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 57 p.

Bucharova, A. 2017. Assisted migration within species range ignores biotic interactions and lacks evidence. Restoration Ecology. 25: 14–18.

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017a. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p.

Chambers, J.C.; Bradley, B.A.; Brown, C.A.; [et al.]. 2014b. Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. invasion in the cold desert shrublands of western North America. Ecosystems. 17: 360–375.

Chambers, J.C.; Maestas, J.D.; Pyke, D.A.; [et al.]. 2017c. Using resilience and resistance concepts to manage persistent threats to sagebrush ecosystems and Greater sage-grouse. Rangeland Ecology and Management. 70: 149–164.

Chambers, J.C.; Miller, R.F.; Board, D.I.; [et al.]. 2014d. Resilience and resistance of sagebrush ecosystems: Implications for state and transition models and management treatments. Rangeland Ecology and Management. 67: 440–454.

Conroy, M.J.; Runge, M.C.; Nichols, J.D.; [et al.]. 2011. Conservation in the face of climate change: The role of alternative models, monitoring, and adaptation in confronting and reducing uncertainty. Biological Conservation. 144: 1204–1213. https://doi.org/10.1016/j.biocon.2010.10.019

Davies, K.W. 2008. Medusahead dispersal and establishment in sagebrush steppe plant communities. Rangeland Ecology and Management. 61: 110–115.

Doherty, K.E.; Evans, J.S.; Coates, P.S.; [et al.]. 2016. Importance of regional variation in conservation planning: A range-wide example of the Greater sage-grouse. Ecosphere. 7(10): Article e01462.

Goldstein, M.I.; Suring, L.H.; Vojta, C.D.; [et al.]. 2013. Developing a habitat monitoring program: Three examples from National Forest planning. Chapter 10. In: Rowland, M.M.; Vojta, C.D., tech. eds. A technical guide for monitoring wildlife habitat. Gen. Tech. Rep. WO-GTR-89. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 74 p.

Hampson, B.A.; de Laat, M.A.; Mills, P.C.; [et al.]. 2010. Distances travelled by feral horses in 'outback.' Australia. Equine Veterinary Journal. 42: 582–586.

Hardegree, S.P.; Abatzoglou, J.T.; Brunson, M.W.; [et al.] 2017. Weather-centric rangeland revegetation planning. Rangeland Ecology and Management. 71: 1–11.

Knick, S.T.; Hanser, S.E.; Miller, R.F.; [et al.]. 2011. Ecological influence and pathways of land use in sagebrush. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 203–251.

Knick, S.T.; Hanser, S.E.; Preston, K.L. 2013. Modeling ecological minimum requirements for distribution of greater sage-grouse leks: Implications for population connectivity across their western range, U.S.A. Ecology and Evolution. 3: 1539–1551.

Knutson, K.C.; Pyke, D.A.; Wirth, T.A.; [et al.]. 2014. Long-term effects of reseeding after wildfire on vegetation composition in the Great Basin shrub steppe. Journal of Applied Ecology. 51(5): 1414–1424.

Maestas, J.D.; Campbell, S.B.; Chambers, J.C.; [et al.]. 2016. Tapping soil survey information for rapid assessment of sagebrush ecosystem resilience and resistance. Rangelands. 38: 120–128.

McDonald, T.; Gann, G.D.; Jonson, J.; Dixon, K.W. 2016. International standards for the practice of ecological restoration—Including principles and key concepts. Washington, DC: Society for Ecological Restoration. http://www.ser.org/?page=SERStandards. [Accessed May 23, 2018].

Miller, R.F.; Chambers, J.C.; Pellant, M. 2014. A field guide to selecting the most appropriate treatments in sagebrush and pinyon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322-rev. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2015. A field guide for rapid assessment of post-wildfire recovery potential in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep.

RMRS-GTR-338. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 68 p.

Pyke, D.A.; Shaff, S.E.; Lindgren, A.; [et al.]. 2014. Region-wide ecological responses of arid Wyoming big sagebrush communities to fuel treatments. Rangeland Ecology and Management. 67: 455–467.

Sage Grouse Initiative [SGI]. 2015. Reducing cultivation of grazing lands conserves sage grouse. Science to Solutions Series Number 8. Washington, DC: U.S. Department of Agriculture, Natural Resources Conservation Service, Sage Grouse Initiative. 4 p. http://www.sagegrouseinitiative.com/

Shinneman, D.J.; Aldridge, C.L.; Coates, P.S.; [et al.]. 2018. A conservation paradox in the Great Basin—Altering sagebrush landscapes with fuel breaks to reduce habitat loss from wildfire. Open-File Report 2018–1034. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. 70 p. https://doi.org/10.3133/ofr20181034. [Accessed May 22, 2018].

Smith, J.T.; Evans, J.S.; Baruch-Mordo, S.; [et al.]. 2016. Reducing cropland conversion risk to sage-grouse through strategic conservation of working rangelands. Biological Conservation. 201: 10–19.

Stiver, S.J.; Apa, A.D.; Bohne, J.R.; [et al.]. 2006. Greater sage-grouse comprehensive conservation strategy. Cheyenne, WY: Western Association of Fish and Wildlife Agencies. 442 p. https://wdfw.wa.gov/publications/01317/. [Accessed Sept. 12, 2018].

U.S. Department of the Interior [USDOI]. 2009. Adaptive management: The U.S. Department of the Interior Technical Guide. Adaptive Management Working Group. Washington, DC: U.S. Department of the Interior. 84 p.

U.S. Department of the Interior [USDOI]. 2016. Safeguarding America's lands and waters from invasive species: a national framework for early detection and rapid response. Washington, DC: U.S. Department of the Interior. 55 p. https://www.doi.gov/sites/doi.gov/files/National%20EDRR%20Framework.pdf. [Accessed July 17, 2016].

U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2013. Greater sage-grouse (*Centrocercus urophasianus*) Conservation Objectives: Final Report. Denver, CO: U.S. Department of the Interior, Fish and Wildlife Service. 91 p. https://www.fws.gov/greatersagegrouse/documents/COT-Report-with-Dear-Interested-Reader-Letter.pdf. [Accessed Aug. 8, 2016].

Walters, C. 1986. Adaptive management of renewable resources. New York, NY: MacMillan. 374 p.

Williams, B.K.; Nichols, J.D.; Conroy, M.J. 2002. Analysis and management of animal populations. San Diego, CA: Elsevier-Academic.

Wirth, T.A.; Pyke, D.A. 2009. Final report for emergency stabilization and rehabilitation treatment monitoring of the Keeney Pass, Cow Hollow, Double Mountain, and Farewell Bend Fires. USGS Open-File Report 2009-1152. Corvallis, OR: U.S. Department of the Interior, U.S. Geological Survey.

Xian, G.; Homer, C.; Meyer, D.; Granneman, B. 2013. An approach for characterizing the distribution of shrubland ecosystem components as continuous fields as part of NLCD. ISPRS Journal of Photogrammetry and Remote Sensing. 86: 136–149.

Appendix 1—Definitions of Terms Used in This Document

Adaptive management—A structured, iterative process of robust decisionmaking in the face of uncertainty, with the aim of reducing uncertainty over time via system monitoring.

At-risk community phase—A community phase that can be designated within the reference state and also in alternative states. This community phase is the most vulnerable to transition to an alternative state (Caudle et al. 2013).

Biological control—The use of natural enemies—predators, parasites, pathogens, and competitors—to control invasive plants over multiple years. Invasive plants have many natural enemies including insects and plant pathogens.

Biopesticide—A pesticide derived from such natural materials as animals, plants, bacteria, and certain minerals. Fungal pathogens and bacterial agents are potential biopesticides for cheatgrass.

Change agents—Disturbances and management actions that influence resource conditions (or status) and trends and subsequent outcomes of conservation and restoration actions.

Community phase—A unique assemblage of plants and associated soil properties that can occur within a state (Caudle et al. 2013).

Cool season/warm season grasses—Cool season or C3 grasses grow during cooler times of the year, typically when temperatures are 40 to 75 °F [4–24 °C], and include wheatgrasses, needle grasses, brome grasses, and blue grasses. Warm season or C4 grasses grow during warmer periods when temperatures are 70 to 95 °F [21–35 °C] and include blue grama, buffalograss, and bluestems. Warm season grasses use soil moisture more efficiently than cool season species and often can withstand drought conditions. For a detailed explanation, see OSU 2017.

Deferred livestock grazing—The dropping of an allotment from the normal scheduled use or rotation for use at a later time.

Early Detection and Rapid Response (EDRR)—A management approach to minimize the establishment and spread of new invasive plant species through a coordinated framework of public and private partners and a process that includes detection and reporting, identification and vouchering, rapid assessment, planning, and rapid response. An overview of the National Framework for Early Detection and Rapid Response (USDOI 2016) is available on the National Invasive Species Council website (https://www.doi.gov/invasivespecies/edrr).

Ecological niche—A species' ecological niche is a function of the environmental conditions under which the species can establish and persist. It depends on the species' physiological and life history requirements for establishment, growth, and reproduction, and its interactions with the native perennial plant community including interspecific competition and response to herbivory and pathogens.

Ecological site (ES)—A conceptual division of the landscape that is defined as a distinctive kind of land based on recurring soil, landform, geology, and climate characteristics that differs from other kinds of land in its ability to produce distinctive kinds and amounts of vegetation and in its ability to respond similarly to management actions and natural disturbances (Caudle et al. 2013).

Ecological site description (ESD)—Documentation of the characteristics of an ecological site. The documentation includes the data used to define the distinctive properties and characteristics of the ecological site, the abiotic and biotic characteristics that differentiate the site (i.e., climate, physiographic characteristics, soil characteristics, plant communities), and the ecological dynamics of the site that describes how changes in disturbance processes and management can affect the site. An ESD also provides interpretations about the land uses and ecosystem services that a particular ecological site can support and management alternatives for achieving land management goals (Caudle et al. 2013).

Ecological type—A category of land with a distinctive (i.e., mappable) combination of landscape elements. The elements making up an ecological type are climate, geology, geomorphology, soils, and potential natural vegetation. Ecological types differ from each other in their ability to produce vegetation and respond to management and natural disturbances (Winthers et al. 2005). In the Science Framework, ecological type is used in a broad sense and refers to ecological type or ecological site groups as described in Chambers et al. 2017: Appendix 3.

Ecosystem services—The direct and indirect contributions of ecosystems to human well-being.

Fire regime—The patterns of fire seasonality, frequency, size, spatial continuity, intensity, type (crown fire, surface fire, or ground fire), and severity in a particular area or ecosystem (Agee 1994; Heinselman 1973; Sugihara et al. 2006). A fire regime is a generalization based on the characteristics of fires that have occurred over a long period. Fire regimes are often described as cycles or rotations because some parts of the fire histories usually get repeated, and the repetitions can be counted and measured.

Focal species—Sagebrush obligate, near-obligate, dependent, or associated species identified as having one or more of the following characteristics: (1) at-risk, (2) influencing management actions and regional economies, (3) potentially being negatively influenced by management actions, or (4) serving as indicators of habitat quality or habitat niches such as riparian areas in sagebrush ecosystems.

Fuel break—A natural or manmade change in fuel characteristics which affects fire behavior so that fires burning into them can be more readily controlled (NWCG 2018).

Greater sage-grouse habitat designations

- Priority Areas of Conservation—Key habitat areas identified and delineated in the sage-grouse conservation plans for each State or through other sage-grouse conservation efforts (USDOI FWS 2013).
- Priority Habitat Management Areas—A Federal habitat designation that includes areas identified as having the highest habitat value for maintaining sustainable GRSG populations including breeding, late brood-rearing, and winter concentration areas.
- General Habitat Management Areas—A Federal habitat designation that identifies areas that are occupied seasonally or year-round and are outside of Priority Habitat Management Areas.
- Important Habitat Management Areas (Idaho only)—Areas in Idaho that provide a management buffer for and that connect patches of Priority Habitat Management Areas. Important Habitat Management Areas encompass areas that are generally moderate to high conservation value habitat or populations but that are not as important as Priority Habitat Management Areas.

- Other Habitat Management Areas (Nevada and northeastern California only)—Areas in Nevada and northeastern California identified as unmapped habitat in the Proposed Resource Management Plan or Final Environmental Impact Statement that are within the Planning Area and contain seasonal or connectivity habitat areas.

Green stripping—The practice of establishing or using patterns of fire tolerant vegetation or other material to reduce wildfire occurrence and size (St. John and Ogle 2009; USDOI BLM 1987). A green strip can be a fuel break as defined by the National Wildfire Coordinating Group (NWCG 2018).

Habitat connectivity—The degree to which the landscape facilitates animal movement and other ecological flows.

Improper livestock grazing—Grazing that impedes progress toward or maintenance of ecological processes and the desired plant community composition and structure within a given set of site conditions and the natural range of variability, including climatic variability and natural disturbance regimes, expected within a management planning time horizon.

Invasive plant species—An invasive species is (1) nonnative (or alien) to the ecosystem under consideration, and (2) its introduction causes or is likely to cause economic or environmental harm or harm to human health (Presidential Executive Order 13112, 1999).

Local adaptation—A population is locally adapted if organisms in that population have differentially evolved as compared to other populations within their species in response to selective pressures imposed by some aspect of their local environment. Locally adapted restoration species or seed collections are likely to perform better than species or collections from outside the local environment.

Major Land Resource Area—A geographic area, usually several thousand acres in extent, that is characterized by a particular pattern of soils, climate, water resources, land uses, and type of agriculture.

Management strategies—Coordinated management activities conducted at mid- to local scales to achieve vegetation and habitat objectives (e.g., strategically locating firefighting resources to protect habitat, coordinating Early Detection and Rapid Response activities for invasive plant species, positioning treatments to increase connectivity).

Metapopulation—A group of populations that are separated by space but consist of the same species. These spatially separated populations interact as individual members move from one population to another.

Monitoring attributes—Ecosystem attributes, such as soil stability and health, hydrologic function, water flow and quality, and biotic integrity, monitored to determine ecosystem status at local, mid-, and broad scales.

Monitoring benchmarks—Indicator values, or ranges of values, that establish desired conditions and are meaningful for management.

Monitoring indicators—Indicators of ecosystem attributes that can be measured and can account for changes in the resource within a realistic timeframe and budget given the site potential and spatial scale of the area being managed. For example, bare ground, vegetation composition, and soil aggregate stability are indicators of hydrologic function.

Monitoring triggers—Levels of environmental conditions that can provide an early warning of possible thresholds and of management changes that may be necessary to maintain the desired environmental conditions (Briske et al. 2008; Goldstein et al. 2013).

Persistent ecosystem threats—Threats that include invasion of nonnative invasive plant species, altered fire regimes, and conifer expansion; are difficult to regulate; and are managed using ecologically based approaches (Evans et al. 2013; Boyd et al. 2014).

Prescribed fire—Any fire intentionally ignited by management actions in accordance with applicable laws, policies, and regulations to meet specific objectives (NWCG 2018). A prescribed fire is also sometimes called a "controlled burn" or "prescribed burn." Prescribed fires consider the safety of the public and fire staff, weather, and probability of meeting the burn objectives (see also Wildfire, Wildland Fire).

Projects—Projects consist of multiple land treatments (see also Treatments).

Reference state—Ecological potential and natural or historical range of variability of the ecological site.

Resilience—Capacity of an ecosystem to reorganize and regain its fundamental structure, processes, and functioning when altered by stressors such as invasive plant species and disturbances such as improper livestock grazing and altered fire regimes (based on Angeler and Allen 2016; Holling 1973).

Resistance—Capacity of an ecosystem to retain its fundamental structure, processes, and functioning (or remain largely unchanged) despite stresses, disturbances, or invasive species (Angeler and Allen 2016; Folke et al. 2004).

Resistance to invasion—Abiotic and biotic attributes and ecological processes of an ecosystem that limit the population growth of an invading species (D'Antonio and Thomsen 2004).

Restoration pathways—A description of the environmental conditions and practices that are required to recover a state that has undergone a transition (Caudle et al. 2013).

Seed zone—An area of relative climatic similarity within which plant materials can be transferred with little risk of being poorly adapted to their new location.

State—A suite of community phases and their inherent soil properties that interact with the abiotic and biotic environment to produce persistent functional and structural attributes associated with a characteristic range of variability (adapted from Briske et al. 2008).

State-and-transition model—A method to organize and communicate complex information about the relationships among vegetation, soil, animals, hydrology, disturbances (fire, lack of fire, herbivory, drought, unusually wet periods, insects and disease), and management actions on an ecological site (Caudle et al. 2013).

Targeted grazing—Application of a specific kind of livestock at a determined season, duration, and intensity to accomplish defined vegetation or landscape goals (Launchbaugh and Walker 2006).

Thresholds—Conditions sufficient to modify ecosystem structure and function beyond the limits of ecological resilience and result in transitions to alternative states (Briske et al. 2008).

Transition—Transitions describe the biotic or abiotic variables or events, acting independently or in combination, that contribute directly to loss of state resilience and result in shifts between states. Transitions are often triggered by disturbances, including natural events (climatic events or fire) and management actions (grazing, prescribed fire, fire suppression). They can occur quickly as in the case of catastrophic events like fire or flood, or over a long period of time as in the case of a gradual shift in climate patterns or repeated stresses like frequent fires (Caudle et al. 2013).

Treatments—Local scale management actions that directly manipulate vegetation to achieve a vegetation or habitat objective (e.g., conifer removals, invasive annual grass controls, fuel treatments, or revegetation).

Wildfire—An unplanned, unwanted wildland fire including unauthorized human-caused fires, escaped wildland fire use events, escaped prescribed fire projects, and all other wildland fires where the objective is to put the fire out (NWCG 2018). See also Prescribed Fire, Wildland Fire.

Wildland fire—Any non-structure fire that occurs in vegetation or natural fuels. Wildland fire includes prescribed fire and wildfire (NWCG 2018). See also Prescribed Fire, Wildfire.

Wildland-Urban Interface—The line, area, or zone where structures and other human development meet or intermingle with undeveloped wildland or vegetative fuels (NWCG 2017, 2018). Describes an area within or adjacent to private and public property where mitigation actions can prevent damage or loss from wildfire.

Woodland (juniper and piñon) phase I, II, III—In phase I trees are present, but shrubs and herbs are the dominant vegetation influencing ecological processes on the site; in phase II trees are codominant with shrubs and herbs and all three vegetation layers influence ecological processes; in phase III trees are the dominant vegetation on the site and the primary plant layer influencing ecological processes on the site (Miller et al. 2005, 2014).

References

Agee, J.K. 1994. Fire and weather disturbances in terrestrial ecosystems of the eastern Cascades. Gen. Tech. Rep. PNW-GTR-320. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 52 p.

Angeler, D.G.; Allen, C.R. 2016. Quantifying resilience. Journal of Applied Ecology. 53: 617–624.

Boyd, C.S.; Johnson, D.D.; Kerby, J.D.; [et al.]. 2014. Of grouse and golden eggs: Can ecosystems be managed within a species-based regulatory framework? Rangeland Ecology and Management. 67: 358–368.

Briske, D.D.; Bestelmeyer, B.T.; Stringham, T.K.; [et al.]. 2008. Recommendations for development of resilience-based state-and-transition models. Rangeland Ecology and Management. 61: 359–367.

Caudle, D.; DiBenedetto, J.; Karl, M.; [et al.]. 2013. Interagency ecological site handbook for rangelands. Washington, DC: U.S. Department of the Interior, Bureau of Land Management; U.S. Department of Agriculture, Forest Service and Natural Resources Conservation Service. http://directives.sc.egov.usda.gov/viewerFS.aspx?hid=33150. [Accessed May 23, 2018].

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p. https://www.treesearch.fs.fed.us/pubs/53983. [Accessed May 23, 2018].

D'Antonio, C.M.; Thomsen, M. 2004. Ecological resistance in theory and practice. Weed Technology. 18: 1572–1577.

Evans, D.; Goble, D.D.; Scott, J.M. 2013. New priorities as the Endangered Species Act turns 40. Frontiers in Ecology and the Environment. 11: 519.

Folke, C.; Carpenter, S.; Walker, B.; [et al.]. 2004. Regime shifts, resilience, and biodiversity in ecosystem management. Annual Review of Ecology and Systematics. 35: 557–581.

Goldstein, M.I.; Suring, L.H.; Vojta, C.D.; [et al.]. 2013. Chapter 10. Developing a habitat monitoring program: three examples from National Forest planning. In: Rowland, M.M.; Vojta, C.D., tech. eds. A technical guide for monitoring wildlife habitat. Gen. Tech. Rep. WO-GTR-89. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 74 p.

Heinselman, M.L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quaternary Research. 3: 329–382.

Holling, C.S. 1973. Resilience and stability in ecological systems. Annual Review of Ecology and Systematics. 4: 1–23.

Launchbaugh, K.; Walker, J. 2006. Chapter 1. Targeted grazing–a new paradigm for livestock management. In: Launchbaugh, K.; Walker, J., eds. Targeted grazing: A natural approach to vegetation management and landscape enhancement. Englewood, CO: American Sheep Industry Association: 2–8.

Miller, R.F; Bates, J.D.; Svejcar, T.J.; [et al.]. 2005. Biology, ecology, and management of western juniper. Tech. Bull. 152. Corvallis, OR: Oregon State University, Agricultural Experiment Station. 82 p.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2014. A field guide to selecting the most appropriate treatments in sagebrush and pinyon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322-rev. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 66 p.

National Wildfire Coordinating Group [NWCG]. 2017. Wildland urban interface wildfire mitigation desk reference guide. PMS 051. Boise, ID: National Wildfire Coordinating Group. 12 p. https://www.nwcg.gov/sites/default/files/publications/pms051.pdf. [Accessed May 24, 2018].

National Wildfire Coordinating Group [NWCG]. 2018. Glossary of wildland fire terminology. Boise, ID: National Wildfire Coordinating Group. https://www.nwcg.gov/glossary/a-z. [Accessed May 24, 2018].

Oregon State University [OSU]. 2017. Differentiate warm-season from cool-season grasses. National Forage and Grasslands Curriculum. Corvallis, OR: Oregon State University. http://forages.oregonstate.edu/nfgc/eo/onlineforagecurriculum/instructormaterials/availabletopics/grasses/differentiate. [Accessed May 24, 2018].

Presidential Executive Order 13112. 1999. Invasive species. Federal Register.

Vol. 64, No. 25. Monday, February 8, 1999. Authenticated U.S. Government Information. Washington, DC: U.S. Government Printing Office. https://www.gpo.gov/fdsys/pkg/FR-1999-02-08/pdf/99-3184.pdf. [Accessed Dec 19, 2016].

St. John, Loren; Ogle, Dan G. 2009. Green strips or vegetative fuel breaks. Tech. Note No. 16. Aberdeen, ID: U.S. Department of Agriculture, Natural Resources Conservation Service, Aberdeen Plant Materials Center. 16 p.

Sugihara, N.G.; van Wagtendonk, J.W.; Fites-Kaufman, J. 2006. Fire as an ecological process. In: Sugihara, N.G.; van Wagtendonk, J.W.; Shaffer, K.E.; [et al.], eds. Fire in California's ecosystems. Berkeley, CA: University of California Press: 58–74.

U.S. Department of the Interior [USDOI]. 2016. Safeguarding America's lands and waters from invasive species: a national framework for early detection and rapid response. Washington, DC: U.S. Department of the Interior. 55 p. https://www.doi.gov/sites/doi.gov/files/National%20EDRR%20Framework.pdf. [Accessed May 24, 2018].

U.S. Department of the Interior, Bureau of Land Management [USDOI BLM]. 1987. Greenstripping Handbook. Boise, ID: U.S. Department of the Interior, Bureau of Land Management. 80 p.

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2013. Greater sage-grouse (Centrocercus urophasianus) conservation objectives: Final report. Denver, CO: U.S. Department of the Interior, Fish and Wildlife Service. 115 p. https://www.fws.gov/greatersagegrouse/documents/COT-Report-with-Dear-Interested-Reader-Letter.pdf [Accessed May 24, 2018].

Winthers, E.; Fallon, D.; Haglund, J.; [et al.]. 2005. Terrestrial ecological unit inventory technical guide. Gen. Tech. Rep. WO-GTR-68. Washington, DC: U.S. Department of Agriculture, Forest Service, Ecosystem Management Coordination Staff, Washington Office. 245 p.

Appendix 2—Websites and Resources for Climate Adaptation and Mitigation

Websites

Climate Change Resources Center (CCRC)

The CCRC is a USDA Forest Service sponsored portal. It is a web-based, nationwide resource that connects land managers and decisionmakers with usable science to address climate change in planning and application. The website contains links to numerous reports, papers, tools, and data for assessing climate change and climate change impacts. Website: http://www.fs.usda.gov/ccrc/home.

Conservation in a Changing Climate

This website is sponsored by the USDOI Fish and Wildlife Service (FWS) and provides information on climate change and the impacts of climate change on wildlife within each FWS region. The website provides information on the FWS response to climate change, including the U.S. Department of the Interior's Strategy for addressing climate change and the FWS Strategic Plan for managing in a time of uncertainty. In addition, ways that individuals can help mitigate the effects of climate change and support wildlife conservation are available. Website: https://www.fws.gov/home/climatechange/.

Climate Data and Analysis Tools

Historical and projected climate and climate change impacts data are available through a wide variety of sources and at different scales, although data at the mid-scale are the most common. In some cases, data may be limited to part of the sagebrush biome.

Climate Impacts Group (CIG)

Hosted by the University of Washington, the CIG provides climate data and analyses of potential climate change impacts at a variety of scales, ranging from local communities to the western United States. Most of the work to date is focused on the Pacific Northwest. Website: https://cig.uw.edu/.

Climate Adaptation Science Centers (CASCs)

The CASCs comprise eight regional CASCs covering the continental United States, Alaska, Hawaii, and U.S. Affiliated Pacific Islands. Each CASC is based at a host university in its region. Most are composed of multi-institution consortia, which include university and non-university partners. The CASCs provide scientific information, decision-support tools, and techniques needed to effectively manage natural and cultural resources and build resilient communities. The website allows individuals to search for climate science research and topics in the region of interest and provides a variety of resources including funding opportunities, webinars, and available education and training. Website: https://nccwsc.usgs.gov/.

Conservation Biology Institute (CBI) Integrated Climate Scenarios

The CBI provides projected climate change scenarios for climate, hydrology, and vegetation in the Northwest (Oregon, Washington, Idaho, western Montana) using downscaled climate projections based on multivariate adapted constructed analogs (MACA) in combination with the MC2 dynamic vegetation model. Model results are available for the entire area or by ecoregion. The site provides guidance and answers to frequently asked questions to assist users. Website: http://consbio.webfactional.com/integratedscenarios/.

Multivariate Adapted Constructed Analogs (MACA)

The MACA site is hosted by the University of Idaho and provides statistically downscaled climate projections for the continental United States using the most current emissions scenarios, several global climate models, and multi-model means. The website provides a number of options for viewing and downloading the data. Website: http://maca.northwestknowledge.net/.

PRISM Historical Climate Data

PRISM uses weather and climate observations from a wide range of monitoring networks to create wall-to-wall spatial climate datasets from 1895 to the present. PRISM datasets are widely used in a variety of climate and natural resource studies to describe historical climate. Website: http://www.prism.oregonstate.edu/.

State Climate Offices

Nearly every State has a climate office that provides access to State and local climate data from a variety of weather stations such as the Community Collaborative Rain, Hail and Snow Network, or CoCoRaHS (https://www.cocorahs.org/), and the Agricultural Meteorological Network (AgMet).

WestMap Climate Analysis Toolbox

WestMap delivers PRISM historical climate data at a variety of spatial scales ranging from Westwide to a single pixel, including user created polygons, and a variety of temporal scales. Climate data provided are precipitation, mean temperature, maximum temperature, and minimum temperature. Website: http://www.cefa.dri.edu/Westmap/westmappass.php.

Western Regional Climate Center (WRCC)

The WRCC provides access to climate and weather data across the western United States from several weather sources, include the NOAA co-op network, remote automated weather stations (RAWS), the Snotel network, and the Community Collaborative Rain, Hail and Snow Network (CoCoRaHS). Website: http://www.wrcc.dri.edu/.

Weather and Climate Tools for Sagebrush Managers

This website was developed by the Conservation Biology Institute to deliver the types of weather and climate data that land managers in sagebrush ecosystems of the northern Great Basin identified as desirable. The website provides graphics and descriptions of historical climate and weather data, including temperature, seasonal precipitation, aridity, and potential evapotranspiration. Also provided are near-term and short-term forecasts for use in planning projects such as postfire seeding and on projected climate change for 2016 to 2045 and 2046 to 2075 based on the MC2 model. The data and information cover the sagebrush biome, but are intended for use at the local scale. Website: http://climateconsole.org/sagebrush.

Great Basin Weather Applications for Rangeland Restoration

The Great Basin Weather Applications website was developed by the Agricultural Research Service in cooperation with the University of Idaho, U.S. Geological Survey, Utah State University, and the Great Basin Fire Science Exchange. This website provides access to restoration-specific weather and microclimatic information that can be used for (1) analyzing historical planting data, (2) expanding inferences derived from short-term field studies, and (3) developing long-term contingency-based adaptive management plans for rangeland restoration. This site provides historical time-series of site-specific weather and seedbed microclimatic information, rankings of year and seasonal weather effects, and detailed assessments of year-specific seasonal favorability for seedling establishment. Educational modules are being developed in collaboration with Brigham Young University for training restoration professionals in the use of weather and climate information for field planning and management. Additional future applications include using seasonal forecasts for real-time management planning and developing probabilistic future weather scenarios for determining adaptation and mitigation strategies under potential future climate regimes. Website: http://greatbasinweatherapplications.org/.

Carbon Storage Tools

Because of the emphasis on forest management in climate change programs, and the fact that most research and information on carbon storage focus on the mid- to biome scale, field personnel in semiarid lands generally lack the baseline information and impact estimation tools they need to conduct either quantitative or qualitative analyses. The U.S. Geological Survey, through its LandCarbon website (https://www2.usgs.gov/climate_landuse/land_carbon/), and Natural Resources Conservation Survey, through its CarbonScapes website (http://carbonscapes.org/), attempt to provide baseline carbon storage information. The LandCarbon site attempts to project how carbon storage may change by mid-century under different greenhouse gas emissions scenarios. Limitations are that the scales of the data provided by LandCarbon and CarbonScapes are too coarse for land use plan and project scales, and data provided by LandCarbon are outdated (2005 vintage). Further, data provided by CarbonScapes use only Forest Service Forest Inventory and Analysis (FIA) data for aboveground carbon, and watershed-scale data in CarbonScapes are not universally available due to lack of completed soil surveys.

The Fire and Fuels Tools (http://www.fs.fed.us/pnw/fera/fft/index.shtml) and First Order Fire Effects Model (FOFEM) (https://www.firelab.org/project/fofem) provide estimates of aboveground carbon by carbon pool for standardized fuel beds and community types. Users can adjust the estimated fuel loadings manually based on local information or plot data. Both tools predict changes in aboveground carbon storage and greenhouse gas emissions from burning. However, these tools are designed to operate at the treatment block scale and cover only fire. Batch processing is theoretically possible with Fire and Fuels Tools, but can be difficult to conduct.

Appendix 3—Invasive Plants to Include in Early Detection and Rapid Response Programs in Sagebrush Ecosystems

Nonnative invasive plants in sagebrush ecosystems listed from highly invasive to weakly invasive (modified from Ielmini et al. 2015: tables 2 and 4), followed by the States where there is still only no, patchy, or limited presence of the species in sagebrush habitat, and then the habitat characteristics and impacts of the invasive plant (based on Sheley and Petroff 1999 and DiTomaso et al. 2013). If a State is not listed, then the species is already established in sagebrush habitat, but there still may be potential for Early Detection and Rapid Response (EDRR) (USDOI 2016) in limited regional and local EDRR areas. For example, Idaho has significant populations of yellow starthistle, but there are still regional areas and land management units that are uninvaded and suitable for local EDRR strategies. Assistance in developing the list was provided by State Weed Coordinators from State Departments of Agriculture.

Certain problem species were noted but not included. For example, perennial pepperweed or tall whitetop (*Lepidium latifolium*) is a major concern in sagebrush ecosystems in California. This species prefers pastures and areas with greater water availability than typically occurs in sagebrush ecosystems, but significant sagebrush areas are on the margins of riparian or wetland zones that are being heavily invaded by perennial pepperweed. There are similar concerns about saltcedar (*Tamarix* spp.). North Dakota did not include any of the listed species because of the small amount of sagebrush habitat in the State.

Plant	Scientific name	EDRR potential in sagebrush habitat	Habitat	Negative impacts
Medusahead	*Taeniatherum caput-medusae*	CA, CO, MT, UT, WY, ID, NV, WA, SD	Occurs in sagebrush-grass or bunchgrass communities that receive at least 9–12 inches [23–30 centimeters] precipitation. Often invades after disturbance. Does well in clay soils that shrink, swell, and crack and openings in chaparral vegetation types.	Low palatability for livestock due to high silica content, which confers competitive advantage over native plants. Awns can injure eyes and mouths of animals. Dense, long-lasting litter layer creates fire risk and reduces seed germination of other species.
Cheatgrass	*Bromus tectorum*	Local and regional EDRR potential	Wide ecological amplitude from salt desert in the Great Basin to coniferous forests in the Rocky Mountains. Areas in which most precipitation arrives in late winter or early spring are most susceptible. Often occurs in disturbed areas and areas with dry sandy soils with little competition.	Increases fine fuels and fire risk. Can outcompete many perennial native plant species and replace many annual species. Reduces production of perennial grasses for livestock forage, but can be grazed in winter or spring. Sharp seeds may cause eye injuries.
North African wiregrass	*Ventenata dubia*	CA, MT, CO, ID, UT, WY, NV, WA, SD	Occurs in bunchgrass, sagebrush, and meadow communities.	Can outcompete perennial bunchgrasses. Low palatability for livestock due to high silica content. Matures early in the season and is likely to pose fire risks.

(Continued)

Plant	Scientific name	EDRR potential in sagebrush habitat	Habitat	Negative impacts
Spotted knapweed	*Centaurea maculosa*	CA, UT, NV, WA, SD, OR*	Occurs over a wide range of elevation and annual precipitation. Does well in forest-grassland interface on deep, well-developed soils, with dense stands occurring in moist areas on well-drained soils including fields, roadsides, and disturbed and degraded rangeland.	Very competitive and can form dense stands that result in higher surface water runoff and soil erosion. Excludes desirable vegetation, thereby reducing livestock and wildlife forage.
Yellow starthistle	*Centaurea solstitialis*	CA, CO, MT, UT, WY, NV, SD, OR*	Occurs on deep, loamy soils and south-facing slopes with 12–25 inches [30–64 centimeters] precipitation. Found in open disturbed sites, rangeland, roadsides, and open woodlands.	Highly competitive and develops dense, impenetrable stands. Reduces forage production for livestock and wildlife. Can be grazed before spine development, but poisonous to horses.
Iberian starthistle	*Centaurea iberica*	CA, CO, ID, MT, UT, WY, NV, WA, SD, OR	Occurs on riverine banks, along watercourses, and in other moist areas.	Unpalatable—spines restrict access to the plant and deter grazing.
Purple starthistle	*Centaurea calcitrapa*	CA, CO, ID, MT, UT, WY, NV, WA, SD, OR	Can inhabit a wide range of conditions, including fertile alluvial soils, pasture, range, open forest, and riparian areas.	Unpalatable—spines restrict access to the plant and deter grazing.
Diffuse knapweed	*Centaurea diffusa*	CA, UT, NV, SD, OR*	Wide ecological amplitude for elevation, aspect, slope, and soil properties. Maximum invasiveness is in shrub steppe, rangelands, and forested benchlands. Often occurs on well-drained soils.	Increases soil erosion and surface runoff. Replaces wildlife and livestock forage, but has some forage value through the bolting stage. Dispersal similar to tumbleweeds.
Leafy spurge	*Euphorbia esula*	CA, UT, NV, WA, OR*	Found in disturbed sites, roadsides, rangelands, and riparian areas with semiarid to mesic conditions. It has wide ecological amplitude and occurs on many soil types. High genetic variability allows it to easily adapt to local growing conditions.	Highly competitive and can form dense clones that suppress native plants and reduce forage. Milky sap is toxic and can irritate skin, eyes, and digestive tracts of humans and other animals. Sheep and goats graze it and can tolerate the toxins.
Rush skeletonweed	*Chondrilla juncea*	CA, CO, MT, WY, NV, SD	Found in rangelands and pastures and along roadsides. Occurs in very dry to very wet environments on disturbed soils and well-drained, sandy-textured, or rocky soils.	Can form dense monocultures and displace native plants, reduce livestock forage, and spread from rangeland to adjacent cropland. Wiry stems can interfere with harvest machinery.
Dalmatian toadflax	*Linaria dalmatica*	CA, NV, WA, SD	Tolerates many soil types and is found on well-drained, coarse-textured soils and sandy loams, as well as heavier soils. Does best in cool, semiarid climates on dry, coarse soils with neutral to slightly alkaline pH and south- to southeast-facing slopes. Occurs in rangelands, disturbed areas, roadsides, and forest clearings. Can move into undisturbed prairies and riparian habitats.	Aggressive invader capable of forming dense colonies and outcompeting native grasses and other perennials. Decreases forage for livestock and wildlife. If sufficient quantities are ingested, quinazoline alkaloids can pose toxicity problems to livestock, but goats and sheep are tolerant. Can increase soil erosion, surface runoff, and sediment yield in invaded bunchgrass communities.

(Continued)

Plant	Scientific name	EDRR potential in sagebrush habitat	Habitat	Negative impacts
Sulphur cinquefoil	*Potentilla recta*	CA, UT, ID, WY, NV, WA, SD	Wide ecological amplitude. Found in conifer, grassland, shrubland, and seasonal wetland ecosystems. Occurs along roadsides and in other disturbed sites, but also will invade low-disturbance sites.	Low palatability for grazing animals, possibly due to phenolic tannins in leaves and stems. Can become a dominant component of plant communities.
Russian knapweed	*Acroptilon repens*	NV, WA, OR*	Found in pastures, in rangelands, and along streambanks and roadsides. Will invade croplands. Occurs on many soil types, but prefers moist soils that are not excessively wet.	Allelopathic and very competitive, forming dense stands. Reduces forage for livestock; low palatability for livestock and toxic to horses.
Squarrose knapweed	*Centaurea virgata*	CA, CO, ID, MT, WY, NV, WA, SD, OR	Found in fields, roadsides, disturbed sites, grasslands, and big sagebrush bunchgrass- and juniper-dominated rangelands. Extends into salt desert shrub, particularly in sandy or gravelly washes, and on dry, rocky, south-facing slopes. Will invade fairly pristine mountain brush types and juniper-Idaho fescue rangeland. Also will invade abandoned dry wheat fields, crested wheatgrass seedings, burned areas, and improperly grazed areas.	Highly competitive. Can endure drought at either temperature extreme, is fire tolerant, and has excellent seed dispersal and rapid response to soil resources released by fire. Rosettes grow slowly for years before flowering, creating basically a vegetative seedbank. Similar palatability and nutritive value to diffuse or spotted knapweed. Sheep and cattle may graze it when other annual forage is sparse. Dense stands can exclude desirable vegetation and wildlife in natural areas.
Whitetop, hoary cress	*Cardaria* spp.	Not listed as an EDRR species by any of the States	Found in disturbed open sites, on ditch banks, and along roadsides. Well-adapted to moist habitats, especially sub-irrigated rangeland, pastures, wetlands, and riparian areas. Tolerates a wide range of soil types and moisture conditions; often found in disturbed areas with other invasive species.	Can form dense monocultures, and is difficult to control due to large and deep roots and rhizomes. Can dramatically reduce biodiversity and forage production and can invade cropland and reduce yields. Plants contain glucosinolates, which can form toxic compounds. Unpalatable to livestock.
Yellow toadflax	*Linaria vulgaris*	CA, UT, SD	Found in riparian areas, rangeland, disturbed areas, roadsides, and forest clearings. Often occurs on moister sites. Tolerates many soil types varying from coarse gravels to sandy loams, but is also found in heavier soils. Can move into undisturbed prairies and riparian habitats.	Highly competitive for soil moisture with winter annuals and shallow-rooted perennials. Aggressive invader capable of forming dense colonies and outcompeting native grasses and perennials. Decreases forage for livestock and wildlife. If sufficient quantities are ingested, quinazoline alkaloids can pose toxicity problems to livestock, but goats and sheep are tolerant.
Dyer's woad	*Isatis tinctoria*	CO, MT, UT, WY, NV, WA, SD	Occurs in disturbed sites, roadsides, pastures, forests, and rangeland often on dry, rocky, or sandy soils. Invades undisturbed natural areas as well as alfalfa and small grain fields. Also found along waterways. Adapted to the arid climate and alkaline soils of the West.	Palatable to cattle only before bolting; grazing can be done before flowering to minimize seed production. Can spread at an annual rate of 14% and reduce grazing capacity by an average of 38%. Capable of invading and increasing density on well-vegetated range sites even in the absence of grazing or disturbance.

(Continued)

Plant	Scientific name	EDRR potential in sagebrush habitat	Habitat	Negative impacts
Mediterranean sage	*Salvia aethiopis*	CO, ID, MT, UT, WY, NV, WA	Found in degraded big sagebrush communities, rangeland, openings in ponderosa pine, and disturbed sites, including roadsides. Also occurs in floodplain and riparian areas following overgrazing, excessive trampling, and soil erosion. Often inhabits moderate to deeper soils with good drainage. Often associated with sites dominated by annual grasses.	Unpalatable to grazing animals, and although not considered toxic, reduces forage production on rangeland and pastures. Tumbleweed-mobility facilitates rapid spread in degraded communities. May attain understory dominance in sagebrush/cheatgrass communities.
Scotch thistle	*Onopordum acanthium*	CA, WA	Found in disturbed areas, rangeland, forest clearings, abandoned cropland, areas of high rodent activity, and along river and stream corridors and roadsides. Best suited to areas with high soil moisture during germination. Often associated with cheatgrass.	Can form dense stands over large acreages and decrease desirable forage. Sharp spines deter livestock and wildlife from grazing. Dense stands can prevent movement by livestock, wildlife, and humans. Grazing of young plants may occur in early stages of infestation, but overgrazing promotes scotch thistle.
Halogeton	*Halogeton glomeratus*	CA, NV, WA, SD	Occurs in dry, arid regions, and is adapted primarily to alkaline and saline soils.	Foliage contains soluble sodium oxalates and can be toxic to livestock, especially sheep, when large quantities are ingested.
Musk thistle	*Carduus nutans*	CA, WA	Found in disturbed open sites, roadsides, pastures, and annual grasslands. Occurs over a wide range of environmental conditions, ranging from saline soils in low elevation valleys to acidic soils in high elevations. Potentially intolerant of shading from neighboring plants.	Can form dense stands over large areas and decrease desirable forage. Sharp spines deter livestock and wildlife from grazing. Dense stands can prevent movement by livestock, wildlife, and humans. Allelopathy can reduce growth of desirable pasture species in an area much greater in diameter than the musk thistles themselves. May take 15 years of treatment to decrease germination.
Common crupina	*Crupina vulgaris*	CA, CO, ID, MT, UT, WY, NV, WA, SD	Occurs in grasslands, pastures, rangeland, canyons, disturbed riparian areas, and gravel pits. Adapted to many temperature and moisture regimes and soil types. Infests sites with cheatgrass.	Highly competitive for limited soil moisture. Dense populations reduce and displace desirable forage species for livestock and wildlife and can contaminate hay. Seeds can survive ingestion by animals and remain viable in soil up to 3 years. Most livestock avoid grazing it. Can displace perennial bunchgrasses and lead to soil erosion because of less effective soil stabilization.

Note.—*Oregon species that is a State-listed B-Noxious Weed and is established in some areas. However, in areas that are currently known to lack the listed invader, it is considered and EDRR species.

References

DiTomaso, J.M.; Kyser, G.B.; Oneto, S.R.; [et al.]. 2013. Weed control in natural areas in the western United States. Davis, CA: UC Davis Weed Research and Information Center. 544 p. http://www.cal-ipc.org/resources/library/publications/weedcontrol/. [Accessed May 23, 2018].

Ielmini, M.R.; Hopkins, T.E.; Mayer, K.E.; [et al.]. 2015. Invasive plant management and greater sage-grouse conservation: A review and status report with strategic recommendations for improvement. Cheyenne, WY: Western Association of Fish and Wildlife Agencies. 47 p. https://www.wafwa.org/Documents%20and%20Settings/37/Site%20Documents/Initiatives/Sage%20Grouse/WAFWA%20Invasive%20Plant%20Management%20and%20Greater%20Sage-Grouse%20Report%20FINAL%203-28-15.pdf. [Accessed May 23, 2018].

Sheley, R.L.; Petroff, J.K., eds. 1999. Biology and management of noxious rangeland weeds. Corvallis, OR: Oregon State University Press. 438 p.

U.S. Department of the Interior [USDOI]. 2016. Safeguarding America's lands and waters from invasive species: A national framework for early detection and rapid response. Washington, DC: U.S. Department of the Interior. 55 p. https://www.doi.gov/sites/doi.gov/files/National%20EDRR%20Framework.pdf. [Accessed May 23, 2018].

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Importance of regional variation in conservation planning: a rangewide example of the Greater Sage-Grouse

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Abstract. We developed rangewide population and habitat models for Greater Sage-Grouse (*Centrocercus urophasianus*) that account for regional variation in habitat selection and relative densities of birds for use in conservation planning and risk assessments. We developed a probabilistic model of occupied breeding habitat by statistically linking habitat characteristics within 4 miles of an occupied lek using a nonlinear machine learning technique (Random Forests). Habitat characteristics used were quantified in GIS and represent standard abiotic and biotic variables related to sage-grouse biology. Statistical model fit was high (mean correctly classified = 82.0%, range = 75.4–88.0%) as were cross-validation statistics (mean = 80.9%, range = 75.1–85.8%). We also developed a spatially explicit model to quantify the relative density of breeding birds across each Greater Sage-Grouse management zone. The models demonstrate distinct clustering of relative abundance of sage-grouse populations across all management zones. On average, approximately half of the breeding population is predicted to be within 10% of the occupied range. We also found that 80% of sage-grouse populations were contained in 25–34% of the occupied range within each management zone. Our rangewide population and habitat models account for regional variation in habitat selection and the relative densities of birds, and thus, they can serve as a consistent and common currency to assess how sage-grouse habitat and populations overlap with conservation actions or threats over the entire sage-grouse range. We also quantified differences in functional habitat responses and disturbance thresholds across the Western Association of Fish and Wildlife Agencies (WAFWA) management zones using statistical relationships identified during habitat modeling. Even for a species as specialized as Greater Sage-Grouse, our results show that ecological context matters in both the strength of habitat selection (i.e., functional response curves) and response to disturbance.

Key words: breeding habitat; conservation planning; ecological variation; function habitat response; Greater Sage-Grouse; landscape context; population index; resource selection function; spatial modeling; thresholds.

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Introduction

In an increasingly anthropogenic world where funding for conservation activities is limited, effective landscape-scale conservation planning tools have been progressively embraced by resource management agencies to both maximize conservation investments and reduce impacts of anthropogenic disturbances. This has corresponded with rapid expansion of landscape-scale, spatially explicit models of species habitat, such as resource selection functions (RSF) (Boyce and McDonald 1999, Johnson et al. 2006, 2013), which simultaneously give insight into the ecology of species and can be used to produce maps to help guide where conservation actions should be most effective. Often, RSF models do not encompass the entire range of a focal species, and therefore, biological relationships are extrapolated to novel areas, not included in the development of the RSF models, when decisions must be made. Extrapolating known relationships often represents the best available information to decision makers, but should be done with caution because the accuracy of extrapolated models relies on similar habitat availability in the novel area (Mladenoff et al. 1999, Aarts et al. 2008).

A species response to particular habitat components can change as a function of the prevalence of the resource, which is referred to as the functional response of a species (Mysterud and Ims 1998). Understanding functional responses related to habitat selection through RSF modeling can elucidate threshold values for habitat quantity and quality, tolerance to perturbations, and cumulative effects (Rhodes et al. 2008). Understanding functional responses is important as conservation plans generally require targets for the amount of habitat required for specific species in order for managers to make cost-effective decisions and balance competing interests (Carwardine et al. 2008). Unfortunately, setting conservation targets based upon thresholds defined for other regions is precarious (Rhodes et al. 2008) because thresholds can vary tremendously across species and landscapes (van der Hoek et al. 2015). Landscape-scale modeling across broad extents is important in understanding how functional responses may vary for wide-ranging species, as landscapes are seldom homogeneous across large extents.

Data on the abundance of individuals are rare for most taxa, yet if available, they can provide baseline data for monitoring populations and conservation actions (Sagarin et al. 2006). Abundance is often clustered across the range of a species, typically being high in relatively few sites and low in the majority (Murphy et al. 2006). Knowledge and mapping of population centers or "hotspots" can be critically important for conservation planning as many species with broad distributions occur in densities of several orders of magnitude higher in hotspots compared with occupied habitats outside of hotspot boundaries (Brown et al. 1995). Locations of population centers of many species can be stable over several decades even while population sizes fluctuate (Brown et al. 1995). Consequently, habitat protection can affect drastically different proportions of target populations depending on overlap with population centers.

Ideally, conservation planning makes the best use of information related to population abundance and habitat requirements while accounting for regional gradients and differences in functional responses. When broad-scale population survey data exist, probabilistic surfaces of density indices and habitat selection indices can be integrated to create analytical tools across broad spatial scales (Coates et al. 2015). This type of integrative methodology can create composite, spatially explicit indices that reflect demographic and habitat information and make predictions to guide landscape-level conservation actions. Unfortunately, such data are rare in conservation planning because the broad-scale population surveys are lacking for many species and habitat modeling, by necessity, is often conducted at scales smaller than a species range.

Greater Sage-Grouse (*Centrocercus urophasianus*; hereafter sage-grouse) is a wide-ranging species of conservation concern that occurs throughout the sagebrush ecosystem in the Intermountain West of the United States (Schroeder et al. 2004: see Figure 1). Sage-grouse occupy approximately one-half of their historical distribution, and populations have declined concomitantly with the loss of sagebrush since pre-European settlement of the West (Schroeder et al. 2004). Currently, sage-grouse are considered "not-warranted" for listing under the United States Endangered Species Act of 1973 (ESA; U.S. Fish and Wildlife Service [USFWS] 2015), with a 5-year review to the decision scheduled for September 2020. Because of the wide-reaching implications of an ESA listing on western lands within North America, monitoring sage-grouse populations is imperative to help inform land and wildlife management agencies responsible for regulatory actions and policies. Lek sites (traditional breeding grounds) provide opportunity to count sage-grouse annually and monitor population response. Leks are typically located in nesting habitat where males are most likely to encounter females for breeding opportunities (Gibson and Langen 1996), and several studies support this hypothesis for both

Greater Prairie chickens (*Tympanuchus cupido*) and sage-grouse (Schroeder and White 1993, Gibson 1996*b*, Holloran and Anderson 2005, Doherty et al. 2010*b*, 2011, Coates et al. 2013). Although sage-grouse leks have been counted each year since the 1950s, wildlife agencies have drastically increased their efforts in surveying known leks and searching for new lek sites since the mid-1990s, with almost exponential increases in survey effort during the last decade (WAFWA 2015). Broad-scale sage-grouse lek survey data managed by each state with sage-grouse provide a unique opportunity to identify sources of temporal and spatial variation in functional responses across the entire range of a species that inhabits most of the western United States. Furthermore, findings from such analysis could be used to target thresholds for conservation planning activities for a species of increasingly high conservation concern.

Knowledge of high-abundance population centers for priority species represents a starting point to frame regional conservation initiatives and can direct management actions to landscapes where they will have the largest benefit to regional populations (Sanderson et al. 2002, Groves 2003). We developed a model to quantify the relative density of breeding birds within each sage-grouse management zone. This was motivated by past work across the range that showed sage-grouse populations are highly clustered (Connelly et al. 2004, Stiver et al. 2006, Doherty et al. 2011). Fortunately, sage-grouse are one of the few species in which extensive data sets exist on distribution and relative abundance across their entire breeding distribution, making an analysis of this scale possible (Connelly et al. 2004, Schroeder et al. 2004). We had two primary objectives within this study: (1) To develop rangewide habitat and population models that identify regional variation in habitat selection and relative densities of sage-grouse for use in conservation planning and risk assessments and (2) to assess the importance of variability in habitat selection and thresholds of disturbance and to identify differences in functional responses across the range of sage-grouse.

Study Area

Our study area includes the entire range of North American sage-grouse populations with the exception of six active leks located in Canada (Fig. 1). Canadian leks were not included in our modeling because of significant differences in available spatial data between the United States and Canada. Loss and degradation of native vegetation have affected much of the sagebrush (*Artemisia* spp.) ecosystem in western North America, and this ecosystem has become increasingly fragmented because of conifer encroachment, exotic annual grass invasion, and anthropogenic development (Knick et al. 2003). The Western Association of Fish and Wildlife Agencies (WAFWA) Conservation Strategy for Greater Sage-Grouse (Stiver et al. 2006) delineated seven sage-grouse management zones to guide conservation and management (Table 1). The boundaries of these management zones were delineated based on differences in ecological and biological attributes (i.e., floristic provinces) rather than on arbitrary political boundaries (Stiver et al. 2006) (Fig. 1). Maps representing the major ecological gradients and subsequent dominant land cover types are shown in Appendix S1. We stratified our analyses by sage-grouse management zones because spatial partitioning of data improves model fit where regional niche variation occurs (Murphy and Lovett-Doust 2007) because of fundamental differences in the ecological gradients and different functional responses at regional scales.

Methods

Breeding habitat model

We developed a binomial probabilistic model of occupied breeding habitat by quantifying habitat characteristics, within 6.4 km (4 miles) of both occupied sage-grouse leks and pseudoabsence points using a classification instance of the nonparametric model Random Forests (Cutler et al. 2007, Olden et al. 2008, Evans et al. 2011, Baruch-Mordo et al. 2013). Model predictions produce an estimated probability of sage-grouse lek occurrence for each 120 × 120 m grid cell within each sage-grouse management zone. Components of sage-grouse habitat were compiled into a GIS database from various sources, but generally represent standard abiotic and biotic variables used in past work to represent sage-grouse habitat (Table 2). Sage-grouse habitat use has been investigated extensively across

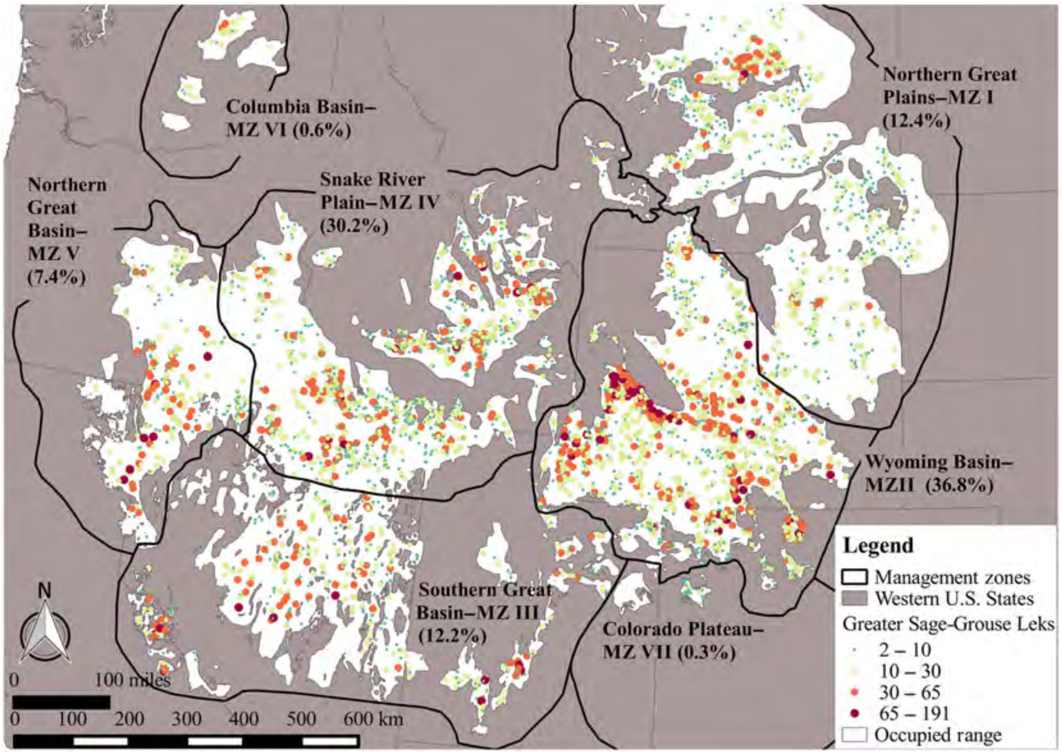

Fig. 1. Location of Greater Sage-Grouse management zones used to spatially subset analyses and the location of active Greater Sage-Grouse leks counted during 2010–2014. Percentages are derived from the sum of the mean peak count of displaying sage-grouse at individual leks during 2010–2014 within each management zone divided by the rangewide total, to give context to the amount of known populations within each management zone.

the range. For brevity, we provide the justification for variables, predicted relationships, and relevant citations in table format, rather than extensive in-text descriptions (Table 2).

Lek survey data

We compared active lek locations with pseudo-absence locations to generate models of predicted breeding sage-grouse habitat across the range. The hotspot hypothesis of lek evolution suggests that leks are typically located in close proximity to nesting habitat where males will most likely encounter prenesting females who are attracted by important habitat features (Schroeder and White 1993, Gibson 1996*a*), such as forbs required for prebreeding (Barnett and Crawford 1994) and sagebrush cover for nesting (Connelly et al. 2000). Additionally, 79–95% of sage-grouse nesting locations are located within 6.4 km of a lek (Holloran and Anderson 2005, Doherty et al. 2011, Coates et al. 2013). Further, recent studies have shown that telemetry-based models of nesting sage-grouse predicted almost two times more nesting habitat around leks than at random locations (Doherty et al. 2010*b*, Fedy et al. 2014). We therefore believe that sage-grouse lek locations are a good predictor of important breeding areas. We used lek data assembled and proofed by WAFWA to develop both our breeding habitat model and breeding population index model. For the purposes of both models, a lek was defined as active if greater than two males were counted during a single counting visit during 2010–2014 and the last count was not a zero.

Pseudoabsence data

Recent lek survey efforts have been intensive enough that although not all leks have been identified, we are confident that the spatial processes governing lek locations and sage-grouse abundance were well represented in the data. To generate pseudoabsence (i.e., background) locations, we

Table 1. Ecological descriptions of Western Association of Fish and Wildlife Agencies Greater Sage-Grouse Management Zones.

Zone	Description
Northern Great Plains (MZ I)	The Northern Great Plains includes the northeastern portions of the sage-grouse range. This management zone experiences the most precipitation, and thus, it contains larger portions of the landscape dominated by grasslands, smaller patches of sagebrush, and more silver sagebrush (Artemisia cana var. cana) than other management zones. MZ I also has the highest amount of land in private ownership, and compared with other management zones, it has the highest amount of cropland
Wyoming Basin (MZ II)	The Wyoming Basin is characterized by large expanses of Wyoming big sagebrush (*Artemisia tridentata* var. *wyomingensis*) with little fragmentation; however, it experiences the greatest amount of oil and gas development. Most of the precipitation in this management zone comes in the form of winter snowfall. MZ II contains the highest densities of sage-grouse across their range
Southern Great Basin (MZ III)	The Southern Great Basin includes the southern- and westernmost populations of sage-grouse. MZ III is the most arid of all the management zones and includes a mix of Wyoming big sagebrush, mountain big sagebrush (*A. tridentata* var. *vaseyana*), low sagebrush (*A. arbuscula*), and black sagebrush (*A. nova*). Topography is rugged with sagebrush on many of the valley floors transitioning to arid coniferous forests at higher elevations on the mountain slopes
Snake River Plain (MZ IV)	The Snake River Plain encompasses the north-central populations of sage-grouse. Like MZ III and MZ V, it is characterized by salt deserts in the lower elevations and conifer forests at higher elevations. Wyoming big sagebrush and basin big sagebrush (*A. tridentata* var. *tridentata*) are the dominant species, with mountain big sagebrush at higher elevations. MZ IV contains the second highest density of sage-grouse across the species range. The Snake River Plains management zone also experiences dense cropland areas; however, they are clustered at lower elevations
Northern Great Basin (MZ V)	The Northern Great Basin is similar to the Southern Great Basin, but it is less arid with precipitation occurring primarily in the winter and spring. Similar to MZ III and MZ IV, lower elevations are dominated by salt deserts and higher elevations are dominated by conifer forest
Columbia Basin (MZ VI)	The Colombia Basin is isolated from the rest of the sage-grouse range and is contained entirely within Washington state. Wyoming big sagebrush and basin big sagebrush are predominate species. MZ VI contains the lowest elevation sagebrush across the range and experiences high amounts of cropland in comparison with all other management zones with the exception of the Northern Great Plains
Colorado Plateau (MZ VII)	The Colorado Plateau is the southeastern-most management zone and contains a small fraction of the overall sage-grouse populations. It is similar to the Southern Great Basin MZ, but it receives more precipitation. Soil types within the Colorado Plateau greatly restrict the sagebrush distribution, and it contains a very small portion of the overall occupied habitat

Notes: Descriptions of management zones were originally summarized (Miller and Eddleman 2001) and adapted by WAFWA for analyses for both the 2004 Conservation Assessment of Greater Sage-Grouse and Sagebrush Habitats (Connelly et al. 2004) and 2006 Greater Sage-Grouse Comprehensive Conservation Strategy (Stiver et al. 2006). We created maps of the ecological gradients and major land cover types between Greater Sage-Grouse management zones for further reference in Appendix S1. Maps focused on the major ecological gradients and subsequent land cover (Figs. 3–9).

modeled the spatial process of known leks, using an isotropic kernel estimate (Diggle 1985), and used the inverse of the density estimate to weight samples. A gradient function allowed for a tension parameter to control the proximity of pseudoabsence locations in relation to known lek locations. We utilized the pseudoabsence model available in the spatialEco library (Evans 2015) and defined the sigma (distance smoothing for the kernel; bandwidth) as 18 km and the gradient as 1, thus providing no weighting to the pseudoabsence diffusion process. This ensured that we were sampling the range of habitat variation within each sage-grouse management zone. To avoid class imbalance (Evans and Cushman 2009) (i.e., zero inflation) issues, we generated an equal ratio of pseudoabsence to lek locations and compared resulting sample variation against population data (rasters) to evaluate whether we had an adequate sample to support model fit, spatial estimation, and inference. We chose an 18-km bandwidth because recent research has shown that this scale represents the scale at which breeding populations move across the landscape to fulfill other seasonal habitat needs (Fedy et al. 2012) and because we specifically designed our study to capture large first-order habitat selection. To accurately define first-order sage-grouse habitat availability extents, we matched the spatial scale of availability to the desired scale of inference because matching such scales is critical to obtaining reliable estimates on selection behavior (Northrup et al. 2013).

Statistical model

Nonparametric methods are becoming much more common in ecological modeling, supporting

Table 2. Description of explanatory variables used to predict the occupied Greater Sage-Grouse breeding habitat across 11 western U.S. States during 2010–2014.

Name	Source (years)	Native pixel (m)	Resampled pixel (m)	Description†	Justification (references)
General habitat predictor group					
Low sagebrush	LANDFIRE EVT 1.2 (2010)‡	30 × 30	120 × 120	% of grid cells classified as low sagebrush	Established positive relationship between sage-grouse abundance and sagebrush (Patterson 1952)
Tall sagebrush	LANDFIRE EVT 1.2 (2010)‡	30 × 30	120 × 120	% of grid cells classified as tall sagebrush	Established positive relationship between sage-grouse abundance and sagebrush (Patterson 1952)
All sagebrush	LANDFIRE EVT 1.2 (2010)‡	30 × 30	120 × 120	% of grid cells classified as all sagebrush	Established positive relationship between sage-grouse abundance and sagebrush (Patterson 1952)
Canopy cover	LANDFIRE Fuels 1.2 (2010)	30 × 30	120 × 120	% canopy cover in 10% increments from 15% to 95%	Established negative relationship between sage-grouse and conifers (Doherty et al. 2008, Baruch-Mordo et al. 2013, Fedy et al. 2014)
Grassland/ herbaceous	LANDFIRE Fuels 1.2 (2010)	30 × 30	120 × 120	% of grid cells classified as grassland	Established negative relationship between sage-grouse abundance and grasslands (Patterson 1952)
Perennial water	National Hydrological Dataset NHD (2012)	Vector of Lines and Polygons	120 × 120	NHD perennial flow lines within a 6440-m moving window, multiplied by the average line length per cell (133.2 m)	Established negative relationship of riparian areas with nest site selection (Crawford et al. 2004) and established positive relationship between sage-grouse populations and riparian habitats (Blomberg et al. 2014)
Intermittent water	NHD (2012)	Vector of Lines and Polygons	120 × 120	See perennial water	See perennial water
Springs and seeps	NHD (2012)	Vector of Lines and Polygons	120 × 120	See perennial water	See perennial water
Topographic wetness index	NHD (2012) and NED elevation Data (2013)	30 × 30	120 × 120	Index of wetness	See perennial water
Climatic data predictor group§					
Gross primary production	MODIS NASA EODP (2009–2013)	1 × 1 km	120 × 120	Index of early brood-rearing habitat (mean of GPP from 5–15 through 6–15)	Forbs are important predictors of early brood survival and habitat selection (Crawford et al. 2004)
Degree days > 5°C	USFS (1961–1990) (Rehfeldt et al. 2006)	1 × 1 km	120 × 120	The number of days that reach a temperature ≥5°C	Large-scale ecological driver of land types. Hypothesized regional-scale relationship between sagebrush landscapes with higher production. Documented carryover effects (Blomberg et al. 2014)
Mean annual precipitation	USFS (1961–1990) (Rehfeldt et al. 2006)	1 × 1 km	120 × 120	Mean annual precipitation (mm)	Large-scale ecological driver of land types. Hypothesized regional-scale relationship between sagebrush landscapes with higher production. Documented carryover effects (Blomberg et al. 2014)
Annual drought index	USFS (1961–1990) (Rehfeldt et al. 2006)	1 × 1 km	120 × 120	Ratio = dd5/map	Large-scale ecological driver of land types. Hypothesized regional-scale relationship between sagebrush landscapes with higher production. Documented carryover effects (Blomberg et al. 2014)

Table 2. Continued.

Name	Source (years)	Native pixel (m)	Resampled pixel (m)	Description†	Justification (references)
Landform variables predictor group					
Roughness	National Elevation Data NED (2013)	30 × 30	120 × 120	SD in elevation within a 6440-m buffer of a grid cell	Established negative relationship between sage-grouse and rough terrain (Doherty et al. 2008, Fedy et al. 2014)
Elevation	NED (2013)	30 × 30	120 × 120	Average elevation within a 6440-m buffer of the grid cell	Hypothesized relationship between grouse populations and areas with higher productivity because of elevation
Steep	NED (2013)	30 × 30	120 × 120	% of landscape classified as steep using Theobald LCAP tool	Established negative relationship between sage-grouse and rough terrain (Doherty et al. 2008, Fedy et al. 2014)
Disturbance variables predictor group					
Human disturbance index	NLCD Disturbed Classes¶ (2011)	30 × 30	120 × 120	Land cover types associated with human presence	Established negative relationship between sage-grouse and human activity (Tack 2009, Naugle et al. 2011*a*)
Oil and gas wells	IHS oil and gas database (1920's–2014)	Point	120 × 120	Density of oil and gas well locations†	Established negative relationship between sage-grouse and oil and gas development (Naugle et al. 2011*b*, Gregory and Beck 2014)
Burned landscapes	WFDSS-GeoMac Fire Perimeters (2000–2008, 2009–2013, 1984–2013)	Vector of Polygons	120 × 120	Proportion of grid cells that are burned within a 6440-m area	Established negative relationship between fire and sagebrush habitat (Nelle et al. 2000, Hess and Beck 2012)
Agriculture lands	NASS (2008–2014)	30 × 30	120 × 120	Proportion of grid cells that have been tilled since 2008 within a 6440-m area	Established negative relationship between sage-grouse and cropland (Knick et al. 2013, Fedy et al. 2014)

Note: All variables with the exception of the climate date predictor group were quantified using a 6.4-km buffer moving window (130.1 km^2).

† All variables were resampled to a 120 × 120 m pixel. All moving windows were calculated at a 6440-m (4-mile) buffer. Oil and gas layers were also calculated at a 2-mile moving window because of variations in the distance the impact was detected (Naugle et al. 2011*a*). We did not use the 120 × 120 m pixels for modeling because leks are a surrogate of habitat at a larger scale.

‡ Landfire vegetation groupings defined in Johnson et al. (2011) SAB.

§ Because climate grids of native resolution change at a 1-km scale and are highly spatially correlated, we did not resample the grids using a 6440-m moving window.

¶ NLCD urban development classes: developed, high intensity; developed low intensity; developed medium intensity; developed, open space; and NLCD impervious surfaces. The index also included roads (TIGER), oil and gas wells (compiled by each state), wind turbines (FCC obstruction database), transmission lines (Ventyx), and pipelines (Ventyx).

inference of nonlinear and spatial dynamics (Cutler et al. 2007, Olden et al. 2008, Evans et al. 2011, Baruch-Mordo et al. 2013). Random Forests uses multiple realizations of the data, with no distributional assumptions, that effectively converge on a stable estimate in very high-dimensional statistical spaces (Murphy et al. 2010, Evans et al. 2011). Model interpretation and inference were supported following the methods presented in Cutler et al. (2007), Murphy et al. (2010), and Evans et al. (2011). The expected complexity in interaction effects, potential latent variables, high spatial variability representing both global and local effects, and nonlinear relationships all support a nonlinear model such as Random Forests as an appropriate choice.

We modeled selection of breeding season habitat within the species range (Johnson 1980, Meyer and Thuiller 2006) using Random Forests, which is a bootstrapped classification and regression tree (CART) approach (Hastie et al. 2008). Random Forests is based on the principle of weak learning, where a set of weak subsample models converge on a stable global model. This method has been shown to provide stable estimates while being robust to many of the issues associated with spatial data (Cutler et al. 2007, Evans et al. 2011) such as autocorrelation and nonstationarity (i.e., nonconstant mean and variance). It also fits complex, nonlinear relationships, accounts for high-dimensional interaction effects, and accounts for hierarchically structured data inherent in nonstationary processes (Cutler et al. 2007, Evans et al. 2011). We expected both global trends in sage-grouse habitat selection and localized variation in habitat selection within each of the seven sage-grouse management zones. First- and second-order variations are addressed in the hierarchical nature of the iterative node partitioning, making this a good model to implement when global trend and local variations (Cressie 1991) are expected to occur in the same model (Evans et al. 2011). Analysis was conducted in program R (R Core Team 2012) using the rgdal (Bivand et al. 2013), sp (Bivand et al. 2008), and raster (Hijmans and Etten 2013) libraries to read spatial data, assign values from spatial covariates to the point observations of our dependent variable, and make spatial predictions. We used the implementation of Random Forests (Breiman 2001) in the R library Random Forest (Liaw and Wiener 2002) and followed the model selection method introduced in Murphy et al. (2010) using the rfUtilities library (Evans and Murphy 2014). Parsimony in Random Forests is important not only for producing a more interpretable model but also for reducing any fitting of the model to statistical noise, thus providing a better model fit (Murphy et al. 2010, Evans et al. 2011).

Evaluation of model fit and spatial predictions

To assess model fit, we used OOB (out-of-bag) error and confusion matrixes (Liaw and Wiener 2002). The OOB error represents the internal evaluation of global and class error against the withheld data from the bootstrap and represents an error distribution across all bootstrap replicates in the ensemble where the median error is used to represent the OOB error. We evaluated model stability and performance using cross-validation methods (Evans et al. 2011), where 10% of data were withheld from training the model and used as a validation data set. Overfitting was assessed by comparing error rates between OOB and cross-validation.

We also tested the sensitivity of the fitted model to errors in classification between used vs. available locations in the rfUtilities library (Evans and Murphy 2014) by randomly changing known lek locations to pseudoabsence points and evaluating cross-classification errors. We systematically changed known lek locations to zeros in 5% increments to understand the influence of pseudoabsence errors on overall error rates and model stability. This was performed because an unknown portion of our pseudoabsence locations were expected to fall within suitable sage-grouse breeding habitat. The primary motivation behind implementing a sensitivity test was to address model sensitivity to any lack of independence. A pseudoreplication problem would also affect the independence (correlation) of the bootstraps and potentially overfit the model. Because ensemble models are based on the premise of weak learning and variation in the bootstrap, if the data are homogenous, the bootstraps would not be independent and the ensemble would exhibit considerable correlation and effectively overfit the model. In evaluating model fit and convergence, we did not observe any indication of ensemble correlation. The sensitivity test allowed better understanding of overall error rates within our model, and more importantly, it allowed the assessment of model stability and prediction congruency across a range of lek locations that are misclassified as pseudoabsence.

Regional variation in habitat selection and disturbance thresholds

We used probability partial plots to elucidate habitat relationships of the modeled covariates after partialing out (holding constant) the other variables in the model. To improve interpretability, we plotted each given covariate for all management zones on the same plot. The probability partial plots were derived using the rfUtilities library (Evans and Murphy 2014).

Management zone VII

Management zone VII, while modeled, has a very small sample size (~0.3% of counted birds between 2010 and 2014) and only contains

652 km^2 of the 192,381 km^2 modeled breeding habitat (Table 5). Therefore, we did not focus on these results in the general manuscript or include MZ VII in figures highlighting functional habitat responses.

Breeding population index model

To map high-abundance population centers, we followed the methods and logic very similar to the models developed by the U.S. Geological Survey (USGS) for the Bi-State Distinct Population Segment of sage-grouse (Coates et al. 2015). Distribution models that combine information about habitat quality and abundance of sage-grouse from multiple data sources are valuable given recent intensification of sage-grouse management and policymaking (Coates et al. 2015). We modified their methods (Coates et al. 2015) to better represent a sage-grouse population index, because their original technique was developed to highlight management priority areas. Our final population index model incorporated two standardized kernel-based point density models, representing local and regional scales and our breeding habitat model described earlier. The results of our models are grids that represent an index to the relative amount of breeding birds for each 120 × 120 m area within each management zone. Our final population index model incorporates spatial patterns of sage-grouse habitat selection with contemporary information of abundance allowing the use of the available data, as proxies for management (Stephens et al. 2015). Population indices, such as ours, allow conservation actions to be targeted to the right landscapes, and help identify threats to a species that are occurring in areas that could impact large proportions of sage-grouse populations.

Kernel density function.—Kernel density functions have been commonly used in ecology to delineate home ranges of individual animals and to map concentrated areas of use by populations (Silverman 1986, Worton 1989). Within our study, we used the kernel density function to group cells of concentrated use by attributing count data to a grid placed over top of a sage-grouse management zone (Silverman 1986, Worton 1989). Using kernels to define population concentrations is consistent with past work defining core areas for sage-grouse (Doherty et al. 2011). We created two kernel models based on two separate bandwidth values (i.e., 6.4 and 18 km), which reflect published information on sage-grouse movement and seasonal space-use patterns. The 6.4-km bandwidth was chosen to correspond with utilization distribution of areas conducive for reproduction in relation to lek sites (e.g., breeding, nesting, brood-rearing), as demonstrated in populations at multiple sites (Holloran and Anderson 2005, Doherty et al. 2011, Coates et al. 2013). Although leking areas generally serve as hubs for nesting and are usually centered across seasonal areas (Coates et al. 2013), some sage-grouse move relatively long distances to access wintering areas (Fedy et al. 2012, Coates et al. 2013). Thus, we incorporated the larger spatial scale of 18 km to reflect these life history patterns (Fedy et al. 2012). Combining the scales appropriately placed greater emphasis on adjacent areas, thus preventing oversmoothing, but still allowed for the representation of sage-grouse occurrence at further distances. We used SAGA-GIS version 2.1.0 (SAGA-GIS 2015) to create two Gaussian kernel density functions. The same set of active lek locations from our habitat model defined the point density for our kernel models, and each point was weighted by the mean peak count of displaying sage-grouse from 2010 to 2014. Following the logic of Coates et al. (2014), we standardized each kernel using a row standardization. We then added each grid together and divided by 2, using the raster library (Hijmans and Etten 2013) in R. The output is a 120 × 120 m raster that represents a multiscale density process of sage-grouse lek counts across two biologically meaningful scales (Eq. 1).

$$\text{Kernel Index} = (\text{standardized 6.4-km kernel} + \text{standardized 18-km kernel})/2 \quad (1)$$

Population index.—Our Kernel Index summarizes the best available information on the relative density of birds across the entire sage-grouse range. We selected bandwidths to correspond with linear movement distance of sage-grouse within the breeding season (Doherty et al. 2011), as well as movements between breeding and other seasonal habitats (Fedy et al. 2012). We believe that the combination of both kernels into a single Kernel Index represents ecologically meaningful areas for sage-grouse. However, kernel functions are inherently an estimator of the spatial point density process, and thus, they are not explicitly linked to habitat features.

We wanted to create a population index to further refine our Kernel Index. First, we wanted a method that would reduce the importance of lands with low probabilities of being habitat based upon known sage-grouse habitat relationships. Secondly, we wanted to increase the value of lands with high probabilities of being occupied habitat, but further away from known leks, thus having lower value in the Kernel Index. We did this by multiplying the Kernel Index by the probability of our breeding habitat model (Eq. 2).

$$\text{Population index} = (\text{Kernel Index} \times \text{breeding habitat model}) \tag{2}$$

Highest population index values arise where high breeding habitat probabilities co-occur with landscapes having higher lek counts. The use of this equation also effectively reduces the value of landscapes near larger sage-grouse leks, which are effectively nonhabitat based upon the prediction of the breeding habitat model. Lastly, multiplying the Kernel Index by the breeding habitat model increases the value of lands further from known sage-grouse leks that have high probabilities of containing breeding sage-grouse. We thought that this was important because our data set utilized all known sage-grouse population survey data across their range; however, our survey data do not represent all leks.

Aggregation using population index volumes

We ordered all population index values from each grid cell within a management zone from the highest to lowest density. We selected the highest density cells in order until they summed to 10% of the total population index within a management zone. We repeated the selection process in 10% increments selecting the highest remaining grid cell densities first until we had 10 bins (i.e., highest density bin represented the top 10% of the population, 100% bin representing all breeding areas identified in modeling). Results are cumulative, such that all bins contain all preceding bins of 10% increments. We then calculated the percentage of the occupied distribution within each incremental 10% population bin.

Results

Breeding habitat model

On average, our breeding habitat model correctly classified 82.0% (range: 75.4–88.0%) of hold-out data from OOB samples (Table 3). Our models also correctly classified independent *K*-fold hold-out data (mean across management zones = 80.9%; range: 75.0–85.8%) (Table 3). General agreement between OOB error rates and *K*-fold cross-validation indicates stability in our model to predict independent data and lack of overfitting (Table 3). We documented higher error rates within pseudoabsence classes compared with our active lek class (Table 4); however, simulations indicated that estimates were stable across a wide range of pseudoabsence errors (0–30% simulated errors in 5% increments). For example, the mean SE across the seven management zones with 20% simulated pseudoabsence errors is 0.032. Low SE indicates model stability and the ability of the Random Forests to predict through statistical noise arising from points that were modeled as absences, which in fact supports lek formation (i.e., false absence). We documented a ~3% error increase for every 5% increase in false absences.

Table 3. Percentage of *K*-fold cross-validation hold-out data set locations (10%) that were correctly classified by a model built with 90% of the data set.

Management zone	1—out-of-bag error	*K*-fold cross-validation % correctly classified
MZ I—Northern Great Plains	76.3	75.9
MZ II—Wyoming Basin	75.4	75.0
MZ III—Southern Great Basin	85.9	85.3
MZ IV—Snake River Plain	83.9	83.6
MZ V—Northern Great Basin	76.3	75.1
MZ VI—Columbia Basin	88.0	85.8
MZ VII—Colorado Plateau	88.0	85.4
Average	82.0	80.9

Note: These results are compared with internal model fit statistics generated via bootstrap resampling (1—out-of-bag error bootstrap error rates).

Table 4. Classification confusion error rates for leks and pseudoabsence locations.

Management zone	Pseudoabsence (%)	Leks (%)
MZ I—Northern Great Plains	29.8	16.9
MZ II—Wyoming Basin	32.7	16.5
MZ III—Southern Great Basin	18.0	9.9
MZ IV—Snake River Plain	21.0	11.3
MZ V—Northern Great Basin	29.4	19.7
MZ VI—Columbia Basin	12.0	12.0
MZ VII—Colorado Plateau	12.0	10.0

Notes: Error rates were generated from bootstrap resampling. Across management zones, there was a general pattern of higher errors in the pseudoabsence class, with the exception of the two smallest management zones, the Columbia Basin and the Colorado Plateau.

Models demonstrate that breeding habitat is highly condensed within the current occupied range of sage-grouse (Fig. 2). All currently active leks occurred on probabilities >0.65; we therefore used this threshold to quantify the amount of breeding habitat. When we use this threshold value, 26% of the current occupied range is predicted to be breeding habitat (Table 5, Fig. 2). Across the range of sage-grouse, general habitat variables and climatic gradient variables had greater importance than disturbance variables in predicting occupied breeding habitat (Table 6; Appendix S2). Not surprisingly, a positive association with the percentage of a landscape dominated by sagebrush within 130.1 km^2 (50.24 $mile^2$; 32,153 acres) was the top variable in four of the seven models and was in the top five variables for all models (Table 6). We documented variation in habitat selection for sagebrush but also show similar patterns across the range (Fig. 3). However, functional habitat selection for sagebrush modeled for the Northern Great Plains and Columbia Basin management zones diverged from results for the rest of the management zones, because sage-grouse were modeled to occupy habitats

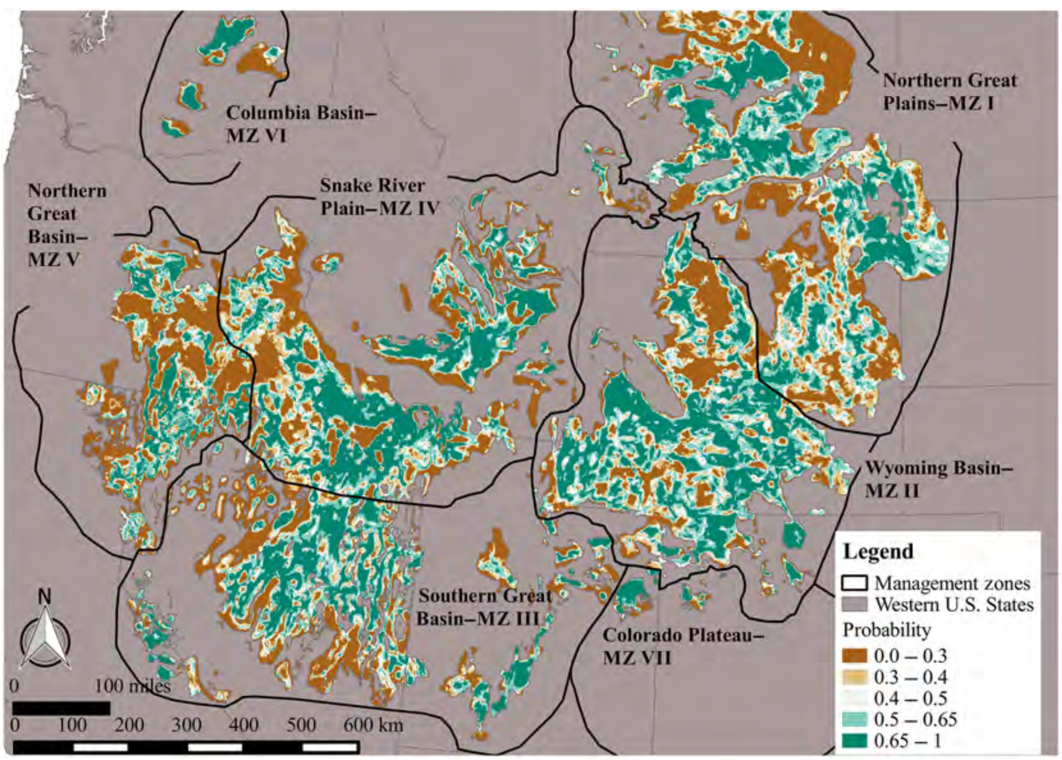

Fig. 2. Breeding habitat model of Greater Sage-Grouse developed within each of the seven management zones. The breeding habitat model is a spatially explicit probability prediction that the surrounding landscape will contain enough breeding habitat to support Greater Sage-Grouse lek formation. All active leks within the sage-grouse range (2010–2014) occurred on probabilities >0.65.

Table 5. Area (km^2) of occupied range (Schroeder et al. 2004) and modeled breeding habitat across the Greater Sage-Grouse range in North America.

Management zone	Occupied range	Modeled breeding habitat	Percentage of occupied range
MZ I—Northern Great Plains†	186,480	41,731	22
MZ II—Wyoming Basin	149,820	48,189	32
MZ III—Southern Great Basin	124,057	36,629	30
MZ IV—Snake River Plain	156,360	46,700	30
MZ V—Northern Great Basin	78,293	14,018	18
MZ VI—Columbia Basin	11,161	4462	40
MZ VII—Colorado Plateau	4777	652	14
Rangewide†	710,948	192,381	26

Note: Breeding habitat probabilities were calculated using a 0.65 threshold, because all current active leks had a probability >0.65.

† Does not include the Canadian portion of the range.

with lower proportions of sagebrush in zones I and VI (Fig. 3). All sage-grouse breeding habitats showed strong avoidance of tree cover; however, strength of avoidance varied between management zones (Fig. 4). The human disturbance index was selected within models for all management zones except the Northern Great Basin with a variable importance range (0.48 for Colorado Plateau to 0.09 for Southern Great Basin, Table 6; Appendix S2). While threshold values between management zones varied similar to tree canopy cover, models documented clear thresholds in amount of landscape-level disturbance tolerated and exhibited the sharpest declines in probability distributions once thresholds were crossed (Fig. 5). Models showed that Northern Great Plains management zone had the lowest threshold for the human disturbance index (2.9% when P ~ 0.65, Fig. 5). Models also documented variability and

Table 6. Top five variables and their importance values selected for each management zone from 2010 to 2014.

Management zone	First variable	Second variable	Third variable	Fourth variable	Fifth variable
Northern Great Plains	Canopy cover	All sagebrush	Roughness	Topographic wetness index	Gross primary production
(I)	1.00	0.63	0.57	0.55	0.45
Wyoming Basin	All sagebrush	Canopy cover	Annual drought index	Degree days > 5°C	Mean annual precipitation
(II)	1.00	0.73	0.68	0.59	0.49
Southern Great Basin	All sagebrush	Degree days > 5°C	Elevation	Annual drought index	Canopy cover
(III)	1.00	0.79	0.70	0.54	0.48
Snake River Plain	Canopy cover	Annual drought index	All sagebrush	Degree days > 5°C	Gross primary production
(IV)	1.00	0.60	0.59	0.51	0.50
Northern Great Basin	All sagebrush	Annual drought index	Low sagebrush	Mean annual precipitation	Degree days > 5°C
(V)	1.00	0.96	0.91	0.79	0.65
Columbia Basin	Elevation	Degree days > 5°C	Grassland/ herbaceous	Annual drought index	All sagebrush
(VI)	1.00	0.42	0.41	0.27	0.22
Colorado Plateau	All sagebrush	Low sagebrush	Human disturbance index	Oil and gas wells	
(VII)	1.00	0.67	0.48	0.40	

Notes: Importance values are scaled by management zone, so that the top variable equals 1 and the remaining variables are a proportion derived by dividing by the top variable, and are derived from probability-scaled partial plots in the Random Forest package in R. Variable importance values for the remaining retained variables (6th to 10th) are in Appendix S2 and, in some cases, explain similar amounts of variation as the fifth variable.

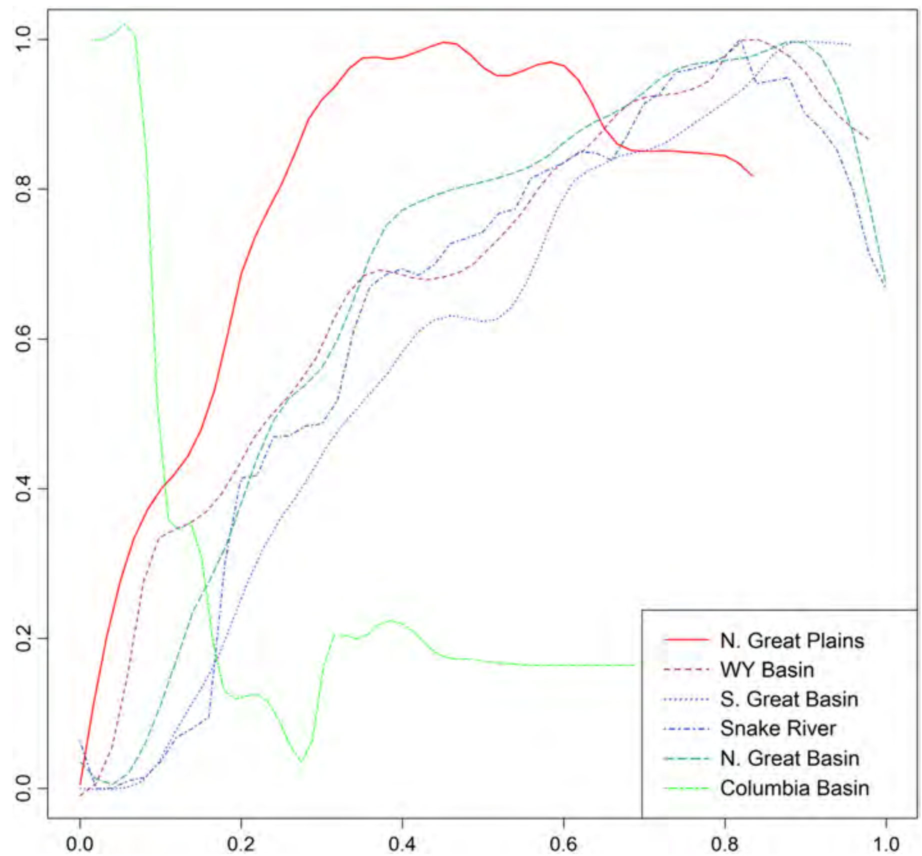

Fig. 3. Functional habitat response between the percentage of all sagebrush cover types (*x*-axis) within a 6.4-km buffer (130.1 km^2) and the probability (*y*-axis) a landscape will contain enough breeding habitat to support Greater Sage-Grouse lek formation within each management zone (2010–2014). Functional response curves were generated using partial probability plots to explore the influence of a given variable on the probability of occurrence while partialing out the average effects of all other variables in the final model.

differences in threshold values for the amount of tillage in the landscape, with sage-grouse in management zone I showing the least tolerance for tilled landscapes (Fig. 6). Despite variability in disturbance and nonhabitat thresholds, we found similar patterns in the peaks of probability distributions ($P > 0.8$) for our two strongest historic climatic predictors (annual drought index [Fig. 7] and degree days > 5°C [Fig. 8]). A current measure of climate as measured by gross primary production had lower variable importance than our historic climate envelopes in model selection (Table 6). We documented similar patterns of selection for gross primary production, although peaks varied across the range with the lowest selected range of gross primary production in the Northern Great Basin and the highest in the Northern Great Plains (Fig. 9).

Breeding population index model

We demonstrate distinct clustering in the relative abundance of sage-grouse populations within each management zone (Figs. 10 and 11). On average, approximately half of the breeding population is predicted to be within 10% of the occupied range. Across all management zones, all populations visually demonstrated asymptotic properties between each additional 10% of the population and the area required to contain those populations (Fig. 11). For example, to go from 80% of the population index to 90%, increased the area required by 44% on average (range: 41% MZ II to 50% MZ I; Fig. 11).

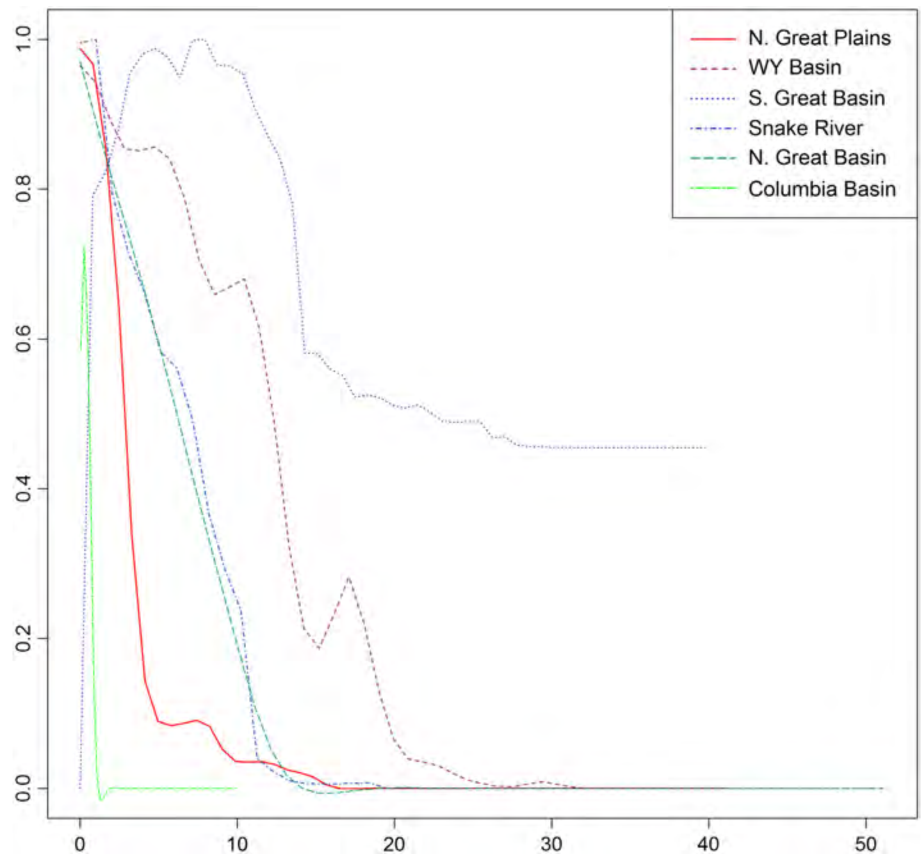

Fig. 4. Functional habitat response between tree canopy cover (x-axis) within a 6.4-km buffer (130.1 km^2) and the probability (y-axis) a landscape will contain enough breeding habitat to support Greater Sage-Grouse lek formation within each management zone (2010–2014). Functional response curves were generated using partial probability plots to explore the influence of a given variable on the probability of occurrence while partialing out the average effects of all other variables in the final model.

Discussion

Clustering of populations is a common ecological phenomenon (e.g., Brown et al. 1995, Murphy et al. 2006). Knowledge of these high-value areas can direct management actions to landscapes where they will have the largest benefit to regional populations (Sanderson et al. 2002, Groves 2003). We documented pronounced clustering in the relative abundance of sage-grouse populations within each management zone (Figs. 10 and 11), consistent with past work at regional (Doherty et al. 2010*a*, Coates et al. 2014), state (Fedy et al. 2014), and local scales (Aldridge and Boyce 2007, Doherty et al. 2010*b*). Our results indicate that approximately half of the breeding population is within ~10% of the range. We also found that 80% of sage-grouse populations were contained in 25–34% of the occupied range within each management zone. Across all management zones, all populations showed an exponential increase in the area required to contain each additional 10% of the population (Fig. 11). Because sage-grouse exhibit markedly clustered populations, if landscape-level risks occur in high-density areas they could negatively affect large proportions of the populations. Conversely, focusing conservation efforts into landscapes that contain higher proportions of birds may demonstrate substantially higher biological returns for conservation investments of similar acreages. Therefore we suggest that, birds, not acres or dollars spent, would be the best currency in conservation plans, because identical acreages of

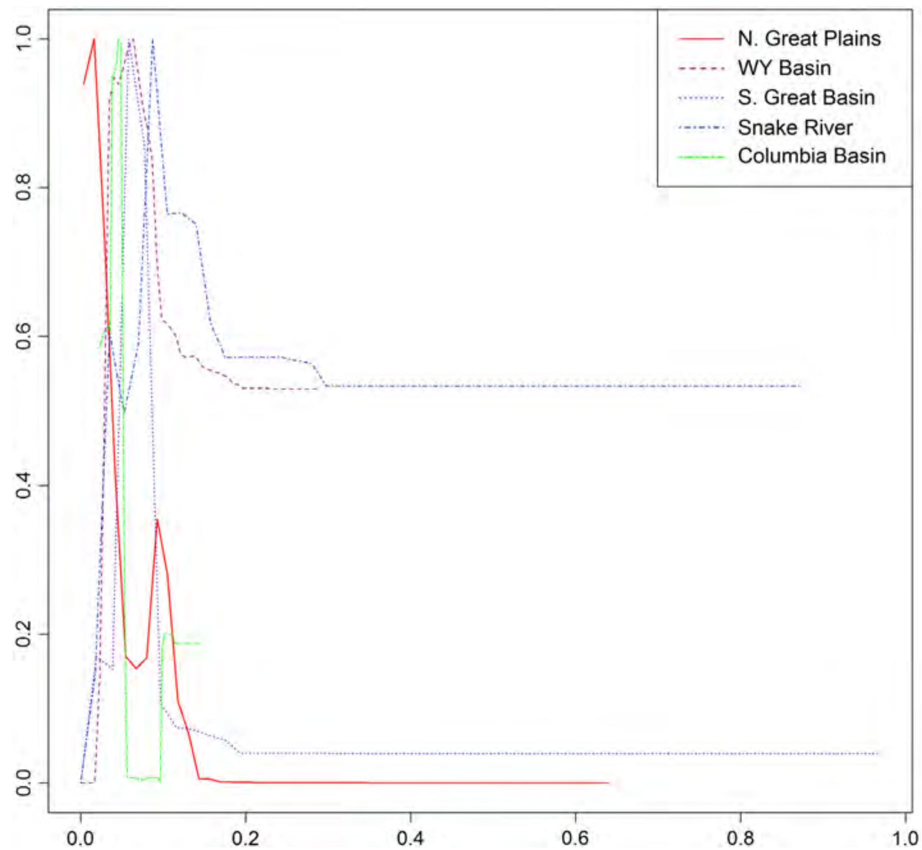

Fig. 5. Functional habitat response between the amount of human disturbance index (*x*-axis) within a 6.4-km buffer (130.1 km^2) and the probability (*y*-axis) a landscape will contain enough breeding habitat to support Greater Sage-Grouse lek formation within each management zone (2010–2014). Functional response curves were generated using partial probability plots to explore the influence of a given variable on the probability of occurrence while partialing out the average effects of all other variables in the final model.

conservation actions can overlap with vastly different numbers of sage-grouse. Our population index models can be used to quantify the relative percentage of sage-grouse populations that overlap management areas, providing regional population context to decisions and the relative ranking of landscape importance for sage-grouse. Simple spatial overlap analyses using our model are a first step in bringing context to the potential population-level effects of both deleterious and beneficial management decisions.

A trade-off exists between model prediction and generalized biological understanding when selecting the appropriate spatial extent for the development of RSF models (Elith and Leathwick 2009). Reducing extent can increase model accuracy (Fedy et al. 2014), but at the cost of generalizability as the models explain variation over a smaller parameter space. Careful thought must be given to study objectives. Sage-grouse management zones are based on unique floristic provinces (Stiver et al. 2006, Appendix S1). The management zone extent represents a good trade-off for our goals because this extent allowed generalized broad-scale biological understanding across far-reaching extents and still retained high spatial predictive capabilities within management zones.

The desired geographic scale of understanding is paramount in studies aimed at obtaining inference on selection behavior. Our study was specifically designed to assess first-order selection of sage-grouse seasonal home ranges (Johnson 1980, Meyer and Thuiller 2006). The rationale for

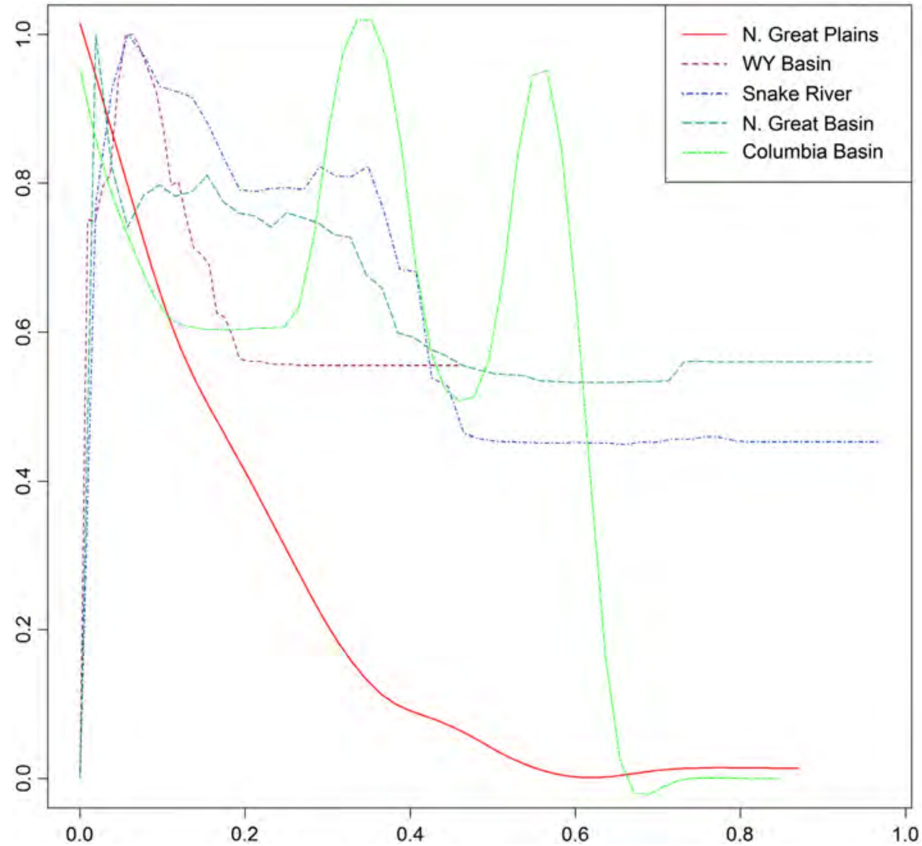

Fig. 6. Functional habitat response between the amount of tilled cropland (*x*-axis) within a 6.4-km buffer (130.1 km^2) and the probability (*y*-axis) a landscape will contain enough breeding habitat to support Greater Sage-Grouse lek formation within each management zone (2010–2014). Functional response curves were generated using partial probability plots to explore the influence of a given variable on the probability of occurrence while partialing out the average effects of all other variables in the final model.

using the first-order scale was twofold: (1) The primary objective was to develop population and habitat models that account for regional variation within each sage-grouse management zone and (2) broad-scale lek data represent locations of populations and are not adequate to appropriately model second- or third-order habitat selection (Johnson 1980, Meyer and Thuiller 2006). Lower orders of habitat selection are generally derived from finer-scale telemetry data at the individual level. Using first-order assessments here that produce a relative probability for each 120 × 120 m grid cell across the range of the species allows for later integration with other research at finer scales (e.g., second to fourth orders). We believe that investigating first-order habitat selection across the entire sage-grouse range is important, because understanding landscape context can elucidate why the results of second- and third-order habitat selection studies can seemingly give conflicting results and varying thresholds, even for well-studied topics (Donovan et al. 1997).

While generating biological insight into species habitat selection is obviously important, one can argue that measures of model prediction and stability are even more important for conservation planning and risk analyses, especially when they may be utilized by agencies to spatially assess species risk, delineate priority areas, or direct resource allocation (see review in Elith and Leathwick 2009). Our models demonstrated high statistical model fit and demonstrated stability to withheld data (Tables 3 and 4). On average, our

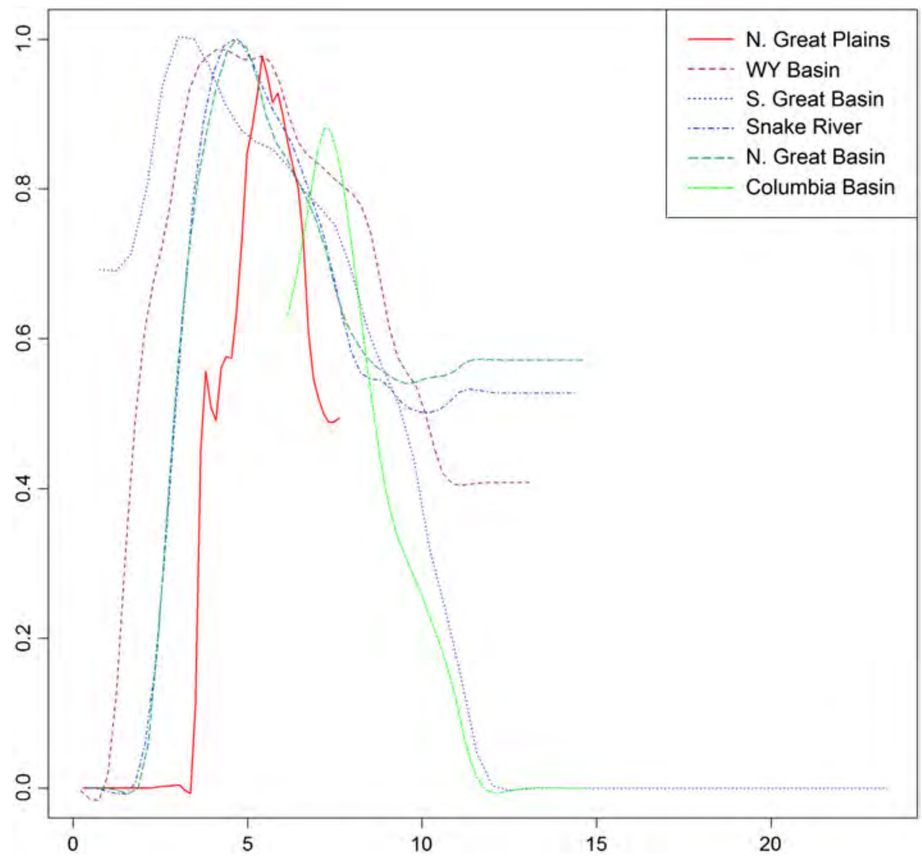

Fig. 7. Functional habitat response between the average annual drought index (*x*-axis) within a 6.4-km buffer (130.1 km^2) and the probability (*y*-axis) a landscape will contain enough breeding habitat to support Greater Sage-Grouse lek formation within each management zone (2010–2014). Functional response curves were generated using partial probability plots to explore the influence of a given variable on the probability of occurrence while partialing out the average effects of all other variables in the final model.

breeding habitat model correctly classified 82.0% (range: 75.4–88.0%) of hold-out data from OOB bootstrap samples and also correctly classified independent *K*-fold hold-out data (mean across management zones = 80.9%; range: 75.0–85.8%, Table 3). General agreement between OOB error rates and *K*-fold cross-validation indicates stability in our model to predict independent data and lack of overfitting (Shmueli 2010). Demonstrated model fit and validations were important, as this modeling effort was directly intended to assess sage-grouse spatial overlap with landscape-level risks.

Sage-grouse are a unique species in wildlife management as we have broad-scale population surveys across the species range that follow a common survey protocol (i.e., lek counts, Connelly et al. 2000). Additionally, although birds require unique habitat components throughout their annual cycle, they do not migrate long distances and, with the exception of peripheral populations in Alberta and Saskatchewan, do not cross international borders. These characteristics simplify many management strategies and facilitate consistency in survey protocols that allowed research into regional variation in functional responses. Additionally, broad-scale population data facilitate the development of integrative methodologies to create composite spatially explicit indices that reflect demographic and habitat information within this study and others (Coates et al. 2014). Indices such as these—particularly those that can be predicted spatially—can help guide

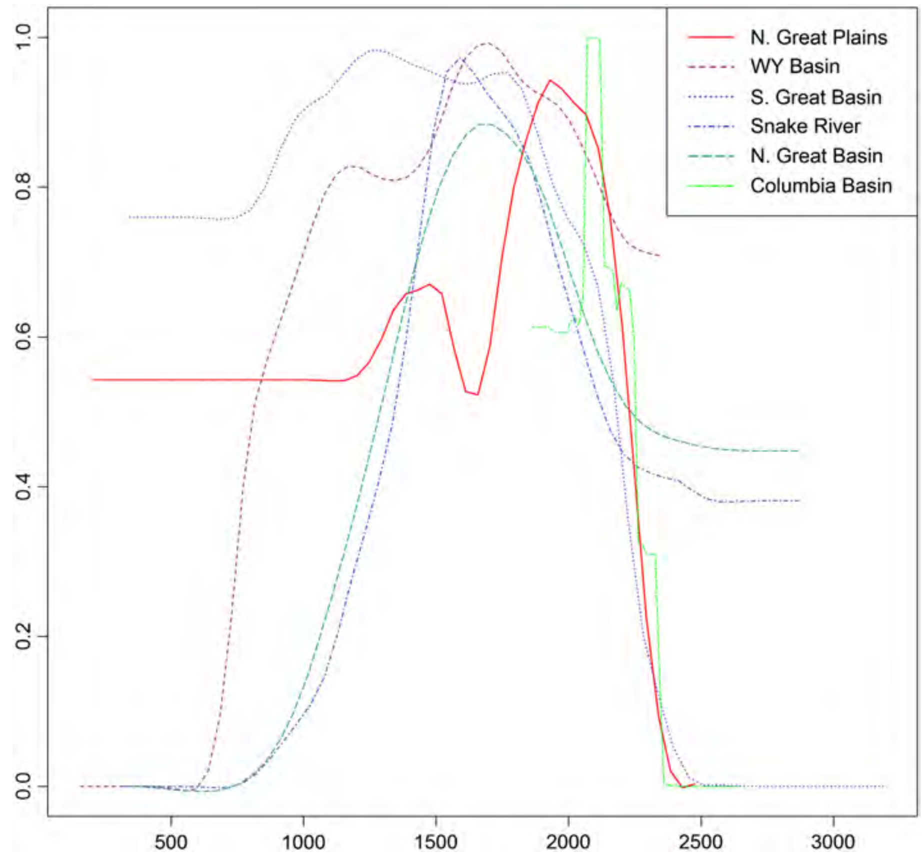

Fig. 8. Functional habitat response between the average degree day 5°C (*x*-axis) within a 6.4-km buffer (130.1 km^2) and the probability (*y*-axis) a landscape will contain enough breeding habitat to support Greater Sage-Grouse lek formation within each management zone (2010–2014). Functional response curves were generated using partial probability plots to explore the influence of a given variable on the probability of occurrence while partialing out the average effects of all other variables in the final model.

landscape-level conservation actions (Stephens et al. 2015). However, in the context of our models, broad-scale lek data are inadequate for modeling second- or third-order habitat selection (Johnson 1980, Meyer and Thuiller 2006), which are known to be important determinants of sage-grouse habitat selection (Connelly et al. 2000). Past research has used lek data as an independent data source to validate landscape-level spatial predictions of second- and third-order habitat selection models generated from telemetry data in both Greater Sage-Grouse (Doherty et al. 2010*b*, Fedy et al. 2014) and Gunnison Sage-Grouse (Aldridge et al. 2012). Thus, first-order habitat selection models will give regional context to priority breeding areas, but should not be viewed as prescriptive at the site level. It should be expected that some priority areas identified at the first-order scale will lack appropriate habitats at the second- or third-order scale and therefore may be unoccupied. Site-scale recommendations will require input from local biologist as well as finer-resolution data (e.g., telemetry data, GPS movements, soil types, local vegetation).

Lek data seem to represent the overall spatial process of relative abundance for sage-grouse, particularly in recent years due to the dramatic increases in survey effort over the last decade (WAFWA 2015). However, sage-grouse lek surveys follow a common survey protocol; they do not follow a statistical design. A design-based survey with a dual-frame sampling protocol (Haines and Pollock 1998, Royle et al. 2005) would strengthen analyses allowing estimation

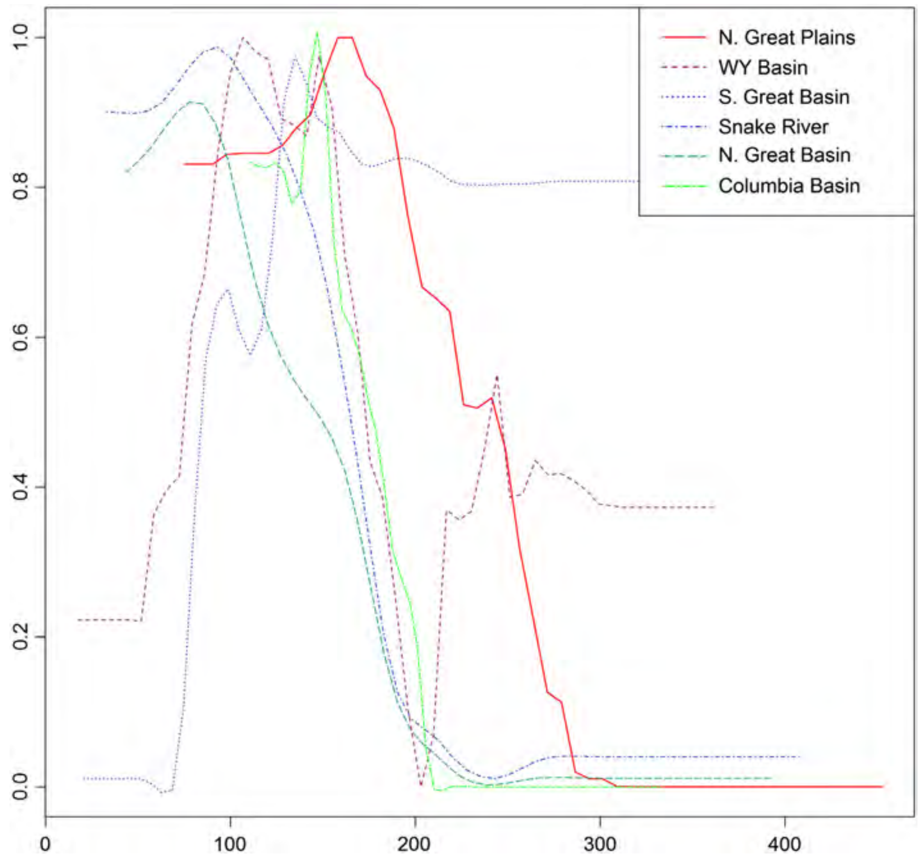

Fig. 9. Functional habitat response between the gross primary production (*x*-axis) within a 6.4-km buffer (130.1 km^2) and the probability (*y*-axis) a landscape will contain enough breeding habitat to support Greater Sage-Grouse lek formation within each management zone (2010–2014). Functional response curves were generated using partial probability plots to explore the influence of a given variable on the probability of occurrence while partialing out the average effects of all other variables in the final model.

and understanding of implications of currently inestimable parameters, such as the effects of sampling variation and detection probabilities on count estimates, or the proportion of leks surveyed each year (Blomberg et al. 2013*a*). The latter parameter is one of the most important breakthroughs, because it could allow more robust estimates of population size with associated variances, vs. the reasonable, but ad hoc approaches used to generate current minimum population estimates (WAFWA 2015). A recent example of a statistically rigorous framework for estimating populations in a similar species, lesser prairie chickens (*Tympanuchus pallidicinctus*), could provide guidance for such an approach in sage-grouse (McDonald et al. 2014). Ultimately, the above limitations affect the scale of inference. Models developed using these data should be viewed as regional indices for conservation planning and risk assessment. Because our goal was to provide regional context and relative ranking of landscape importance for sage-grouse, the use of lek count data was appropriate.

Landscapes are seldom homogeneous across large extents. Thus, landscape-scale modeling is important to understand how functional responses vary for wide-ranging species. Variation in functional response to particular habitat components has been documented in ungulates (Godvik et al. 2009, Herfindal et al. 2009, Moreau et al. 2012, Beyer et al. 2013), wolves (Hebblewhite and Merrill 2008, Houle et al. 2010, Matthiopoulos et al. 2011), and other large mammals (Gillies et al. 2006, Roever et al. 2012).

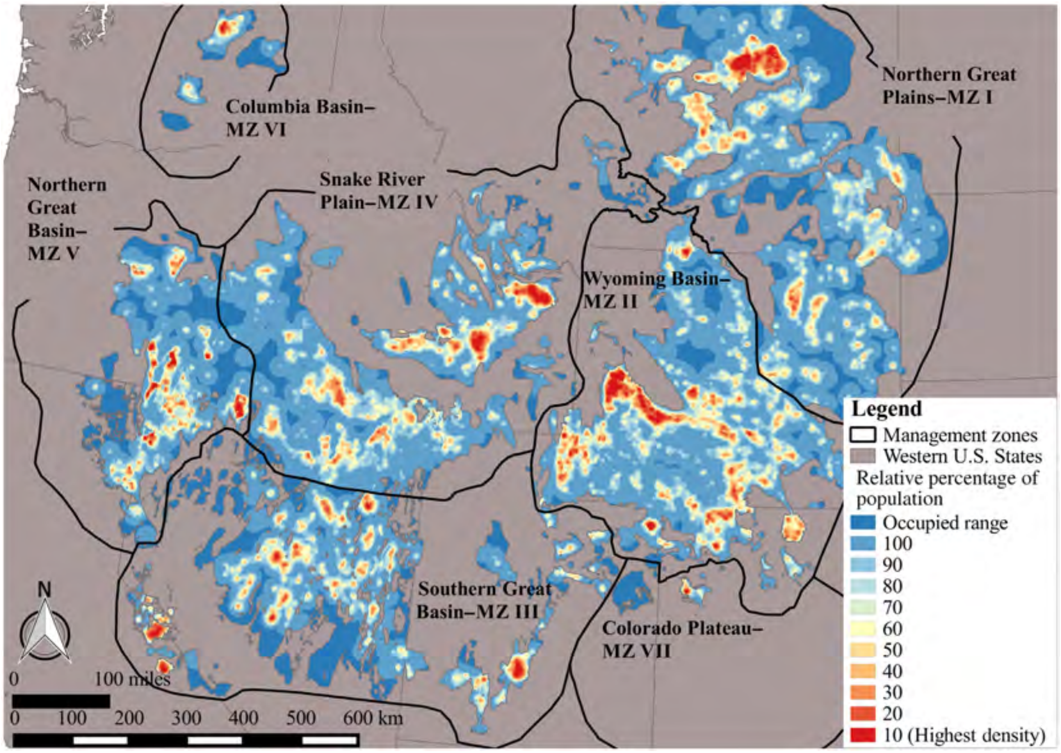

Fig. 10. Breeding population index model of Greater Sage-Grouse within each of the seven management zones. Our population index model provides spatial insight into the relative importance of specific areas to the overall management zonewide breeding abundance of Greater Sage-Grouse during 2010–2014. Population index values are relative within each management zone. Sage-grouse population index areas represent spatial locations of the known breeding population in 10% bins differentiated by color. The darkest red areas contain 10% of the breeding population in the smallest area. Because bins are additive, red and orange hue areas combined capture 50% of the population, etc.

Typically, these studies focused on the spatial scales of inter-home range variation within one to several subpopulations (Mysterud and Ims 1998, Gillies et al. 2006, Hebblewhite and Merrill 2008, Herfindal et al. 2009, Houle et al. 2010). Some studies have moved further along the hierarchical order of habitat selection (Johnson 1980) and focused on within-home range variation in functional response (Houle et al. 2010, Moreau et al. 2012). However, almost all these studies were driven by high-input and high-detail Global Positioning System (GPS) radiotelemetry data, with few exceptions (Mysterud and Ims 1998, Herfindal et al. 2009). We demonstrate that less detailed data (i.e., lek survey data), collected across large extents (i.e., the entire species range), can also highlight the regional variation in functional responses. Our analyses clearly highlight that understanding regional variation in habitat selection is critical to designing effective conservation plans for sage-grouse.

We documented variability in sage-grouse functional response to sagebrush across the range (Fig. 3). Not surprisingly, sage-grouse showed strong selection for landscape-level sagebrush with the exception of the Columbia Basin (see variable justification in Table 2 and Fig. 3). While landscape-scale extents differed, our sagebrush functional response curves broadly agreed with other landscape-level assessments, which recommended >50% sagebrush cover (Wisdom et al. 2011) and >65% sagebrush cover (Aldridge et al. 2008). More recent analyses documented that 90% of the active leks in the western range of sage-grouse occurred in landscapes with at least 40% sagebrush (Knick et al. 2013). All active

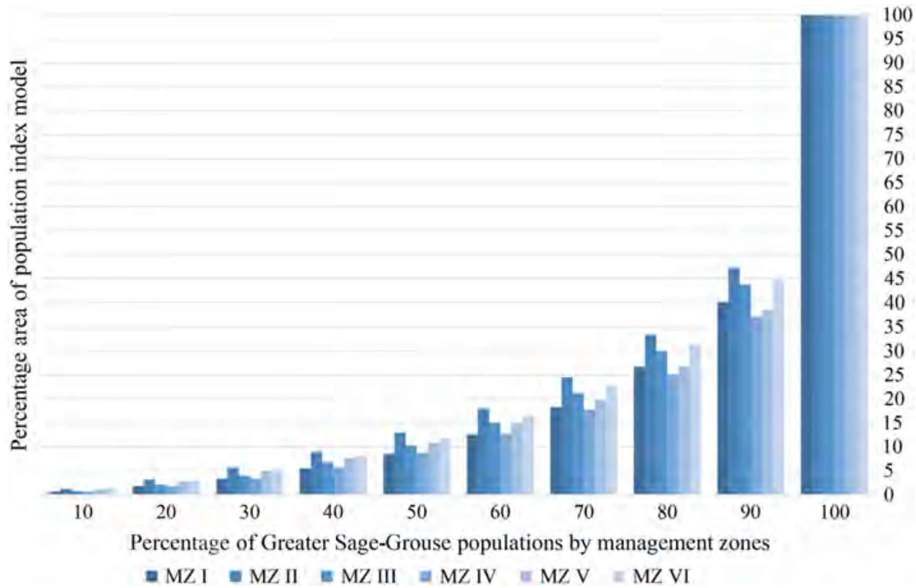

Fig. 11. The percentage of the population index model and resulting percentage area of the entire population index model by management zones during 2010–2014.

leks within our habitat selection model had predicted occupancy probabilities >0.65. When the probability of occurrence crossed this threshold, we documented sagebrush cover ranging from ~35% (Wyoming Basin) to ~55% (Southern Great Basin) (Fig. 3). These threshold values are lower than previously documented; however, contemporary thresholds of occurrence could be lower than recommendations for rangewide averaged responses for long-term persistence (Aldridge et al. 2008, Wisdom et al. 2011). Additionally, our threshold estimates are similar to estimates of ecological minimum requirements (Knick et al. 2013). These past analyses averaged habitat or population responses of sage-grouse at rangewide (Aldridge et al. 2008, Wisdom et al. 2011) or the entire western-range (MZ III–VI) extents (Knick et al. 2013). Generating averaged functional habitat response across very large extents clearly furthers biological understanding, yet by design they cannot elucidate regional differences in functional responses if they exist. We also documented divergence in the Northern Great Plains, which crossed the probability of occurrence threshold at ~20% sagebrush cover (Fig. 3). We showed variation in functional responses across a wide range of variables within all management zones; however, the Columbia Basin and Northern Great Plains consistently showed the greatest divergence in functional habitat selection. Spatial interpolation or extrapolation of habitat selection models is most accurate when the availability of habitats is approximately the same in the novel areas (Mladenoff et al. 1999, Aarts et al. 2008) because of the functional response in habitat use (Mysterud and Ims 1998). Similarly, models averaging habitat responses across the range of sage-grouse are likely to misclassify important habitats when landscape context diverges from rangewide averaged habitat conditions such as the Columbia Basin and Northern Great Plains management zones.

Parallel to many other recent studies at landscape scales (Aldridge and Boyce 2007, Doherty et al. 2008, Knick et al. 2013, Coates et al. 2014, Fedy et al. 2014), our research also suggests that sage-grouse occupancy is more complicated than just sagebrush abundance; other core environmental conditions must be met for sage-grouse landscape occupancy. We also documented variability in thresholds of disturbance factors (i.e., tillage, conifer, human disturbance index) across the range of sage-grouse (Figs. 4–6). Sage-grouse are well known to avoid human disturbances (e.g., Naugle et al. 2011*b*) and other nonhabitat features such as conifers encroaching into sagebrush-dominated landscapes (Baruch-Mordo et al. 2013). We showed strong negative relationships between sage-grouse occurrence and tree canopy cover across each management zone, which was consistent with the highest resolution and definitive study on the effects of conifer encroachment (Baruch-Mordo et al. 2013). Yet, our results indicate that caution should be used when extrapolating results from small portions of the Northern Great Basin to other sage-grouse population across the range, because of regional variation in functional responses to tree cover. Sage-grouse populations in the Northern Great Plains and the Columbia Basin exhibited more pronounced avoidance of tree canopy cover than in the Northern Great Basin, while the more contiguous habitats of the Wyoming Basin and Southern Great Basin demonstrated more tolerance (Fig. 4). The human disturbance index was included in every management zone but one; however, human disturbance was less important in explaining occurrence than landscape-level sagebrush or climatic envelope variables (Table 6; Appendix S2). However, the human disturbance index exhibited the sharpest declines in probability distributions once thresholds were crossed, suggesting important tipping points for human disturbance in proximity to leks (Fig. 5). The variation in the functional response to human disturbance was substantial among management zones, demonstrating that a one-size-fits-all approach to acceptable disturbance thresholds around leks should exercise precaution and target the lowest threshold, or potentially adjust regionally. Avoidance of areas with relatively small amounts of human disturbance is consistent with past research (Knick et al. 2013); however, direct comparison of rates across studies is not possible as the suite of variables and the scale at which they were quantified differed between studies.

If habitat fragmentation is a key determinant of where thresholds occur, we would expect to see habitat thresholds occurring earliest in landscapes with the highest levels of fragmentation (Hill and Caswell 1999, Fahrig 2003). Consistently across the range, the two most fragmented populations with the highest amounts of agriculture (Northern Great Plains and Columbian Basin; Appendix S1) had the lowest tolerance to human disturbance (Fig. 5). Increased impact of disturbance within fragmented habitats was also documented in relation to oil and gas development across the state of Wyoming. Sage-grouse within the more fragmented habitats of northern Wyoming (Great Plains management zone) showed increased population-level impacts within the same oil and gas development density categories, compared with the more contiguous habitats of southern Wyoming (Wyoming Basin management zone) (Doherty et al. 2010*a*). Variation in disturbance thresholds is also known to vary with habitat quality. For example, wolves (*Canis lupis*) in the boreal forest avoided anthropogenic development as disturbance densities increased (Lesmerises et al. 2012, Ehlers et al. 2014), but showed more tolerance of disturbance in high-quality prey habitats (Lesmerises et al. 2012). Further, habitat quality is more than simply food availability. For example, mule deer (*Odocoileus hemionus*) within the Piceance and Upper Green River Basins showed avoidance of oil and gas development; however, effect sizes were larger in the Upper Green River Basin (Sawyer et al. 2006, Northrup et al. 2015). The authors hypothesized that the more rugged areas of the Piceance Basin provided more security cover than the flatter areas of Upper Green River Basin (Northrup et al. 2015). This hypothesis was generated by observing behavior differences in which mule deer showed less avoidance of infrastructure when they had the security cover of darkness at night (Northrup et al. 2015). It is likely that the variation in sage-grouse response to disturbance observed in this study is influenced by mechanisms related to fragmentation, habitat quality, or others. However, the finer-scale data to test each of these hypotheses are not available rangewide. Regardless, the relevant point is that understanding variation in habitat selection and disturbance thresholds across large spatial extents is necessary to inform land-use management decisions that try to balance trade-offs among competing interests.

Management implications

Our work is an improvement over past rangewide population models (Doherty et al. 2011) because it represents a comprehensive integration of both habitat and population information at a rangewide scale for sage-grouse while accounting for regional variation in habitat selection and bird densities. Our models can serve as a consistent currency to assess the overlap of sage-grouse habitats (Fig. 2) and populations (Fig. 10) with conservation actions or threats.

We also document the importance of regional variation in habitat selection and varying thresholds in response to disturbance across the range. Partial probability plots highlight how ecological gradients (Appendix S1) across the range (Table 5) can change functional habitat responses and ultimately predictions of breeding habitat. Our work highlights the need for careful consideration when extrapolating results of studies in one management zone into others, especially if they have vastly different ecological context. Our study extent was rangewide and addressed first-order selection of habitats within management zones. Thus, our results should apply to questions and management at that scale. However, management may require actions at smaller scales of selection, and we caution against implementing smaller-scale actions based on the results presented here. Our results suggest that multiscale (first- to third-order) and cumulative effects should be investigated simultaneously in future research.

The complexities of ecological context fundamentally influence how species respond to other components in the system. In other words, where you draw your study boundaries fundamentally determines what you learn about the ecology of the species. We show this is true, even for a species as specialized as sage-grouse. Often, models do not encompass the entire range of a focal species, and therefore, biological relationships or thresholds of disturbance are extrapolated to novel areas not included in the development of the models. Extrapolation of results into novel landscapes is often required as managers are mandated by law to make decisions based upon the best available scientific information. Unfortunately, setting conservation targets based upon thresholds defined in other regions is precarious (Rhodes et al. 2008) because thresholds can vary tremendously across species and landscapes (van der Hoek et al. 2015). For example, our results indicate that at the first-order level, disturbance thresholds defined in the Great Basin management zones would likely exceed sage-grouse occupancy requirements if extrapolated to the Great Plains and Columbia Basin management zones. When potential for conflict is high and thresholds are extrapolated into novel landscapes, clearly defined adaptive management goals and monitoring systems would be prudent. Within this adaptive management framework, it is also critical that assumptions are stated explicitly and tested with data whenever possible.

Acknowledgments

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Literature Cited

Aarts, G., M. MacKenzie, B. McConnell, M. Fedak, and J. Matthiopoulos. 2008. Estimating space-use and habitat preference from wildlife telemetry data. Ecography 31:140–160.

Aldridge, C. L., and M. S. Boyce. 2007. Linking occurrence and fitness to persistence: habitat-based approach for endangered greater sage-grouse. Ecological Applications 17:508–526.

Aldridge, C. L., S. E. Nielsen, H. L. Beyer, M. S. Boyce, J. W. Connelly, S. T. Knick, and M. A. Schroeder. 2008. Range-wide patterns of greater sage-grouse persistence. Diversity and Distributions 14:983–994.

Aldridge, C. L., D. J. Saher, T. M. Childers, K. E. Stahlnecker, and Z. H. Bowen. 2012. Crucial nesting habitat for Gunnison sage-grouse: a spatially explicit hierarchical approach. Journal of Wildlife Management 76:391–406.

Barnett, J. K., and J. A. Crawford. 1994. Pre-laying nutrition of sage-grouse hens in Oregon. Journal of Range Management 47:114–118.

Baruch-Mordo, S., J. S. Evans, J. P. Severson, D. E. Naugle, J. D. Maestas, J. M. Kiesecker, M. J. Falkowski, C. A. Hagen, and K. P. Reese. 2013. Saving sage-grouse from the trees: a proactive solution to reducing a key threat to a candidate species. Biological Conservation 167:233–241.

Beyer, H. L., R. Ung, D. L. Murray, and M. J. Fortin. 2013. Functional responses, seasonal variation and thresholds in behavioural responses of moose to road density. Journal of Applied Ecology 50: 286–294.

Bivand, R. S., T. Keitt, and B. Rowlingson. 2013. rgdal: bindings for the Geospatial Data Abstraction Library. R package version 0.8-9. http://CRAN.R-project.org/package=rgdal

Bivand, R. S., E. J. Pebesma, and V. Gomez-Rubio. 2008. Applied spatial data analysis with R. Springer, New York, New York, USA. http://www.asdar-book.org/

Blomberg, E. J., J. S. Sedinger, D. Gibson, P. S. Coates, and M. L. Casazza. 2014. Carryover effects and climatic conditions influence the postfledging survival of greater sage-grouse. Ecology and Evolution 4:4488–4499.

Blomberg, E. J., J. S. Sedinger, D. V. Nonne, and M. T. Atamian. 2013*a*. Annual male lek attendance influences count-based population indices of greater sage-grouse. Journal of Wildlife Management 77:1583–1592.

Blomberg, E. J., J. S. Sedinger, D. V. Nonne, and M. T. Atamian. 2013*b*. Seasonal reproductive costs contribute to reduced survival of female greater sage-grouse. Journal of Avian Biology 44:149–158.

Boyce, M. S., and L. L. McDonald. 1999. Relating populations to habitats using resource selection functions. Trends in Ecology & Evolution 14: 268–272.

Breiman, L. 2001. Random forests. Machine Learning 45:5–32.

Brown, J. H., D. W. Mehlman, and G. C. Stevens. 1995. Spatial variation in abundance. Ecology 76: 2028–2043.

Carwardine, J., K. A. Wilson, M. Watts, A. Etter, C. J. Klein, and H. P. Possingham. 2008. Avoiding costly conservation mistakes: the importance of defining actions and costs in spatial priority setting. PLoS ONE 3:e2586.

Coates, P. S., M. L. Casazza, E. J. Blomberg, S. C. Gardner, S. P. Espinosa, J. L. Yee, L. Wiechman, and B. J. Halstead. 2013. Evaluating greater sage-grouse seasonal space use relative to leks: implications for surface use designations in sagebrush ecosystems. Journal of Wildlife Management 77:1598–1609.

Coates, P. S., et al. 2015. Integrating spatially explicit indices of abundance and habitat quality: an applied example for greater sage-grouse management. Journal of Applied Ecology 53:83–95.

Coates, P. S., B. J. Halstead, E. J. Blomberg, B. Brussee, K. B. Howe, L. Wiechman, J. Tebbenkamp, K. P. Reese, S. C. Gardner, and M. L. Casazza. 2014. A hierarchical integrated population model for greater sage-grouse (*Centrocercus urophasianus*) in the bi-state distinct population segment, California and Nevada. US Geological Survey Open-File Report 2014–1165.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming, USA.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967–985.

Crawford, J. A., R. A. Olson, N. E. West, J. C. Mosley, M. A. Schroeder, T. D. Whitson, R. F. Miller, M. A. Gregg, and C. S. Boyd. 2004. Ecology and management of sage-grouse and sage-grouse habitat. Rangeland Ecology & Management 57:2–19.

Cressie, N. 1991. Statistics for spatial data. Wiley-Interscience, New York, New York, USA.

Cutler, D. R., T. C. Edwards, K. H. Beard, A. Cutler, K. T. Hess, J. Gibson, and J. J. Lawler. 2007. Random forests for classification in ecology. Ecology 88:2783–2792.

Diggle, P. 1985. A kernel method for smoothing point process data. Applied statistics 34:138–147.

Doherty, K. E., D. E. Naugle, H. E. Copeland, A. Pocewicz, and J. M. Kiesecker. 2011. Energy development and conservation tradeoffs: systematic planning for greater sage-grouse in their eastern range. Pages 505–516 *in* S. T. Knick and J. W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology. Volume 38. University of California Press, Berkeley, California, USA.

Doherty, K. E., D. E. Naugle, and J. S. Evans. 2010*a*. A currency for offsetting energy development impacts: horse-trading sage-grouse on the open market. PLoS ONE 5:e10339.

Doherty, K. E., D. E. Naugle, and B. L. Walker. 2010*b*. Greater sage-grouse nesting habitat: the importance of managing at multiple scales. Journal of Wildlife Management 74:1544–1553.

Doherty, K. E., D. E. Naugle, B. L. Walker, and J. M. Graham. 2008. Greater sage-grouse winter habitat selection and energy development. Journal of Wildlife Management 72:187–195.

Donovan, T. M., P. W. Jones, E. M. Annand, and F. R. Thompson. 1997. Variation in local-scale edge effects: mechanisms and landscape context. Ecology 78:2064–2075.

Ehlers, L. W., C. Johnson, and D. Seip. 2014. Movement ecology of wolves across an industrial landscape supporting threatened populations of woodland caribou. Landscape Ecology 29: 451–465.

Elith, J., and J. R. Leathwick. 2009. Species distribution models: ecological explanation and prediction across space and time. Annual Review of Ecology, Evolution, and Systematics 40:677–697.

Evans, J. S. 2015. spatialEco: an R package for spatial analysis and modeling. R package version 0.1-1. http://cran.r-project.org/package=spatialEco

Evans, J. S., and S. A. Cushman. 2009. Gradient modeling of conifer species using random forests. Landscape Ecology 24:673–683.

Evans, J. S., and M. A. Murphy. 2014. rfUtilities. R package version 1.0-0, http://CRAN.R-project.org/package=rfUtilities

Evans, J. S., M. A. Murphy, Z. A. Holden, and S. A. Cushman. 2011. Modeling species distribution and change using random forests in predictive species and habitat modeling. Pages 139–159 *in* C. A. Drew, Y. F. Wiersman, and F. Huettmann, editors. Landscape ecology: concepts and applications. Springer, New York, New York, USA.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution, and Systematics 34:487–515.

Fedy, B. C., K. E. Doherty, C. L. Aldridge, M. O'Donnell, J. L. Beck, B. Bedrosian, D. Gummer, M. J. Holloran, G. D. Johnson, and N. W. Kaczor. 2014. Habitat prioritization across large landscapes, multiple seasons, and novel areas: an example using greater sage-grouse in Wyoming. Wildlife Monographs 190:1–39.

Fedy, B. C., et al. 2012. Interseasonal movements of greater sage-grouse, migratory behavior, and an assessment of the core regions concept in Wyoming. Journal of Wildlife Management 76: 1062–1071.

Gibson, R. M. 1996*a*. Female choice in sage grouse: the roles of attraction and active comparison. Behavioral Ecology and Sociobiology 39:55–59.

Gibson, R. M. 1996*b*. A re-evaluation of hotspot settlement in lekking sage grouse. Animal Behaviour 52:993–1005.

Gibson, R. M., and T. A. Langen. 1996. How do animals choose their mates? Trends in Ecology & Evolution 11:468–470.

Gillies, C. S., M. Hebblewhite, S. E. Nielsen, M. A. Krawchuk, C. L. Aldridge, J. L. Frair, D. J. Saher, C. E. Stevens, and C. L. Jerde. 2006. Application of random effects to the study of resource selection by animals. Journal of Animal Ecology 75:887–898.

Godvik, I. M. R., L. E. Loe, J. O. Vik, V. Veiberg, R. Langvatn, and A. Mysterud. 2009. Temporal scales, trade-offs, and functional responses in red deer habitat selection. Ecology 90:699–710.

Gregory, A. J., and J. L. Beck. 2014. Spatial heterogeneity in response of male greater sage-grouse lek attendance to energy development. PLoS ONE 9:e97132.

Groves, C. R. 2003. Drafting a conservation blueprint: a practitioner's guide to planning for biodiversity. Island Press, Washington, DC, USA.

Haines, D. E., and K. H. Pollock. 1998. Estimating the number of active and successful bald eagle nests: an application of the dual frame method. Environmental and Ecological Statistics 5:245–256.

Hastie, T., R. Tibshirani, and J. Friedman. 2008. The elements of statistical learning. Second edition. Springer-Verlag, New York, New York, USA.

Hebblewhite, M., and E. Merrill. 2008. Modelling wildlife–human relationships for social species with mixed-effects resource selection models. Journal of Applied Ecology 45:834–844.

Herfindal, I., J. P. Tremblay, B. B. Hansen, E. J. Solberg, M. Heim, and B. E. Saether. 2009. Scale dependency and functional response in moose habitat selection. Ecography 32:849–859.

Hess, J. E., and J. L. Beck. 2012. Disturbance factors influencing greater sage-grouse lek abandonment in north-central Wyoming. Journal of Wildlife Management 76:1625–1634.

Hijmans, R. J., and J. V. Etten. 2013. raster: geographic data analysis and modeling. R package version 2.1-25. http://CRAN.R-project.org/package=raster

Hill, M., and H. Caswell. 1999. Habitat fragmentation and extinction thresholds on fractal landscapes. Ecology Letters 2:121–127.

Holloran, M. J., and S. H. Anderson. 2005. Spatial distribution of greater sage-grouse nests in relatively contiguous sagebrush habitats. Condor 107:742–752.

Houle, M. L., D. Fortin, C. Dussault, R. H. Courtois, and J. P. Ouellet. 2010. Cumulative effects of forestry on habitat use by gray wolf (*Canis lupus*) in the boreal forest. Landscape Ecology 25:419–433.

Johnson, C. J., S. E. Nielsen, E. H. Merrill, T. L. McDonald, and M. S. Boyce. 2006. Resource selection functions based on use–availability data: theoretical motivation and evaluation methods. Journal of Wildlife Management 70:347–357.

Johnson, D., M. Holloran, J. Connelly, S. Hanser, C. Amundson, and S. Knick. 2011. Influences of environmental and anthropogenic features on greater sage-grouse populations, 1997–2007. Studies in Avian Biology 38:407–450.

Johnson, D. H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology 61:65–71.

Johnson, D. S., M. B. Hooten, and C. E. Kuhn. 2013. Estimating animal resource selection from telemetry data using point process models. Journal of Animal Ecology 82:1155–1164.

Knick, S. T., D. S. Dobkin, J. T. Rotenberry, M. A. Schroeder, W. M. Vander Haegen, and C. van Riper III. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. Condor 105:611–634.

Knick, S. T., S. E. Hanser, and K. L. Preston. 2013. Modeling ecological minimum requirements for distribution of greater sage-grouse leks: implications for population connectivity across their western range, USA. Ecology and Evolution 3: 1539–1551.

Lesmerises, F., C. Dussault, and M.H. St-Laurent. 2012. Wolf habitat selection is shaped by human activities in a highly managed boreal forest. Forest Ecology and Management 276:125–131.

Liaw, A., and M. Wiener. 2002. Classification and regression by randomForest. R News 2:18–22.

Matthiopoulos, J., M. Hebblewhite, G. Aarts, and J. Fieberg. 2011. Generalized functional responses for species distributions. Ecological Applications 92:583–589.

McDonald, L., G. Beauprez, G. Gardner, J. Griswold, C. Hagen, F. Hornsby, D. Klute, S. Kyle, J. Pitman, and T. Rintz. 2014. Range-wide population size of the lesser prairie-chicken: 2012 and 2013. Wildlife Society Bulletin 38:536–546.

Meyer, C. B., and W. Thuiller. 2006. Accuracy of resource selection functions across spatial scales. Diversity and Distributions 12:288–297.

Miller, R. F., and L. L. Eddleman. 2001. Spatial and temporal changes of sage grouse habitat in the sagebrush biome. Technical Bulletin 151. Oregon State University Agricultural Experiment Station. Corvallis, Oregon, USA, 39 pp.

Mladenoff, D. J., T. A. Sickley, and A. P. Wydeven. 1999. Predicting gray wolf landscape recolonization: logistic regression models vs. new field data. Ecological Applications 9:37–44.

Moreau, G., D. Fortin, S. Couturier, and T. Duchesne. 2012. Multi-level functional responses for wildlife conservation: the case of threatened caribou in managed boreal forests. Journal of Applied Ecology 49:611–620.

Murphy, H. T., and J. Lovett-Doust. 2007. Accounting for regional niche variation in habitat suitability models. Oikos 116:99–110.

Murphy, H. T., J. VanDerWal, and J. Lovett-Doust. 2006. Distribution of abundance across the range in eastern North American trees. Global Ecology and Biogeography 15:63–71.

Murphy, M. A., J. S. Evans, and A. Storfer. 2010. Quantifying *Bufo boreas* connectivity in Yellowstone National Park with landscape genetics. Ecology 91:252–261.

Mysterud, A., and R. A. Ims. 1998. Functional responses in habitat use: Availability influences relative use in trade-off situations. Ecology 79:1435–1441.

Naugle, D. E., K. E. Doherty, B. L. Walker, H. E. Copeland, M. J. Holloran, and J. D. Tack. 2011*a*. Sage-grouse and cumulative impacts of energy development. Pages 55–70 *in* D. Naugle, editor. Energy development and wildlife conservation in Western North America. Island Press, Washington, DC, USA.

Naugle, D. E., K. E. Doherty, B. L. Walker, H. E. Copeland, and J. D. Tack. 2011*b*. Sage-grouse and cumulative impacts of energy development. Pages 213–225 *in* P. L. Krausman and L. K. Harris, editors. Cumulative effects in wildlife management. CRC Press, New York, New York, USA.

Nelle, P. J., K. P. Reese, and J. W. Connelly. 2000. Long-term effects of fire on sage grouse habitat. Journal of Range Management 53:586–591.

Northrup, J. M., C. R. Anderson, and G. Wittemyer. 2015. Quantifying spatial habitat loss from hydrocarbon development through assessing habitat selection patterns of mule deer. Global Change Biology 21:3961–3970.

Northrup, J. M., M. B. Hooten, C. R. Anderson, and G. Wittemyer. 2013. Practical guidance on characterizing availability in resource selection functions under a use–availability design. Ecology 94:1456–1463.

Olden, J. D., J. J. Lawler, and N. L. Poff. 2008. Machine learning without tears: a primer for ecologist. The Quarterly Review of Biology 83:171–193.

Patterson, R. L. 1952. The sage grouse in Wyoming. Wyoming Game and Fish Commission, Sage Books Inc., Denver, Colorado, USA.

R Core Team. 2012 R: a language and environment for statistical computing. R. Foundation for Statistical

Computing, Vienna, Austria. http://www.R-project.org/

Rehfeldt, G. E., N. L. Crookston, M. V. Warwell, and J. S. Evans. 2006. Empirical analyses of plant-climate relationships for the Western United States. International Journal of Plant Sciences 167: 1123–1150.

Rhodes, J. R., J. G. Callaghan, C. A. McAlpine, C. D. Jong, M. E. Bowen, D. L. Mitchell, D. Lunney, and H. P. Possingham. 2008. Regional variation in habitat-occupancy thresholds: a warning for conservation planning. Journal of Applied Ecology 45:549–557.

Roever, C. L., R. J. van Aarde, and K. Leggett. 2012. Functional responses in the habitat selection of a generalist mega-herbivore, the African savannah elephant. Ecography 35:972–982.

Royle, J. A., J. D. Nichols, and M. Kéry. 2005. Modelling occurrence and abundance of species when detection is imperfect. Oikos 110:353–359.

SAGA-GIS. 2015. version 2.1.0. http://www.saga-gis.org/

Sagarin, R. D., S. D. Gaines, and B. Gaylord. 2006. Moving beyond assumptions to understand abundance distributions across the ranges of species. Trends in Ecology & Evolution 21:524–530.

Sanderson, E. W., K. H. Redford, A. Vedder, P. B. Coppolillo, and S. E. Ward. 2002. A conceptual model for conservation planning based on landscape species requirements. Landscape and Urban Planning 58:41–56.

Sawyer, H., R. M. Nielson, F. Lindzey, and L. L. McDonald. 2006. Winter habitat selection of mule deer before and during development of a natural gas field. Journal of Wildlife Management 70: 396–403.

Schroeder, M. A., C. L. Aldridge, A. D. Apa, J. R. Bohne, C. E. Braun, S. D. Bunnell, J. W. Connelly, P. A. Deibert, S. C. Gardner, and M. A. Hilliard. 2004. Distribution of sage-grouse in North America. Condor 106:363–376.

Schroeder, M. A., and G. C. White. 1993. Dispersion of greater prairie chicken nests in relation to lek location: evaluation of the hot-spot hypothesis of lek evolution. Behavioral Ecology 4:266–270.

Shmueli, G. 2010. To explain or to predict? Statistical Science 25:289–310.

Silverman, B. W. 1986. Density estimation for statistics and data analysis. CRC Press, New York, New York, USA.

Stephens, P. A., N. Pettorelli, J. Barlow, M. J. Whittingham, and M. W. Cadotte. 2015. Management by proxy? The use of indices in applied ecology. Journal of Applied Ecology 52:1–6.

Stiver, S., A. Apa, J. Bohne, S. Bunnell, P. Deibert, S. Gardner, M. Hilliard, C. McCarthy, and M. Schroeder. 2006. Greater sage-grouse comprehensive conservation strategy. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming, USA.

Tack, J. 2009. Sage-grouse and the human footprint: implications for conservation of small and declining populations. University of Montana, Missoula, Montana, USA.

van der Hoek, Y., A. M. Wilson, R. Renfrew, J. Walsh, P. G. Rodewald, J. Baldy, and L. L. Manne. 2015. Regional variability in extinction thresholds for forest birds in the north-eastern United States: an examination of potential drivers using long-term breeding bird atlas datasets. Diversity and Distributions 21:686–697.

WAFWA. 2015. Greater sage-grouse population trends: an analysis of lek count databases 1965–2015. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming, USA, 55 pp.

Wisdom, M. J., C. W. Meinke, S. T. Knick, and M. A. Schroeder. 2011. Factors associated with extirpation of sage-grouse. Studies in Avian Biology 38:451–472.

Worton, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. Ecology 70:164–168.

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RESEARCH ARTICLE

Nesting habitat selection influences nest and early offspring survival in Greater Sage-Grouse

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ABSTRACT

Adaptive habitat-selection theory predicts that individuals should use habitats that maximize lifetime fitness. However, trade-offs between life-history stages, environmental variability, and predator–prey dynamics can interact with individual preferences, which may result in individuals selecting suboptimal habitats. Understanding the distinction between adaptive and maladaptive animal use of habitat is central to effective species conservation, because use of maladaptive habitat is counter to conservation objectives. Our objectives were to assess whether habitat characteristics selected by Greater Sage-Grouse (*Centrocercus urophasianus*) were correlated with increased production of fledged young. We monitored 411 nests and 120 broods from 234 females between 2004 and 2012 in central Nevada, USA. We determined which habitat characteristics were selected as nesting habitat and assessed whether these characteristics influenced nest success and early offspring survival. The relationships between characteristics selected at nest sites and metrics of reproductive success were variable, in that certain characteristics (e.g., forb cover, amount of pinyon–juniper woodlands) were correlated with higher nest survival and chick survival, but other characteristics (e.g., amount of sagebrush, residual grass height) did not improve reproductive success. Despite variability among predictor variables, we found a positive effect of selection of fine-scale habitat characteristics on nest ($\beta_{NS\text{-}Local} = 0.14$, 85% confidence interval [CI]: 0.04–0.23) and chick survival ($\beta_{CS\text{-}Local} = 0.39$, 85% CI: 0.27–0.50); however, we did not find that selection of broad-scale habitat characteristics predicted reproductive success ($\beta_{NS\text{-}Landscape} = -0.04$, 85% CI: –0.15 to 0.06; $\beta_{CS\text{-}Landscape} = 0.06$, 85% CI: –0.06 to 0.18). Additionally, nest-site selection was more predictive of chick survival than of nest survival, which suggests that females' selection of nesting habitat was based primarily on its qualities as brood-rearing habitat. Together, these findings suggest that nest-site selection may be influenced by more than increased reproductive success, or that there is a landscape-level pattern to local-scale habitat characteristics.

Keywords: Centrocercus urophasianus, Greater Sage-Grouse, habitat selection, reproductive success

La selección del hábitat de anidación influencia la supervivencia del nido y de los polluelos recién nacidos en *Centrocercus urophasianus*

RESUMEN

La teoría de la selección adaptativa del hábitat predice que los individuos deberían usar hábitats que maximicen su adecuación biológica en el transcurso de toda su vida. Sin embargo, las soluciones de compromiso entre las diferentes etapas de la historia de vida, la variabilidad ambiental y la dinámica depredador-presa pueden interactuar con las preferencias individuales, resultando en la selección de hábitats sub-óptimos por parte de los individuos. Entender la distinción entre el uso animal adaptativo y maladaptativo de los hábitats es central para la conservación efectiva de las especies, debido a que el uso maladaptativo de los hábitats es contraproducente para los objetivos de conservación. Nuestro objetivo fue evaluar si las características del hábitat seleccionadas por *Centrocercus urophasianus* estuvieron correlacionadas con un aumento en la producción de juveniles emplumados. Seguimos 411 nidos y 120 nidadas provenientes de 234 hembras de *C. urophasianus* entre 2004 y 2012 en el centro de Nevada, EEUU. Determinamos las condiciones de hábitat que fueron seleccionadas como hábitat de anidación y evaluamos si estas características influenciaron el éxito de anidación y la supervivencia de los polluelos recién nacidos de *C. urophasianus*. Las relaciones entre las características seleccionadas en los sitios de anidación y las métricas del éxito reproductivo fueron variables ya que ciertas características (e.g., cobertura de forbias, cantidad de bosques de piñón y enebro) estuvieron correlacionadas con una supervivencia más alta del nido y de los polluelos, pero otras características (e.g., cantidad de artemisa, altura del pasto residual) no mejoraron el éxito reproductivo. A pesar de la variabilidad entre las variables predictivas, encontramos un efecto positivo de la selección de características del hábitat de escala fina sobre el nido ($\beta_{NS\text{-}Local} = 0.14$, 85% intervalo de confianza [IC]: 0.04–0.23) y la supervivencia de los polluelos ($\beta_{CS\text{-}Local} = 0.39$, 85% IC: 0.27–0.50); sin embargo, no encontramos que la selección de las características del hábitat de escala gruesa predijeran el éxito reproductivo ($\beta_{NS\text{-}Landscape} = -0.04$, 85% IC: –0.15 a 0.06; $\beta_{CS\text{-}Landscape} = 0.06$, 85% IC: –0.06 a 0.18). Adicionalmente, la selección del sitio de anidación fue más predictiva con la supervivencia del polluelo que la supervivencia del nido, lo que sugiere que las hembras pueden estar seleccionando el hábitat de anidación basadas principalmente en sus cualidades como hábitat de cría de la nidada. En conjunto, estos hallazgos sugieren que la selección del sitio de anidación puede estar influenciada por más que el incremento en el éxito reproductivo, o que hay un patrón a nivel de paisaje a la escala local de las características del hábitat.

Palabras clave: Centrocercus urophasianus, éxito reproductivo, selección de hábitat

INTRODUCTION

Habitat heterogeneity is a key attribute of most landscapes, which likely results in spatial (Newton 1991) or temporal (Franklin et al. 2000) variability in fitness among individuals. Adaptive habitat-selection theory predicts that individuals should select areas with habitat characteristics that maximize their fitness, or lifetime reproductive success (Orians and Wittenberger 1991). The relationship between habitat selection (i.e. disproportionate use of a resource in relation to availability; Johnson 1980) and fitness is complex, however, and may also be affected, in part, by intraspecific (Morris 1989) and interspecific interactions (Fretwell and Lucas 1969), climate variability (Martin 2001), and habitat degradation (Feary et al. 2007). Further complicating matters is the fact that individual habitat use may be constrained by aspects of the species' life history. For example, maladaptive preferences may be associated with behaviors of the species, such as site fidelity (Ganter and Cooke 1998), that limit the absolute amount of habitat perceived to be available to an individual.

Understanding the distinction between adaptive and maladaptive animal use of habitat is central to effective species conservation, because animal use of maladaptive habitat is clearly counter to conservation objectives. Maladaptive habitat–organism relationships are thought to be more common in disturbed environments because of a mismatch between evolved strategies and current environmental conditions (Battin 2004). Because species of conservation concern often occur in disturbed environments, patterns in habitat use in these species may not always be indicative of the habitat conditions required for population stability, yet "use" is often used as a primary metric for delineating conservation areas (Kautz et al. 2006). For example, in the United States, when a species is protected under the Endangered Species Act, an early step in recovery planning is designation of "critical habitat," which is defined as the "geographic area occupied by the species" (U.S. Department of the Interior 2014). However, a multitiered approach that first identifies use and then demonstrates organism success will provide the greatest conservation benefit, because population persistence is ultimately determined by both the probability of occurrence and the fitness of individuals using a particular area (Aldridge and Boyce 2007).

Despite the recognized eco-evolutionary importance of habitat selection in human-modified ecosystems (Morris 2011), assessments of the relationships between habitat selection and fitness have been difficult to quantify because they must consider future reproductive value of adults (Nicolai and Sedinger 2012), recruitment of young into the breeding population (Kazantzidis et al. 1997), and the future reproductive success of the offspring (Pierotti 1982). In reality, however, studies often assess the influence of habitat use on a single fitness component (e.g., nest survival) that is considered an index of overall reproductive success (Newlon and Saab 2011). For many birds, nest survival is arguably the most widely reported metric of reproductive success (e.g., Sedinger 2007). The selection of a nest site, however, not only influences the viability of eggs, but also influences hatchling survival directly by determining the environmental characteristics (e.g., food sources and predator communities) that offspring encounter during early life (Kolbe and Janzen 2001). Therefore, using nest success itself as an index of reproductive success may fail to fully assess whether nesting-habitat preferences are adaptive (Jones 2001), because individuals may use habitat that influences multiple fitness components, including those in later life-history stages (e.g., selection of nest sites that improves offspring survival but not nest survival; Chalfoun and Martin 2007).

Greater Sage-Grouse (*Centrocercus urophasianus*; hereafter "sage-grouse") are a species of conservation concern because their distribution and abundance have declined substantially since European settlement (Knick and Connelly 2011), which has been attributed primarily to habitat loss or degradation (Garton et al. 2011). Habitat degradation (e.g., introduction of exotic grasses, increased fire frequency and intensity, grazing by nonnative ungulates) has been linked to a reduction in nest survival (Connelly et al. 2011), altered nest-site selection (Kirol et al. 2012), and reduced recruitment of individuals into breeding populations (Blomberg et al. 2012). Although assessments of the environmental features associated with either nest-site selection or nest survival are common (Hagen et al. 2007, Dinkins et al. 2014), few studies have directly modeled the effect of habitat selection on fitness components (Aldridge and Boyce 2007, Kirol et al. 2015). Although positive associations between nest-site selection and nest survival have been reported (Lockyer et al. 2015), sage-grouse chicks are nidifugous and permanently leave the nest with their mothers approximately a day after hatch (Connelly et al. 2011). Therefore, it remains unclear whether nest-site selection influences chick survival after individuals leave the nest.

Our primary goal was to assess whether sage-grouse selected nest-site characteristics that were positively associated with multiple components of reproductive success. Sage-grouse reproductive potential is highly dependent on habitat structure and function (Connelly et al. 2011); therefore, we were interested in whether habitat characteristics selected at nest sites also influenced individual reproductive parameters. As such, our approach was to assess the effect of habitat characteristics selected near nest sites on nest survival, prefledging chick survival, and the total production of fledged young. We hypothesized that (1) nests surrounded by habitat characteristics predictive of nest-site selection should have higher probabilities of hatching; and (2) offspring from these nests should have higher probabilities of survival.

METHODS

Study System

We monitored the reproductive behavior of female sage-grouse from 2004 to 2012 in Eureka County, Nevada, USA (Supplemental Material Figure S1). We focused our research on a subset of breeding leks and other areas of seasonal habitat associated with these leks within Eureka County.

Field Monitoring

We captured female sage-grouse on 11–13 leks during each breeding season (March–May) from 2004 to 2012, and in seasonal high-elevation habitat during each fall from 2005 to 2011. Females were equipped with a 22 g radio with necklace-style attachment (A4060 and A3950; Advanced Telemetry Systems, Isanti, Minnesota, USA) and a size-14 federal band. Radio-tagged females were located approximately twice a week. After discovery of a nest, active nests were visited approximately twice a week until the nest hatched or failed (Gibson et al. 2015). After hatch, we monitored each brood's status through weekly brood flush counts, when we counted all chicks seen after flushing the female. We continued weekly flush counts until 42 days after hatch (hereafter "prefledging period") or until 2 wk of consecutive counts of zero chicks, whichever occurred first.

Nest Vegetation Surveys

We measured vegetation at all nest sites within 3 days of either the predicted or actual date of hatch (Gibson et al. 2016). Predicted hatch dates for depredated nests were estimated by floating eggs, using methods described by Blomberg et al. (2014a). We sampled vegetation characteristics for each nest along 10 m intersecting transects, centered at the nest bowl (hereafter "plot"; Gregg et al. 1994). We measured vegetation using the line-intercept (Canfield 1941) and Daubenmire (1959) frame methods. The line-intercept method was used to estimate proportional shrub cover of various shrub species, which we then aggregated together as total shrub cover, sagebrush (*Artemisia* spp.) shrub cover, and cover of shrubs other than sagebrush (e.g., rubber rabbitbrush [*Ericameria nauseosa*], western serviceberry [*Amelanchier alnifolia*], western snowberry [*Symphoricarpos occidentalis*]; hereafter "other shrub cover").

We used five 20 $\times$ 50 cm Daubenmire frames placed along each transect to classify grass, forb, and total (i.e. grass, forb, and shrub) cover to 7 classes: not present, <5%, 5–25%, 25–50%, 50–75%, 75–95%, and >95%. Height (cm) of the nearest representative of residual (dead) grass, live grass, live forb, and shrub to the northeast corner in each frame was measured and used as a proxy for average height for each group in the plot. We identified each shrub, grass, and forb taxon that occurred within each frame to either species or genus and recorded these to calculate the taxon richness of the aforementioned functional groups within each plot. Each year, we randomly selected 2 points from within a 5 km buffer around each of the 13 study leks (Coates et al. 2013) and conducted vegetation surveys at these points following the same protocols. Given that >95% of nests in our study system were located within 5 km of these study leks, we believed that this was an appropriate distance for classification of available nesting habitat. We began random vegetation surveys after the first predicted hatch date in a given year, and completed them opportunistically throughout the nesting season to coincide with sampling at nest locations.

Evaluating Habitat Characteristics

We used terminology consistent with Doherty et al. (2010), in which predictor variables from fine-scale vegetation surveys (those described above) were defined as "local-scale," and predictor variables derived from more broad-scale Geographic Information Systems data (described below) were defined as "landscape-scale" variables. Local-scale predictor variables included percent total cover, percent forb cover, percent grass cover, forb taxon richness, average forb height, average live grass height, average residual grass height, average shrub height, total percent shrub cover, percent sagebrush cover, and percent non-sagebrush shrub cover within 5 m of the nest.

We developed a suite of landscape-level habitat covariates that characterized plant community structure, topography, and disturbance in nesting habitat at 3 spatial extents (500 m, 1,000 m, and 2,000 m radius circles) surrounding nests and random points. We selected these spatial extents on the basis of previously published assessments of sage-grouse habitat selection (Aldridge and Boyce 2007, Doherty et al. 2010). Landscape-scale predictor variables included proportional cover (based on the proportion of 30 m pixels classified as a certain cover type) of sagebrush, pinyon–juniper woodland, and exotic grassland, as well as nest elevation, slope, aspect, distance from nearest lek, distance from nearest road, and distance from nearest spring, seep, or other water source (for descriptions, see Supplemental Material Table S1).

We randomly generated sampling points at the landscape level, using ArcMap 10.0 with an "available" data layer that we defined by placing a 5,000 m buffer around all spring (April 1–June 15) locations of radio-tagged females collected from 2004 to 2012. The buffered spring telemetry-location layer represented a coarse approximation of land area that was available to nesting sage-grouse for use as habitat within our study system. We randomly generated $\sim$1 point km^{-2} of available land area, which yielded 2,203 random points. Each landscape-level random point was randomly assigned a year and was randomly paired with an actual nest point for the purpose of conforming to general linear mixed-effects analyses described below.

Potential Confounding between Individual Habitat Use and Spatial Distribution of Leks

Sage-grouse have a lek-based mating system; therefore, individual nest-site use may be confounded with the spatial distribution of leks. Leks are not randomly distributed across the landscape and have been hypothesized to occur as a function of patterns in female resource use (Gibson 1992). Thus, leks are likely to be located in proximity to habitat selected by females for nesting or other life stages. Unlike nest locations, "available" or "unused" points in habitat-selection studies are often distributed at random and thus are independent of this confounding effect. We were interested in assessing the influence of female habitat selection on reproductive success, so it was inappropriate to directly compare random and nest locations, because habitat characteristics that covaried with distance from lek (e.g., elevation, sagebrush shrub cover) could be interpreted as habitat selection, even in the absence of individuals preferring a certain habitat characteristic. Therefore, for analyses that assessed individual habitat selection, we controlled for the effect of distance from lek by subsampling our random points to match the distribution of distances from leks of our sample of nest locations (Supplemental Material Figure S2). This approach does not decouple the association between sites used by individuals and distance from their breeding leks, so we cannot determine whether females use habitat because it is near leks, or if leks occur in habitats used by females. However, representing habitat conditions accessible to nesting females based on their proximity to a lek does constrain the available points, which should reduce the occurrence of false positives that could occur if the distribution of leks were influenced by features other than female resource use (e.g., "hotspot" or "preference" hypotheses). Concurrently, we conducted an identical analysis using the full suite of random points to evaluate whether considering the spatial distribution of leks affected resource-selection results. This also allowed us to assess the loss of statistical power related to the subsampling approach, which reduced the number of available points from 2,203 to 471.

Statistical Analyses

Resource-selection functions. We developed resource-selection functions following methods outlined by Boyce and McDonald (1999) and Hebblewhite and Merrill (2008), although we deviated from those methods by estimating resource-selection probability function (RSPF) coefficients using logistic models instead of log-linear models. We interpret these relative probabilities of nest-site selection as an index of habitat suitability because we did not quantify true "unused" habitat. We performed each resource-selection function analysis in a generalized linear mixed-model framework (Zuur et al. 2009) using the lme4 package (Bates and Maechler 2010) in R (R Development Core Team 2012). Both selection analyses incorporated a random intercept for year and individual in all models. Year and individual were used as random effects to account for variation in strength of selection from unexplained sources. Although resource-selection models did not simultaneously integrate covariates from both local and landscape scales, we calculated the correlation (Pearson's *r*) among selected habitat characteristics between spatial scales to assess cross-scale patterns in habitat selection.

Nest survival. We used the nest survival module in Program MARK (White and Burnham 1999) to model variation in daily nest survival (DNS) probabilities based on data collected from nests monitored from 2004 to 2012. We censored observer-related nest abandonments because we were primarily interested in the relationships between habitat characteristics and nest survival, and we considered failure due to abandonment to be independent of nest habitat. However, we corrected DNS post hoc, using the approach outlined by Gibson et al. (2015) to estimate more accurately the number of fledged chicks (see below). We assumed a 37-day nesting period (Blomberg et al. 2015) in our estimates of overall nest survival.

Prefledging chick survival. We used the "survival of young from marked adults" module (Lukacs et al. 2004) in

Program MARK to estimate weekly survival of individual chicks during the prefledging period. This module uses repeated offspring counts and allows for estimation of apparent offspring survival (φ) accounting for imperfect detection (p), conditioned on the assumption that offspring are completely associated with a parent, which is uniquely marked and available for observation. Encounter histories represented weekly counts of chicks observed with their mother from hatch until 6 wk after hatch. We assumed that survival through the sixth week represented prefledging chick survival, but we only modeled the effect of nest-site characteristics in the first 2 wk, which corresponds to the period of time when chicks were least mobile.

Estimated number of fledged chicks. We converted estimates of nest survival and prefledging chick survival of a specific nest (i) to reflect the estimated number of chicks using the following equation:

$$\text{Number of fledged chicks}_i = \text{nest survival}_i \times \text{prefledging chick survival}_i \times \text{mean clutch size}$$

We used the estimate of mean clutch size for all nest attempts (~7.2 eggs) reported in Blomberg et al. (2014a). We assumed no covariance among the specified parameters and calculated the variance of their product by the parametric bootstrap method (Zhou 2002).

Model Selection

We used an information-theoretic approach to evaluate support for candidate models (Burnham and Anderson 2002) and considered covariate effects to be meaningful if 85% confidence intervals (CI) of β coefficients did not overlap 0.0 (Arnold 2010). For the nest-selection analysis, we used an iterative process in model creation in which we applied individual covariates to assess potential sources of heterogeneity in nest-site selection and reproductive success. Covariates were grouped by type (e.g., understory vegetation, overstory vegetation) and were added to a model one covariate at a time. Covariates that were substantially correlated with each other (Pearson's $r > 0.50$) were not included simultaneously in a single model. A priori, we developed a few interactions between covariate effects that we hypothesized might be biologically meaningful, which were generally related to assessing synergistic effects between vertical and horizontal cover. We considered interactions between forb height and forb cover, grass height and grass cover, residual grass height and grass cover, sagebrush shrub cover and other shrub cover, total shrub cover and shrub height, and slope and elevation. We also assessed nonlinear relationships for certain characteristics of interest, which included sagebrush shrub cover, other shrub cover, distance from nearest spring, shrub height, amount of habitat classified as sagebrush, and amount of habitat classified as pinyon–juniper woodlands.

Covariates that improved model fit were retained in the model structure. The top model from each covariate grouping was used to develop a global model that included informative covariates from each covariate type. We then performed backward model selection; each covariate in the current global model was removed singularly to assess covariance between it and the remaining covariates. If exclusion of a single covariate resulted in a decrease in AIC_c, that covariate was removed from consideration, simplifying the current global model. For the nest-survival and prefledging chick-survival analyses, we developed a candidate model set from covariates supported by the nest-site-selection analysis. Additionally, to assess further the influence of the association between habitat use and lek distribution, we considered models that included distance from nearest lek as a linear and nonlinear covariate. All covariates were z-standardized $\bar{x} = 0.0$ (SD $= 1.0$; White and Burnham 1999). We evaluated the fit of our RSPF models following methods outlined by Aldridge et al. (2012) with the use of Kendall's c-index (Harrell et al. 1996) to estimate a model's concordance of relative probabilities of nest-site selection among nest locations and available points.

RESULTS

We discovered 411 nests made by 242 unique females over 9 yr. After excluding abandoned nests, we based the nest-survival analysis on 350 nests, among which 133 were successful and 217 were depredated. We monitored 120 broods (862 chicks) from 99 unique females over a period of 8 yr, at least 163 chicks of which survived until 42 days after hatch.

Nest-site Selection

In each RSPF analysis, inclusion of individual as a random intercept did not improve model fit, so we removed individual as a random intercept in all subsequent models. However, in each analysis, inclusion of year as a random effect substantially improved model fit over the null model (Supplemental Material Table S2); therefore, we incorporated it in all models.

Local-scale nest-site selection. The concordance of relative probability of habitat selection between used and available points suggested that the most competitive local model was fairly predictive (Kendall's $c = 0.72$). Our top-ranked model from the local RSPF analysis (Table 1; Supplemental Material Table S3 and Figure S3) suggested that females selected for both sagebrush shrub cover ($\beta_{SBC} = 1.12$, 85% CI: 0.91–1.32) and other shrub cover ($\beta_{OSC} =$

TABLE 1. Performance of resource-selection functions based on generalized linear mixed models to assess the influence of local-scale vegetation characteristics on Greater Sage-Grouse nest-site selection in Eureka County, Nevada, USA, 2004–2012 (K = number of parameters).

Model [a]	ΔAIC_c	w_i	K	Dev.
$OSC_5 * SBC_5 + SH_5 + FH_5 * FC_5 + GH_5 + RGH_5 + FRich_5$	0.00	0.57	13	707.53
$OSC_5 * SBC_5 + SH_5 + FH_5 + FC_5 + GH_5 + RGH_5 + FRich_5$	0.83	0.37	12	710.36
$OSC_5 * SBC_5 + FH_5 + FC_5 + GH_5 + RGH_5 + FRich_5$	4.58	0.06	11	716.10
$OSC_5 * SBC_5 + SH_5$	50.79	0.00	7	770.32
$FH_5 * FC_5 + GH_5 + RGH_5 + FRich_5$	73.42	0.00	9	788.95

[a] Model-selection notation follows Burnham and Anderson (2002). All models include random intercepts for year and individual. Subscripts denote the scale of the variable (i.e. area within 5 m of a point). Horizontal cover variables include sagebrush shrub cover (SBC), other shrub cover (i.e. not sagebrush; OSC), and forb cover (FC). Vertical cover variables include average shrub height (SH), average forb height (FH), average live grass height (GH), and average residual grass height (RGH). "FRich" represents forb taxon richness within a given plot. All covariates were z-standardized prior to analysis. Main effects are included in models in which an interaction is specified. Full model results are presented in the Supplemental Material.

1.03, 85% CI: 0.72–1.34) as nesting habitat, but areas comprising high amounts of both sagebrush and non-sagebrush shrub cover ($\beta_{SBC*OSC} = 0.40$, 85% CI: 0.19–0.61) were more likely to be selected than areas of similar cover that comprised a less diverse shrub community. We also found that females selected for both the horizontal ($\beta_{FC} = 0.47$, 85% CI: 0.26–0.67) and vertical ($\beta_{FH} = 0.41$, 85% CI: 0.20–0.62) properties of the forb communities, but areas that comprised both taller forbs and proportionally larger coverage ($\beta_{FH*FC} = 0.33$, 85% CI: 0.02–0.65) were more likely to be selected than tall but sparse, or short but abundant, forb communities (Supplemental Material Figure S4). In addition to the dimensional properties of the forb community, females also selected for areas with a more diverse forb community ($\beta_{FR} = 0.33$, 85% CI: 0.18–0.48; Supplemental Material Figure S7). Additionally, females selected areas with taller residual grasses ($\beta_{RGH} = 0.33$, 85% CI: 0.14–0.52; Supplemental Material Figure S5) or live grasses ($\beta_{GH} = 0.21$, 85% CI: 0.04–0.38; Supplemental Material Figure S6), which suggests that females also selected for areas with greater vertical cover from grasses near nests. We also found that females avoided areas dominated by taller shrubs ($\beta_{SH} = -0.26$, 85% CI: –0.10 to –0.46; Supplemental Material Figure S8), which was potentially related to an avoidance of decadent sagebrush stands.

Landscape-scale nest-site selection. The concordance of relative probability of habitat selection between nest locations and the available points for both the full and the subsampled data suggested that the most competitive landscape model was predictive (Kendall's $c_{sub} = 0.79$ vs. Kendall's $c_{full} = 0.79$), and we did not lose explanatory power with the reduction in available points used for the subsampled analysis. We did not observe a substantial amount of variation in either model performance or beta parameter coefficients between subsampled and full random-point datasets (Figure 1). However, certain parameter estimates differed between the analyses (e.g., parameters for elevation, proportion of area classified as sagebrush), which is potentially indicative of absolute spatial constraints on individual habitat selection. Because of similarities among the analyses, we only report parameter estimates from the analysis based on subsampled data. However, we include parameter estimates for both analyses in Supplemental Material Table S7.

Our top-ranked model from the landscape RSPF analyses (Supplemental Material Tables S4, S5, and Figure S9) suggested that individuals selected for areas that had a greater amount of the surrounding area classified as sagebrush ($\beta_{Sage1,000} = 1.89$, 85% CI: 1.66–2.13). Additionally, individuals selected for areas at moderate elevation and on slopes (Supplemental Material Figure S10; $\beta_{Slope*Elev} = -0.36$, 85% CI: –0.26 to –0.45; $\beta_{Slope} = 0.53$, 85% CI: 0.39–0.68; $\beta_{Elev} = 0.01$, 85% CI: –0.16 to 0.18). We found that areas became increasingly unsuitable the farther they were located from a spring or water source (Supplemental Material Figure S11; $\beta_{Spring} = -0.12$, 85% CI: –0.02 to –0.23; $\beta_{Spring}{}^2 = -0.26$, 85% CI: –0.17 to –0.34). We also found that individuals selected for areas with low to moderate proportions of pinyon–juniper forest (10–30%), whereas areas with moderate to high proportions of pinyon–juniper forests nearby (>30%; Supplemental Material Figure S12) were avoided ($\beta_{PJ1,000} = 0.95$, 85% CI: 0.72–1.34; $\beta_{PJ1,000}{}^2 = -0.32$, 85% CI: –0.14 to –0.39).

Relationships between Habitat Selection and Reproductive Success

The relationships between habitat characteristics that were selected at nest sites and metrics of reproductive success were variable. Specific cover-type features (e.g., other shrub cover, forb cover, presence of pinyon–juniper woodlands) were correlated with higher nest survival and chick survival (Figure 1; Supplemental Material Tables S7, S8). We also found that some selected habitat characteristics (e.g., slope, sagebrush shrub cover, average forb

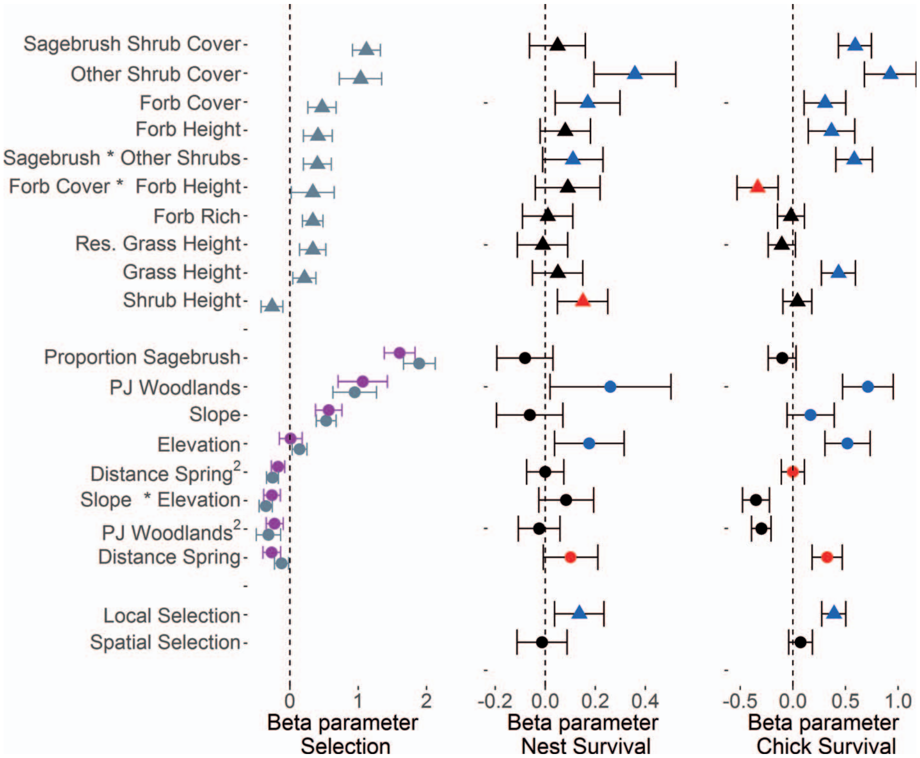

FIGURE 1. Parameter estimates and associated error of habitat characteristics selected as nest sites by female Greater Sage-Grouse in Eureka County, Nevada, USA, 2004–2012 (left), their influence on nest survival (middle), and prefledging chick survival (right). In the left graph, color denotes whether the random points were constrained to match the distribution of distances-from-lek observed from the sample of nest locations (violet) or not (gray). In the middle and left graphs, color denotes whether the preferences for each variable were adaptive (blue), maladaptive (red), or not supported (black). Triangles represent variables measured at the local scale, and circles represent variables measured at the landscape scale. Superscripts denote quadratic relationships. Landscape covariates and local selection represent the relative probability that a given nest would be suitable as a nest site. For additional information on habitat variables, see Supplemental Material.

height, average shrub height) only influenced one reproductive component and had a neutral effect on the other (i.e. nest survival or prefledging chick survival). Model support for many of the selected habitat characteristics in both the nest-survival and chick-survival analyses, however, was weak, and confidence intervals for the parameter coefficient widely overlapped zero (e.g., proportion of habitat classified as sagebrush within 1,000 m, forb richness, residual grass height), which suggested that some selected characteristics did not provide an appreciable benefit to reproductive success. Additionally, and contrary to our hypotheses, we found that certain habitat features selected at nest sites were actually negatively associated with annual reproductive success (e.g., distance to spring).

Despite variability among individual predictor variables, we found an overall positive effect of habitat selection at the local scale on nest survival ($\beta_{NS\text{-}Local} = 0.14$, 85% CI: 0.04–0.23; Figure 2) and prefledging chick survival ($\beta_{CS\text{-}Local} = 0.39$, 85% CI: 0.27–0.50), which supported our hypothesis that habitat selection positively influenced reproductive success. However, we found no support for an effect of nest-site selection at the landscape scale on nest survival ($\beta_{NS\text{-}Landscape} = -0.04$, 85% CI: –0.15 to 0.06) or prefledging chick survival ($\beta_{CS\text{-}Landscape} = 0.06$, 85% CI: –0.06 to 0.18), which suggests that the overall suite of habitat characteristics selected at the landscape scale was not directly predictive of reproductive performance (Figures 1 and 2).

Our results suggest that use of habitat characteristics at one scale was correlated with habitat use at the other scale (Figure 3). The direction of these cross-scale relationships was predominantly associated with the influence of the correlated habitat characteristics on reproductive success. In Figure 3, values that fall in the gray regions suggest that use of a landscape-scale habitat characteristic is positively associated with use of local-scale habitat characteristics, and that correlation between these variables is positively related to the habitat-selection coefficient of the local-scale habitat characteristic. Values that fall in the white regions suggest the opposite pattern—that is, the use of a landscape-scale habitat characteristic was negatively associated with use of local-scale habitat characteristic, and the

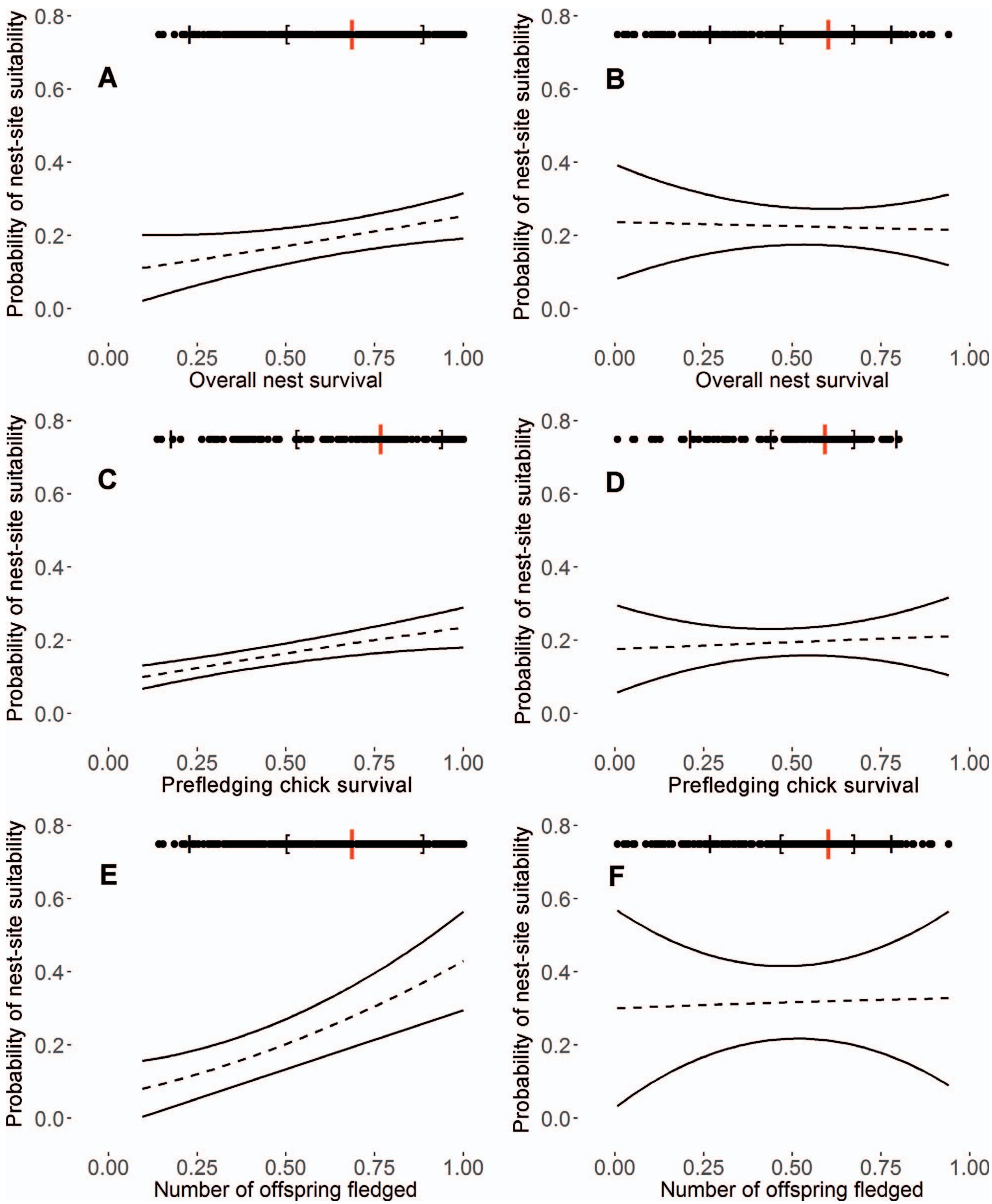

FIGURE 2. Influence of the relative probability of nest use from local-scale and landscape-scale resource-selection models on overall probabilities of (**A**) local and (**B**) landscape nest survival; (**C**) local and (**D**) landscape prefledging chick survival; and (**E**) local and (**F**) landscape number of chicks fledged in Greater Sage-Grouse in Eureka County, Nevada, USA, 2004 -2012. Error represented by 95% confidence intervals. The distribution of successful nest locations (**A, B, E, F**), or brood locations (**C, D**) is delineated by the median (red line), 1st quartile ([ ), and third quartile ( ]), with the solid circles representing individual data points. Error (solid lines) represented by 95% confidence intervals.

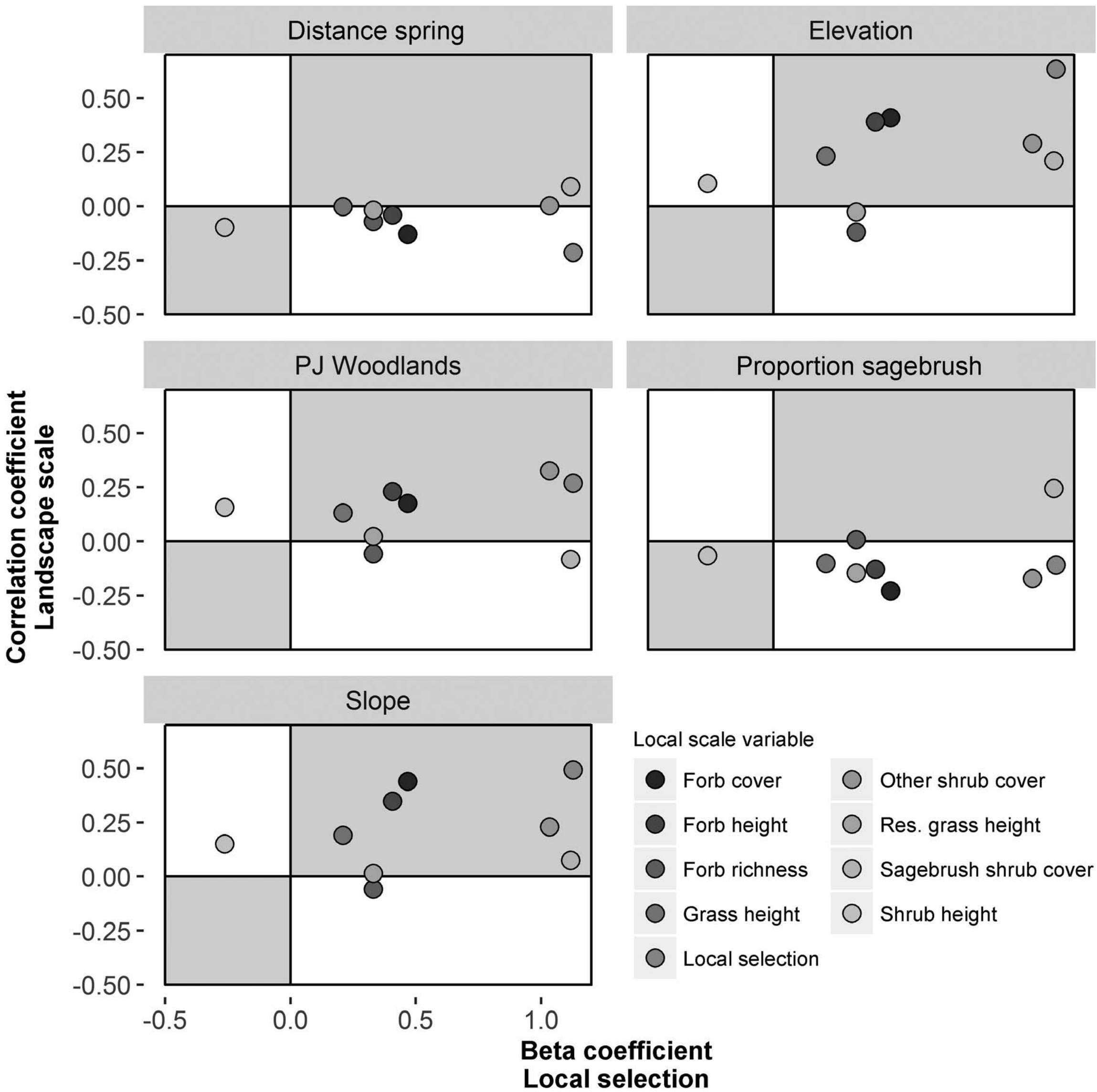

FIGURE 3. Relationship between parameter coefficients for habitat characteristics selected at the local scale (*x*-axis), and the correlation coefficients between selected habitat variables at the landscape scale and selected habitat variables at the local scale (*y*-axis), in Greater Sage-Grouse in Eureka County, Nevada, USA, 2004–2012. Points that fall in the gray sections indicate a positive association between the specified landscape-scale and local-scale habitat characteristics, whereas points that fall in the white sections indicate a negative association.

correlation between these variables is negatively related to the habitat-selection coefficient of the local-scale habitat characteristic. Landscape features (e.g., elevation, pinyon–juniper woodlands) that were positively associated with reproductive success were also positively correlated with multiple local-scale habitat characteristics that were, in turn, positively correlated with reproductive success and their habitat-selection coefficients (Figure 3). However, habitat characteristics selected at the landscape scale that were not positively associated with reproductive success (e.g., distance from spring, proportion of habitat classified as sagebrush) were predominantly negatively correlated with the habitat-selection coefficients of features selected at the local scale that improved reproductive success.

Although we did not find support for a relationship between nest survival and distance from lek ($\beta_{\mathrm{DLek}} = 0.08$, 85% CI: –0.02 to 0.19), we did find a positive relationship between distance from nearest lek and prefledging chick survival ($\beta_{\mathrm{DLek}} = 0.15$, 85% CI: 0.00–0.30; $\beta_{\mathrm{DLek}} = -0.20$, 85% CI: –0.12 to –0.28; Figure 4), which suggests that reproductive success was greater for individuals more distant from leks, peaking at ~5 km from leks. However,

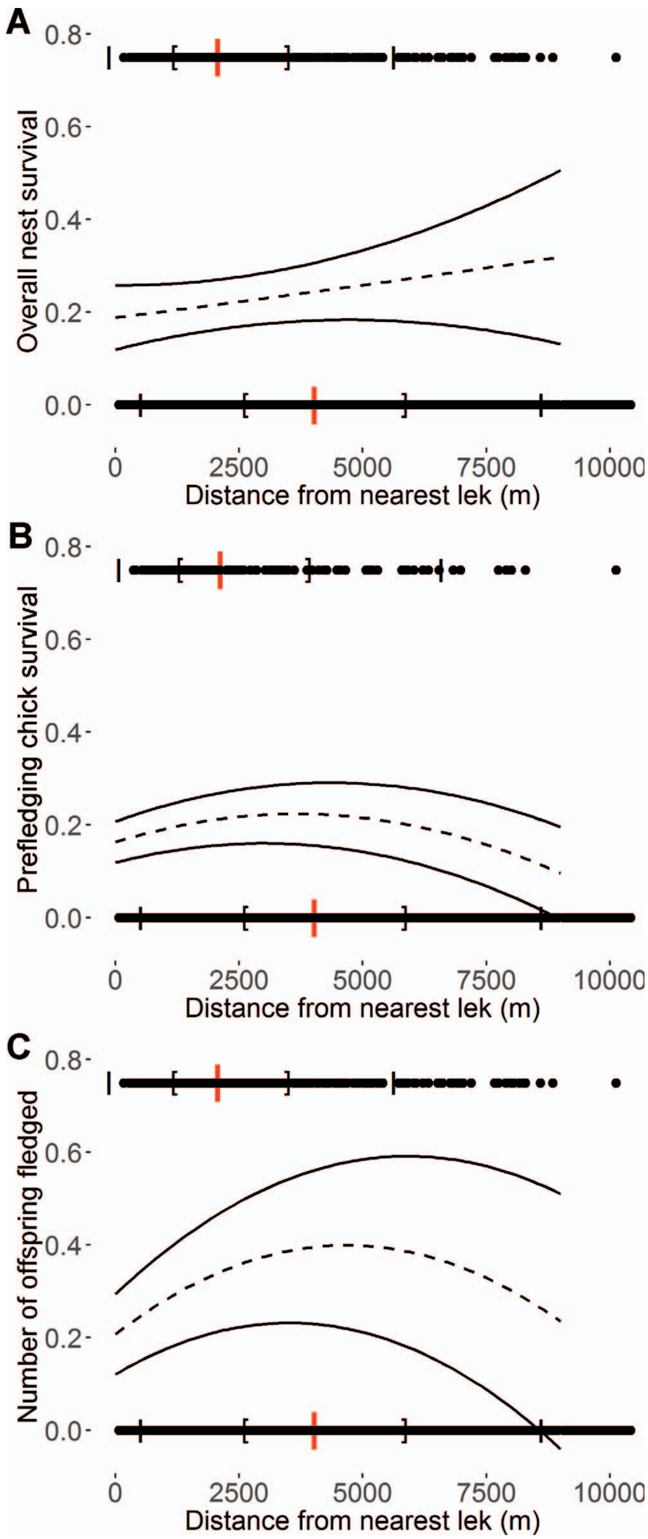

FIGURE 4. Influence of distance from nearest known active breeding lek on (**A**) probability of overall nest survival, (**B**) probability of prefledging chick survival if a nest successfully hatched, and (**C**) predicted number of offspring that survived until fledging. The distributions of nest locations (**A** and **C**: top), successful nest locations (**B**: top), and random points (**A, B, C**: bottom) are delineated by the median (red dash), first quartile ([ ), and third quartile ( ]), with the solid circles representing individual data points. Solid lines represent 95% confidence intervals.

broods were located closer to leks than expected at random (Figure 4), which resulted in a relative reduction in reproductive success of 12.5% between the average brood location and random location.

DISCUSSION

Consistent with our a priori hypothesis, we found that habitat selection at the local scale appeared to be adaptive, in that female nest-site selection at this scale was correlated with reproductive success. This suggested that female sage-grouse made adaptive breeding decisions despite recent habitat degradation and population declines (Blomberg et al. 2012). However, we also found that nest-site selection at the landscape scale did not predict reproductive success. Although we found that selection for certain individual habitat characteristics at the landscape scale was positively associated with reproductive success (e.g., selection of mid-elevation, mid-slope habitats), individuals disproportionately selected other habitat characteristics associated with reduced annual reproductive success (e.g., selection of nest sites near water sources and leks), which counteracted the adaptive behaviors. Additionally, we found that use of habitat characteristics selected at the landscape scale that were positively associated with reproductive success was also positively associated with the selection coefficients for local-scale habitat characteristics, which suggests that individuals may select habitats at the landscape scale as a function of local-scale habitat conditions. Conversely, use of landscape-scale habitat characteristics that were not positively associated with reproductive success was mostly negatively correlated with local-scale habitat characteristics. Together, these findings suggest that at the landscape scale, nest-site selection may be influenced by more than the increased reproductive success associated with certain habitat characteristics, or that there is a landscape-level pattern to local-scale habitat characteristics.

Nesting farther from leks appeared to result in higher reproductive success. This may have been related to density-dependent processes such as food competition among chicks of different broods. For example, Gregg and Crawford (2009) reported that areas with high densities of hatchlings near leks had lower survival. Furthermore, the consistent presence of leks across space and time allows nest predators to respond numerically or functionally to areas of greater prey availability (i.e. sage-grouse and their nests or offspring). Similarly, the negative relationship between distance from springs and reproductive success may also be related to increased mammalian nest-predator abundance, most notably coyotes (*Canis latrans*), which are positively associated with water availability (Kozlowski et al. 2008). We also speculate that natural selection has possibly not had enough time to fully modify nesting habitat selection in response to recent habitat degradation in this system (e.g., expansion of exotic grasslands or more frequent and persistent drought; Blomberg et al. 2012, 2014b). Furthermore, the time lag for natural selection to modify habitat selection is likely influenced by temporal heterogeneity in environmental conditions that influence fitness (Borash et al. 1998), such as drought conditions (Blomberg et al. 2012) and species-specific barriers to dispersal (e.g., natal site fidelity; Jahner et al. 2016).

We also found that across scales, nest-site selection was more predictive (Figures 1 and 2) of early chick survival than of nest survival, which suggests that females' selection of nesting habitat is based primarily on its qualities as early brood-rearing habitat (e.g., diverse food availability; Blomberg et al. 2013). We speculate that this relationship may be related both to brood-rearing habitat being limited in overall area compared with other habitats in our study system (Atamian et al. 2010) and offspring survival being negatively associated with the distance required to reach brood-rearing habitats. Therefore, we would expect females to prefer nesting habitats in proximity to brood-rearing habitats.

In summary, we found that female sage-grouse tended to favor habitat characteristics for nesting that increased their production of fledged young, even though some of these habitat variables did not positively influence nest success. The overall direction of the effect of habitat selection on reproductive success, however, varied depending on the scale of the analysis; selection for local vegetation features substantially improved reproductive success, whereas selection for landscape-scale habitat characteristics was not associated with reproductive success. We speculate that at finer scales, females selected habitat features that appeared to improve reproductive success; but at larger scales, habitat selection could be constrained by life history (e.g., site fidelity) or by limitations on the locations (e.g., distance from leks) or availability of better nest sites (Zimmerman et al. 2007), which limited the potential for an individual to select habitat that optimized reproductive success.

Conservation implications. Our results demonstrate the utility of using models of habitat selection along with models of multiple reproductive rates to better inform management of species of conservation concern. This approach to inference allows practitioners to assess relationships between fecundity and individual habitat characteristics as well as overall patterns in habitat use at multiple scales, which provides information regarding the habitat characteristics (e.g., forb biomass, shrub cover) and geographic areas (e.g., along an elevation gradient) that are critical for population sustainability. This approach, ultimately, allows managers to move beyond designating areas as critical habitat solely on the basis of observed species occupancies, by incorporating measures of reproductive success, which should provide increased power and resolution for ranking habitat in terms of both its importance to an individual species and its management priority.

A nontrivial byproduct of these analyses was revisiting theory about the formation of leks (Bradbury and Gibson 1983, Beehler and Foster 1988) and the innate influence of lek location on sage-grouse habitat use and demography. Although we reported that individual habitat selection was only slightly affected by the spatial distribution of breeding leks, individuals were spatially constrained by the location of leks, which ultimately influenced their reproductive success; this suggests that restoration of currently unoccupied habitats with the goal of enticing sage-grouse to disperse may be unsuccessful because this approach ignores sage-grouse behavior (e.g., site fidelity; Jahner et al. 2016). We propose that geographic areas with active breeding leks in habitats composed of the selected habitat characteristics associated with high reproductive success should be candidates for designation as critical habitat. Other areas near active leks that are composed of less selected habitat, however, may be candidates for conservation actions, because these areas may be required to maintain gene flow among populations (Davis et al. 2015, Oyler-McCance et al. 2015).

Contrary to conventional wisdom (Baruch-Mordo et al. 2013), we found no negative influence of pinyon–juniper woodland on either nest or offspring survival (Supplemental Material Figure S12). Furthermore, our results suggest that areas with low amounts of pinyon–juniper woodland were more likely to be used as nesting habitat than similar areas with no pinyon–juniper woodland in the surrounding landscape. This finding is most likely driven by female selection for habitat characteristics associated with productive mountain shrub communities, which often occur near pinyon–juniper woodlands in our study system, and not selection for pinyon–juniper woodland itself (Figure 3). This result does, however, provide additional support for the hypothesis that sage-grouse prefer heterogeneous shrub communities over a homogeneous sagebrush environment. Most importantly, it also suggests that the presence of pinyon–juniper woodland alone does not ensure habitat avoidance or the loss of reproductive performance. We suspect that other factors, such as predator community composition or abundance, and habitat community composition, may interact with the effects of proximity to pinyon–juniper. Our results did indicate that areas comprising moderate to high amounts of pinyon–juniper (>30% of surrounding land cover) were not suitable as nesting habitat, which is most likely related to the loss of the shrub and forb communities in these areas (Miller et al. 2000). This suggests that in systems such as this, large-scale pinyon–juniper removal projects may not be a cost-effective means to improve sage-grouse reproductive success and may, in fact, reduce population sustainability through encroachment of invasive exotic grasses (Roundy et al. 2014). Our results, however, do not address the potential negative effect of sagebrush habitat lost as a result of pinyon–juniper expansion into high-elevation sagebrush communities. We speculate that if high-elevation mountain shrub communities are limited, their loss could be detrimental to sage-grouse populations within the Great Basin.

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Ethics statement:
Capture and handling of sage-grouse were approved by the University of Nevada Reno Institutional Animal Care and Use Committee (protocol nos. A02/03-22, A05/06-22, A07/08-22, A09/10-22).

Author contributions:
D.G., E.B., M.A., and J.S. conceived the study and designed the methods. D.G., E.B., and M.A. collected data and conducted the research. D.G. analyzed the data. D.G., E.B., and J.S. wrote the paper. J.S. contributed substantial resources.

LITERATURE CITED

Aldridge, C. L., and M. S. Boyce (2007). Linking occurrence and fitness to persistence: Habitat-based approach for endangered Greater Sage-Grouse. Ecological Applications 17:508–526.

Aldridge, C. L., D. J. Saher, T. M. Childers, K. E. Stahlnecker, and Z. H. Bowen (2012). Crucial nesting habitat for Gunnison Sage-Grouse: A spatially explicit hierarchical approach. Journal of Wildlife Management 76:391–406.

Arnold, T. W. (2010). Uninformative parameters and model selection using Akaike's Information Criterion. Journal of Wildlife Management 74:1175–1178.

Atamian, M. T., J. S. Sedinger, J. S. Heaton, and E. J. Blomberg (2010). Landscape-level assessment of brood rearing habitat for Greater Sage-Grouse in Nevada. Journal of Wildlife Management 74:1533–1543.

Baruch-Mordo, S., J. S. Evans, J. P. Severson, D. E. Naugle, J. D. Maestas, J. M. Kiesecker, M. J. Falkowski, C. A. Hagen, and K. P. Reese (2013). Saving sage-grouse from the trees: A proactive solution to reducing a key threat to a candidate species. Biological Conservation 167:233–241.

Bates, D. M., and M. Maechler (2010). lme4: Linear mixed-effects models using S4 classes. R package version 0.999375-36/r1083. http://R-Forge.R-project.org/projects/lme4/

Battin, J. (2004). When good animals love bad habitats: Ecological traps and the conservation of animal populations. Conservation Biology 18:1482–1491.

Beehler, B. M., and M. S. Foster (1988). Hotshots, hotspots, and female preference in the organization of lek mating systems. The American Naturalist 131:203–219.

Blomberg, E. J., D. Gibson, M. T. Atamian, and J. S. Sedinger (2014a). Individual and environmental effects on egg allocations of female Greater Sage-Grouse. The Auk: Ornithological Advances 131:507–523.

Blomberg, E. J., D. Gibson, and J. S. Sedinger (2015). Biases in nest survival associated with choice of exposure period: A case study in North American upland game birds. The Condor: Ornithological Applications 117:577–588.

Blomberg, E. J., S. R. Poulson, J. S. Sedinger, and D. Gibson (2013). Prefledging diet is correlated with individual growth in Greater Sage-Grouse (*Centrocercus urophasianus*). The Auk 130:715–724.

Blomberg, E. J., J. S. Sedinger, M. T. Atamian, and D. V. Nonne (2012). Characteristics of climate and landscape disturbance influence the dynamics of Greater Sage-Grouse populations. Ecosphere 3(6):55.

Blomberg, E. J., J. S. Sedinger, D. Gibson, P. S. Coates, and M. L. Casazza (2014b). Carryover effects and climatic conditions influence the postfledging survival of Greater Sage-Grouse. Ecology and Evolution 4:4488–4499.

Borash, D. J., A. G. Gibbs, A. Joshi, and L. D. Mueller (1998). A genetic polymorphism maintained by natural selection in a temporally varying environment. The American Naturalist 151:148–156.

Boyce, M. S., and L. L. McDonald (1999). Relating populations to habitats using resource selection functions. Trends in Ecology & Evolution 14:268–272.

Bradbury, J. W., and R. M. Gibson (1983). Leks and mate choice. In Mate Choice (P. Bateson, Editor). Cambridge University Press, Cambridge, UK. pp. 109–138.

Burnham, K. P., and D. R. Anderson (2002). Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach, second edition. Springer, New York, NY, USA.

Canfield, R. H. (1941). Application of the line interception method in sampling range vegetation. Journal of Forestry 39:388–394.

Chalfoun, A. D., and T. E. Martin (2007). Assessments of habitat preferences and quality depend on spatial scale and metrics of fitness. Journal of Applied Ecology 44:983–992.

Coates, P. S., M. L. Casazza, E. J. Blomberg, S. C. Gardner, S. P. Espinosa, J. L. Yee, L. Wiechman, and B. J. Halstead (2013). Evaluating Greater Sage-Grouse seasonal space use relative to leks: Implications for surface use designations in sagebrush ecosystems. Journal of Wildlife Management 77:1598–1609.

Connelly, J. W., C. A. Hagen, and M. A. Schroeder (2011). Characteristics and dynamics of Greater Sage-Grouse populations. In Greater Sage-Grouse: Ecology and Conservation of a Landscape Species and Its Habitats (S. T. Knick and J. W. Connelly, Editors). Studies in Avian Biology 38:53–67.

Daubenmire, R. F. (1959). Canopy coverage method of vegetation analysis. Northwest Science 33:43–64.

Davis, D. M., K. P. Reese, S. C. Gardner, and K. L. Bird (2015). Genetic structure of Greater Sage-Grouse (*Centrocercus urophasianus*) in a declining, peripheral population. The Condor: Ornithological Applications 117:530–544.

Dinkins, J. B., M. R. Conover, C. P. Kirol, J. L. Beck, and S. N. Frey (2014). Greater Sage-Grouse (*Centrocercus urophasianus*) select habitat based on avian predators, landscape composition, and anthropogenic features. The Condor: Ornithological Applications 116:629–642.

Doherty, K. E., D. E. Naugle, and B. L. Walker (2010). Greater Sage-Grouse nesting habitat: The importance of managing at multiple scales. Journal of Wildlife Management 74:1544–1553.

Feary, D. A., G. R. Almany, M. I. McCormick, and G. P. Jones (2007). Habitat choice, recruitment and the response of coral reef fishes to coral degradation. Oecologia 153:727–737.

Franklin, A. B., D. R. Anderson, R. J. Gutiérrez, and K. P. Burnham (2000). Climate, habitat quality, and fitness in Northern Spotted Owl populations in northwestern California. Ecological Monographs 70:539–590.

Fretwell, S. D., and H. L. Lucas, Jr. (1969). On territorial behavior and other factors influencing habitat distribution in birds. I. Theoretical development. Acta Biotheoretica 19:16–36.

Ganter, B., and F. Cooke (1998). Colonial nesters in a deteriorating habitat: Site fidelity and colony dynamics of Lesser Snow Geese. The Auk 115:642–652.

Garton, E. O., J. W. Connelly, J. S. Horne, C. A. Hagen, A. Moser, and M. A. Schroeder (2011). Greater Sage-Grouse population dynamics and probability of persistence. In Greater Sage-Grouse: Ecology and Conservation of a Landscape Species and Its Habitats (S. T. Knick and J. W. Connelly, Editors). Studies in Avian Biology 38:293–381.

Gibson, D., E. J. Blomberg, M. T. Atamian, and J. S. Sedinger (2015). Observer effects strongly influence estimates of daily nest survival probability but do not substantially increase rates of nest failure in Greater Sage-Grouse. The Auk: Ornithological Advances 132:397–407.

Gibson, D., E. J. Blomberg, and J. S. Sedinger (2016). Evaluating vegetation effects on demographic rates: The role of plant phenology and sampling bias. Ecology and Evolution 6:3621–3631.

Gibson, R. M. (1992). Lek formation in Sage Grouse: The effect of female choice on male territory settlement. Animal Behaviour 43:443–450.

Gregg, M. A., and J. A. Crawford (2009). Survival of Greater Sage-Grouse chicks and broods in the northern Great Basin. Journal of Wildlife Management 73:904–913.

Gregg, M. A., J. A. Crawford, M. S. Drut, and A. K. Delong (1994). Vegetational cover and predation of Sage Grouse nests in Oregon. Journal of Wildlife Management 58:162–166.

Hagen, C. A., J. W. Connelly, and M. A. Schroeder (2007). A meta-analysis of Greater Sage-Grouse *Centrocercus urophasianus* nesting and brood-rearing habitats. Wildlife Biology 13 (Supplement 1):42–50.

Harrell, F. E., Jr., K. L. Lee, and D. B. Mark (1996). Multivariable prognostic models: Issues in developing models, evaluating assumptions and adequacy, and measuring and reducing errors. Statistics in Medicine 15:361–387.

Hebblewhite, M., and E. Merrill (2008). Modelling wildlife–human relationships for social species with mixed-effects resource selection models. Journal of Applied Ecology 45:834–844.

Jahner, J. P., D. Gibson, C. L. Weitzman, E. J. Blomberg, J. S. Sedinger, and T. L. Parchman (2016). Fine-scale genetic structure among Greater Sage-Grouse leks in central Nevada. BMC Evolutionary Biology 16:127.

Johnson, D. H. (1980). The comparison of usage and availability measurements for evaluating resource preference. Ecology 61:65–71.

Jones, J. (2001). Habitat selection studies in avian ecology: A critical review. The Auk 118:557–562.

Kautz, R., R. Kawula, T. Hoctor, J. Comiskey, D. Jansen, D. Jennings, J. Kasbohm, F. Mazzotti, R. McBride, L. Richardson, and K. Root (2006). How much is enough? Landscape-scale conservation for the Florida panther. Biological Conservation 130:118–133.

Kazantzidis, S., V. Goutner, M. Pyrovetsi, and A. Sinis (1997). Comparative nest site selection and breeding success in two sympatric Ardeids, Black-crowned Night-Heron (*Nycticorax nycticorax*) and Little Egret (*Egretta garzetta*) in the Aries Delta, Macedonia, Greece. Colonial Waterbirds 20:505–517.

Kirol, C. P., J. L. Beck, J. B. Dinkins, and M. R. Conover (2012). Microhabitat selection for nesting and brood-rearing by the Greater Sage-Grouse in xeric big sagebrush. The Condor 114: 75–89.

Kirol, C. P., J. L. Beck, S. V. Huzurbazar, M. J. Holloran, and S. N. Miller (2015). Identifying Greater Sage-Grouse source and sink habitats for conservation planning in an energy development landscape. Ecological Applications 25:968–990.

Knick, S. T., and J. W. Connelly (2011). Greater Sage-Grouse and sagebrush: An introduction to the landscape. In Greater Sage-Grouse: Ecology and Conservation of a Landscape Species and Its Habitats (S. T. Knick and J. W. Connelly, Editors). Studies in Avian Biology 38:1–9.

Kolbe, J. J., and F. J. Janzen (2001). The influence of propagule size and maternal nest-site selection on survival and behaviour of neonate turtles. Functional Ecology 15:772–781.

Kozlowski, A. J., E. M. Gese, and W. M. Arjo (2008). Niche overlap and resource partitioning between sympatric kit foxes and coyotes in the Great Basin Desert of western Utah. The American Midland Naturalist 160:191–208.

Lockyer, Z. B., P. S. Coates, M. L. Casazza, S. Espinosa, and D. J. Delehanty (2015). Nest-site selection and reproductive success of Greater Sage-Grouse in a fire-affected habitat of northwestern Nevada. Journal of Wildlife Management 79: 785–797.

Lukacs, P. M., V. J. Dreitz, F. L. Knopf, and K. P. Burnham (2004). Estimating survival probabilities of unmarked dependent young when detection is imperfect. The Condor 106:926–931.

Martin, T. E. (2001). Abiotic vs. biotic influences on habitat selection of coexisting species: Climate change impacts? Ecology 82:175–188.

Miller, R. F., T. J. Svejcar, and J. A. Rose (2000). Impacts of western juniper on plant community composition and structure. Journal of Range Management 53:574–585.

Morris, D. W. (1989). Density-dependent habitat selection: Testing the theory with fitness data. Evolutionary Ecology 3:80–94.

Morris, D. W. (2011). Adaptation and habitat selection in the eco-evolutionary process. Proceedings of the Royal Society of London, Series B 278:2401–2411.

Newlon, K. R., and V. A. Saab (2011). Nest-site selection and nest survival of Lewis's Woodpecker in aspen riparian woodlands. The Condor 113:183–193.
Newton, I. (1991). Habitat variation and population regulation in sparrowhawks. Ibis 133 (Supplement S1):76–88.
Nicolai, C. A., and J. S. Sedinger (2012). Trade-offs between offspring fitness and future reproduction of adult female Black Brent. Journal of Animal Ecology 81:798–805.
Orians, G. H., and J. F. Wittenberger (1991). Spatial and temporal scales in habitat selection. The American Naturalist 137:29–49.
Oyler-McCance, S. J., R. S. Cornman, K. L. Jones, and J. A. Fike (2015). Genomic single-nucleotide polymorphisms confirm that Gunnison and Greater sage-grouse are genetically well differentiated and that the Bi-State population is distinct. The Condor: Ornithological Applications 117:217–227.
Pierotti, R. (1982). Habitat selection and its effect on reproductive output in the Herring Gull in Newfoundland. Ecology 63: 854–868.
R Development Core Team (2012). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
Roundy, B. A., R. F. Miller, R. J. Tausch, K. Young, A. Hulet, B. Rau, B. Jessop, J. C. Chambers, and D. Eggett (2014). Understory cover responses to piñon–juniper treatments across tree dominance gradients in the Great Basin. Rangeland Ecology & Management 67:482–494.
Sedinger, J. S. (2007). Improving understanding and assessment of Greater Sage-Grouse populations. University of Idaho College of Natural Resources Experiment Station Bulletin 88: 43–56.
U.S. Department of the Interior (2014). Listing endangered and threatened species and designating critical habitat; implementing changes to the regulations for designating critical habitat. Federal Register 79:27066–27078.
White, G. C., and K. P. Burnham (1999). Program MARK: Survival estimation from populations of marked animals. Bird Study 46 (Supplement):S120–S139.
Zhou, S. (2002). Estimating parameters of derived random variables: Comparison of the delta and parametric bootstrap methods. Transactions of the American Fisheries Society 131: 667–675.
Zuur, A. F., E. N. Ieno, N. J. Walker, A. A. Saveliev, and G. M. Smith (2009). Mixed Effects Models and Extensions in Ecology with R. Springer, New York, NY, USA.

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Evaluating vegetation effects on animal demographics: the role of plant phenology and sampling bias

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Abstract

Plant phenological processes produce temporal variation in the height and cover of vegetation. Key aspects of animal life cycles, such as reproduction, often coincide with the growing season and therefore may inherently covary with plant growth. When evaluating the influence of vegetation variables on demographic rates, the decision about when to measure vegetation relative to the timing of demographic events is important to avoid confounding between the demographic rate of interest and vegetation covariates. Such confounding could bias estimated effect sizes or produce results that are entirely spurious. We investigated how the timing of vegetation sampling affected the modeled relationship between vegetation structure and nest survival of greater sage-grouse (*Centrocercus urophasianus*), using both simulated and observational data. We used the height of live grasses surrounding nests as an explanatory covariate, and analyzed its effect on daily nest survival. We compared results between models that included grass height measured at the time of nest fate (hatch or failure) with models where grass height was measured on a standardized date – that of predicted hatch date. Parameters linking grass height to nest survival based on measurements at nest fate produced more competitive models, but slope coefficients of grass height effects were biased high relative to truth in simulated scenarios. In contrast, measurements taken at predicted hatch date accurately predicted the influence of grass height on nest survival. Observational data produced similar results. Our results demonstrate the importance of properly considering confounding between demographic traits and plant phenology. Not doing so can produce results that are plausible, but ultimately inaccurate.

Introduction

For many animals, vegetation represents an important habitat feature, and thus as a component of the environment plays a critical role in affecting both ecological and evolutionary processes. For this reason, understanding how vegetation structure and composition affect animal demographics, individual fitness, and population growth, is a key to both basic and applied research. Modern methods of demographic analysis (e.g., White and Burnham 1999; Dinsmore et al. 2002; Kery and Schaub 2012) have substantially improved our ability to understand the influence of vegetation and other environmental variables on demographic rates.

Many demographic analyses are ultimately rooted in regression-based techniques, where a response variable (the demographic rate) is explained as a function of one or more predictor terms (e.g., a vegetation covariate) based on a modeled relationship (e.g., a logistic regression). Various criteria, such as *P*-values, AIC scores, or credible intervals, are used to establish whether the modeled relationship has statistical support, which in turn is presumed to indicate a true biological relationship between the vegetation metric and the demographic rate. Central to this conclusion is an implicit assumption that the covariance between the covariate and response variable is driven by the true underlying mechanism, and not by sampling bias associated with the design of data collection. In the case of a relationship between an environmental variable (e.g., vegetation height) and a demographic rate, we therefore assume that a positive or negative association between the two was driven by the influence of vegetation on the demographic rate, and not by a spurious correlation caused by measurement error. Such error is often associated with imprecision of an instrument or operator, but it can also be influenced by selection bias and preferential sampling (Diggle et al. 2010), which may lead to biased parameter estimates (Muff et al. 2015).

The implications of random sampling error for model convergence, fit, and accuracy have been discussed thoroughly in general (Anderson and Gerbing 1984) and in ecological scenarios specifically (Walters and Ludwig 1981; Ostermiller and Hawkins 2004; Staples et al. 2004). These efforts collectively suggest that most issues regarding random sampling error can be largely ameliorated through increased sample sizes. In contrast, directional (nonrandom) sampling error is predominantly ignored in ecological contexts but can substantially bias results. This may be particularly true for longitudinal data where loss of individuals through time (e.g., through mortality) may affect the distribution of the remaining sampled population (Alexander et al. 1986; Goodman and Blum 1996). More importantly, spurious relationships can be observed between measured variables and demographic rates if individual sampling is somehow associated with the individual's attrition (i.e., how long an individual remains in the sample; Goodman and Blum 1996).

During the growing season, plant phenology produces temporal variation in vegetation composition, structure, and total biomass. For many animals, key aspects of the annual life cycle often coincide with the growing season (Jones and Cresswell 2010; Miller-Rushing et al. 2010), and are therefore inherently coincident with plant growth. In birds, nesting represents the central element of annual reproduction, and parents often time nesting to produce young at the height of the growing season when food resources are most abundant (Nussey et al. 2005). In studies of avian nesting ecology, investigators frequently measure a suite of vegetation features to provide covariates for nest survival analysis (e.g., Pitman et al. 2005; Gregory et al. 2011; Davis et al. 2014). Typically vegetation is sampled subsequent to nest fate (success or failure) to minimize disturbance at active nests. However, there is inconsistency among studies with respect to how successful and failed nests are sampled; some investigators elect to sample vegetation at or near the timing of fate, whereas others sample on a standardized date, such as the predicted date of hatch (see Fig. 1). Given that nesting often coincides with plant growth, the decision about when nests are sampled is potentially very important, as sampling at nest fate results in successful nests being sampled later in the growing season, on average, than failed nests.

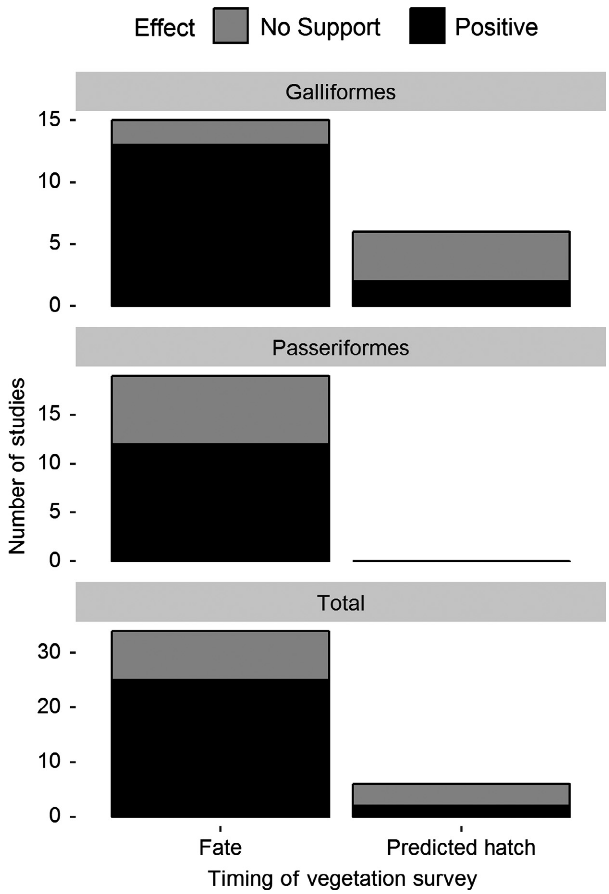

Figure 1. Summary of literature review assessing variation in study design for studies measuring nest site vegetation for Galliformes (top panel) and Passeriformes (center panel) in grasslands and shrublands. Two common survey protocols included sampling nest vegetation at nest fate (i.e., hatch or failure) or on a predicted hatch date, and publications reported positive (black) and no support for an effect (gray) of grass.

Our goal was to evaluate how timing of vegetation surveys influenced the ability to detect the effects of vegetation covariates on nest survival. We addressed this question using simulated data, as well as observational data collected on nesting greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse), using average grass height surrounding a nest as an example vegetation covariate. For both simulated and real data sets, we assessed statistical relationships between average grass height covariates and nest survival, and compared results for models that included grass height measured at nest fate with models that included grass height measured at a predicted hatch date. We also compared three different scenarios: (1) grass height had no influence on nest survival, (2) grass height positively influenced nest survival, and (3) grass height negatively influenced nest survival, to further assess the implications of study design for the direction and magnitude of modeled effects. Although we use grass height as an example, the principle we address here applies to any environmental variable that covaries temporally, or is otherwise confounded, with a demographic fate.

Methods

Literature review

We performed an informal literature review for ground-nesting birds nesting in grasslands and shrublands to determine the frequency with which researchers measured vegetation at either the observed date of nest fate, or on a predicted hatch date. We focused exclusively on grassland and shrubland literature as the general hypothesis in these regions is that increased herbaceous ground cover should positively influence nesting success through visual concealment (see review). We only considered publications that assessed the influence of grass height or cover on nest success. We used Google Scholar (http://scholar.google.com) to search for the following key words: nest success, nest survival, grass height, grassland, shrub-steppe, and we further explored manuscripts that were cited within publications that met the necessary criteria. We excluded publications in which we could not determine the relative timing of the vegetation survey, as well as studies outside of grasslands and shrublands. We report the timing of the vegetation survey (at fate vs. predicted hatch), species, and the general direction of the reported effect of grass height or cover for each publication.

Simulated Data

Encounter histories

We developed three different scenarios that varied only in how nest site vegetation (i.e., grass height) influenced the underling survival probability of nests within each data set (i.e., no effect, positive effect, and negative effect). We created 500 replicate encounter histories under each scenario, where each consisted of a maximum of 400 nests that were monitored over an 80-day nesting season. The nesting season consisted of two phases: (1) nest initiation and (2) nest activity. The nest initiation phase was 40 days long, and we allowed 10 nests to be initiated each day. The nest activity phase lasted the entire 80-day nesting season, or as long as at least one simulated nest remained active. The length of exposure varied among nests, began with the interval immediately after a nest was initiated, and ended when the nest failed or hatched, whichever came first. Hatch occurred after 37 days of nonfailure. We also included a nest observation term, where each nest had a 0.33 probability of being visited during a given day, and its current nesting state (active/terminated) accurately assigned. The number of nests within each encounter history was therefore variable ($\bar{x}_{\text{Ind}}$ = 328.43, SD = 7.55), because some nests failed prior to detection. This allowed us to integrate a realistic degree of stochasticity in the age at which nests entered the sample, which was related to imperfect nest detection. We made the following additional constraints designed to meet model specifications for nest survival models (Dinsmore and Dinsmore 2007). The date of nest discovery was the first observation of an active nest. Nests that failed before an initial observation were censored from the history because they were never discovered. Nests that failed before hatching, but that were observed at least once, were assigned a "last alive" date that coincided with the most recent date the nest was both active and visited by an observer. Failed nests were randomly assigned a "last check" date, which occurred 1–3 days after the nest's true failure date. Nests that hatched were assigned both a last alive and a last check date that were equal to the hatch date of the nest. The average daily nest survival probability (DNS) across all nests in each scenario was constrained to be 0.96, such that DNS was constant among nests within the 'no-effect' scenario and therefore was independent of grass height. For the 'positive-effect' and 'negative-effect' scenarios, we allowed DNS to vary among each nest, i, as a function of a linear effect (on the logit scale) of grass height

$$\text{logit}(\text{DNS}_i) = (\beta_{\text{Intercept}} + \beta_{\text{grass}} \times \text{Initial Grass Height}_i)$$

where for the 'positive-effect' scenario, β_{grass} = 0.25, and for the 'negative-effect' scenario, $\beta_{\text{grass}} = -0.25$. Initial grass height was the simulated grass height at each nest (described below). $\beta_{\text{Intercept}}$ was given as

$$\beta_{\text{Intercept}} = \text{logit}(0.96).$$

Simulated grass height covariate

We assigned each nest an initial grass height value, which was calculated as a function of nest initiation date (ID), daily grass growth (Daily Growth), base grass height (Base), and random variation (ε).

$$\text{Initial Grass Height}_i = \text{Base}_i + \text{Daily Growth}_i \times \text{ID}_i + \varepsilon_i.$$

We estimated grass growth between nest initiation and hatch using grass growth rates given by Hausleitner et al. (2005), who reported that average grass height at sage-grouse nests changed from 10.0 cm at nest initiation to 15.6 cm at nest hatch, or approximately 0.156 cm/day (mean no. days = 36). We assumed that the mean initiation grass height (10.0 cm) corresponded with the mean nest initiation date (ID = 20), and constrained grass height to grow by 0.156 cm/day, which yielded initial grass heights ranging from 7.04 to 13.11 cm corresponding to nests initiated on days 1–40. We also incorporated a stochastic feature for each nest, where variation was drawn from a normal distribution ($\bar{x}_\varepsilon$ = 0 cm, SD = 2 cm) and added to the initial grass height measurement. We then allowed grass to grow throughout the nest activity phase, and assigned each nest two additional grass height values: (1) grass height at the date associated with the date last checked (hereafter, fate) and (2) grass height 37 days after nest initiation (hereafter, hatch). These latter values reflected grass heights that would have been recorded if vegetation surveys were conducted at nest fate or on a predicted hatch date, respectively.

Real Data

We used data collected from nests monitored from 2004 to 2012 in Eureka County, NV, USA (Gibson et al. 2015) to further assess the potential bias in assessing grass height effects associated with timing of vegetation surveys. We measured grass height at all nest sites within 3 days of either the predicted or actual date of hatch. Predicted hatch dates for failed nests were estimated by floating eggs using methods described in detail by Blomberg et al. (2013) and Gibson et al. (2015). Grass height was sampled along 10 m intersecting transects, centered at the nest bowl (Gregg et al. 1994). We used five 20 × 50 cm Daubenmire frames placed along each transect and measured the height (cm) of the nearest representative of live grass to the northeast corner in each frame. We averaged these measurements to estimate mean live grass in the plot associated with each nest. To estimate grass height at nest fate (GH_{Fate}), we regressed the measured average grass heights ($\text{GH}_{\text{Survey}}$) against the ordinal date of the vegetation survey ($\text{Date}_{\text{Survey}}$) to develop a grass height correction factor (b_1) based on the difference between the ordinal date a nest terminated ($\text{Date}_{\text{Fate}}$) and $\text{Date}_{\text{Survey}}$

$$\text{GH}_{\text{Fate}} = \text{GH}_{\text{Survey}} - (\text{Date}_{\text{Survey}} - \text{Date}_{\text{Fate}}) \times b_1$$

which yielded predicted grass height measurements as though surveys had been completed immediately after nest fate, assuming a linear growth rate for grass.

Analysis and model selection

We used RMark (Laake 2013) in R (R Core Team 2012) to call the nest survival module in the program MARK (White and Burnham 1999), which we used to estimate the effect size of grass height on daily nest survival for each simulated and real data set. For both simulated and real data sets, we considered three models: (1) constant DNS (null model); (2) DNS varied by the average grass height on the date of recorded nest fate; and (3) DNS varied by the average grass height on the predicted hatch date. We used an information theoretic approach to evaluate support for candidate models (Burnham and Anderson 2002), and considered covariate effects to be meaningful if 85% confidence intervals of β coefficients did not overlap 0.0 (Arnold 2010). For the simulated scenarios, we used the mean and standard deviation of parameter estimates, as well as mean ΔAIC_c and AIC_c model weights (w_i). We focus our assessment of results on the extent to which simulated scenarios deviate from our known effect sizes, and in the case of the real data, how our inferences related to grass height effects changed depending on how we incorporated measures of nest vegetation.

Results

We reviewed 28 publications involving 19 species, and found that 22 (~79% of studies) sampled vegetation relative to nest fate, whereas six sampled vegetation relative to a predicted hatch date (Table 1). Some publications reported independent effects for multiple species, and we report effects relative to the total number of analyses (n = 45) across all publications. Of the analyses based on vegetation data sampled at nest fate, 25 (~74%) of studies reported a positive effect of grass height or cover on nest survival, while nine (~26%) analyses lacked support for an effect of grass height or cover. Of the analyses based on vegetation data sampled at predicted hatch date, two (~33%) reported a positive effect of grass height or cover on nest survival, while four (~67%) analyses lacked support for an effect of grass height or cover (Fig. 1, Table 1).

Simulated data

Our simulations suggested that measuring grass height at nest fate resulted in effect sizes that were positively biased relative to true effects. Under the negative effect of grass height on nest survival scenario, measurement at fate produced a positive effect ($\bar{\beta}_{\text{fate}}$ = 0.50, SD = 0.06) whereas measurement at hatch produced a negative effect that was close to the true effect ($\bar{\beta}_{\text{hatch}}$ = −0.23, SD = 0.06). Under the no-effect scenario, measurement at fate also produced a positive effect

Table 1. Summary of literature review assessing variation in study design for studies assessing the influence of nest site grass height or cover on nest survival for grassland or shrubland bird species. Two common survey protocols included sampling nest vegetation at nest fate (i.e., hatch or failure) or on a predicted hatch date, and publications reported positive and no support for an effect of grass. For studies that considered multiple species of bird, values in ( ) represent the number of species reported to have the specified relationship between grass height and nest survival.

Species	Timing of survey	Direction of effect	Source
Grasshopper sparrow (*Ammodramus savannarum*)	Fate	Positive	Lyons (2013)
Clay-colored sparrow (*Spizella pallida*), Savannah sparrow (*Passerculus sandwichensis*), and bobolink (*Dolichonyx oryzivorus*)	Fate	Positive (1), no support (2)	Kerns et al. (2010)
Clay-colored sparrow (*Spizella pallida*), Savannah sparrow (*Passerculus sandwichensis*), and bobolink (*Dolichonyx oryzivorus*)	Fate	Positive (3)	Winter et al. (2005)
Sprague's Pipit (*Anthus spragueii*), Savannah Sparrow (*Passerculus sandwichensis*), Baird's Sparrow (*Ammodramus bairdii*), Chestnut-collared Longspur (*Calcarius ornatus*), and Western Meadowlark (*Sturnellaneglecta*)	Fate	Positive (4), no support (1)	Davis (2005)
Brewer's Sparrows (*Spizella breweri*), Horned Lark (*Eremophila alpestris*), Sage Thrasher (*Oreoscoptes montanus*), Savannah Sparrow (*Passerculus sandwichensis*), Vesper Sparrow (*Pooecetes gramineus*), and Western Meadowlarks (*Sturnella neglecta*)	Fate	Positive (2), no support (4)	Vander Haegen et al. (2015)
Brewer's Sparrows (*Spizella breweri*), Lark Sparrows (*Chondestes grammacus*), Vesper Sparrows (*Pooecetes gramineus*), and Western Meadowlarks (*Sturnella neglecta*)	Fate	Positive	Knight et al. (2014)
Vesper sparrow (*Pooecetes gramineus*)	Fate	Positive	Sadoti et al. (2014)
Greater Prairie Chicken (*Tympanuchus cupido*)	Fate	Positive	McKee et al. (1998)
Greater Prairie Chicken (*Tympanuchus cupido*)	Fate	Positive	McNew et al. (2014)
Greater Sage-grouse (*Centrocercus urophasianus*)	Fate	Positive	Doherty et al. (2014)
Greater Sage-grouse (*Centrocercus urophasianus*)	Fate	Positive	Doherty et al. (2011)
Greater Sage-grouse (*Centrocercus urophasianus*)	Fate	Positive	Coates & Delehanty (2010)
Greater Sage-grouse (*Centrocercus urophasianus*)	Fate	No support	Kolada et al. (2009)
Greater Sage-grouse (*Centrocercus urophasianus*)	Fate	Positive	Lockyer et al. (2015)
Greater Sage-grouse (*Centrocercus urophasianus*)	Fate	Positive	Popham & Gutierrez (2003)
Greater Sage-grouse (*Centrocercus urophasianus*)	Fate	Positive	Wing (2014)
Greater Sage-grouse (*Centrocercus urophasianus*)	Fate	Positive	Kaczor et al. (2011)
Greater Sage-grouse (*Centrocercus urophasianus*)	Fate	Positive	Bell (2011)
Greater Sage-grouse (*Centrocercus urophasianus*)	Fate	Positive	Rebholz (2008)
Gunnison Sage-grouse (*Centrocercus minimus*)	Fate	Positive	Stanley et al. (2015)
Lesser Prairie Chicken (*Tympanuchus pallidicinctus*)	Fate	Positive	Pitman et al. (2005)
Greater Sage-grouse (*Centrocercus urophasianus*)	Predicted Hatch	No support	Gibson (2015)
Greater Sage-grouse (*Centrocercus urophasianus*)	Predicted Hatch	Positive	Gregg et al. (1994)
Greater Sage-grouse (*Centrocercus urophasianus*)	Predicted Hatch	No support	Davis et al. (2014)
Greater Sage-grouse (*Centrocercus urophasianus*)	Predicted Hatch	Positive	Sveum et al. (1998)
Lesser Prairie Chicken (*Tympanuchus pallidicinctus*)	Predicted Hatch	No support	Davis (2009)
Northern Bobwhite (*Colinus virginianus*)	Fate	No support	Rader et al. (2007)
Long-billed Curlew (*Numenius americanus*)	Predicted Hatch	No support	Gregory et al. (2011)

($\bar{\beta}_{\text{fate}}$ = 0.60, SD = 0.06) whereas measurement at hatch correctly predicted no effect ($\bar{\beta}_{\text{hatch}}$ = 0.00, SD = 0.06). Under the positive-effect scenario, measurement at fate correctly identified a positive effect, but the effect magnitude was more than three times greater ($\bar{\beta}_{\text{fate}}$ = 0.80, SD = 0.06) than the true effect size, which again was correctly approximated by measurement at hatch ($\bar{\beta}_{\text{hatch}}$ = 0.23, SD = 0.06); Fig. 2).

In addition to producing biased effect sizes, models based on measurement at fate were better supported in model selection, substantially outcompeting the grass height at hatch models (which accurately described the

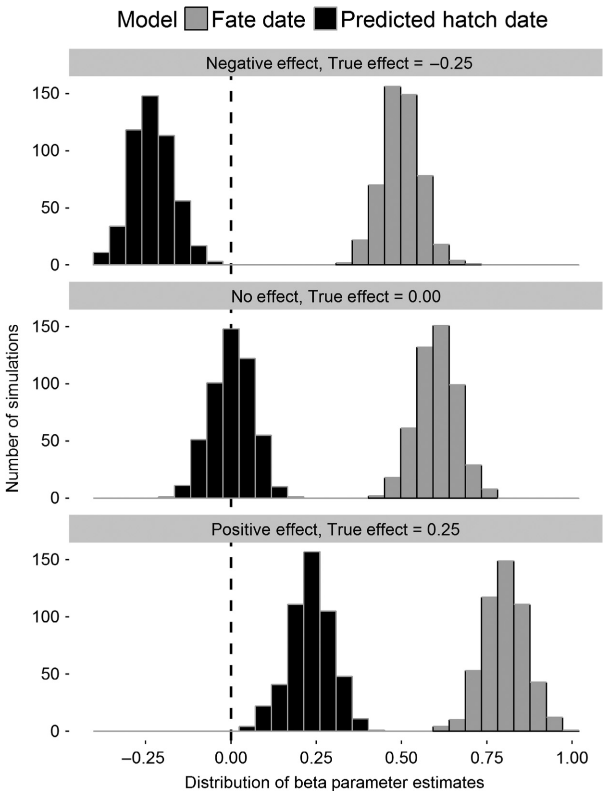

Figure 2. The distribution of parameter coefficient estimates for nest survival models that differed based on whether vegetation was measured on a predicted nest hatch date (black) or on the date of nest fate (gray). Three scenarios were considered where grass height reduced nest survival (top panel), had no influence on nest survival (center panel) and where grass height positively influenced nest survival (bottom panel). Each scenario was evaluated using 500 iterations of simulated nest survival data.

true demographic mechanism in the data; Tables 2–4). Thus, our simulations indicate that models of covariate effects based on measurement at fate will be favored as predictive based on established model selection procedures (Burnham and Anderson 2002), even in situations where no effect of the covariate exists in nature. Furthermore, our results suggest that measuring grass height at nest fate may result in a false-positive reporting of effects when no effect exists (Fig. 2).

Although we found that parameter coefficients were biased relative to timing of vegetation sampling, mean estimates of daily nest survival (DNS) were identical for models that measured vegetation at fate ($\overline{\mathrm{DNS}}_{\mathrm{fate}} = 0.964$, SE = 0.002) and hatch ($\overline{\mathrm{DNS}}_{\mathrm{hatch}} = 0.964$, SE = 0.002), and also matched our simulated conditions (DNS = 0.96). Thus bias associated with measurement error was associated with parameter coefficients and predicted DNS for a specific covariate value, but estimates of nest survival for the sample as a whole were unaffected. Or in other words, the mean estimate (i.e., the intercept) of nest survival was not influenced by the confounding effect of measurement error, as the sampling bias was solely attributed to the specified parameter coefficient (i.e., slope).

Table 2. Summary of the performance of nest survival models in Program MARK used to assess the influence of timing of vegetation surveys at nest sites on nest survival. Results are based on 500 iterations, each with unique encounter histories in which the underlying daily nest survival was positively influenced by grass height. All reported results are average values across all iterations. We do not report the average model deviance as it would be uninformative.

Model[a]	$\Delta \mathrm{AIC}_c$	w_i	No. par	β	SE
Grass $\mathrm{Height}_{\mathrm{Fate}}$	0.01	1.00	2	0.80	0.07
Grass $\mathrm{Height}_{\mathrm{Hatch}}$	80.78	0.00	1	0.23	0.07
Constant	91.39	0.00	1	3.28	0.07

[a]Model selection notation follows Burnham and Anderson (2002). All models included an intercept-term. Grass $\mathrm{Height}_{\mathrm{Fate}}$ represents models that included a covariate based on the simulated grass height at a nest on the date the nest's fate was assigned. Grass $\mathrm{Height}_{\mathrm{Hatch}}$ represents models that included a covariate based on the simulated grass height at a nest on the date a nest hatched, or was supposed to hatch, if unsuccessful. Constant represents the null, or intercept-only model.

Table 3. Summary of the performance of nest survival models in Program MARK used to assess the influence of timing of vegetation surveys at nest sites on nest survival. Results are based on 500 iterations, each with unique encounter histories in which the underlying daily nest survival was negatively influenced by grass height. All reported results are average values across all iterations. We do not report the average model deviance as it would be uninformative.

Model[a]	$\Delta \mathrm{AIC}_c$	w_i	No. par	β	SE
Grass $\mathrm{Height}_{\mathrm{Fate}}$	0.04	0.99	2	0.50	0.07
Grass $\mathrm{Height}_{\mathrm{Hatch}}$	39.32	0.00	2	−0.23	0.07
Constant	49.73	0.00	1	3.28	0.07

[a]Model selection notation follows Burnham and Anderson (2002). All models included an intercept-term. Grass $\mathrm{Height}_{\mathrm{Fate}}$ represents models that included a covariate based on the simulated grass height at a nest on the date the nest's fate was assigned. Grass $\mathrm{Height}_{\mathrm{Hatch}}$ represents models that included a covariate based on the simulated grass height at a nest on the date a nest hatched, or was supposed to hatch, if unsuccessful. Constant represents the null, or intercept-only model.

Real data

Model selection and parameter estimates from the sage-grouse nest survival analysis based on real data mirrored results from the simulated data. These results also suggested that the derived $\mathrm{GH}_{\mathrm{fate}}$ variable produced effect sizes that were greater ($\beta_{\mathrm{fate}} = 0.47$; $\mathrm{SE}_{\mathrm{fate}} = 0.09$) than that of the $\mathrm{GH}_{\mathrm{hatch}}$ variable ($\beta_{\mathrm{hatch}} = 0.06$; $\mathrm{SE}_{\mathrm{hatch}} = 0.06$).

Table 4. Summary of the performance of nest survival models in Program MARK used to assess the influence of timing of vegetation surveys at nest sites on nest survival. Results are based on 500 iterations, each with unique encounter histories in which the underlying daily nest survival was not influenced by grass height. All reported results are average values across all iterations. We do not report the average model deviance as it would be uninformative.

Model[a]	ΔAIC_c	w_i	No. par	β	SE
Grass $Height_{Fate}$	0.00	1.00	2	0.60	0.07
Constant	72.90	0.00	1	3.28	0.07
Grass $Height_{Hatch}$	74.05	0.00	2	0.00	0.07

[a]Model selection notation follows Burnham and Anderson (2002). All models included an intercept-term. Grass $Height_{Fate}$ represents models that included a covariate based on the simulated grass height at a nest on the date the nest's fate was assigned. Grass $Height_{Hatch}$ represents models that included a covariate based on the simulated grass height at a nest on the date a nest hatched, or was supposed to hatch, if unsuccessful. Constant represents the null, or intercept-only model.

Similar to the simulated results, the GH_{fate} variable was also better supported in model selection, outcompeting the GH_{hatch} model by more than 30 AIC units and receiving all of the AIC model weight (Table 5). Interpretation of grass height influence on sage-grouse nest survival was also different between the two metrics; the GH_{fate} variable suggested a strong positive effect of grass height on sage-grouse nest survival, whereas GH_{hatch} suggested only a very weak effect (Fig. 3).

Discussion

Our results provide multiple lines of evidence that confounding between plant phonology and demographic processes can have important implications when evaluating vegetation effects on vital rates, in our case nest survival. We found that studies of grassland and shrubland birds predominantly used nest site vegetation metrics sampled at the time of nest fate, which was more likely to yield a positive effect of grass height or cover when compared to data collected on a standardized date, such as predicted hatch. Our simulations, and evidence from real data, also show that grass height measurements recorded at nest fate produced effect sizes that were biased high relative to the true effect of grass height, and in some cases, this bias was sufficient to change the overall direction of the effect as well as its magnitude. These results are undoubtedly related to the inherent confounding between grass growth and timing of fate for failed versus successful nests; the fate of successful nests occurs inherently later in the season, therefore vegetation biomass will increase prior to sampling for successful nests when

Table 5. Performance of nest survival models in Program MARK used to assess the influence of timing of vegetation surveys at nest sites on Greater sage-grouse nest survival in Eureka County, NV, 2004–2012.

Model[a]	ΔAIC_c	AIC_c w_i	No. par	Dev.	β	SE
Grass $Height_{Fate}$	0.00	1.00	2	1478.72	0.47	0.09
Constant	31.61	0.00	1	1512.34	2.97	0.06
Grass $Height_{Hatch}$	32.59	0.00	2	1511.31	0.06	0.06

[a]Model selection notation follows Burnham and Anderson (2002). All models included an intercept-term. Grass $Height_{Hatch}$ represents a model that included a covariate based on the measured grass height at a nest on the date a nest hatched, or was supposed to hatch, if unsuccessful. Grass $Height_{Fate}$ represents a model that included a covariate based on the estimated grass height at a nest on the date the nest's fate was assigned, which was derived from Grass $Height_{Hatch}$ and estimated daily grass growth. Constant represents the null, or intercept-only model.

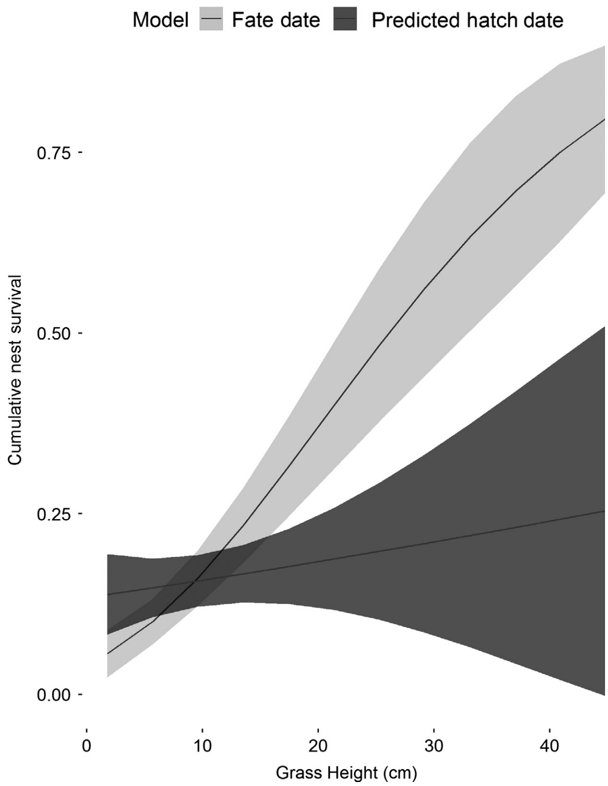

Figure 3. Estimated probability of cumulative nest survival relative to the average grass height within 100 m^2 of a nest, where grass was measured on the predicted hatch date of a nest (gray line, dark gray ribbon) or was predicted based on the date of nest fate (black line, light gray ribbon) for Greater Sage-Grouse in Eureka, Nevada, USA from 2004–2012. Predicted grass height at fate was estimated by regressing average grass heights against ordinal dates of vegetation surveys to correct grass height measurements based on daily growth rates.

compared with unsuccessful nests, which fail and are sampled earlier. Most notably, models based on vegetation data collected at nest fate were more parsimonious (i.e., lower AIC value) relative to models based on vegetation data collected at hatch. This was completely an artifact, however, driven by the aforementioned confounding between grass growth and nest fate, and the increase in explanatory power was related to the covariate accounting for the confounding between fate and the value of the covariate introduced into the model by sampling at nest fate. Sampling nests on a standardized date that is also biologically meaningful, such as the predicted hatch date, appear to overcome these issues and produce robust estimates of vegetation effects on nest survival.

Although we have focused on grass height and nest survival, confounding among plant phenology and demographic process is clearly a more general issue because of the role that vegetation structure and composition play in the study of animal ecology from both basic and applied perspectives (e.g., Fisher and Davis 2011; Germain and Arcese 2014). Key to our results is an inherent confounding between the timing of nest fate and the timing of vegetation sampling when measured at fate. Other demographic traits, such as age-specific survival or breeding probabilities, could be similarly affected if careful consideration is not given to potential confounding factors during study design and data analysis. This would include both the timing of vegetation sampling, and the temporal resolution of the demographic estimate, as well as the potential correspondence of these two measures. For example, age- or stage-specific survival is commonly expressed as a probability value that reflects the likelihood an individual will survive a given time interval (e.g., a week, month, year, etc.). If vegetation is sampled at a finer time interval (e.g., weekly), and then is applied as a covariate to an interval that is more coarse (e.g., monthly survival probability), and if vegetation is likely to change due to growth or senescence throughout the monthly interval, mean vegetation measures may differ for individuals that die early in the month compared to those that survive the duration of the month. In this case, the inherent confounding between vegetation and demographics can be resolved by estimating survival probability at a temporal resolution that matches that of the vegetation sampling (e.g., a weekly survival probability), and by including vegetation measures as time-varying covariates during data analysis (Bonner et al. 2010).

Increased grass height or cover is correlated with reduced visibility (Carlyle et al. 2010), and we agree that a positive association between grass biomass and nest survival is intuitive, especially for ground-nesting species. In our positive-effect simulations, grass height measured at nest fate correctly identified the positive association between grass and nest survival. However, the magnitude of the effect, as evidenced by the modeled parameter coefficient, was inflated relative to the true effect. Depending on study objectives, the magnitude of an effect may be as important for biological interpretation as the fact that the effect exists in the first place. This consideration may be particularly important to identify conservation guidelines or targets for vegetation management (e.g., Connelly et al. 2000; Hagen et al. 2004) because modeled parameter coefficients can be used to identify management thresholds, and will be inflated in a scenario of positive sampling bias (Fig. 3). Additionally, as conservation plans are often inadequately funded, the appropriation of resources toward management objectives based on habitat metric that has not reliably been demonstrated to improve reproductive performance has additional consequences as it reduces the amount of resources available for more meaningful restoration efforts. We also appreciate the rationale for measuring vegetation at nest fate; nests presumably fail because of conditions that are present at the time of failure (e.g., vegetation failing to conceal a nest from a predator) and so measuring those conditions are somewhat intuitive. This approach, however, cannot disentangle the confounding between the timing of vegetation sampling and nest fate from the true demographic mechanism associated with vegetation concealment, as our simulations demonstrate.

When designing future research, we recommend that investigators carefully consider confounding between plant phenology and their demographic rate of interest, and conduct vegetation sampling accordingly. In situations where vegetation sampling was conducted at nest fate, measurements can be date-corrected to remove the potential confounding between the timing of nest fate and vegetation measurement. This can be accomplished by regressing the vegetation measurement on the ordinal date of the survey, and using the model residuals as date-corrected estimates. Alternatively, we have outlined above an approach to predict grass height at nest fate based on measured grass height on a predicted hatch date. A slight modification could also be used to forecast vegetation measurements based on system-specific growth rates and timing of surveys at fate relative to a predicted hatch date. This latter approach also has the advantage of creating date-corrected vegetation measurements that fall within the same range as values likely to be measured in the field, as opposed to the residual-based approach which would yield both positive and negative values relative to the modeled regression line. Although these two approaches are not perfect, they should serve to disentangle confounding between plant growth and timing of nest failure, effectively removing a source of sampling bias from the data. Both approaches assume that vegetation growth is linear with respect to ordinal date, however nonlinear growth could be easily incorporated using quadratic effect terms. Lastly, we speculate that future advancements in automated aerial technology, spatial imaging resolution and classification may allow researchers to quantify fine-scale habitat characteristics while nests are active without excessive disturbance, effectively disentangling plant growth and timing of nest fate (Connelly et al. 2000; Drever et al. 2015).

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References

Alexander, R. A., G. V. Barrett, G. M. Alliger, and K. P. Carson. 1986. Towards a general-model of nonrandom sampling and the impact on population correlation – generalizations of berkson fallacy and restriction of range. Br. J. Math. Stat. Psychol. 39:90–105.

Anderson, J. C., and D. W. Gerbing. 1984. The effect of sampling error on convergence, improper solutions, and goodness-of-fit indexes for maximum-likelihood confirmatory factor-analysis. Psychometrika 49:155–173.

Arnold, T. W. 2010. Uninformative parameters and model selection using Akaike's information criterion. J. Wildl. Manage. 74:1175–1178.

Bell, C. B. 2011. Nest site characteristics and nest success of translocated and resident greater sage grouse at Clear Lake National Wildlife Refuge. Master of Science, Humboldt State University, Arcata.

Blomberg, E. J., S. R. Poulson, J. S. Sedinger, and D. Gibson. 2013. Prefledging diet is correlated with individual growth in greater sage-grouse (*Centrocercus urophasianus*). Auk 130:715–724.

Bonner, S. J., B. J. T. Morgan, and R. King. 2010. Continuous covariates in mark-recapture-recovery analysis: a comparison of methods. Biometrics 66:1256–1265.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer, New York.

Carlyle, C. N., L. H. Fraser, C. M. Haddow, B. A. Bings, and W. Harrower. 2010. The use of digital photos to assess visual cover for wildlife in rangelands. J. Environ. Manage. 91:1366–1370.

Coates, P. S., and D. J. Delehanty. 2010. Nest predation of greater sage-grouse in relation to microhabitat factors and predators. J. Wildl. Manage. 74:240–248.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildl. Soc. Bull. 28:967–985.

Davis, S. K. 2005. Nest-site selection patterns and the influence of vegetation on nest survival of mixed-grass prairie passerines. Condor 107:605–616.

Davis, D. M. 2009. Nesting ecology and reproductive success of lesser prairie-chickens in shinnery oak-dominated rangelands. Wilson J. Ornithol. 121:322–327.

Davis, D. M., K. P. Reese, and S. C. Gardner. 2014. Demography, reproductive ecology, and variation in survival of greater sage-grouse in northeastern California. J. Wildl. Manage. 78:1343–1355.

Diggle, P. J., R. Menezes, and T. L. Su. 2010. Geostatistical inference under preferential sampling. J. R. Stat. Soc. Ser. C Appl. Stat. 59:191–232.

Dinsmore, S. J., and J. J. Dinsmore. 2007. Modeling avian nest survival in program mark. Stud. Avian Biol. 34:73–83.

Dinsmore, S. J., G. C. White, and F. L. Knopf. 2002. Advanced techniques for modeling avian nest survival. Ecology 83:3476–3488.

Doherty, K. E., J. L. Beck, and D. E. Naugle. 2011. Comparing ecological site descriptions to habitat characteristics influencing greater sage-grouse nest site occurrence and success. Rangeland Ecol. Manag. 64:344–351.

Doherty, K. E., D. E. Naugle, J. D. Tack, B. L. Walker, J. M. Graham, and J. L. Beck. 2014. Linking conservation actions to demography: grass height explains variation in greater sage-grouse nest survival. Wildl. Biol. 20:320–325.

Drever, M. C., D. Chabot, P. D. O'Hara, J. D. Thomas, A. Breault, and R. L. Millikin. 2015. Evaluation of an unmanned rotorcraft to monitor wintering waterbirds and coastal habitats in British Columbia, Canada. J. Unmanned Vehicle Syst. 3:256–267.

Fisher, R. J., and S. K. Davis. 2011. Post-fledging dispersal, habitat use, and survival of Sprague's pipits: are planted grasslands a good substitute for native? Biol. Conserv. 144:263–271.

Germain, R. R., and P. Arcese. 2014. Distinguishing individual quality from habitat preference and quality in a territorial passerine. Ecology 95:436–445.

Gibson, D. 2015. The role of environmental stochasticity on population demography of greater sage-grouse in central Nevada, USA. Doctoral Dissertation, University of Nevada, Reno.

Gibson, D., E. J. Blomberg, M. T. Atamian, and J. S. Sedinger. 2015. Observer effects strongly influence estimates of daily nest survival probability but do not substantially increase rates of nest failure in greater sage-grouse. Auk: Ornithol. Adv. 132:397–407.

Goodman, J. S., and T. C. Blum. 1996. Assessing the non-random sampling effects of subject attrition in longitudinal research. J. Manage. 22:627–652.

Gregg, M. A., J. A. Crawford, M. S. Drut, and A. K. Delong. 1994. Vegetational cover and predation of sage grouse nests in oregon. J. Wildl. Manage. 58:162–166.

Gregory, C. J., S. J. Dinsmore, L. A. Powell, and J. G. Jorgensen. 2011. Nest survival of long-billed curlew in Nebraska. Wader Study Group Bulletin 118:109–113.

Hagen, C. A., B. E. Jamison, K. M. Giesen, and T. Z. Riley. 2004. Guidelines for managing lesser prairie-chicken populations and their habitats. Wildl. Soc. Bull. 32:69–82.

Hausleitner, D., K. P. Reese, and A. D. Apa. 2005. Timing of vegetation sampling at greater sage-grouse nests. Rangeland Ecol. Manag. 58:553–556.

Jones, T., and W. Cresswell. 2010. The phenology mismatch hypothesis: are declines of migrant birds linked to uneven global climate change? J. Anim. Ecol. 79:98–108.

Kaczor, N. W., K. C. Jensen, R. W. Klaver, M. A. Rumble, K. M. Herman-Brunson, and C. C. Swanson. 2011. Nesting success and resource selection of greater sage-grouse. Ecol. Conser. Manag. Grouse 39:107–118.

Kerns, C. K., M. R. Ryan, R. K. Murphy, F. R. Thompson III, and C. S. Rubin. 2010. Factors affecting songbird nest survival in northern mixed-grass prairie. J. Wildl. Manage. 74:257–264.

Kery, M., and M. Schaub. 2012. Bayesian population analysis using WinBUGS – a hierarchical perspective. Academic Press, Burlington.

Knight, E. C., N. A. Mahony, and D. J. Green. 2014. Crop type influences edge effects on the reproduction of songbirds in sagebrush habitat near agriculture. Avian Conser. Ecol. 9:8.

Kolada, E. J., M. L. Casazza, and J. S. Sedinger. 2009. Ecological factors influencing nest survival of greater sage-grouse in Mono County, California. J. Wildl. Manage. 73:1341–1347.

Laake, J. L. 2013. RMark: an R interface for analysis of capture-recapture data with MARK. AFSC Processed Rep. 2013–01:25.

Lockyer, Z. B., P. S. Coates, M. L. Casazza, S. Espinosa, and D. J. Delehanty. 2015. Nest-site selection and reproductive success of greater sage-grouse in a fire-affected habitat of northwestern Nevada. J. Wildl. Manage. 79:785–797.

Lyons, T. 2013. Nest predation and habitat selection in the grasshopper sparrow (*Ammodramus savannarum*). Master of Science, University of Illinois at Urbana-Champaign, Urbana-Champaign.

McKee, G. W., M. R. Ryan, and L. M. Mechlin. 1998. Predicting greater prairie-chicken nest success from vegetation and landscape characteristics. J. Wildl. Manage. 62:314–321.

McNew, L. B., L. M. Hunt, A. J. Gregory, S. M. Wisely, and B. K. Sandercock. 2014. Effects of wind energy development on nesting ecology of greater prairie-chickens in fragmented grasslands. Conserv. Biol. 28:1089–1099.

Miller-Rushing, A. J., T. T. Hoye, D. W. Inouye, and E. Post. 2010. The effects of phenological mismatches on demography. Philos. Trans. R. Soc. Lond. B Biol. Sci. 365:3177–3186.

Muff, S., A. Riebler, L. Held, H. Rue, and P. Saner. 2015. Bayesian analysis of measurement error models using integrated nested Laplace approximations. J. R. Stat. Soc. Ser. C Appl. Stat. 64:231–252.

Nussey, D. H., E. Postma, P. Gienapp, and M. E. Visser. 2005. Selection on heritable phenotypic plasticity in a wild bird population. Science 310:304–306.

Ostermiller, J. D., and C. P. Hawkins. 2004. Effects of sampling error on bioassessments of stream ecosystems: application to RIVPACS-type models. J. N. Am. Benthol. Soc. 23:363–382.

Pitman, J. C., C. A. Hagen, R. J. Robel, T. M. Loughin, and R. D. Applegate. 2005. Location and success of lesser prairie-chicken nests in relation to vegetation and human disturbance. J. Wildl. Manage. 69:1259–1269.

Popham, G. P., and R. J. Gutierrez. 2003. Greater sage-grouse *Centrocercus urophasianus* nesting success and habitat use in northeastern California. Wildl. Biol. 9:327–334.

Rader, M. J., L. A. Brennan, F. Hernandez, N. J. Silvy, and W. Ben. 2007. Nest-site selection and nest survival of Northern Bobwhite in Southern Texas. Wilson J. Ornithol. 119:392–399.

Rebholz, J. L. 2008. Influence of habitat characteristics on Greater sage-grouse reproductive success in the Montana Mountains. Master of Science, Oregon State University, Nevada.

Sadoti, G., M. G. Pollock, K. T. Vierling, T. P. Albright, and E. K. Strand. 2014. Variogram models reveal habitat gradients predicting patterns of territory occupancy and nest survival among vesper sparrows. Wildl. Biol. 20:97–107.

Stanley, T. R., C. L. Aldridge, D. J. Saher, and T. M. Childers. 2015. Daily nest survival rates of Gunnison sage-grouse (*Centrocercus minimus*): assessing local- and landscape-scale drivers. Wilson J. Ornithol. 127:59–71.

Staples, D. F., M. L. Taper, and B. Dennis. 2004. Estimating population trend and process variation for PVA in the presence of sampling error. Ecology 85:923–929.

Sveum, C. M., W. D. Edge, and J. A. Crawford. 1998. Nesting habitat selection by sage grouse in south-central Washington. J. Range Manag. 51:265–269.

Vander Haegen, W. M., M. A. Schroeder, W.-Y. Chang, and S. M. Knapp. 2015. Avian abundance and reproductive success in the intermountain west: local-scale response to the conservation reserve program. Wildl. Soc. Bull. 9999:1–16.

Walters, C. J., and D. Ludwig. 1981. Effects of measurement errors on the assessment of stock-recruitment relationships. Can. J. Fish Aquat. Sci. 38:704–710.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46:120–139.

Wing, B. R. 2014. The role of vegetation structure, composition, and nutrition in greater sage-grouse ecology in northwestern Utah. Master of Science, Utah State University, Logan.

Winter, M., D. H. Johnson, and J. A. Shaffer. 2005. Variability in vegetation effects on density and nesting success of grassland birds. J. Wildl. Manage. 69:185–197.

M. Scott, eds. Estimating numbers of terrestrial birds. Cooper Ornithol. Soc., Stud. Avian Biol. 6.

U.S. FISH AND WILDLIFE SERVICE. 1988. Nongame bird strategies. Off. Migr. Bird Manage., Washington, D.C. 23pp.

WILEY, J. W. 1979. The white-crowned pigeon in Puerto Rico: status, distribution and movements. J. Wildl. Manage. 43:402–413.

———, AND B. N. WILEY. 1979. The biology of the white-crowned pigeon. Wildl. Monogr. 64. 54pp.

WOOD, D. A. 1992. Official lists of endangered and potentially endangered fauna and flora in Florida. Florida Game and Freshwater Fish Comm., Tallahassee. 25pp.

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VEGETATIONAL COVER AND PREDATION OF SAGE GROUSE NESTS IN OREGON

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Abstract: Because of long-term declines in sage grouse (*Centrocercus urophasianus*) abundance and productivity in Oregon, we investigated the relationship between vegetational cover and nesting by sage grouse in 2 study areas. Medium height (40–80 cm) shrub cover was greater ($P < 0.001$) at nonpredated ($\bar{x} = 41\%$, $n = 18$) and predated ($\bar{x} = 29\%$, $n = 106$) nests than in areas immediately surrounding nests ($\bar{x} = 15$ and 10%, $n = 18$ and 106, nonpredated and predated, respectively) or random locations ($\bar{x} = 8\%$, $n = 499$). Tall (>18 cm), residual grass cover was greater ($P < 0.001$) at nonpredated nests ($\bar{x} = 18\%$) than in areas surrounding nonpredated nests ($\bar{x} = 6\%$) or random locations ($\bar{x} = 3\%$). There was no difference ($P > 0.05$) in grass cover among predated nests, nest areas, and random sites. However, nonpredated nests had greater ($P < 0.001$) cover of tall, residual grasses ($\bar{x} = 18\%$) and medium height shrubs ($\bar{x} = 41\%$) than predated nests ($\bar{x} = 5$ and 29% for grasses and shrubs, respectively). Removal of tall grass cover and medium height shrub cover may negatively influence sage grouse productivity.

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Key words: *Centrocercus urophasianus*, habitat, nesting, Oregon, predation, reproduction, sage grouse, selection.

Sage grouse populations declined in several western states from the 1950s through the 1980s (Crawford and Lutz 1985, Klebenow 1985). In Oregon, the decrease in abundance of sage grouse was attributed to impaired productivity (Crawford and Lutz 1985). Reduced productivity may result from several factors, including excessive nest predation (Autenrieth 1981:39). Batterson and Morse (1948) and Nelson (1955) identified predation as the primary factor directly influencing sage grouse nesting success in Oregon. Although predators may be the immediate cause of nest loss, the amount and composition of vegetational cover at nests may influence predation (Bowman and Harris 1980, Redmond et al. 1982). We hypothesized that predation of sage grouse nests in Oregon was related to amount and composition of vegetational structural components surrounding nests. Our objective was to identify vegetational characteristics at nonpredated and predated sage grouse nest sites in comparison with randomly selected locations in 2 areas of southeastern Oregon.

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STUDY AREAS

We conducted the study in 2 areas of southeastern Oregon: Hart Mountain National Antelope Refuge (Lake County) and Jackass Creek (Harney County). Topography of both areas consisted of flat sagebrush plains interrupted by rolling hills, ridges, and draws. Elevations ranged from 1,500 to 2,450 m at Hart Mountain and from 1,200 to 1,700 m at Jackass Creek. Mean maximum temperature (Mar–Aug) was 21 C at Hart Mountain and 24 C at Jackass Creek. Annual precipitation averaged 29 cm in both areas.

Vegetation at Hart Mountain and Jackass Creek consisted of low sagebrush (*Artemisia arbuscula*), big sagebrush (*A. tridentata*), green rabbitbrush (*Chrysothamnus viscidiflorus*), and western juniper (*Juniperus occidentalis*). Stands of aspen (*Populus tremuloides*), curl-leaf mountain-mahogany (*Cercocarpus ledifolius*), and bitter-brush (*Purshia tridentata*) occurred only at Hart Mountain. Common annual and perennial forbs included mountain-dandelion (*Agoseris* spp.), hawksbeard (*Crepis* spp.), milk-vetch (*Astragalus* spp.), lupine (*Lupinus* spp.), and phlox (*Phlox* spp.). Grasses consisted mainly of bluegrass (*Poa* spp.), bluebunch wheatgrass (*Agropyron spicatum*), needlegrass (*Stipa* spp.), fescue (*Festuca* spp.), giant wildrye (*Elymus cinereus*), and bottlebrush squirreltail (*Sitanion hystrix*) (plant nomenclature from Hitchcock and Cronquist [1987]).

METHODS

From summer 1988 through spring 1991, we captured (Giesen et al. 1982) female sage grouse during July–August near watering areas and during March–April on and near leks. We fitted each hen with an aluminum leg band and a poncho-mounted, solar-powered radio transmitter with a nickel-cadmium battery (Amstrup 1980). The radio package (radio and poncho) weighed approximately 25 g. Juvenile females captured during summer were not marked with radios. We monitored radio-marked hens 3 times weekly throughout the nesting season with a hand-held antenna and portable receiver. When monitoring indicated a hen initiated a nest, visual confirmation was made without intentionally flushing the hen. Subsequently, we monitored hens remotely to avoid disturbance. When monitoring indicated a hen had ceased nesting efforts, we determined nest fate. We classified nests as nonpredated if ≥ 1 egg hatched or if incubation exceeded 30 days. Predated nests were identified by the presence of firmly attached shell membranes in broken eggs or by missing eggs.

We measured vegetation in a 78-m^2 area (circular area with a radius of 5 m) at nonpredated nest sites after completion of incubation and at predated nest sites on predicted hatch dates. We measured vegetation at randomly selected locations during early May. We located random sites with a random numbers table, which was used to determine starting points, compass bearing, and distance traveled. The number of random locations sampled in each study area was determined by canopy cover of sagebrush and sample size requirements (Snedecor and Cochran 1967:516). We measured canopy cover (%) of shrubs by line-intercept (Canfield 1941) along 2 10-m perpendicular transects intersecting at the nest or random location. The position of the first transect was determined from a randomly selected compass bearing. We placed each intercepted shrub into 1 of 3 height classes: short (<40 cm), medium (40–80 cm), or tall (>80 cm). We based height classes on results of previous studies (Nelson 1955, Wallestad and Pyrah 1974, Autenrieth 1981:17, Wakkinen 1990). Canopy cover of shrubs was recorded separately for each height class. We estimated cover (%) of forbs and grasses in 5 20- × 50-cm plots spaced equidistantly along each transect (Daubenmire 1959). We measured maximum droop height (excluding flowering stalks) of grasses at the nest bush and at random locations throughout each study area and classified grass genera as short (<18 cm) or tall (>18 cm), following results of Wakkinen (1990). We identified shrubs to species and forbs and grasses to genus.

To determine the relationship between vegetational features and predation of sage grouse nests, we apportioned the 78-m^2 area in which vegetational measurements were taken at each nest into 2 components: a 3-m^2 area at the nest and a 75-m^2 area immediately surrounding the nest. We used a factorial analysis of variance (ANOVA) and Student-Newman-Keuls multiple range tests adjusted for unequal sample sizes (Zar 1974:154) to compare vegetational characteristics among plot types (nonpredated nest and nest area, predated nest and nest area, and random location). Study area and year were additional factors in the ANOVA model to account for variation associated with spatial and temporal differences. The only interactions were those for plot type by study area for forb (P = 0.009) and tall grass (P < 0.001) cover. However, individual ANOVAs coupled with Student-Newman-Keuls multiple range tests for these 2 variables by study area revealed identical patterns of mean separation, which indicated that these vegetational characteristics were not confounded by study area. Consequently, we assumed plot type was independent of study area. We detected no other interactions for any vegetational characteristic. Pearson correlation coefficients were used to test for intercorrelation among variables. All data were normally distributed, and we considered results significant if $P \leq 0.05$.

RESULTS

During 3 years, we located 124 sage grouse nests (57 at Hart Mountain and 67 at Jackass Creek); 18 of these were nonpredated (11 and 7 at Hart Mountain and Jackass Creek, respectively). Sage grouse nested in big sagebrush, low sagebrush, and mixed sagebrush (mosaic of big and low sagebrush) stands. Of 18 nonpredated nests, 13 were in big sagebrush stands, whereas only 3 and 2 nonpredated nests were in low and mixed sagebrush stands, respectively. Ninety-four percent of all nests from radio-marked hens were under sagebrush. Other vegetation used for nesting included rabbitbrush (n = 5), bitterbrush (n = 1), and giant wildrye (n = 1). Sagebrush collectively represented 87% of the shrub component in both study areas. Other shrubs included bitter-brush (6%), rabbitbrush (4%), horsebrush (*Tetradymia* spp.) (1%), and mountain snowberry (*Symphoricarpos oreophilus*) (1%). Tall grass genera included giant wildrye, wheatgrass, fescue, and needlegrass. Short grass genera consisted of bottlebrush squirreltail, junegrass (*Koleria cristata*), brome (*Bromus* spp.), and bluegrass.

Cover of tall grasses was greater (P < 0.001) at nonpredated nests than at predated nests or random locations (Table 1). No differences in grass cover were detected between predated nests and random sites. Except for one case, tall grasses at nonpredated nests were composed of residual cover.

For all nests, shrub cover of medium height was greater (P < 0.001) at nests than in the immediate area surrounding nests or random locations (Table 1). However, cover of medium height shrubs was greater (P < 0.001) at nonpredated nests than at predated nests. Furthermore, the immediate area surrounding nonpredated nest sites had greater (P < 0.001) cover of medium height shrubs than random locations. Shrub cover of short height was greater (P = 0.02) at predated nests than at random locations. Amount of tall grass was not correlated with short (r = −0.06) or medium (r = 0.12) shrub cover.

DISCUSSION

We found a relationship between vegetational cover and predation of sage grouse nests. Nonpredated nests had greater cover of tall, residual grasses and medium height shrubs than predated nests. No previous research demonstrated the value of residual grass cover at sage grouse nests, although its importance was suggested by Pyrah (1971) and Wakkinen (1990). Wakkinen (1990) reported data about grass height and nest fate but found no relationships. Our data, however, indicated that tall, residual grass cover may enhance sage grouse nest success. Grass cover was identified as an important nesting habitat component for other galliformes, including California quail (*Callipepla californica*) (Leopold 1977: 168), Attwater prairie-chickens (*Tympanuchus cupido attwateri*) (Lehman 1941:14), and plains sharp-tailed grouse (*T. phasianellus jamesi*) (Hillman and Jackson 1973:24). Lehman (1941: 14) noted that all prairie-chicken nests he located were in residual grass cover. The presence of tall, residual grass cover influenced nest site selection and nest predation rates of gray partridge (*Perdix perdix*) in Great Britain (Rands 1982).

We also demonstrated the importance of medium height shrub cover to successful nesting sage grouse. Wallestad and Pyrah (1974) found that successful nests had greater sagebrush cover than unsuccessful nests. Contrastingly, Autenrieth (1981:20) and Wakkinen (1990) found no relationship between canopy cover of sagebrush and nest fate. Hulet et al. (1986) reported that successful nests were located in areas of less shrub cover and shorter height sagebrush than nests that were predated.

Tall, dense, vegetational cover may provide scent, visual, and physical barriers between predators and nests of ground-nesting birds (Bowman and Harris 1980, Redmond et al. 1982, Sugden and Beyersergen 1987, Crabtree et al. 1989). Greater amounts of tall grasses and medium height shrubs at successful sage grouse

Table 1. Vegetational characteristics (% cover) at nonpredated and predated nests and areas immediately surrounding nests of radio-marked sage grouse, and random locations in southeastern Oregon, 1989–91.

Characteristic	Nonpredated (n = 18) Nest[a] $\bar{x}$	SE	Nonpredated Nest area[b] $\bar{x}$	SE	Predated (n = 106) Nest $\bar{x}$	SE	Predated Nest area $\bar{x}$	SE	Random (n = 499) $\bar{x}$	SE
Grass cover										
Short, <18 cm	6A[c]	1.1	7A	1.2	6A	0.7	8A	0.5	8A	0.3
Tall, >18 cm	18A	5.5	6B	2.0	5B	1.2	3B	0.6	3B	0.2
Forb cover	8A	1.2	10A	1.4	9A	0.9	9A	0.5	9A	0.3
Shrub cover										
Short, <40 cm	14AB	3.9	15AB	2.7	19B	1.9	17AB	1.0	14A	0.4
Medium, 40–80 cm	41A	5.2	15B	3.3	29C	2.1	10BD	1.0	8D	0.4
Tall, >80 cm	1A	0.7	1A	0.7	4A	1.2	1A	0.3	3A	0.3

[a] 3-m^2 area at nest.
[b] 75-m^2 area immediately surrounding nest.
[c] Means with same letter within rows were not different $P \geq 0.05$.

nests likely provided the lateral and overhead concealment needed for security from predators. Nests lacking adequate cover were more likely to be predated. Our results confirmed the hypothesis of a relationship between vegetational cover and predation, but further investigation, in the form of controlled experimental tests, is needed to elucidate this principle.

MANAGEMENT IMPLICATIONS

Land management practices that decrease tall grass and medium height shrub cover at potential nest sites may be detrimental to sage grouse populations because of increased nest predation. Livestock grazing remains the most common and widespread use of rangelands in Oregon and is the principal land management practice and proximate factor that affects grass cover and height (Rickard et al. 1975). Grazing of tall grasses to <18 cm would decrease their value for nest concealment. Land management practices that affect medium height shrub cover include eradication of sagebrush for agricultural production, increased livestock forage, urban development, and mining activities (Klebenow 1972, 1985; Braun et al. 1977). Habitats that support the amount and type of grass cover needed for successful sage grouse nesting typically contain 8–12% shrub cover in Wyoming big sagebrush (*A. t. wyomingensis*) stands and 15–20% shrub cover in mountain (*A. t. vaseyana*) or basin (*A. t. tridentata*) big sagebrush stands (Winward 1991). Management activities should allow for maintenance of tall, residual grasses or, where necessary, restoration of grass cover within these stands.

LITERATURE CITED

AMSTRUP, S. C. 1980. A radio-collar for game birds. J. Wildl. Manage. 44:214–217.

AUTENRIETH, R. E. 1981. Sage grouse management in Idaho. Idaho Dep. Fish and Game, Boise. Wildl. Bull. 9. 238pp.

BATTERSON, W. M., AND W. B. MORSE. 1948. Oregon sage grouse. Oregon State Game Comm., Portland. Fauna Ser. 1. 29pp.

BOWMAN, G. B., AND L. D. HARRIS. 1980. Effect of spatial heterogeneity on ground-nest depredation. Ecol. Monogr. 38:269–308.

BRAUN, C. E., T. BRITT, AND R. O. WALLESTAD. 1977. Guidelines for maintenance of sage grouse habitats. Wildl. Soc. Bull. 5:99–106.

CANFIELD, R. H. 1941. Application of the line interception method in sampling of range vegetation. J. For. 39:386–394.

CRABTREE, R. L., L. S. BROOME, AND M. L. WOLFE. 1989. Effects of habitat characteristics on gadwall nest predators and nest-site selection. J. Wildl. Manage. 53:129–137.

CRAWFORD, J. A., AND R. S. LUTZ. 1985. Sage grouse population trends in Oregon, 1941–1983. Murrelet 66:69–74.

DAUBENMIRE, R. F. 1959. A canopy-coverage method of vegetation analysis. Northwest Sci. 33: 224–227.

GIESEN, K. M., T. J. SCHOENBERG, AND C. E. BRAUN. 1982. Methods for trapping sage grouse in Colorado. Wildl. Soc. Bull. 10:224–231.

HILLMAN, C. N., AND W. W. JACKSON. 1973. The sharp-tailed grouse in South Dakota. South Dakota Dep. Game, Fish, and Parks, Pierre. Tech. Bull. 3. 64pp.

HITCHCOCK, C. L., AND A. CRONQUIST. 1987. Flora of the Pacific Northwest. Sixth ed. Univ. Washington Press, Seattle. 730pp.

HULET, B. V., J. T. FLINDERS, J. S. GREEN, AND R. B. MURRAY. 1986. Seasonal movements and habitat selection of sage grouse in southern Idaho. Pages 168–175 *in* E. D. McArthur and B. L. Welch, eds. Proc. symposium of the biology of

Artemisia and *Chrysothamnus*. USDA For. Serv. Gen. Tech. Rep. INT-200. 398pp.

KLEBENOW, D. A. 1972. The habitat requirements of sage grouse and the role of fire in management. Proc. Tall Timbers Tire Ecol. Conf. 12:305–315.

———. 1985. Habitat management for sage grouse in Nevada. World Pheasant Assoc. J. 10:34–46.

LEHMAN, V. W. 1941. Attwater's prairie chicken: its life history and management. U.S. Fish and Wildl. Serv. North Am. Fauna 57. 63pp.

LEOPOLD, A. S. 1977. The California quail. Univ. California Press, Berkeley. 281pp.

NELSON, O. C. 1955. A field study of the sage grouse in southeastern Oregon with special reference to reproduction and survival. M.S. Thesis, Oregon State Univ., Corvallis. 113pp.

PYRAH, D. B. 1971. Sage grouse habitat research in central Montana. Proc. West. Assoc. State Game and Fish Comm. 51:293–300.

RANDS, M. R. 1982. The influence of habitat on the population ecology of partridge. Ph.D. Thesis, Univ. Oxford, U.K. 350pp.

REDMOND, G. W., D. M. KEPPIE, AND P. W. HERZOG. 1982. Vegetative structure, concealment, and success at nests of two races of spruce grouse. Can. J. Zool. 60:670–675.

RICKARD, W. H., D. W. URESK, AND J. F. CLINE. 1975. Impact of cattle grazing on three perennial grasses in south-central Washington. J. Range Manage. 28:108–112.

SNEDECOR, G. W., AND W. G. COCHRAN. 1967. Statistical methods. Sixth ed. Iowa State Univ. Press, Ames. 507pp.

SUGDEN, L. G., AND G. W. BEYERSERGEN. 1987. Effect of nesting cover and density on American crow predation on simulated duck nests. J. Wildl. Manage. 51:481–485.

WAKKINEN, W. L. 1990. Nest site characteristics and spring-summer movements of migratory sage grouse in southeastern Idaho. M.S. Thesis, Univ. Idaho, Moscow. 57pp.

WALLESTAD, R. O., AND D. B. PYRAH. 1974. Movement and nesting of sage grouse hens in central Montana. J. Wildl. Manage. 38:630–633.

WINWARD, A. H. 1991. A renewed commitment to management of sagebrush grasslands. Pages 2–7 *in* Management in the sagebrush steppe. Agric. Exp. Stn. Spec. Rep. 880. Oregon State Univ., Corvallis. 48pp.

ZAR, J. H. 1974. Biostatistical analysis. Prentice-Hall, Englewood Cliffs, N.J. 620pp.

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A meta-analysis of greater sage-grouse *Centrocercus urophasianus* nesting and brood-rearing habitats

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Hagen, C.A., Connelly, J.W. & Schroeder, M.A. 2007: A meta-analysis of greater sage-grouse *Centrocercus urophasianus* nesting and brood-rearing habitats. - Wildl. Biol. 13 (Suppl. 1): 42-50.

The distribution and range of the greater sage-grouse *Centrocercus urophasianus* have been reduced by 56% since the European settlement of western North America. Although there is an unprecedented effort to conserve the species, there is still considerable debate about the vegetation composition and structure required for nesting and brood-rearing habitat. We conducted a meta-analysis of vegetation characteristics recorded in studies at nest sites (N = 24) and brood habitats (N = 8) to determine if there was an overall effect (Hedge's d) of habitat selection and to estimate average canopy cover of sagebrush *Artemisia* spp., grass and forbs, and also height of grass at nest sites and brood-rearing areas. We estimated effect sizes from the difference between use (nests and brood areas) and random sampling points for each study, and derived an overall effect size across all studies. Sagebrush cover (d_{++} = 0.39; 95% C.I.: 0.19-0.54) and grass height (d_{++}= 0.28; 95% C.I.: 0.13-0.42) were greater at nest sites than at random locations. Vegetation at brood areas had less sagebrush cover (d_{++} = -0.17; 95% C.I.: -0.44 - +0.18), significantly taller grasses (d_{++} = 0.31; 95% C.I.: 0.14-0.45), greater forb (d_{++} = 0.48; 95% C.I.: 0.30-0.67) and grass cover (d_{++}= 0.17; 95% C.I.: 0.08-0.27) than at random locations. These patterns were especially evident when we examined early (< 6 weeks post hatching) and late brood-rearing habitats separately. The overall estimates of nest and brood area vegetation variables were consistent with those provided in published guidelines for the management of greater sage-grouse.

Key words: Artemisia spp., breeding habitat, effect size, greater sage-grouse, Hedges' d, meta-analysis, sagebrush

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The distribution and range of greater sage-grouse *Centrocercus urophasianus* have been reduced by 56% since the European settlement of western North America (Connelly & Braun 1997, Schroeder et al. 2004). Although loss and fragmentation of sagebrush *Artemisia* spp. habitats have been cited as the primary causes for the decline of the species, degradation of existing habitat also has been considered an important factor (Braun 1998). Guidelines for protection and management of nesting and brood-rearing habitat have been provided to land managers (Connelly et al. 2000). In general, a range of 15-25% sagebrush, > 10% forb, > 15 % grass canopy cover and, a herbaceous height of 18 cm are needed for breeding habitats of greater sage-grouse.

Techniques used to measure vegetation characteristics have not always been consistent (Wamboldt et al. 2006). Additionally, some researchers and managers have questioned the applicability of management guidelines (Connelly et al. 2000) across the range of the greater sage-grouse, as well as the techniques used to derive the earlier estimates of vegetative cover and height (Bates et al. 2004, Schultz 2004). In particular, subsequent debate over the quantitative properties of the recommended vegetative characteristics required for greater sage-grouse has become a hindrance to implementing conservation actions. To address these concerns and examine the relevance of management guidelines additional analyses are needed. One potential analytical method that was not used when producing the earlier guidelines (Braun et al. 1977, Connelly et al. 2000) was the research synthesis or meta-analysis, which allows an evaluation of the generality of a given effect as a result of combining parameter estimates (effect sizes) from a set of studies (Hall et al. 1994). The use of meta-analysis can advance our knowledge and understanding of observed findings, and contribute to the advancement of more theoretical issues (Hedges & Olkin 1985).

Schultz (2004) analysed the data set in Connelly et al. (2000) and used the analysis to critique the published guidelines. However, since these articles were published, more data have become available. Because the interpretation of earlier research is a fundamental tool in the development of appropriate guidelines to management, we employed meta-analytic techniques to the research summarized by Connelly et al. (2000) as well as research conducted more recently. The purpose of our meta-analysis was to estimate the effect of habitat selection of breeding habitats (i.e. nesting and brood rearing) of greater sage-grouse. To this end we compared vegetation characteristics at use sites to random points, to evaluate the similarity of effect sizes across studies, and to determine if the overall effect size for each vegetation characteristic is statistically or biologically meaningful.

Methods

Literature review and data selection

We reviewed peer-refereed articles and graduate research theses (N = 15) and non-refereed agency reports (N = 4) that pertained to greater sage-grouse habitat use during the nesting and brood-rearing periods (Tables 1 and 2). Because studies reported significant differences in vegetation between years (Fischer 1994, Apa 1998, Sveum et al. 1998, Holloran 1999) or study areas (Gregg 1991, Drut 1992, Slater 2003) we estimated effect size for each significant unit. We included estimates from studies that reported actual cover values (e.g. 32.3%) and excluded values from one study (Klott et al. 1993) that used ranked cover values (e.g. 1-5 from Daubenmire (1959) readings). In some studies, a limited number of vegetative characteristics were recorded, thus sample sizes in Tables 1 and 2 vary for each estimate of effect size. We examined the relationship of sagebrush cover, grass cover, forb cover and grass height at nest sites and brood-use sites compared to their respective random points. These variables were consistently reported across studies and provided the largest sample sizes for our comparisons. Several articles reported only shrub cover (e.g. Drut 1992, Gregg 1993, Fischer 1994, Hanf et al. 1994, Sveum et al. 1998), which may have included a mix of sagebrush and other shrubs. Because of limited sample sizes, we estimated effect sizes and parameter estimates for sagebrush only and shrub cover (i.e. sagebrush and other shrub cover) and present results for each. Canopy cover was sometimes estimated with line-intercept or quadrats. However, because we used a standardized metric in our meta-analysis, we could compare studies that used these different methodologies (Hedges & Olkin 1985, Gurevitch & Hedges 1999). Because brood survival rates and habitat use differ between 0-6 weeks post hatching and > 6 weeks post hatching (Holloran 1999, Lyon 2000), we estimated effect sizes for brood-use by early and late periods for studies that differentiated between them. We estimated a pooled effect size for studies that did not differentiate between early and late brood-rearing periods.

Data analysis

A general equation for an effect size is the treatment mean minus control mean divided by the pooled variance (Hedges 1982). The effect size for each study serves as a dependent variable that can be modeled as a function of discrete or continuous explanatory

Table 1. Studies and vegetation data used in meta-analyses of greater sage-grouse nesting habitats throughout North America. Sagebrush (shrub), grass and forb canopy cover (in %) and grass height (in cm) were vegetation variables considered in the analyses. Vegetation community was described in each study as silver sagebrush (SS), mountain big sagebrush (MT) or Wyoming big sagebrush (WY). ND means that no data were available or had been reported in a manner that was usable in the meta-analysis.

	Nest site vegetation									
Study	Vegetation community	N	Shrub cover	SD	Grass cover	SD	Forb cover	SD	Grass height	SD
Aldridge 2005	SS	93	25.46	18.52	19.56	16.59	3.82	5.30	33.94	20.25
Aldridge & Brigham 2002	SS	29	31.90	21.92	31.90	21.33	8.10	6.03	30.90	19.28
Apa 1998 (1989)	MT	11	22.00	12.60	16.20	9.95	11.50	5.64	23.00	4.97
Apa 1998 (1990)	MT	10	18.80	6.32	17.00	6.01	9.00	5.06	32.40	6.01
Apa 1998 (1991)	MT	18	16.70	7.64	13.50	5.09	8.60	12.73	41.90	7.64
Fischer 1994 (Postburn)	WY	67	17.90	38.08	29.30	10.64	4.30	4.09	22.10	7.37
Fischer 1994 (Preburn)	WY	71	29.00	1.20	7.20	25.85	ND	ND	19.80	6.74
Gregg 1991(Jackass Creek)	WY	51	56.00	22.00	11.10	10.00	12.80	11.00	ND	ND
Gregg 1991 (Hart Mountain)	MT	47	51.00	15.00	18.00	20.00	6.50	5.00	ND	ND
Hanf et al. 1994	WY	20	44.00	8.90	15.00	8.94	5.00	8.94	22.00	13.42
Hausleitner 2003	MT	93	26.90	13.50	3.70	3.86	6.90	7.71	13.80	6.75
Heath et al. 1998	WY	42	19.00	12.90	8.20	4.73	2.04	2.33	16.60	3.56
Holloran 1999 (1997)	WY	32	24.90	11.80	5.50	3.53	6.70	3.64	20.80	4.25
Holloran 1999 (1998)	WY	45	25.20	9.72	4.10	1.74	7.80	3.65	17.10	2.73
Klott et al. 1993	WY	8	24.47	15.75	ND	ND	ND	ND	16.69	8.70
Lyon 2000	WY	50	25.60	991	10.60	11.70	8.20	9.21	21.30	4.25
Popham & Gutiérrez 2003	WY	40	14.50	18.97	12.50	15.81	ND	ND	23.10	18.97
Schroeder 1995	WY	78	17.24	9.76	51.03	15.94	20.64	13.35	107.88	28.62
Slater 2003 (Collett Creek)	WY	64	22.24	11.68	6.23	3.36	7.96	6.88	18.21	3.04
Slater 2003 (Salt Creek)	WY	21	24.80	8.29	3.26	2.84	1.33	1.47	16.23	3.16
Sveum et al. 1998 (1992)	WY	21	51.00	27.50	26.00	20.62	12.00	13.75	ND	ND
Sveum et al. 1998 (1993)	WY	45	59.00	26.83	27.00	20.12	21.00	20.12	ND	ND
Wakkinen 1990	WY	49	21.50	41.08	6.50	24.65	ND	ND	18.20	7.00
Wik 2002	WY	38	21.00	8.63	58.00	17.88	ND	ND	25.00	7.40

variables or used to estimate a cumulative effect size. The effect size magnitude can be ranked small (0.2), medium (0.5) or large (0.8) standard deviations from a null effect size of zero, as a general rule (Cohen 1969).

We used Hedges' d (Hedges 1982) to estimate effect sizes for sagebrush cover, grass height, grass cover and forb cover for each study because it is conducive to estimating an effect between paired treatments. With E as the treatment group and C as the control, Hedges' d was calculated as:

$$d = \frac{\overline{X}^E - \overline{X}^c}{S} J$$

where S is the pooled standard deviation and the variance ($v = \sqrt{S}$) of Hedges' d is:

$$v = \frac{N^c + N^E}{N^c N^E} + \frac{d^2}{2(N^c + N^E)}$$

and J is the correction for small sample sizes:

$$J = 1 - \frac{3}{4(N^c + N^E - 2) - 1}$$

We estimated cumulative effect size d_{++} as:

$$d_{++} = \frac{\sum_{i=1}^{n} w_i d_i}{\sum_{i=1}^{n} w_i}$$

where the weight w_i for study i is the reciprocal of the variance ($w_i = 1/v$). We used random sites as the 'control' group and use (nests or brood) sites as the 'treatment' group; thus, a positive estimate of d indicates that the variable was greater at use sites than at random points. Confidence limits (95% C.I.) were

Table 2. Studies and vegetation data used in the meta-analyses of greater sage-grouse brood-rearing habitats throughout North America. Sagebrush (shrub), grass and forb canopy cover (in %) and grass height (in cm) were vegetation variables considered in the analyses. Dominant vegetation community was described in each study as silver sagebrush (SS), mountain big sagebrush (MT) and Wyoming big sagebrush (WY). ND means that no data were available or had been reported in a manner that was usable in the meta-analysis.

	Brood-rearing area vegetation									
Brood period/study	Vegetation community	N	Shrub cover	SD	Grass cover	SD	Forb cover	SD	Grass height	SD
Early										
Drut 1992 (Hart Mt)	MT	87	23.00	8.00	15.00	7.00	11.00	7.00	ND	ND
Drut 1992 (Jackass)	WY	84	26.00	8.00	9.00	5.00	13.00	6.00	ND	ND
Hausleitner 2003	MT	31	12.70	10.02	5.80	2.78	7.50	3.90	21.70	5.57
Heath et al. 1998	WY	16	14.40	8.80	12.50	13.20	2.80	2.80	16.10	4.80
Holloran 1999	WY	67	15.83	8.67	5.89	5.74	9.25	4.93	18.59	4.94
Lyon 2000	WY	23	21.50	7.35	14.20	18.10	8.30	9.91	23.30	4.90
Sveum 1995	WY	53	11.00	7.28	17.00	21.84	22.00	14.56	ND	ND
Late										
Drut 1992 (Hart Mt)	MT	38	24.00	9.50	16.00	7.00	20.00	8.00	ND	ND
Drut 1992 (Jackass)	WY	38	29.00	15.00	8.00	5.00	8.00	6.00	ND	ND
Hausleitner 2003	MT	28	8.40	7.41	9.10	9.52	8.90	5.29	20.00	5.82
Heath et al. 1998	WY	22	11.10	10.79	15.60	19.23	10.10	11.73	15.60	6.10
Holloran 1999	WY	59	17.40	12.10	5.26	2.83	9.01	5.17	16.53	4.35
Sveum 1995	WY	19	7.00	8.72	18.00	13.08	23.00	13.08	ND	ND
Both										
Aldridge 2005	SS	139	8.85	7.90	21.20	13.56	8.88	9.08	8.85	7.90
Aldridge & Brigham 2002	SS	91	20.90	15.55	34.20	19.56	10.90	11.45	20.90	15.55
Apa 1998	MT	49	14.10	11.90	10.00	9.80	8.00	11.20	14.10	11.90
Klott et al. 1993	WY	13	16.76	5.72	ND	ND	ND	ND	10.60	11.51
Hausleitner 2003	MT	92	10.60	11.51	6.50	5.75	8.00	6.71	16.48	4.21
Slater 2003	WY	13	13.50	13.41	6.81	5.77	5.45	6.20	13.50	13.41
Wik 2002	WY	46	15.00	10.17	50.00	14.24	16.00	10.17	20.00	6.78

estimated for d, and we used bias-corrected bootstrap sampling to estimate confidence limits for d_{++}, to account for replicate years or areas within studies. We evaluated the plausibility of using additional explanatory variables to explain the observed differences in effect sizes across studies. The Q_T statistic is based on the total sum of squares and specifically tests for equal effect sizes across studies. If Q_T is greater than would be expected at random (χ^2-distribution), then additional variables (e.g. nest success rates) might help explain the observed variation in the data. We assumed that random variation occurred across nesting studies and estimated effect sizes using random effects models (Hedges 1982). However, we used mixed models to identify if there was a common effect size across brood-rearing periods (categorical data) for each cover type. The basic assumption for this analysis is that random variation occurs among effect sizes within a brood period, but may differ between periods (Gurevitch & Hedges 1999). Here the statistic Q_B can be used to assess the amount of variation accounted for between groups. If Q_B is significantly large, it suggests that effect sizes are larger between groups than expected from random. Applications of mixed-model meta-analysis are uncommon in ecological studies, but likely are the most appropriate for such data sets (Gurevitch & Hedges 1999). All meta-analytic calculations were conducted in MetaWin 2.0 (Rosenberg et al. 2000).

The quality of a research synthesis hinges on the quality of the publications available to analyse, as well as on studies not published because of a lack of significant results (Rosenberg 2005). This is referred to as publication bias and can overestimate the effect size if a large number of non-significant studies are not published or accessible. One of the simplest methods to evaluate the potential impact of publication bias is the calculation of a fail-safe number (N_+). A fail-safe number indicates the number of non-significant, unpublished (or missing) studies that would need to be added to a meta-analysis to reduce an overall statistically significant observed result to non-significance (Rosenberg 2005). We estimated fail-safe numbers for each significant effect size using Fail-Safe Number Calculator (Rosenberg 2005), and considered an effect size robust if $N_+ > 5N + 10$, where N is the observed number of studies used to estimate the effect size.

To add biological relevance to the meta-analysis, we used a weighted general linear model (PROC GLM; SAS Institute 2000) and estimated the mean and 95% C.I. for sagebrush cover, grass cover, forb cover and grass height at nest and brood-use sites.

Results

Effect sizes

Greater sage-grouse females selected nest sites with generally more sagebrush cover (d_{++} = 0.39; 95% C.I.: 0.19-0.54) and taller grass height (d_{++} = 0.28; 95% C.I.: 0.15-0.41) than random sites (Fig. 1). Grass (d_{++} = 0.13; 95% C.I.: -0.03 - +0.25) and forb cover (d_{++} = 0.15; 95% C.I.: -0.06 - +0.37) were greater at nest sites, but neither effect was significantly large. An examination of Q_T indicated that d was homogenous ($P > 0.2$) among studies for each variable and that additional information would not explain the observed effect sizes (Table 3). Shrub cover had a larger effect size than sagebrush only (d_{++} = 0.74; 95% C.I.: 0.39-1.13).

Vegetation at brood areas combined among all periods had greater forb cover (d_{++} = 0.46; 95% C.I.: 0.30-0.66), grass cover (d_{++} = 0.19; 95% C.I.: 0.09-0.30), significantly taller grasses (d_{++} = 0.29; 95% C.I.: 0.13-0.42), and less sagebrush cover (d_{++} = -0.17; 95% C.I.: -0.44 - +0.18) than random locations (see Fig. 1). However, females exhibited some variation in habitat selection for sagebrush between these periods (Q_B = 6.12, df = 2, P = 0.046). Generally, early brood-use areas were comprised of greater forb cover (d_{++} = 0.57; 95% C.I.: 0.23-0.80), grass cover (d_{++} = 0.27; 95% C.I.: 0.11-0.50), and taller grass (d_{++} = 0.39; 95% C.I.: 0.26-0.60), but less sagebrush cover (d_{++} = -0.46; 95% C.I.: -0.75 - -0.19) than random sites. Effect size for shrub cover changed moderately when using all studies (d_{++} = -0.61; 95% C.I.: -0.95 - -0.31). During late brood rearing, forb cover (d_{++} = 0.55; 95% C.I.: 0.23-0.79) and grass cover (d_{++} = 0.16; 95% C.I.: 0.05-0.30) were greater at use sites, but sagebrush cover (d_{++} = -0.08; 95% C.I.: -0.48 - +0.12) and shrub cover (d_{++} = -0.04; 95% C.I.: -0.31 - +0.15) were similar between use and random sites. For studies that pooled estimates across both periods, forb cover was greater (d_{++} = 0.27; 95% C.I.: 0.04-0.54) and grass height taller (d_{++} = 0.34; 95% C.I.: 0.20-0.48) than at random sites. Sagebrush cover (d_{++} = 0.15; 95% C.I.: -0.36 - +0.77) and grass cover (d_{++} = 0.11; 95% C.I.: -0.01 - +0.32) were greater at brood use areas but neither of these factors was significant. Examination of Q_T values indicated that effect sizes were homogenous ($P > 0.25$) except for shrub cover, and additional explanatory variables would not explain variation in effect sizes across all studies (see Table 3). The test of heterogeneity is conservative with small sample sizes and therefore interpreted in an appropriately conservative manner.

Publication bias

We conducted fail-safe calculations for 12 effect sizes that were significant (see Table 3). The effect size of disproportional use of sagebrush and grass height was robust for nest sites as was forb cover at early and late brood-rearing areas (see Table 3). Grass cover and height effect sizes for brood-rearing areas were not

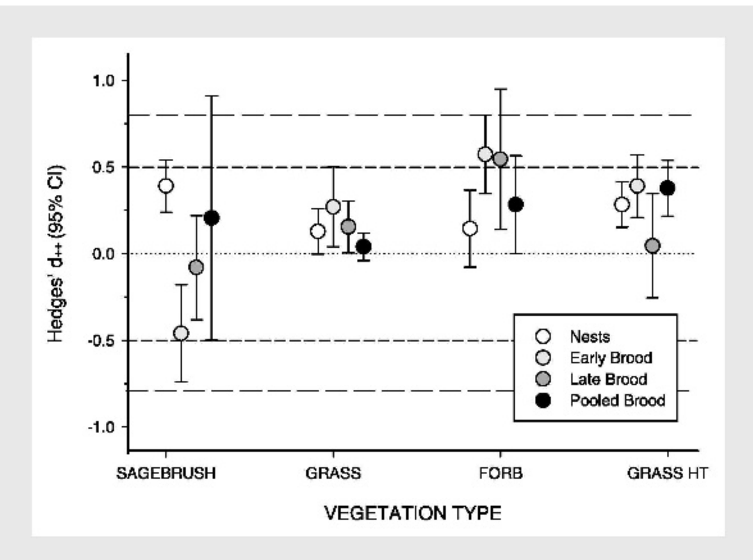

Figure 1. Cumulative effect sizes (d_{++}) by vegetation types and across nesting and brood-rearing habitats. Long-dashed lines indicate large ($d > 0.8$), small-dashed lines indicate medium ($0.8 \geq d > 0.5$), and dotted line indicates small ($0 < d < 0.5$) effects. Significant positive and negative effects indicate selection for or against a vegetation type, respectively. Estimates with 95% C.I. including 0, indicate no effect of habitat selection.

Table 3. Estimates of vegetation characteristics at greater sage-grouse use sites from 19 studies across the species range, and diagnostic statistics (Q_T, N_+) for meta-analysis. Means and confidence intervals were derived from a weighted mean linear model where the inverse of the variance was the weighting factor. The 'early' period was defined as brood habitat used < 6 weeks post hatching, the 'late' period as > 6 weeks post hatching, and 'both' were studies that pooled estimates across both periods. An asterisk (*) indicates that a fail-safe number (N_+) is robust (> 5N + 10). The fail-safe number is equivalent to the number of studies of null effect and mean weight necessary to reduce the observed significance level to $\alpha = 0.05$.

			Parameter estimates		Diagnostics			
Cover type	Period	N	$\bar{x}$	95% C.I.	Q_T	df	P	Fail safe (N_+)
Forb (%)	Nest	19	4.02	2.05-5.99	21.3	18	0.27	NA
	Early	7	6.74	3.91-9.56	4.5	6	0.61	94*
	Late	6	10.78	6.50-15.06	5.3	5	0.38	49*
	Both	6	8.51	2.92-14.10	4.4	5	0.50	13
Grass (%)	Nest	23	6.75	4.53-8.98	25.9	22	0.26	NA
	Early	7	7.56	4.35-10.76	7.5	6	0.28	14
	Late	6	7.57	4.17-10.98	3.6	5	0.61	1
	Both	6	11.44	5.79-17.10	5.4	5	0.38	NA
Sagebrush (%)	Nest	19	21.51	19.91-23.93	13.7	16	0.62	270*
	Early	4	16.84	9.59-24.08	3.2	3	0.37	14
	Late	3	10.92	1.67-20.16	1.9	2	0.38	NA
	Both	7	14.15	8.39-19.92	5.1	6	0.53	NA
Shrub cover (%)	Nest	24	25.13	20.35-29.91	35.3	23	0.05	1133*
	Early	7	18.07	13.31-22.83	5.3	6	0.50	204*
	Late	6	13.71	7.53-19.88	5.3	5	0.38	NA
Grass height (cm)	Nest	20	19.77	17.36-22.18	16.6	19	0.61	193*
	Early	4	19.78	15.91-23.65	2.8	3	0.41	5
	Late	3	17.24	12.58-21.90	1.6	2	0.45	NA
	Both	7	19.16	15.17-23.15	7.5	6	0.28	40

robust for missing studies. However, these were relatively small effect sizes (see Fig. 1). The effect size of sagebrush cover at brood-rearing areas was robust.

Parameter estimates

Sagebrush canopy cover was apparently greater at nest sites (21.5%) than at brood areas (< 16.9%; see Table 3). Combined forb (4.1%) and grass cover (6.5%) was less at nest sites than at brood areas (forb > 6.7%, grass > 7.6%). However, grass height was comparable (~19 cm) in nest and brood areas. During brood rearing, sagebrush cover decreased from early to late periods, forb cover increased, whereas grass cover and height did not change appreciably (see Table 3).

Discussion

Our study provides the first quantitative assessment of available data for greater sage-grouse habitat selection during the nesting and brood-rearing periods. We found a general effect for habitat selection across the range of these studies, as evidenced by low levels of variation in effect sizes across studies and regions. Many of our estimated effect sizes were robust to the potential impacts of publication bias, lending considerable support to the generality of our findings. There was a medium to large effect (d = 0.37-0.74) of selection for vegetation characteristics, with greater sagebrush cover for nest concealment and forb cover for females with broods. There were smaller effects (d ~ 0.2) for selection of grass height and cover by nesting and brood-rearing females. The variation of effect sizes in sagebrush cover was more substantial between brood periods, signifying a seasonal shift in habitat use.

Effect sizes

Because random variation was as expected, we can infer that greater sage-grouse females were selecting for similar nesting vegetation (greater sagebrush cover, grass cover and/or taller grasses) throughout the geographic range of these studies. This quantitative assessment supports earlier qualitative reviews of sage-grouse habitat requirements during the nesting period (Braun et al. 1977, Connelly et al. 2000) that suggested the importance of sagebrush and grass cover as well as grass height. Our study also indicated the importance of reporting sagebrush cover separately from other shrub species as there was a moderate change in effect size and increase in variance of effect size, when comparing studies reporting sagebrush versus shrub cover. Although the measurement of grass height has only recently been standardized (Connelly et al. 2003), we identified an overall selection for taller grasses at nest sites. Additionally, the relatively small selection effect of greater grass cover may have been confounded with grass height. Many short stature grasses may have been included in the estimates of grass cover, and may contribute to the relatively small effect size of grass cover at use sites.

Brood females selected early and late habitats with less sagebrush cover and greater herbaceous cover (grass and forbs) than random sites. This generalized effect for greater herbaceous cover during brood rearing is likely a result of mesic plant communities with an abundance of invertebrates and foods that are critical to the growth and development of chicks (Johnson & Boyce 1991, Drut et al. 1994). Alternatively, this effect may have been correlated with broods seeking habitats with less shrub cover and greater understory in more xeric sites. Taller grasses were selected more so during early brood rearing than during late brood rearing. The proximity of early brood rearing to nesting sites may have contributed to this result, or because females were selecting sites with less sagebrush cover, the use of taller grasses may have provided greater vertical screening and protection. However, as broods mature tall stature grasses appeared to become less important, as did sagebrush cover. For studies that pooled vegetation measurements across both brood periods the effect sizes were generally small and may have been confounded by potential effects between early and late broods. Sagebrush cover was greater at brood use sites for pooled studies and was likely due to selection for silver sagebrush *A. cana* sites in Alberta where the extent of sagebrush could be a limiting factor (Aldridge & Brigham 2002, Aldridge 2005).

Publication bias

Generally, our findings were robust to publication bias with respect to vegetation needs for each life stage. Our evaluation of potential impacts of publication bias indicated that habitat usage by greater sage-grouse at nest sites was robust for sagebrush cover and grass height, each effect requiring two to several hundred studies of 'no effect' to nullify our results. Similarly, our estimated effects of less shrub cover and greater forb cover during brood rearing were robust to publication bias. The effects of grass cover were relatively small and more susceptible to non-significant or missing studies. These findings may help guide future work to identify vegetation characteristics that should be evaluated more carefully and perhaps reduce some of this ambiguity (e.g. grass cover).

Parameter estimates

The weighted average of cover and height values were within the range specified by the greater sage-grouse management guidelines for breeding habitats (Connelly et al. 2000). Our analysis indicated that the range (95% C.I.s) of vegetation measurements encompassed those in the guidelines published by Connelly et al. (2000), recommending 15-25% sagebrush cover, $>$ 10% forb cover, $>$ 15% grass cover and $\geq$ 18-cm grass height (see Table 3). Estimates of sagebrush were not markedly different when we included studies that reported only shrub cover. Despite criticisms of the established guidelines (Bates et al. 2004, Schultz 2004), our quantitative analysis that includes new data published after 2000 strongly suggests that these values for describing breeding habitats are reasonable. Because these measurements are generally recorded over relatively small scales ($<$ 30 m), identifying the appropriate proportions of these vegetative characteristics in a larger landscape is paramount (Bates et al. 2004).

Conclusions and recommendations

The magnitude of effects sizes combined with the parameter estimates in our meta-analyses demonstrated a shift in habitat selection by females between nesting and brood-rearing periods, primarily a shift in sagebrush and forb canopy cover. However, most studies have not quantified the spatial distribution or juxtaposition of these vegetative communities. Understanding the optimum mix and spatial arrangement of these communities and their effects on demographic rates in a landscape could substantially enhance management of the greater sage-grouse. More importantly, studies of breeding habitats need to begin to examine the relationship between vegetative communities, landscape metrics (e.g. habitat patch size, fragmentation and distance to roads) and demographic rates. Similarly, as more studies begin to compare vegetation and other differences between successful and unsuccessful nests, a meta-analysis could prove useful in identifying a general effect for factors contributing to nest success.

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References

Aldridge, C.L. 2005: Identifying habitats for persistence of greater sage-grouse (Centrocercus urophasianus) in Alberta, Canada. - PhD thesis, University of Alberta, Edmonton, Canada, 250 pp.

Aldridge, C.L. & Brigham, R.M. 2002: Sage-grouse nesting and brood habitat use in southern Canada. - Journal of Wildlife Management 66: 433-444.

Apa, A.D. 1998: Habitat use and movements of sympatric sage and Columbian sharp-tailed grouse in southeastern Idaho. - PhD thesis, University of Idaho, Moscow, USA, 199 pp.

Bates, J., Davies, K. & Miller, R. 2004: Ecology of the Wyoming big sagebrush alliance in the northern Great Basin: 2004 Progress Report. - Eastern Oregon Agricultural Research Center, Burns, Oregon, USA, 65 pp.

Braun, C.E. 1998: Sage grouse declines in western North America: what are the problems? - Transactions of Western Association of Fish and Wildlife Agencies 78: 139-156.

Braun, C.E., Britt, T. & Wallestad, R.O. 1977: Guidelines for maintenance of sage grouse habitats. - Wildlife Society Bulletin 5: 99-106.

Cohen, J. 1969: Statistical power analysis for the behavioral sciences. - Erlbaum, Hillsdale, New Jersey, USA, 567 pp.

Connelly, J.W. & Braun, C.E. 1997: Long-term changes in sage grouse Centrocercus urophasianus populations in western North America. - Wildlife Biology 3: 229-234.

Connelly, J.W., Schroeder, M.A., Sands, A.R. & Braun, C.E. 2000: Guidelines to manage sage grouse populations and their habitats. - Wildlife Society Bulletin 28: 967-985.

Connelly, J.W., Reese, K.P. & Schroeder, M.A. 2003: Monitoring of greater sage-grouse habitats and populations. - College of Natural Resources Experiment Station Bulletin 80, University of Idaho, Moscow, USA, 50 pp.

Daubenmire, R. 1959: A canopy-coverage method of vegetational analysis. - Northwest Science 33: 43-64.

Drut, M.S. 1992: Habitat use and selection by sage grouse broods in southeastern Oregon. - M.Sc. thesis, Oregon State University, Corvallis, USA, 44 pp.

Drut, M.S., Crawford, J.A. & Gregg, M.A. 1994: Brood habitat use by sage grouse in Oregon. - Great Basin Naturalist 54: 170-176.

Fischer, R.A. 1994: The effects of prescribed fire on the ecology of migratory sage grouse in southeastern Idaho. - PhD thesis, University of Idaho, Moscow, USA, 150 pp.

Gregg, M.A. 1991: Use and selection of nesting habitat by sage grouse in Oregon. - M.Sc. thesis, Oregon State University, Corvallis, USA, 46 pp.

Gurevitch, J. & Hedges, L.V. 1999: Statistical issues in ecological meta-analyses. - Ecology 80: 1142-1149.

Hall, J.A., Rosenthal, R., Tickle-Degnen, L. & Mosteller, F. 1994: Hypotheses and problems in research synthesis. - In: Cooper, H. & Hedges, L.V. (Eds.); The Handbook of research synthesis. Russell Sage Foundation, New York, USA, pp. 17-28.

Hanf, J.M., Schmidt, P.A. & Groshens, E.B. 1994: Sage grouse in the high desert of central Oregon: results of a study, 1988-1993. - Bureau of Land Management, Portland, Oregon, USA, 67 pp.

Hausleitner, D. 2003: Population dynamics, habitat use and movements of greater sage-grouse in Moffat County, Colorado. - M.Sc. thesis, University of Idaho, Moscow, USA, 162 pp.

Heath, B.J., Straw, R., Anderson, S.H., Lawson, J. & Holloran, M.J. 1998: Sage-grouse productivity, survival, and seasonal habitat use among three ranches with different livestock grazing, predator control, and harvest management practices. - Project Completion Report, Wyoming Game and Fish Department, Cheyenne, USA, 71 pp.

Hedges, L.V. 1982: Estimation of effect size from a series of independent experiments. - Psychological Bulletin 92: 490-499.

Hedges, L.V. & Olkin, I. 1985: Statistical methods for meta-analysis. - Academic Press, San Diego, California, USA, 369 pp.

Holloran, M.J. 1999: Sage grouse (Centrocercus urophasianus) seasonal habitat use near Casper, Wyoming . - M.Sc. thesis, University of Wyoming, Laramie, USA, 123 pp.

Johnson, G.D. & Boyce, M.S. 1991: Survival, growth, and reproduction of captive-reared sage grouse. - Wildlife Society Bulletin 19: 88-93.

Klott, J.H., Smith, R.B. & Vullo, C. 1993: Sage grouse habitat use in the Brown's Bench area of south-central Idaho. - United States Department of Interior, Bureau of Land Management, Idaho State Office, Technical Bulletin 93-4, Boise, Idaho, USA, 14 pp.

Lyon, A.G. 2000: The potential effects of natural gas development on sage grouse near Pinedale, Wyoming. - M.Sc. thesis, University of Wyoming, Laramie, USA, 121 pp.

Popham, G.P. & Gutiérrez, R.J. 2003: Greater sage-grouse Centrocercus urophasianus nesting success and habitat use in northeastern California. - Wildlife Biology 9: 327-334.

Rosenberg, M.S. 2005: The file-drawer problem revisited: a general weighted method for calculating fail-safe numbers in meta-analysis. - Evolution 59: 464-468.

Rosenberg, M.S., Adams, D.C. & Gurevitch, J. 2000: Metawin: statistical software for Meta-analysis Version 2.0. - Sinauer Associates, Sunderland, Massachusetts, USA, 128 pp.

SAS Institute 2000: SAT/STAT user's guide, Release 8.0 Edition. - SAS Institute, Inc. Cary, North Carolina, USA. Available at: http://v8doc.sas.com/sashtml/

Schroeder, M.A. 1995: Productivity and habitat use of sage grouse in north-central Washington. Job Progress Report W-96-R. -Washington Department of Fish and Wildlife, Olympia, USA, 23 pp.

Schroeder, M.A., Aldridge, C.L., Apa, A.D., Bohne, J.R., Braun, C.E., Bunnell, S.D., Connelly, J.W., Diebert, P., Gardner, S.C., Hilliard, M.A., Kobriger, G.D., McAdam, S.M., McCarthy, C.W., McCarthy, J.J., Mitchell, D.L., Rickerson, E.V. & Stiver, S.J. 2004: Distribution of sage-grouse in North America. - Condor 106: 363-376.

Schultz, B. 2004: Analysis of studies used to develop herbaceous height and cover guidelines for sage grouse nesting habitat. - Cooperative Extension Special Publication SP-04-11, University of Nevada, Reno, USA, 25 pp.

Slater, S.J. 2003: Sage-grouse (Centrocercus urophasianus) use of different-aged burns and the effects of coyote control in southwestern Wyoming. - M.Sc. thesis, University of Wyoming, Laramie, USA, 177 pp.

Sveum, C.M. 1995: Habitat selection by sage-grouse hens during the breeding season in south-central Washington. - M.Sc. thesis, Oregon State University, Corvallis, USA, 86 pp.

Sveum, C.M., Edge, W.D. & Crawford, J.A. 1998: Nesting habitat selection by sage grouse in south-central Washington. - Journal of Range Management 51: 265-269.

Wakkinen, W.L. 1990: Nest site characteristics and spring-summer movements of migratory sage grouse in south-eastern Idaho. - M.Sc. thesis, University of Idaho, Moscow, USA, 57 pp.

Wambolt, C.L., Frisina, M.R., Knapp, S.J. & Frisina, R.M. 2006: Effect of method, site, and taxon on line-intercept estimates of sagebrush cover. - Wildlife Society Bulletin 34: 440-445.

Wik, P.A. 2002: Ecology of greater sage-grouse in south-central Owyhee County, Idaho. - M.Sc. thesis, University of Idaho, Moscow, USA, 141 pp.

NESTING AND BROOD-REARING HABITAT SELECTION OF GREATER SAGE-GROUSE AND ASSOCIATED SURVIVAL OF HENS AND BROODS AT THE EDGE OF THEIR HISTORIC DISTRIBUTION

BY

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NESTING AND BROOD-REARING HABITAT SELECTION OF GREATER SAGE-GROUSE AND ASSOCIATED SURVIVAL OF HENS AND BROODS AT THE EDGE OF THEIR HISTORIC DISRIBUTION

This thesis is approved as a creditable and independent investigation by a candidate for the Master of Science degree and is acceptable for meeting the thesis requirements for this degree. Acceptance of this thesis does not imply that the conclusions reached by the candidate are necessarily the conclusions of the major department.

18 April 2007

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ABSTRACT

NESTING AND BROOD-REARING HABITAT SELECTION OF GREATER SAGE-GROUSE AND ASSOCIATED SURVIVAL OF HENS AND BROODS AT THE EDGE OF THEIR HISTORIC DISRIBUTION

Katie. M Herman-Brunson

May 2007

Greater sage-grouse (*Centrocercus urophasianus*) once occurred in 12 states and 3 Canadian provinces. Sage-grouse populations have declined over the last 60 years due to extensive habitat alteration and loss. Concerns for the management and conservation of greater sage-grouse and their habitats have resulted in petitions to list them under the Endangered Species Act. In North Dakota, sage-grouse are confined to approximately 800 square miles of sagebrush habitat, which is facing severe habitat fragmentation and habitat loss. Sage-grouse in North Dakota are not isolated, but are contiguous with populations in Montana and South Dakota. Annual rates of change suggest a long-term population decline in North Dakota, declining 2.79% per year from 1965 to 2003. The species is listed as a Priority Level 1 Species of Special Concern in the state. The objectives of this study were to estimate nest survival, hen and brood survival, and associated nest and brood-site habitat selection of sage-grouse in southwestern North Dakota. The study was conducted during the spring and summer of 2005 and 2006 in Bowman County, North Dakota. Nest-sites were monitored to determine nest fate and broods were monitored by tracking radio-marked adults that successfully hatched young.

Habitat selection was characterized by comparing vegetation at nest-sites and brood-sites to vegetation points at randomly selected sites. I found 34 nests from 39 female sage-grouse (21 in 2005, 18 in 2006) that were radio-marked. Vegetation measurements were taken at 34 nest-sites and 50 random points. I collected vegetation measurements from 130 brood-sites and 107 random sites. Nest survival averaged 31% (33% in 2005 and 30% in 2006). The best model of nest survival included daily precipitation. Models that contained percent grass cover and grass height from the Robel pole also had substantial support (i.e., < 2 AIC units) to explain nest survival. One model strongly supported characteristics associated with selection of nest-sites that included percent total cover, 1-m VOR, and sagebrush density. Sage-grouse nests were positively associated with more total cover, 1-m VOR, and sagebrush density than were present at random sites. In 2005, hen survival was 84% (95%CI: 0.67 to 1.00, n = 20) from capture date through the brood-rearing season, and 60% (95%CI: 0.44 to 0.76, n = 39) in 2006. I monitored 7 broods in 2005, with an average of 6.86 ± 0.95 chicks/hen at hatch. At 3 weeks post hatch, the average brood size was 2.34 chicks/hen representing 34% apparent survival. In 2006, 6 broods averaged 6.67 ± 1.03 chicks/hen at hatch. At 3 weeks post hatch, the average brood size was 2.83 chicks/hen representing 42% apparent survival. A total of 38 sage-grouse chicks were radio-marked (13 in 2005, and 25 in 2006). Chick survival from hatch date to 3 weeks post hatch, combined with those that survived to 5-6 weeks of age and were able to be captured, 17% of the chicks were estimated to recruit into the population in December 2005 and 13% in December 2006. The majority of identifiable predation events on radio-marked sage-grouse chicks were from canids. One model of

brood site selection was positively associated with more total forb, total grass, and total sagebrush than was present at randomly selected sites, and negatively associated to percent bareground, sagebrush height and sagebrush width. Brood sites consisted of 6-16% forb cover, 29-34 % grass cover, 5% sagebrush cover and approximately 30-38 cm tall sagebrush plants, and 50-53 cm wide sagebrush plants. Percent bareground cover consisted of 11-25% at brood sites. I recommend that managers develop strategies to preserve the integrity of shrubsteppe habitat in southwestern North Dakota. Herbaceous cover in sagebrush habitats is an important component of nesting and brood-rearing habitat for sage-grouse. Thus I recommend management activities that maintain or restore dense, taller residual grass within sage-grouse habitat.

TABLE OF CONTENTS

LIST OF TABLES

Table Page

LIST OF FIGURES

Figure Page

LIST OF APPENDICES

CHAPTER 1-GENERAL INTRODUCTION

INTRODUCTION

Greater sage-grouse (*Centrocercus urophasianus*) populations were once distributed throughout 12 states in western North America, and 3 Canadian provinces. Populations of sage-grouse have undergone long-term population declines due to extensive alteration and loss of sagebrush (*Artemisia* spp.) habitats (Schroeder et al. 2004). Sage-grouse started to decline during the early twentieth century, corresponding with the American westward movement and the arrival of European settlers in the 1800s, increasing numbers of livestock, and intense agriculture practices (Patterson 1952, Gill 1966). Estimates of regional declines in sage-grouse have ranged from 17 to 47% (Connelly and Braun 1997). There was a corresponding decline in sagebrush habitat quality and quantity due to agriculture, invasive exotics (i.e., cheatgrass), overgrazing, energy development, drought, fire, and herbicides (Patterson 1952, Homer et al. 1993, Gregg et al. 1994, Connelly and Braun 1997, Braun 1998, Connelly et al. 2000, Hemstrom et al. 2002). Greater sage-grouse currently occupy 56% of their historic range (Schroeder et al. 2004), and 17% of their prehistoric range in North Dakota.

Greater sage-grouse are obligates of sagebrush ecosystems that dominate most of western North America. Sagebrush is required for food, shelter, and as a water source for sage-grouse (Swenson 1987, Fischer et al. 1996, Schroeder et al. 1999). During the winter months, sagebrush is the only source of food (Hupp and Braun 1989, Welch et al. 1991) with the sage-grouse's diet consisting of leaves and buds (Welch et al. 1991, Homer et al. 1993, Connelly et al. 2000). Sage-grouse are unique among the Galliformes because they lack a well developed gizzard, which makes their dependence on soft vegetation critical. Since their diet is based mostly on herbaceous leaves of sagebrush, there is no need for a highly developed gizzard (McCarthy and Kobriger 2005).

Sagebrush steppe is important as a management indicator for sage-grouse in all shrub-steppe vegetation communities. Sagebrush coexists with understory forbs that are important for female sage-grouse during nesting and brood-rearing (Drut et al. 1994a, Crawford 1997, Connelly et al. 2000). Greater sage-grouse nest beneath sagebrush (Patterson 1952, Gill 1966, Connelly et al. 1991, Musil et al. 1994, Sveum et al. 1998), where females may show nest-site fidelity from year to year (Fischer et al. 1993). Klebenow (1969) and Wallestad (1975) found that sagebrush provided female sage-grouse with nesting cover and early brood-rearing habitat. Females typically chose nest-sites with horizontal cover of greater than 73% (Musil et al. 1994, Connelly et al. 2000), and tall residual grasses of greater than 18 cm and medium shrubs from 40-80 cm of height (Gregg et al. 1994, Sveum et al. 1998, Connelly et al. 2000).

Recent research of nesting sage-grouse emphasizes the importance of herbaceous cover in determining nest fate. Nest-sites coexist in areas of greater than 38% sagebrush cover because of greater amounts of forbs (Klebenow 1969, Connelly et al. 2000). Presence of forbs increased initiation rates of hens and nutrient acquisition by chicks (Johnson and Boyce 1990, Barnett and Crawford 1994, Drut et al. 1994b, Crawford 1997, Sveum et al. 1998, Gregg 2001).

The decline of sage-grouse throughout their range has caused them to be listed as a Priority Level 1 Species of Special Concern in North Dakota. Immediate research and conservation action is necessary for sage-grouse and their habitats (Wambolt et al. 2002, Schroeder et al. 2004). Similar concerns nationally also have led to petitioning the U.S. Fish and Wildlife Service to protect the greater sage-grouse under the Endangered Species Act (ESA), which would have a significant impact on private and federal land management practices within the United States. North Dakota is situated on the eastern edge of distribution of sage-grouse and sagebrush steppe communities; thus this species may not utilize habitats as predictably as in the interior areas of sagebrush country (Smith 2003, Lewis 2004).

Little is known about the finite habitat use or seasonal movements of sage-grouse in North Dakota. Sage-grouse have never been widespread in North Dakota and are currently confined to the southwestern portion of the state, in western Bowman, Slope, and Golden Valley counties (Johnson and Knue 1989, McCarthy and Kobriger 2005). The North Dakota population is contiguous with sage-grouse populations in South Dakota and Montana (McCarthy and Kobriger 2005). My study was conducted to gather data on seasonal habitat use during nesting and brood-rearing, and survival rates of female sage-grouse and chicks in southwestern North Dakota. In North Dakota and other areas of western United States, sage-grouse inhabit areas where *Artemisia tridentaem wyomingensis* and other related forbs and grasses occur (McCarthy and Kobriger 2005). Nesting studies are important to ascertain data in regards to nest success, nesting habitat, and to quantify nest-site vegetation to guide management and conservation activities for sage-grouse habitats.

The objectives of this study were to (1) determine and quantify nesting and brood-rearing habitat selection of radio marked sage-grouse in North Dakota; (2) estimate survival of radio-marked female sage-grouse in southwestern North Dakota and (3) investigate and determine specific causes for observed sage-grouse mortalities. Other objectives were to (4) estimate nest success of radio-marked female sage-grouse in North Dakota, (5) evaluate the cause and timing of nest failures (e.g., abandonment, predation), (6) estimate brood survival of radio-marked female sage-grouse in southwestern North Dakota, and (7) investigate the cause(s) of brood/chick mortality. Addressing these objectives will help resource managers in the development of management recommendations to benefit state and federal wildlife and habitat management agencies that coordinate management of greater sage-grouse and their habitats. These comparisons will benefit managers by providing a measure of management success and failures for sage-grouse. This research also will aid in providing information on habitat selection, movements, and survival of sage-grouse at the eastern fringe range of existence; an area where basic reproductive ecology of the species has not been studied. Data from this study when compared to those from stable populations in the heart of sagebrush range can help elucidate ultimate factors required by sage-grouse.

STUDY AREA

The study area was located in Bowman and Slope counties in southwestern North Dakota (Figure 1). Topography was flat to unglaciated gently-rolling prairie with few buttes and intermittent streams. Soil orders consisted of Entisols, Alfisols, Mollisols, Inceptisols, Mollisols, and Aridisols (Johnson 1976, Kalvels 1982, Johnson 1988, Smith 2003). Annual precipitation ranged from 35.6 cm to 40.6 cm with a majority falling from April to September. Annual summer and winter temperatures ranged from 9.9°C to 27.5°C and from -15.6°C to 0.2°C, respectively (Opdahl et al. 1975, Thompson 1978, Smith 2003). Precipitation for 2005 was 35.88 cm and average January and July temperatures were –10°C and 21°C, respectively (North Dakota Agricultural Weather Network, 2006).

Vegetation was a mixture of shrubland, with an understory of perennial and annual forbs and grasses, with open grassland (Johnson and Larson 1999). Dominant shrub species included silver sagebrush (*A. cana*), big sagebrush (*A. tridentata*), western snowberry (*Symphoricarpos occidentalis*), rubber rabbitbrush (*Chrysothamnus nauseosus*), and greasewood (*Sarcobatus vermiculatus*) (Johnson and Larson 1999).

Dominant grasses in the area consisted of kentucky bluegrass (*Poa pratensis*), western wheatgrass (*Pascopyrum smithii*), japanese brome (*Bromus japonicus*), needle and thread (*Stipa comata*), and junegrass (*Koeleria macrantha*). Dominant forbs were common yarrow (*Achillea millefodium*), common dandelion (*Taraxacum officinale*), and textile onion (*Allim textile*) (Johnson and Larson 1999).

The majority of the land in the study site was publicly owned and under the jurisdiction of the Bureau of Land Management (BLM). The normal stocking rate for grazing in Bowman County is 4-10 acres per AUM, but in areas with rough terrain/poor soils it can be as high as 14 acres per AUM. Allotments differ from low management intensity areas, because they are small tracts of public land surrounded by large blocks of private land, to high intensity areas, which are large blocks of land, with livestock numbers reported accordingly to land availability. Livestock may or may not be rotated on low management areas, but are rotated through grazing pastures on a schedule in larger blocks of public land. Most ranchers do not use the federal land year round, but year-round grazing is allowed on federal lands to provide flexibility to lessee grazing needs (Mitch Iverson, BLM personal communications).

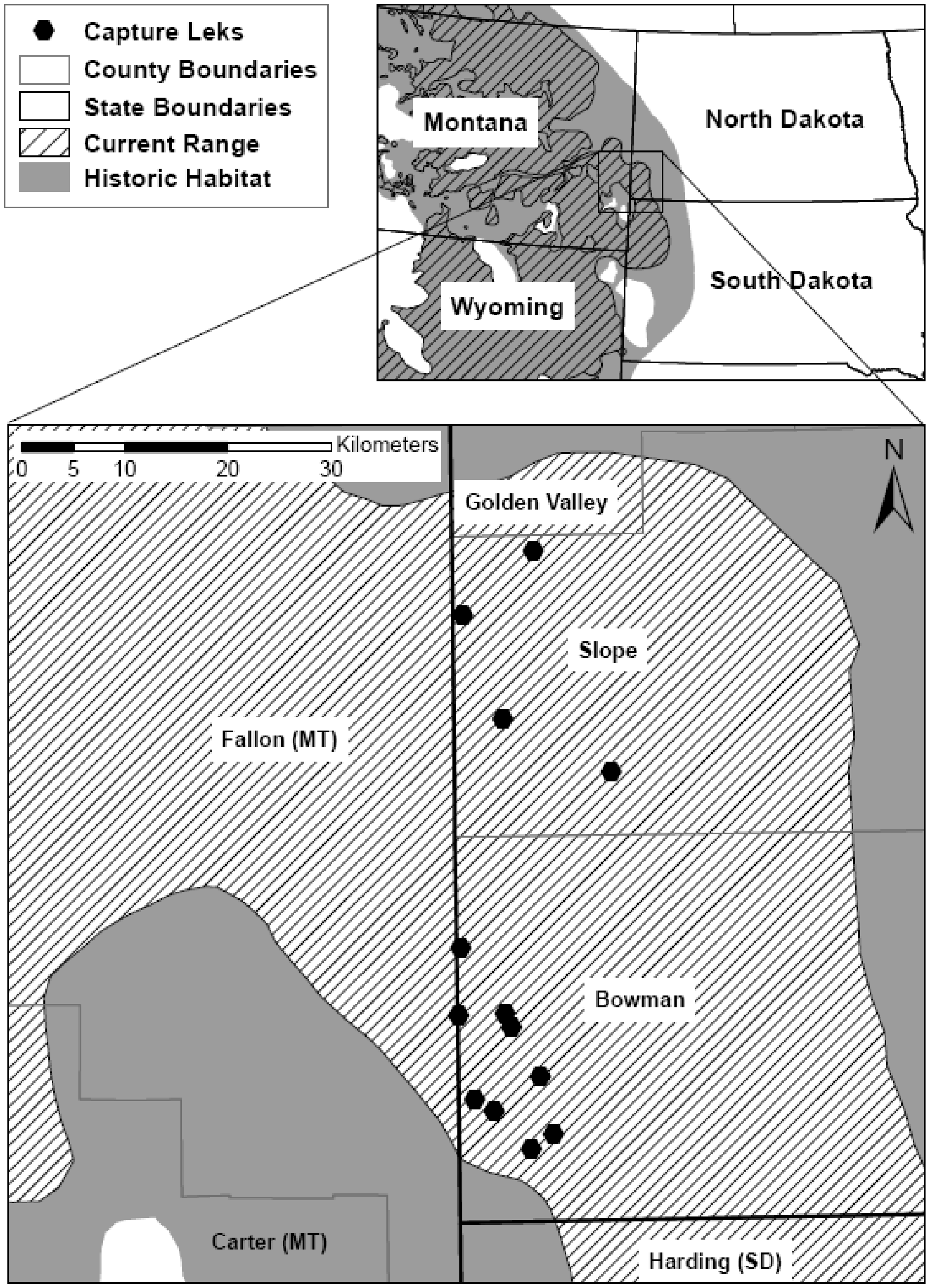


Figure 1. Study area of Bowman, Slope, and Golden Valley counties with capture leks documented during 2005 and 2006.

LITERATURE CITED

Barnett, J.K., and J.A. Crawford. 1994. Pre-laying nutrition of sage grouse hens in Oregon. Journal of Range Management 47:114-118.

Braun, C.E. 1998. Sage grouse declines in western North America: what are the problems? Pages 139-156 *in* Proceedings of the Western Association of Fish and Wildlife Agencies 78th Annual Conference in Jackson, WY, USA.

Connelly, J.W., W.L. Wakkinen, A.D. Apa, and K.P. Reese. 1991. Sage grouse use of nest sites in southeastern Idaho. Journal of Wildlife Management 55:521-524.

Connelly, J.W., and C.E. Braun. 1997. Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. Wildlife Biology 3:229-234.

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:1-19.

Crawford, J.A. 1997. Importance of herbaceous vegetation to female sage grouse *Centrocercus Urophasianus* during the reproductive period: a synthesis of research from Oregon, USA. Wildlife Biology 3:271.

Drut, M.S., J.A. Crawford, and M.A. Gregg. 1994a. Brood habitat use by sage grouse in Oregon. Great Basin Naturalist 54:170-176.

Drut, M.S., W.H. Pyle, and J.A. Crawford. 1994b. Technical note: diets and food selection of sage grouse chicks in Oregon. Journal of Range Management 47:90-93.

Fischer, R.A., A.D. Apa, W.L. Wakkinen, K.P. Reese, and J.W. Connelly. 1993. Nesting-area fidelity of sage grouse in southeastern Idaho. The Condor 95:1038-1041.

Fischer, R.A., K.P. Reese, and J.W. Connelly. 1996. Influence of vegetal moisture content and nest fate on timing of female sage grouse migration. The Condor 98:868-872.

Gill, R.B. 1966. A literature review on sage grouse. Colorado Game, Fish, and Parks, Research Division and Cooperative Wildlife Research Unit, Denver, Colorado, USA.

Gregg, M.A., J.A. Crawford, M.S. Drut, and A.K. Delong. 1994. Vegetational cover and predation of sage grouse nests in Oregon. Journal of Wildlife Management 58:162-166.

Gregg, M.A. 2001. Sage grouse reproductive ecology and habitat associations: applications to management. Pages 8-12 *in* C.S. Boyd and T.C. Westfall, organizers. Proceedings of the ecology and management of sage grouse symposium. 54th Annual Meeting of the Society of Range Management, Kona, Hawii, USA.

Hemstrom, M.A., M.J. Wisdom, W.J. Hann, M.M. Rowland, B.C. Wales, and R.A. Gravenmier. 2002. Sagebrush-steppe vegetation dynamics and restoration potential in the interior Columbia Basin, U.S.A. Conservation Biology 16:1243-1255.

Homer, C.G., T.C. Edwards, Jr., R.D. Ramsey, and K.P. Price. 1993. Use of remote sensing methods in modeling sage grouse winter habitat. Journal of Wildlife Management 57:78-84.

Hupp, J.W. and C.E. Braun. 1989. Topographic distribution of sage grouse foraging in winter. Journal of Wildlife Management 53:823-829.

Johnson, P.R. 1976. Soil survey of Butte County, South Dakota. U.S. Department of Agriculture, South Dakota Agriculture Experiment Station, Brookings, South Dakota, USA.

Johnson, W.F. 1988. Soil survey of Harding County, South Dakota. U.S. Department of Agriculture, South Dakota Agricultural Experiment Station, Brookings, South Dakota, USA.

Johnson, M.D. and J. Knue. 1989. Feathers from the prairie. North Dakota Game and Fish Department, Bismarck, North Dakota, USA.

Johnson, G.D. and M.S. Boyce. 1990. Feeding traits with insects in the diet of sage grouse chicks. Journal of Wildlife Management 54:89-91.

Johnson, J.R., and G.E. Larson. 1999. Grassland plants of South Dakota and the northern Great Plains. South Dakota State University. Brookings, South Dakota, USA.

Kavels, J. 1982. Soil survey of Fall River County, South Dakota. U.S. Department of Agriculture, South Dakota Agricultural Experiment Station, Brookings, South Dakota, USA.

Klebenow, D.A. 1969. Sage grouse nesting and brood habitat in Idaho. Journal of Wildlife Management 33:649-662.

Lewis, A.R. 2004. Sagebrush steppe habitats and their associated bird species on South Dakota, North Dakota, and Wyoming: Life on the edge of the sagebrush ecosystem. PhD Dissertation, South Dakota State University, Brookings, SD. 126pp.

McCarthy, J.J., and J.D. Kobriger. 2005. Management plan and conservation strategies for greater sage-grouse in North Dakota. North Dakota Game and Fish Department. 96pp.

Musil, D.D., K.P. Reese, and J.W. Connelly. 1994. Nesting and summer habitat use by translocated sage grouse (*Centrocercus urophasianus*) in central Idaho. Great Basin Nat. 54:228-233.

North Dakota Agricultural Weather Network. 2006. Climatology Information, Bowman ND. http://ndawn.ndsu.nodak.edu. (Accessed Sept 5, 2006).

Opdahl, D.D., W.F. Freymiller, L.P. Haugan, R.J. Kukowski, B.C. Baker, and J.G. Stevens. 1975. Soil survey of Bowman County, North Dakota. U.S. Department of Agriculture, North Dakota Agricultural Experiment Station, Fargo, North Dakota, USA.

Patterson, R.L. 1952. The sage grouse of Wyoming. Sage Books, Inc. Denver, Colo. 341pp.

Schroeder, M.A., J.R. Young, and C.E. Bruan. 1999. Sage grouse (*Centrocercus urophasianus*). Pages 1-28 *in* A. Poole and F. Gill, editors. The birds of North America, No. 425. The Birds of North America, Inc., Philadelphia, Pennsylvania, USA.

Schroeder, M.A., C.L. Aldridge, A.D. Apa, J.R. Bohne, C.E. Braun, S.D. Bunnell, J.W. Connelly, P.A. Diebert, S.C. Gardner, M.A. Hilliard, G.D. Kobriger, C.W. McCarthy. 2004. Distribution of Sage Grouse in North America. Condor 106:363-376.

Smith, J. 2003. Greater sage-grouse on the edge of their range: leks and surrounding landscapes in the Dakotas. M.S. Thesis, South Dakota State University, Brookings, SD. 213pp.

Sveum, C.M. W.D. Edge, and J.A Crawford. 1998. Nesting habitat selection by sage grouse in southcentral Washington. Journal of Range Management 51:265-269.

Swenson, J.E. 1987. Decrease of sage grouse *Centrocercus urophasianus* after ploughing of sagebrush steppe. Biology Conservation 41:125-132.

Thompson, K.W. 1978. Soil Survey of Slope County, North Dakota. U.S. Department of Agriculture, Soil Conservation Service and Forest Service, North Dakota Agricultural Experiment Station, Fargo, North Dakota, USA.

Wallestad, R.O. 1975. Life history and habitat requirements of sage grouse in central Montana. Montana Game and Fish and Bureau of Land Management, Billings, Montana, USA.

Wambolt, C.L., A.J. Harp, B.L. Welch, N. Shaw, J.W. Connelly, K.P. Reese, C.E. Braun, D.A. Klebenow, E.D. McArthur, J.G. Thompson, L.A. Torell, and J.A. Tanaka. 2002. Conservation of greater sage-grouse on public lands in western U.S.:implications of recovery and management policies. Policy Analysis Center for Western Public Lands, Policy Paper SG-02-02, Caldwell, ID.

Welch, B.L., F.J. Wagstaff, and J.A. Roberson. 1991. Preference of wintering sage grouse for big sagebrush. Journal of Range Management 44:462-465.

CHAPTER 2- NESTING ECOLOGY OF GREATER SAGE-GROUSE AT THE EASTERN EDGE OF THEIR HISTORIC DISTRIBUTION

INTRODUCTION

Following the arrival of European settlers in the 1800s, greater sage-grouse (*Centrocercus urophasianus*) habitat has been changing continuously (Girard 1937, Patterson 1952). Greater sage-grouse experienced population declines from 45- 80% across their range by the 1950s (Braun 1998) and during the 10-year period from 1985-1995 sage-grouse populations declined 33% (Connelly and Braun 1997). Historically, sage-grouse range is limited to sagebrush (*Artemisia* spp.) vegetation types in at least 12 states and 3 Canadian provinces, but currently they reside in 11 states and 2 Canadian provinces (Schroeder et al. 2004). Sage-grouse are sagebrush obligates and degradation and loss of sagebrush resulted in population declines and constriction of the range (Wisdom et al. 2005, Welch 2005).

Discovery of oil and gas throughout the United States in the 1930s and 1940s impacted wildlife habitats in numerous ways. In Colorado, the initial impacts of oil and gas development caused sage-grouse populations to decline drastically from noise, habitat loss, infrastructure and human activities (Braun 1987). The long-term effects is unknown, but there is no evidence that populations of sage-grouse will recover to pre-disturbance populations, and the length of recovery time for these habitats is estimated to range from 2-30 years (Braun 1998). Grazing by domestic livestock, fire, construction, power lines, fences, and drought also contributed to loss of sagebrush (Braun 1998, Schroeder et al. 1999, Welch 2005). These changes have affected nest-site and brood rearing habitats by fragmenting the landscape and causing habitat loss (Connelly and Braun 1997, Beck et al. 2003, Crawford et al. 2004).

The importance of sagebrush for nesting habitat of sage-grouse is well documented (Girard 1937, Patterson 1952, Gill 1965, Wallestad and Pyrah 1974, Gray 1967, Petersen 1980, Autenreith 1981, Connelly et al. 1991, Musil et al. 1994). Understanding which characteristics are important for nest-site habitat selection and the associated factors that affect nest success is critical management, conservation, or rehabilitation of sagebrush habitats to benefit sage-grouse. Braun et al. (1977) reported that female sage-grouse typically nested in stands of medium density sagebrush within 3 km of leks. Dense understory vegetation and overstory cover at nest-sites were critical factors determining nest-site selection (DeLong et al. 1995). Vegetation characteristics at successful nest-sites included shrubs greater than 18 cm tall and > 31% canopy cover (Barnett and Crawford 1994). Despite well understood nesting habitat in the core of sage-grouse range, knowledge of reproductive ecology and habitat selection by sage-grouse occurring at the eastern range of their distribution is limited.

The sage-grouse population in North Dakota is contiguous with populations in Montana and South Dakota (McCarthy and Kobriger 2005). Annual rates of change suggest a long-term population decline of about 2.79% per year from 1965 to 2003 (McCarthy and Kobriger 2005). Current breeding populations in North America are estimated to be 3 to 6 times lower than occurred in the late 1960s to early 1970s (Connelly et al. 2004). Sage-grouse are a Priority Level 1 Species of Special Concern in North Dakota. With this listing, it is recommended that immediate research and conservation actions be taken. Thus, population declines may be related to declining habitat quality which may result in decreased survival and productivity; however, the significance of these factors is unknown. The fragmentation of sagebrush habitats could render them unsuitable as nesting habitat and could contribute to population declines by reducing nest success and overall population productivity.

My objectives were to determine and quantify nest-site habitat selection of sage-grouse in North Dakota, and estimate specific factors that affect sage-grouse nest survival. Other objectives were to estimate nest survival of radio-marked female sage-grouse in North Dakota and evaluate the cause and timing of nest failures (e.g., abandonment, predation), followed by development of models to best explain nest survival and nest-site habitat selection. This data will help in the development of management recommendations to assist state and federal wildlife and habitat management agencies that coordinate management of greater sage-grouse and their habitats.

METHODS

Data Collection

Capture and Marking. – I captured birds at night on or near leks from 31 March – 23 April 2005 and 27 March – 27 April 2006. I used hand-held spotlights to locate birds and approached them while shining the spotlight to confuse them and then used long-handled nets to capture the hens (Giesen et al. 1982). I recorded age, sex, weight, and placed leg-bands and 20-gram necklace type radio transmitters with mortality sensors on each bird (Advanced Telemetry Systems, Isanti, Minnesota). Each bird was released at the point of capture. The transmitters were less than 2% of the bird's body weight.

Monitoring Radio-marked Hens. - I located radio-marked hens from aerial and ground radiotelemetry. Ground telemetry locations were made weekly using a hand-held 3-element yagi antenna. I recorded locations using a hand-held GPS unit when a nest was initiated. I monitored the hen ≥ 2 times each week to determine nest fate. I marked each nest with flagging approximately 20-40 m south. After the hen began incubation, I flushed her from the nest and determined incubation stage by floating (Hays and LeCroy 1971; Appendix A). If the hen was absent from the nest, the nest was examined to determine nest fate. Nests that were predated, I searched the immediate area for hair, tracks, scat, or any other sign that would indicate the species of predator (Sargeant et al. 1998). Successfully hatched nests were determined by membrane conditions of the egg or visual observation of a brood with the radio-marked hen. Nests were considered successful if ≥ 1 egg hatched. I estimated egg hatchability as percentage of eggs present at the time of hatching which produced chicks. I classified nests with eggshell fragments firmly attached to shell membranes or missing eggs as unsuccessful.

Habitat Measurements. - I recorded vegetation measurements at nest-sites and random locations ≤ 3 km of leks during May and June of 2005 and 2006. Coordinates of nests and random sites were entered into a GPS to locate the point in the field. The accuracy of GPS units was usually less than ± 10 m. Because nests are usually located beneath a shrub, the random site was then centered over the nearest shrub. I recorded slope and aspect for each nest-site using a linometer and compass, respectively as the downhill direction from each nest. At each nest and random site, I established four 50-m transects which were centered over the nest or random site. I recorded species, height, length, and width of sagebrush at each nest and random site. At each 10-m interval (n = 20) along each transect I recorded the distance to the nearest sagebrush using the point-centered-quarter method (Cottam and Curtis 1956). For every sagebrush encountered, I also recorded the height, length, width of the sagebrush, and height of grass growing beneath the shrub. I estimated visual obstruction and height of grass using a modified Robel pole delineated in 2.54 cm increments (Robel et al. 1970, Benkobi et al. 2000). In order to avoid trampling on the vegetation, I viewed the pole from 3 directions for the 1 to 5 m measurement intervals. Herbaceous canopy cover was estimated at the nest or random point, and at 10-m intervals along 50-m transects in 0.10 m^2 quadrats (see Appendix B for species identification; Daubenmire 1959). I recorded total cover, total sagebrush, total grass, total forb, litter, bareground and dominant species of grasses and shrubs in each quadrat. I obtained measures of maximum and minimum daily temperature, and daily precipitation from the closest weather station in Bowman County (North Dakota Agricultural Weather Network). Additionally, I assessed the road density in a minimum convex polygon using sage-grouse locations from 2005 to 2007 in North Dakota to estimate road miles per square km of sage-grouse habitat as an index of fragmentation (ESRI, Inc. ArcGIS 9.1, Redlands, CA.).

Data Analyses

Distance. - The distance from each random site to the nearest lek, and from each nest to nearest lek and the distance from each nest to the lek nearest to where the hen was captured (if the hen was captured that year) were calculated, along with distance from nest to nearest lek between successful and unsuccessful nests. To examine nest-site fidelity to specific nesting areas, I compared distances between consecutive-years' nests, and between those that were successful or unsuccessful in 2005 to their distances moved in 2006. I tested the hypotheses that there was no difference in distribution between the distances from random sites to nearest lek, nests to nearest lek and lek of capture, or between successful and unsuccessful nests using multiple response permutation programs (MRPP; Mielke and Berry 2001). Statistical significance was determined at alpha $\leq$ 0.05 for these univariate tests.

Habitat Selection. - Canopy cover values were recoded to mid-point values of the categories and I summarized data to an average value for each variable for the site. In addition, I summarized visual obstruction (VOR) values from the nest and 1-m to 5-m intervals; I also calculated the average VOR for the site. Estimates of sagebrush density were made from maximum likelihood estimates (Pollard 1971). I then used MRPP (Mielke and Berry 2001) to test the distributions of vegetation variables between nests and random sites, and used this as a screening process to distinguish important variables for future analysis with a critical value of alpha $\leq$ 0.05. The variables evaluated included percent total vegetative cover, percent grass cover, percent forb cover, percent sagebrush cover, percent bareground, percent litter, sagebrush height, average sagebrush width, site-

VOR and 1-m intervals to 5-m, grass height as measured on the Robel pole (max-VOR), and sagebrush density (Appendix C). I also tested these variables for similarities in distributions at successful and unsuccessful nests, nests of adults and yearlings, and between years using MRPP as initial screening of variables to be included in other analyses at the critical value for alpha $\leq$ 0.05.

I used information theoretic approach (Burnham and Anderson 2002) with logistic regression to estimate variables selected for by female sage-grouse at nest-sites using SAS JMP (2005 SAS Institute Inc). I developed 10 *a priori* models including variables from the previous MRPP test to predict nest-sites. The candidate models included vegetative variables of percent vegetative cover, percent grass cover, percent forb cover, percent sagebrush cover, sagebrush height, site-VOR, nest-VOR and 1-m VOR, grass height from the Robel pole, and sagebrush density. The variable year was considered as a design variable and was included in all candidate models. Thus, any difference among the models in the candidate sets were due to differences in the vegetative variables. For ease of interpretation, I did not include year in the tables. I tested the strength of the model to predict nest-sites using receiver operating characteristic curve (ROC) used as model fit or discrimination diagnostics (SAS JMP). Receiver operation characteristic values between 0.7 and 0.8 were considered acceptable discrimination, and ROC values between 0.8 and 0.9 were considered excellent discrimination (Hosmer and Lemeshow 2000).

To prevent underfitting or overfitting, Akaike's Information Criterion (AIC) was used as the basis for model selection. Using the log-likelihood values and number of parameters (*k*) provided in the output file from the 10 models within Program JMP. The models were ranked using the equation: $AIC = -2(log\text{-}likelihood) + 2k$. The two components of AIC include; *-2(log-likelihood*), which measures discrepancy of the fit between the data and the model, and (*k*) is a penalty for the number of parameters included in the model to prevent overfitting the models. Unless the sample size is large with respect to the number of parameters estimated, the use of *AICc* is recommended; $AIC + 2K(K + 1)/n – K -1$. The models were ranked using Δ *AICc* (Burnham and Anderson 1992).

Nest Survival and Modeling.- I estimated daily survival rate (DSR) of nests using program MARK (White and Burnham 1999) for the 27 day incubation period. I standardized May 6 as day 1 and numbered all nest check dates sequentially thereafter. Estimates of nest survival between adult and yearling hens and nest survival rates between years were compared using Program MARK.

Factors Influencing Nest Survival.- Nest survival probabilities were estimated as a function of continuous and categorical habitat variables using nest survival analyses in program MARK (White and Burnham 1999). Continuous variables included percent vegetative cover, grass cover, forb cover, sagebrush cover, max-VOR, site-VOR, sagebrush height, and sagebrush density. Categorical variables included bird age, nest age, and year. Time-dependent variables included maximum and minimum daily temperatures and precipitation during the interval since the nest status was determined.

Continuous covariates were standardized as deviations from a mean of 0. Categorical and time-dependent covariates were coded with the actual values so they would not hamper numerical optimization of likelihood (Burnham and Anderson 2002). The nesting period was 54 days beginning on 6 May each year. Nest age (in days) was then coded relative to 6 May. Thus, the covariate nest age had values from -17 to 17 and was modeled as a continuous variable.

Variables combined with constant DSR were compared using Akaike's Information Criterion corrected for a small sample size (*AICc;* Akaike 1973, Burnham and Anderson 2002). Models within 2 units of the minimum *AICc* model were considered best approximating models to explain variation of nest survival (Burnham and Anderson 1998). Variable weights were calculated by adding AICc weights of all models that included variables of interest to assess relative importance of single variables (Burnham and Anderson 2002). I used beta estimates of continuous variables in each set of candidate models to determine direction of effect of that variable on DSR. Because the saturated model fits the data perfectly, there is no need for a goodness-of-fit test between models (Cooch and White 2006).

Hen Survival. - Hen survival was estimated using Kaplan-Meier product-limit method (Kaplan and Meier 1958) modified for staggered entry (Pollock et al. 1989) throughout the nesting and brood-rearing periods.

RESULTS

Capture and Marking Female Sage-Grouse. **-** Thirty-nine hens were captured and fitted with radio-collars during spring and summer 2005 and 2006 (21 during 2005, 18 during 2006); 36% (14/39) were adults. Twenty female sage-grouse were included in analyses of nests in 2005, and nine additional female sage-grouse were used during spring 2006 analyses, for a total of 29 hens to model nest survival.

Monitoring Radio-collared Hens. - Based on morning lek counts and capture data during 2005 and 2006, peak hen visitation to leks occurs between 5 April – 11 April (Figure 2). Sage-grouse began laying eggs on 9 April, 2005 and 11 April, 2006 based on the 27 day incubation and the assumption of 1 egg laid every 1.3 days (Patterson 1952). Average nest initiation date during 2005 and 2006 was 23 April (range 21 April to 25 April) (n = 36). Adults initiated nests approximately 5 days earlier than yearlings. There were 2 renests in 2005; average date of renests on 21 May (Table 1). Average clutch size was 8 eggs per nest (n = 36). Clutch size for 12 successful nests averaged 7.58 ± 0.63 eggs, and 20 unsuccessful nests averaged 8.1 ± 0.49. Clutch size was not significantly different between successful and unsuccessful nests (P = 0.699, MRPP), therefore, I pooled data for further analyses. There was no difference in clutch size between adults and yearlings (P = 0.858, MRPP). Overall probability that an egg present at hatching produced living young (Mayfield Egg Hatchability) was 0.34 (n = 258). Most eggs were predated, abandoned, or infertile (74%).

Nest Attempts .- All radio-marked hens initiated a nest in 2005. In 2006, 13 of 14 adults (93%), and 5 of 7 yearlings (71%) initiated nests. There was no difference in nest initiation rates between years (P = 0.105, MRPP), thus data were pooled. Overall nest initiation for adult hens (n = 20) averaged 95% and was not significantly different (P = 0.578, MRPP) from yearling hens (n = 16) which averaged 88%. Overall nest initiation was 92% for adults and yearlings with both years combined (Table 2). I found that 90% of nest failures of first nest attempts were depredated or abandoned after ≥ 1 week of incubation (Figure 3). Renesting rate during my study was 2 of 21 (10%) during both field seasons. Adults initiated nests approximately 5 days earlier than yearlings, however, there was no significant difference (P = 0.07, MRPP).

Distance Between Nests. - Average distance between individual nests in 2005 to subsequent nests in 2006 for 9 birds was 2.35 ± 0.10 km. Distance between unsuccessful nests in 2005 to subsequent nests in 2006 averaged 2.06 ± 0.99 km (n = 4), and distance between successful nests in 2005 to subsequent nests in 2006 averaged 2.58 ± 1.73 km (n = 5). Hens that were unsuccessful their first nesting season did not move farther from their nests in 2005 then hens that were successful their first year (P = 0.457, MRPP).

Average distance from nests to the lek where a hen was captured was 4.94 ± 4.06 km and average distance from nests to nearest lek was 2.66 ± 2.35 km. Unsuccessful nests averaged 2.75 ± 2.85 km (n = 13) and successful nests averaged 2.53 ± 1.52 km (n = 9) from the nearest lek. Sixty-eight percent of nests were ≤ 3 km from a lek (Figure 4). There was no difference in distribution of distances from nests to nearest leks between unsuccessful and successful nests (P = 0.457, MRPP) or between years (P = 0.449, MRPP), and no difference between age classes (P = 0.767, MRPP).

Nest Survival and Modeling. - Nest survival was 33% in 2005 (n = 14) and 30% in 2006 (n = 15). Overall nest survival was 31%, including 1 successful renest. Constant DSR was the most parsimonious model; thus data were pooled because there was no difference between years or age category of nest survival (Table 3).

Estimates of variables screened from MRPP between successful and unsuccessful nests indicated certain variables might have more explanatory power to model nest survival. Competitive variables incorporated in nest survival models included percent grass cover, percent forb cover, nest-VOR, sagebrush density, and sagebrush height (Table 4). The relationship of each variable in relation to nest survival is incorporated in Table 4. There was little evidence that a particular vegetation characteristic or combination of two characteristics influenced nest survival. Single variable models including percent grass cover and height of grass from Robel pole had about equal weight. Models including percent sagebrush cover, percent grass, percent forb, nest-VOR, and sagebrush density were a second group of models with less influence on nest survival than the previous variables. The model that incorporated daily precipitation was the best predictive model of nest survival (Table 5). After model averaging, DSR was best explained by the most parsimonious model of daily precipitation (Figure 5).

Nest-site Selection. - Most nests were beneath a shrub and 88% were located beneath sagebrush. One sage-grouse nest was beneath four-wing saltbush (*Atriplex canesens*), one nest was beneath eastern red cedar (*Juniperus virginiana*), two nests were in residual cover of sweet clover (*Melilotus officinalis*) from the previous year, and one nest was in wheat stubble (*Triticum* spp.).

The distribution of percent total cover, grass cover, forb cover, sagebrush cover, litter, nest-VOR, 1-m VOR, and sagebrush density differed ($P \leq 0.05$, MRPP) between nests and random sites. Averages for these variables were greater at nests than at random sites (Table 6). Distributions of percent forb cover, sagebrush cover, bareground, grass beneath the sagebrush differed ($P \leq 0.05$, MRPP) between years at both nest and random sites. In addition, all VOR measurements extending out from nests differed between years at random sites. Average values for all these variables were greater in 2005 than 2006. All logistic models included the design variable year (Table 7).

One model strongly was supported with selection of nests that included percent total cover, 1-m VOR, and sagebrush density (Table 8). Sage-grouse nests were positively associated with more percent total cover, 1-m VOR, and sagebrush density than were present at random sites. In the model, increasing VOR by 2.54 cm increased the probability of the site to be a nest by a multiplicative factor of 0.281 ± 0.275 (CI 95%). Increasing total vegetative cover by 10%, increased the probability of the site to be a nest by a multiplicative factor of 0.60 ± 0.52 (CI 95%), and increasing sagebrush density by 50 shrubs/hectare, increased the probability of the site to be a nest by a multiplicative factor of 4.3 ± 0.85 (CI 95%) (Table 9). Classification accuracy of the model was acceptable with an ROC value = 0.76.

Habitat fragmentation. - In North Dakota, I estimated 1.45 km of roads/km^2 in approximately 900 km^2 area of sage-grouse habitat.

Hen Survival.- Hen survival was evaluated throughout nesting and brood rearing periods from time of capture (March – April) through August. In 2005, hen survival was 84% (95%CI: 0.67 to 1.00, n = 20) (Figure 6). In 2006, hen survival was 60% (95%CI: 0.44 to 0.76, n = 39) (Figure 7).

DISCUSSION

Breeding Chronology and Nesting. - Peak hen attendance at leks by greater sage-grouse in southwestern North Dakota was later than in the Columbia Basin and Great Basin states (Bradbury et al. 1989, Schroeder 1997, Connelly et al. 2004), but similar to sage-grouse ranges on the western edge of the Great Plains (Jenni and Hartzler 1978, Aldridge and Brigham 2001, Hausleitner 2003). This effect may be mitigated by milder temperatures and different precipitation between my study area and that at the Great Basin or Columbia Basin. Precipitation in the latter areas occurs most during the fall, winter, and spring and by July most of the Great Basin and Columbia Basin have little green herbaceous vegetation remaining (Bailey 1980). In North Dakota, about 60% of the precipitation occurs between April and July (North Dakota Agricultural Weather Network, 2006). Even though peak hen attendance was later in North Dakota; initiation of incubation in my study was similar to those found in the Columbia and Great Basin states (Schroeder 1997) suggesting that peak attendance was not synchronize with nest initiation.

All of the radio-marked hens attempted to nest in 2005. In 2006, some hens did not nest. In 2006, I had a larger sample size which could account for a greater probability of some hens to not initiate a nest. Reported nest initiation rates vary by region. Averaged across 11 studies, nest initiation rates of hens was 80%. Competition for nest-sites by female sage-grouse in populations that are dense could cause some hens not to nest. However, where populations are low, competition for nest-sites is less likely, and most hens will initiate a nest (Aldridge and Brigham 2001). The observed nest initiation rate may also be influenced by the abundance and distribution of suitable habitat for all aspects of sage-grouse's life history.

Renesting by hens varies regionally from 6–87% (Hanf et al. 1994, Schroeder 1997, Aldridge and Brigham 2001). Renesting in wild turkeys has been related to habitat quality (Rumble and Hodorff 1993, Rumble et al. 2003). My low renesting rates may suggest low habitat quality. Female sage-grouse nested 3 times in Washington, with adults more likely to renest than yearlings (Sveum 1995, Schroeder 1997). Protein may be an important variable for renesting because it is a major nutrient found in eggs (Carey 1996) and could be a limiting factor for egg production in sage-grouse (Moss 1972, Thomas and Popko 1981, Thomas 1982). Protein resources necessary for reproduction originate from the diet (i.e., exogenous sources; Beckerton and Middleton 1982, Carey 1996). Daily intake of proteins during spring, age, date of first nesting attempt, and incubation stage of the lost nest could affect renesting abilities of sage-grouse (Seubert 1952, Gates 1962, Sopuck and Zwicket 1983, Bergerud 1988, and Grand and Flint 1996). Gregg et al. (2006) documented that renesting rates decreased when hens initiated nests later in the nesting season or lost nests later during incubation. Bergerud (1988) suggested that adult hens renest more frequently than yearlings because they tend to nest earlier in the season, and therefore have enough time to initiate a second nest and hatch a successful brood and raise a brood.

Habitat quality and quantity may also be responsible for low nest success by decreasing amount of protein available during early spring nesting seasons. The age of nest at termination could be a factor associated with the lack of renesting in southwestern North Dakota. Hen age did not appear to be an important variable for distinguishing between renesting and non-renesting sage-grouse because 40% of nest failures were by adults, and 50% were yearlings.

Adult hens tend to nest earlier than yearlings (Batterson and Morse 1948, Schlattterer 1960, Petersen 1980, Schroeder 1997), and I noted a similar pattern. Earlier nesting by adults is attributed to adults being more biologically ready to nest than yearlings (Schroeder 1997). I found adults initiated nests approximately 5 days earlier than yearlings.

Average clutch size in my study was consistent with other studies (Wallestad and Pyrah 1974, Sveum 1995). Despite predictions of age-specific differences in clutch size (Wallestad and Pyrah 1974, Petersen 1980), clutch size of adult and yearling nests was similar during my study.

Nest Survival. - Connelly et al. (2004) reported that nest success of female sage-grouse across their range varied from 14-86%. Average nest success across the range was 47.7% (Trueblood 1954, Gregg 1991). Nest success during my study was lower than previous studies; most nests were lost to predation or weather. Nest predation can be higher in fragmented landscapes (Andre´n et al. 1985, Andre and Angelstam 1988, and Kurki et al. 1997). My study area had 1.45 km of roads/km^2 resulting in fragmentation. In the Powder River Basin there have been large-scale modifications of sagebrush habitat associated with oil and gas development that could have important impacts on habitat use or survival rates of sagebrush obligate species (Walker, unpublished data 26th Meeting of the Western Agencies Sage and Columbian Sharp-tailed Grouse Technical Symposium, abstract). Since I was not able to demonstrate that vegetation characteristics surrounding nests influenced nest survival at the scale at which I measured vegetation, predation and other factors that caused nest loss may be random events. Alternatively, the protective quality of habitat could be homogeneously poor (see habitat selection). Due to the limited distribution of dense sagebrush, sage-grouse could be constrained to remaining sagebrush habitats for nest-sites. There was support for models of nest survival that included precipitation (random) and vegetation (nonrandom).

Within fragmented ecosystems, it has been hypothesized that increased levels of moisture during incubation increased nest depredation in wild turkeys (*Meleagris gallopavo*; Roberts et al. 1995, Roberts and Porter 1998, Lehman 2005). My models indicated that precipitation was the best variable to explain nest success in 2005 and 2006. Predators with a keen sense of smell use olfactory cues to locate nests (Storaas 1988), which makes following scent easier during moist conditions supporting the hypothesis that hens are more vulnerable to predation during wet periods than under dryer conditions (Roberts and Porter 1998). Syrotuck (1972) hypothesized that water activated bacteria on the skin of turkeys, and allowed predators to locate incubating hens efficiently, which could be a similar hypothesis for female sage-grouse.

Habitat Selection. - Female sage-grouse in North Dakota selected nests in areas with more vegetative cover and higher sagebrush density than occurred elsewhere in the study area. Within these areas the actual nest-site that was selected was rather small as evidenced by inclusion of 1-m VOR and no other VOR intervals. Both nest and random sites were centered over sagebrush, so it is not surprising nest-VOR was not important. Several studies have established the importance of sagebrush canopy cover (Patterson 1952, Wallestad and Pyrah 1974, Wakkinen 1990, Fischer 1994, Sveum et al. 1998) and herbaceous canopy cover (Wakkinen 1990, Connelly et al. 1991, Sveum et al. 1998) to sage-grouse nesting habitat.

Visual obstruction surrounding more than the bush itself can provide additional concealment from predators. I found average total vegetative cover around nests > 66%. These variables contribute to successfully camouflaging the nest-site (Autenrieth 1981). I found hens selected for more dense sagebrush habitat than what was available. However, sage-grouse habitat includes a wide range of sagebrush density. Sagebrush density at nest-sites in my study was about ½ that reported in Nevada (Klebenow 1969) and 1/3 that reported for Montana (Wallestand and Pyrah 1974), while in south-central Idaho sagebrush density (Connelly 1991) was only slightly greater than in my study. Sagebrush density varies with local conditions and sagebrush species (Davies et al. 2006). Despite low sagebrush density and cover in my study, the amount of grass cover around nests suggests that grass is an important contributor to cover of sage-grouse nests.

Across their range, female sage-grouse usually select sagebrush patches for nests with shrub canopy cover of 15-25%, and avoid sparse or excessively dense patches (Connelly et al. 2000). However, in southwestern North Dakota, hens may have to select different nest-site characteristics to maintain adequate cover because of restricted patches of remaining sagebrush habitats, all of which are similar in habitat quality. This is a topic of high priority for future studies in North Dakota.

Previous studies have noted that hens select nest-sites with the tallest available bush, with the greatest diameter to initiate a nest (Gray 1967, Klebenow 1969, Wallestand and Pyrah 1974, Autenreith 1981). In my study, hens did not select for taller bushes at nests. The lack of selection for tall sagebrush may reflect homogeneity among sagebrush plants in the area. Sage-grouse can inhabit areas of lower sagebrush height and density than reported in the literature if additional cover from grasses is available. Previous studies have also documented the importance of cover from grasses within shrub stands (Wakkinen 1990, Connelly et al. 1991) which is associated to higher nest success rates (Gregg et al. 1994), and can offer additional nest protection. Hens selected nest-sites with grass cover consisting of half the total cover around the nest-site. It is likely that graminoid cover provides alternative nest cover than sagebrush.

Distance.- Most grouse species display fidelity to their nesting areas. The distance between consecutive nests varies from 0.7 to 2.8 km (Fischer et al. 1993, Schroeder 1997). The fidelity of hens in my study was typical of other studies. In my study, successful hens nested farther from their previous nest than unsuccessful hens. However, Fisher et al. (1993) found that unsuccessful hens nested furthur between consecutive years than successful hens. Although fidelity to breeding areas may be advantageous for grouse (Bergerud and Gratson 1988), fidelity to nest-sites could decrease nest success following habitat alterations that make the areas less secure. Habitat fragmentation and habitat alteration throughout southwestern North Dakota from associated agriculture and oil and gas development alter the landscape for sage-grouse every year. Increased fragmentation and low connectivity of sagebrush habitats may explain why some hens are moving exceptionally large distances between nesting attempts in North Dakota.

Previous studies have documented that hens select nest-sites independent of proximity to leks. Nonetheless, most nests occur within 2.5-3.2 km of leks (Wallestad and Pyrah 1974, Bradbury et al. 1989, Wakkinen et al. 1992). During my study, average distance from nests to lek of capture was 4.94 ± 4.06 km. Sixty-eight percent of nests in my study were within 3.2 km of the nearest lek. Autenrieth (1981) suggested that lek to nest distances were inversely correlated to habitat quality. However, the limited distribution and patchiness of sagebrush in North Dakota restrict nesting which is mostly confined to sagebrush to occur near leks which are associated with sagebrush.

Hen Survival. -Survival of female sage-grouse is normally presented on an annual basis. Because there are numerous ways of evaluating survival (i.e., leg-bands, radio transmitters, brood observations), estimates of survival are hard to obtain that are comparable between studies. Previous estimates of annual survival range from 57-78% (Connelly et al. 1991, Aldridge and Brigham 2001, Wik 2002, and Hausleitner 2003), suggesting that mortality of hens during nesting and brood-rearing seasons was not a primary factor affecting the sage-grouse population in North Dakota.

MANAGEMENT IMPLICATIONS

Vegetative trends in sagebrush habitat found in North Dakota are similar to the rest of the sagebrush range. I recommend that managers develop strategies to preserve the integrity of shrubsteppe habitat in southwestern North Dakota. Herbaceous cover in sagebrush habitats is an important component of nesting habitat for female sage-grouse. Thus, I recommend management activities that maintain or restore dense, taller residual grass within nesting habitat. There is little direct evidence associating livestock grazing practices to sage-grouse population levels. However, my results suggest excessive annual grazing within suitable nesting habitat could have a negative impact on the following year's nesting success by reducing residual grass cover, thereby reducing the quality of habitat for nesting birds. Factors such as timing, density, and spatial distribution of grazing should be reevaluated to maximize the protective cover value of the sagebrush. Ensuring proper grazing management on federal and state lands and encouraging participation from local land owners to participate in similar grazing practices with considerations for sage-grouse will help maintain adequate herbaceous understory throughout the nesting season.

I suggest expanding the current 3.2 km rule of the 1988 Resource Management Plan guideline initiated by the BLM in relation to habitat quality around known leks to a 5 km buffer, and encourage strict enforcement of these guidelines. This increased distance under special management would include 86% of nests versus 68% with the current 3.2 km buffer. There currently are no management regulations pertaining to sage-grouse on state owned land in North Dakota. Consequently, activities associated with oil and gas development can occur year round anywhere. I recommend that states implement the same regulation as the BLM and apply it to a 5 km buffer around leks.

The relatively random distribution of nests in relation to leks indicates that habitat management should focus on providing suitable sagebrush habitats wherever possible regardless of their distance to active leks. Efforts should focus on constructing a habitat suitability index to aid in assessing habitat quality of sage-grouse throughout North Dakota. Additionally, future research should identify movement corridors, and assess distribution and quality of sagebrush habitats throughout North Dakota.

Table 1. Mean clutch size and mean initiation dates for first nests and renests of greater sage-grouse in southwestern North Dakota, USA, 2005–2006.

	1st Nest				**Renest**			
Year	**Initiation Date**	**Clutch Size**	**Clutch size range**	***N***	**Initiation Date**	**Clutch Size**	**Clutch size range**	***N***
2005	25 April (9 April – 16 May)	8.1	7-12	17	21 May (8 May- 3 June)	6	4-8	2
2006	21 April (11 April – 14 May)	7.8	1-10	19	-	-	-	-
Avg/ Range	23 April	8.0	1-12	36	-	-	-	-

Table 2. Nest initiation rate of radio-marked adult and yearling sage-grouse in southwestern North Dakota, USA, 2005-2006.

Year	**Adults**	***N***	**Yearlings**	***N***	**Total**
2005	100%	6	100%	9	100% (15 of 15)
2006	93%	14	71%	7	86% (18 of 21)
Total	95%	20	88%	16	92% (33 of 36)

Table 3. Summary of model selection results for nest survival between year and age of greater sage-grouse in southwestern North Dakota, USA, 2005-2006.

Model	**AICc** [a]	**Δ AICc** [b]	**AICc Weight** [c]	***K*** [d]
(.)	112.572	0.00	0.5422	1
(Year)	114.565	1.99	0.20028	2
(Age)	114.590	2.02	0.19770	2
(Year * Age)	116.978	4.41	0.05989	4

[a] Akaike's Information Criterion adjusted for small sample size (*AICc*)
[b] Difference in AICc *(*Δ *AICc*)
[c] Akaike weights (*wi*)
[d] Number of parameters (*K*).

Table 4. Summary of model selection results for nest survival between habitat variables of greater sage-grouse in southwestern North Dakota, USA, 2005-2006.

Model	**AICc** [a]	**Δ AICc** [b]	**AICc Weight** [c]	***K*** [d]
(.)	112.572	0.00	0.1535	1
TOGR (-)	112.651	0.08	0.1467	2
Grass Hgt within shrubs (-)	113.181	0.61	0.1125	2
TOFO (+)	113.545	0.97	0.0938	2
TOSH (+)	113.618	1.05	0.0903	2
TOFO (+) + TOGR (-)	113.725	1.15	0.0857	3
Nest VOR (-)	113.782	1.21	0.0833	2
Shrub Density (+)	113.912	1.34	0.0780	2
Shrub Hgt (-)	114.706	2.13	0.0739	3
Grass Hgt within shrub (-) + Shrub Hgt (-)	114.706	2.13	0.0525	3
TOCO (+) + Nest VOR (-)	115.766	3.19	0.0308	3
Nest Age (+)	162.733	50.16	0.0000	1

[a] Akaike's Information Criterion adjusted for small sample size (AICc)
[b] Difference in AICc (Δ AICc)
[c] Akaike weights (wi)
[d] Number of parameters (*K*).

Table 5. Summary of model selection results for nest survival between time-dependent variables of greater sage-grouse in southwestern North Dakota, USA, 2005-2006

Model	**AICc** [a]	**Δ AICc** [b]	**AICc Weight** [c]	***K*** [d]	***Log-likelihood***
Daily Precip	107.467	0.00	0.3374	2	103.436
Daily Precip + TOGR	107.608	0.14	0.3145	3	101.545
Daily Precip + Grass height with Robel pole	109.051	1.58	0.1528	3	102.988
Year + Daily Precip	109.480	2.01	0.1233	3	103.417
(.)	112.572	5.10	0.0263	1	110.562
Max Temp	112.947	5.48	0.0218	2	108.915
Year + Max Temp	113.896	6.43	0.0136	3	107.833
Min Temp	114.392	6.92	0.0106	2	110.360

[a] Akaike's Information Criterion adjusted for small sample size (AICc)
[b] Difference in AICc (Δ AICc)
[c] Akaike weights (wi)
[d] Number of parameters (*K*).

Table 6. Combined average distributions of vegetation characteristics for nest-sites and random sites of sage-grouse in southwestern North Dakota using MRPP, 2005-2006.

Variable	**Nest $\bar{x}$ (*n* = 34)**	**Random $\bar{x}$ (*n* = 50)**	**p-value**
Total cover (%)	70	54	< 0.001
Total grass (%)	27	19	0.0111
Total forb (%)	15	11	< 0.001
Total sagebrush (%)	10	7	0.003
Bareground (%)	21	33	0.0058
Litter (%)	13	8	< 0.001
Sagebrush density/hectare	2,576.1	1,399.4	< 0.001
Nest-VOR	9.3	7	0.0019

Table 7. Average vegetation characteristics of nest-site and random sites between years for sage-grouse in southwestern North Dakota using MRPP, during 2005-2006.

Variable	**Nest 2005 (*n* = 17)**	**Nest 2006 (*n* = 17)**	**p-value**	**Random 2005 (*n* = 17)**	**Random 2006 (*n* = 33)**	**p-value**
Total Forb (%)	23	8	< 0.001*	16	8	< 0.001*
Total Sage (%)	11	8	0.0242*	9	6	0.0238*
Bareground (%)	27	16	0.0269*			
Grass hgt. in shrub (cm)	35.1	29.9	0.0185*	41.5	32.2	0.0041*
Avg. width of shrubs (cm)	41.5	53	0.0061*	48.5	31.8	< 0.001*
Nest VOR (in)	9.7	8.9	0.6525	23.6	6.7	< 0.001*
VOR 1m	4.1	3.7	0.7094	9.9	2.4	< 0.001*
VOR 2m	3.4	2.5	0.3131	7.8	2.2	< 0.001*
VOR 3m	2.6	2.4	0.2705	6.6	2.1	< 0.001*
VOR 4m	2.2	2.6	0.6016	7.1	2.1	< 0.001*
VOR 5m	2.3	2.1	0.9263	7.3	2.2	< 0.001*
VOR 10m	2.2	2.2	0.8988	8.4	1.8	< 0.001*
VOR 20m	1.6	2.2	0.1289	6.8	1.4	< 0.001*
VOR 30m	2.2	2.2	0.7868	7.3	1.5	< 0.001*
VOR 40m	2.1	2.2	0.6366	6.6	1.5	< 0.001*
VOR 50m				5	1.1	< 0.001*

Asterisks (*) indicates significant difference between nests of 2005 and 2006, and significant differences between random sites compared between 2005 and 2006.

Table 8. Logistic regression models predicting greater sage-grouse nest-sites (n = 34) versus random sites (n = 50) using vegetal data collected in North Dakota, USA, 2005-2006. Log-likelihood (-2 ln [L]), number of parameters including year indicator variable plus 2 (intercept + SE) (K), Akaike's Information Criterion adjusted for small sample size (AICc), difference in AICc (ΔAICc), and Akaike weights (wi). Models with ΔAICc < 2 are highlighted.

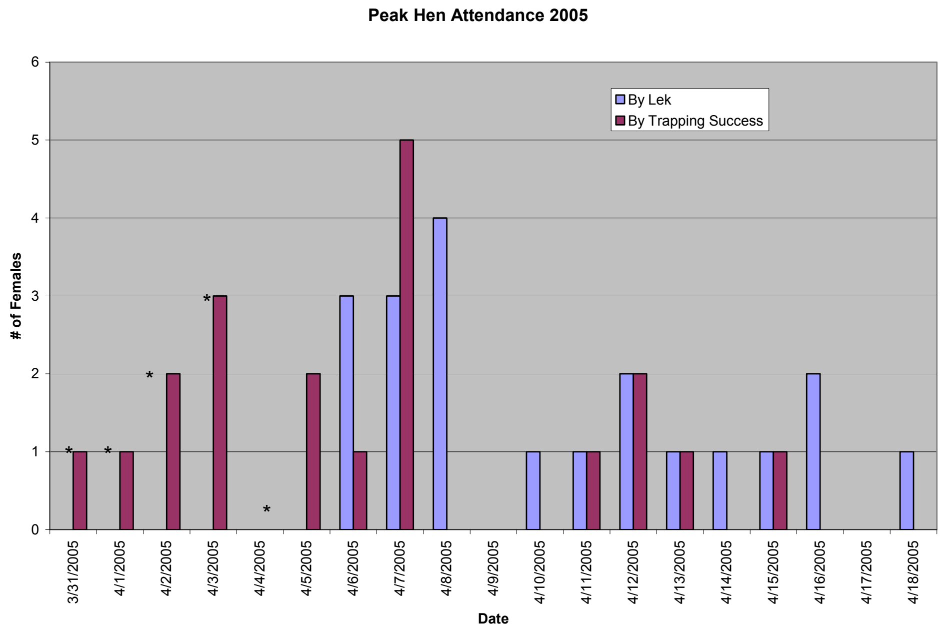
Model	**Log-likelihood**	**K**	**AICc**	**Δ AICc**	**Wi**
TOCO + VOR 1-M + SHRUB DEN	**-45.937434**	**6**	**91.8749**	**0**	**0.873**
TOCO + GRASS HGT + SHRUB HGT + SHRUB DEN	-40.570508	7	96.78808	4.913	0.075
TOCO + GRASS HGT + SHRUB HGT + VOR 0-M + VOR 1-M	-45.857659	8	97.68791	5.813	0.048
TOCO + COVER + SHRUB DEN	-46.019476	6	103.8773	12.002	0.002
TOFO + TOGR + TOSH + SHRUB DEN + VOR 0-M + VOR 1-M	-46.353773	9	105.6413	13.766	0.001
TOCO + HEIGHT + VOR 0-M + VOR 1-M	-46.863904	7	106.4435	14.569	0.001
TOCO + SHRUB DEN + VOR 1-M + VOR 0-M	-48.530504	7	107.2043	15.330	<0.001
GRASS HGT + SHRUB HGT + VOR 0-M + VOR 1-M	-49.759395	7	107.3624	15.487	<0.001
TOFO + TOGR + TOSH	-51.746931	6	110.2784	18.403	<0.001
GRASS HGT + SHRUB HGT + SHRUB DEN	-45.778607	6	112.7362	20.861	<0.001

[a] I included the following habitat variables in my models: total canopy coverage (TOCO), percent forb cover (TOFO), percent grass cover (TOGR), percent sagebrush cover (TOSH), sagebrush height (SHRUB HGT), site-VOR (COVER), 0m-VOR (VOR 0m), and 1-m-VOR (VOR 1m), sagebrush density/hectare (SHRUB DEN), and max grass height surrounding the Robel pole (HEIGHT).

[b] To facilitate interpretation, I excluded year indicator variable from model column.

Table 9. Odds ratio and confidence intervals associated with independent variables that best explain nest-sites in southwestern North Dakota, USA, 2005-2006.

Variable	**Odds Ratio**	**Odds Lower CI**	**Odds Upper CI**
TOCO	0.060	0.006	0.502
Sagebrush density	0.086	0.008	0.732
1-m VOR	0.280	0.017	4.013



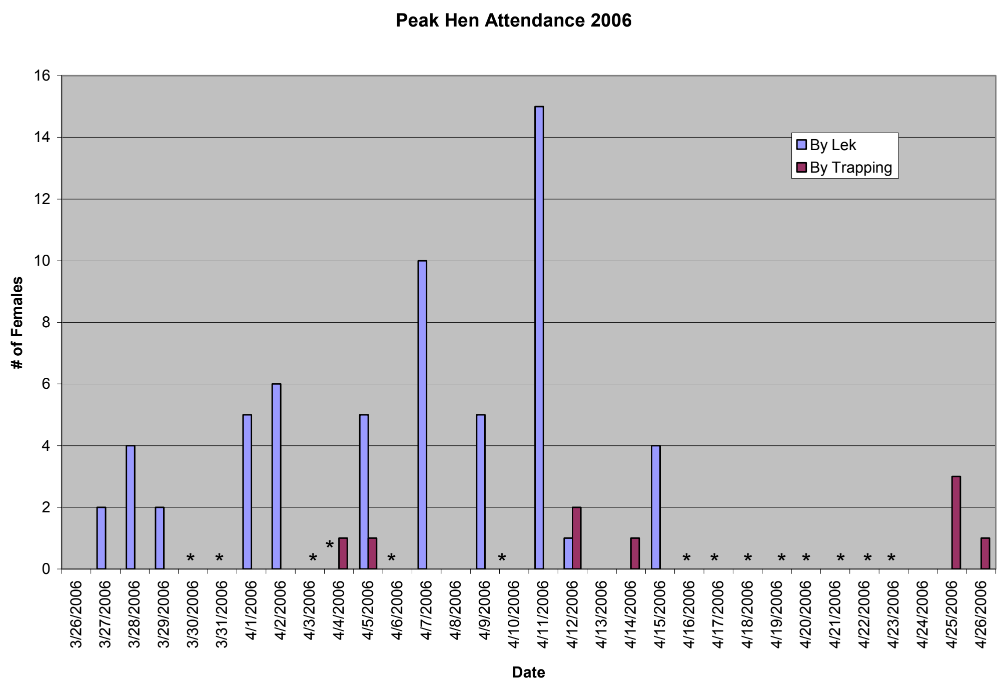


Figure 2. Number of female sage-grouse counted from morning lek counts or trapping success in 2005 and 2006 in relation to date. Asterisks (*) indicates dates lek counts were not conducted.

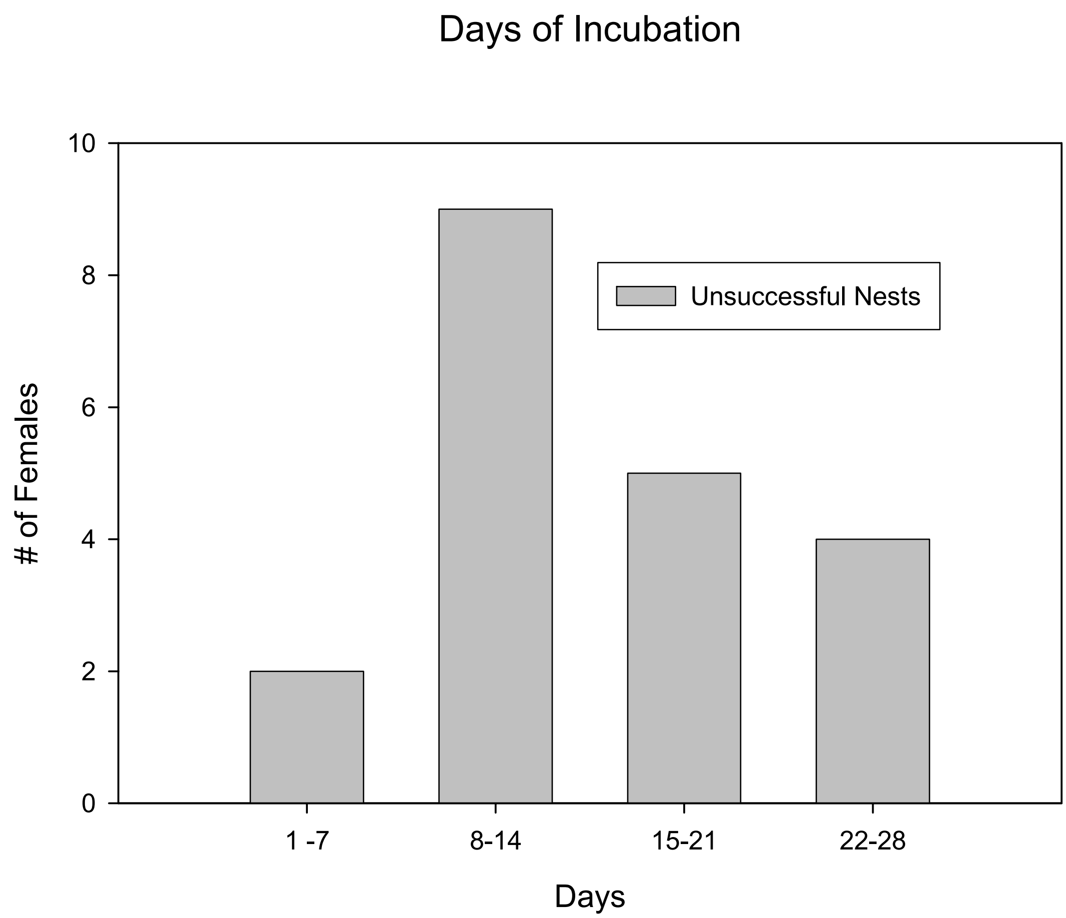


Figure 3. Nest loss period during 4-week incubation for first nesting attempts of greater sage-grouse in southwestern North Dakota, USA, 2005-2006.

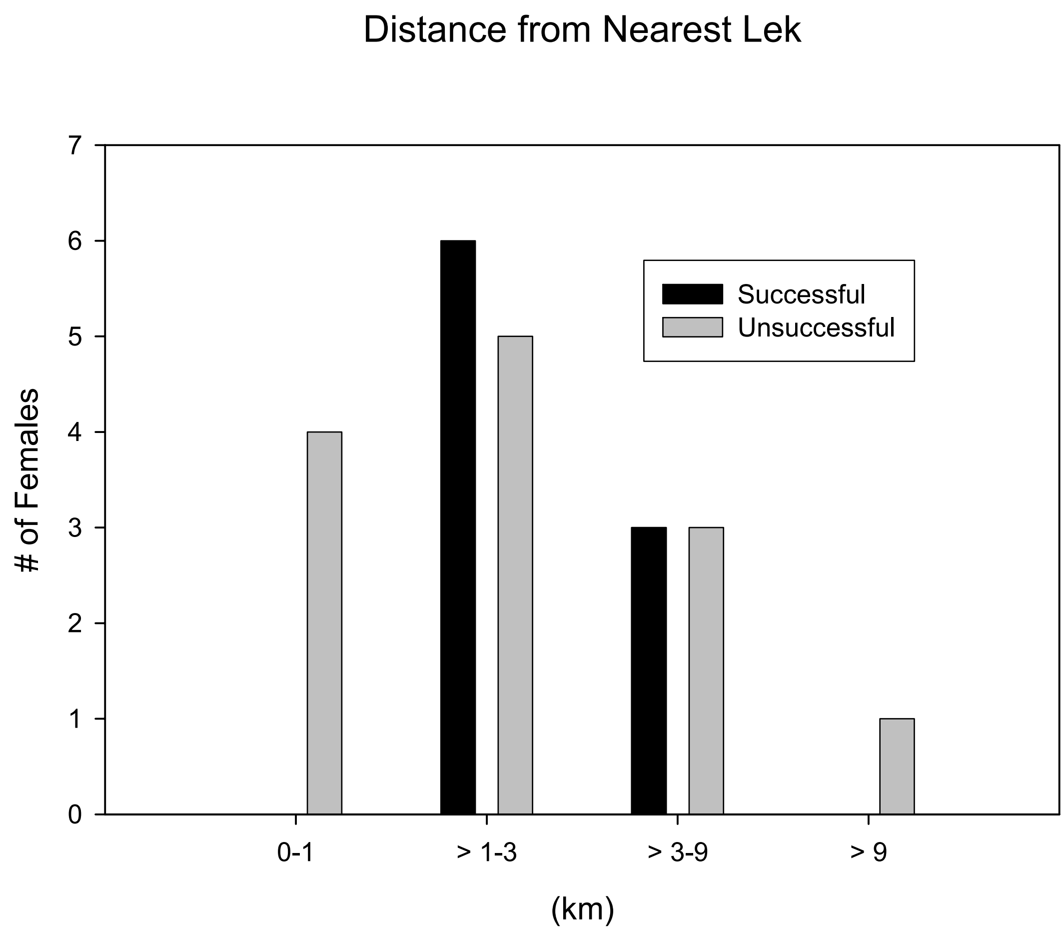


Figure 4. Distribution of distances between 22 pairs of nests to nearest lek distance for greater sage-grouse in southwestern North Dakota, USA, 2005-2006.

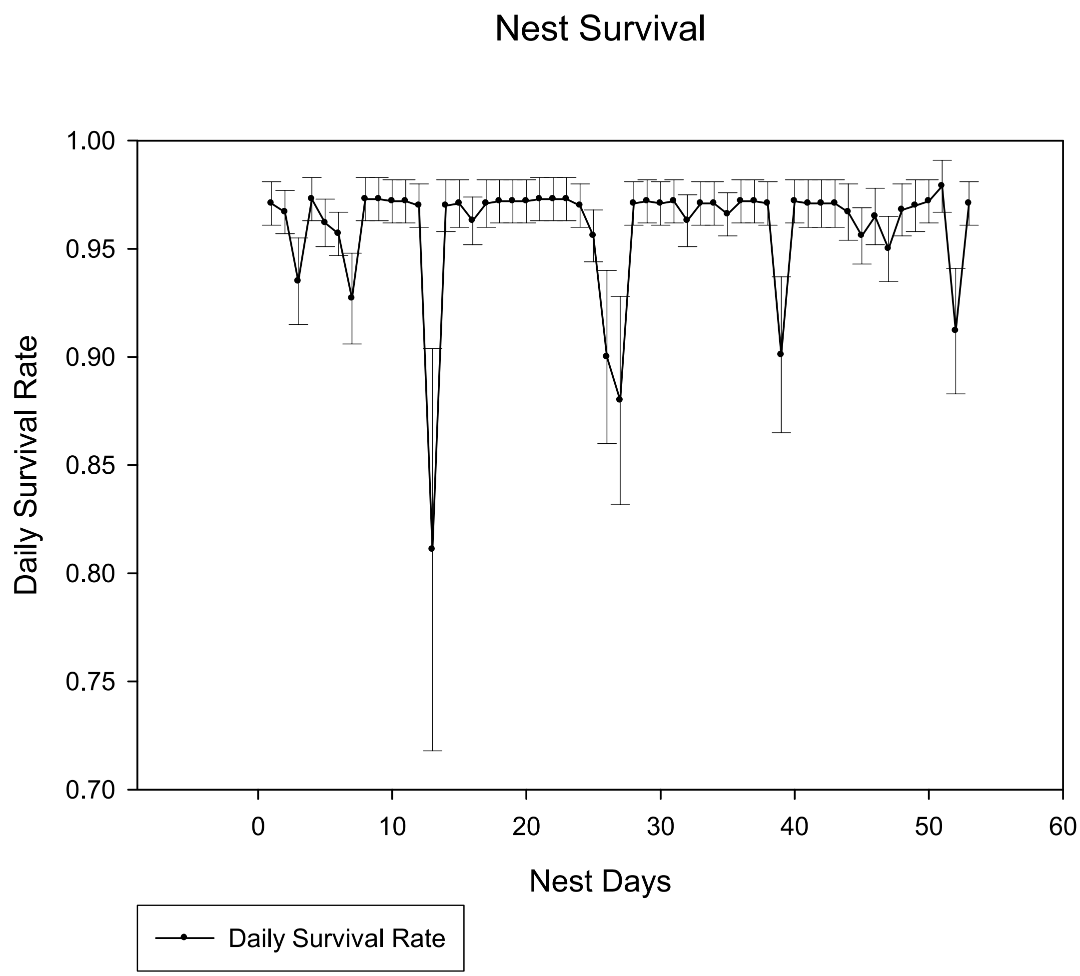


Figure 5. Daily survival rate from model averaging of models < 2 Δ AICc from the most parsimonious model over the 54 day nesting period used in Program MARK to model nest survival in southwestern North Dakota, 2005-2006. The spikes implicate a rain event, with DSR including average values from percent total grass and grass height from the Robel pole.

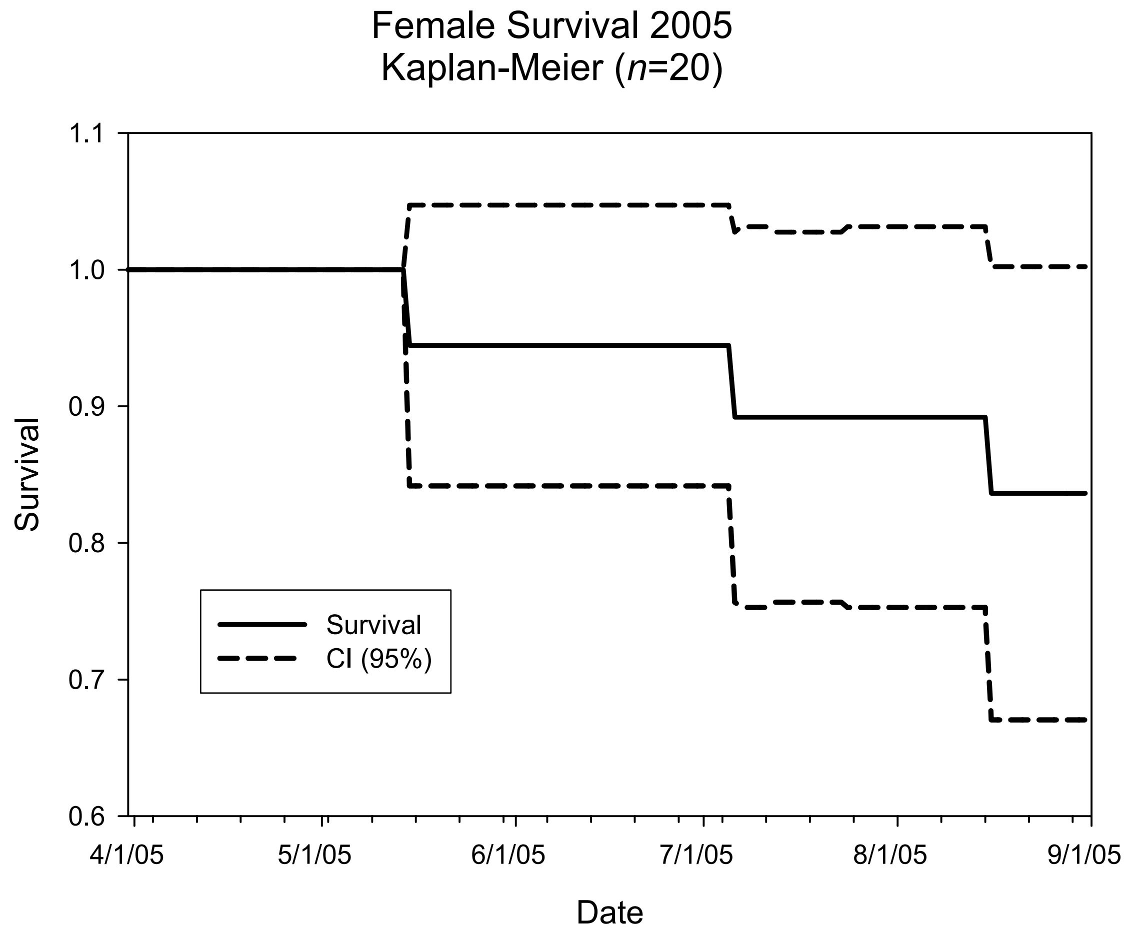


Figure 6. Greater sage-grouse hen survival rate and 95% confidence intervals (dashed lines) during the nesting and brood-rearing season during 2005 in southwestern North Dakota, USA (Kaplan and Meier 1958, Pollock et al. 1989).

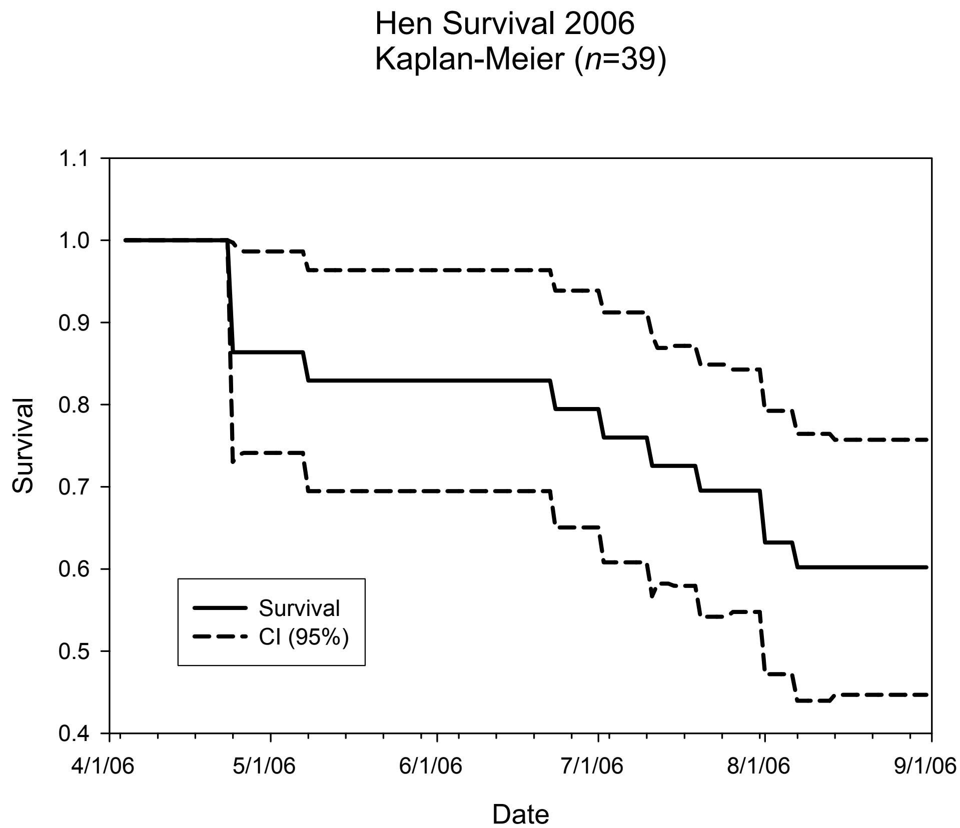


Figure 7. Greater sage-grouse hen survival rate and 95% confidence intervals (dashed lines) during the nesting and brood rearing season from 2006 in southwestern North Dakota, USA (Kaplan and Meier 1958, Pollock et al. 1989).

LITERATURE CITED

Akaike, H. 1973. Information theory as an extension of the maximum likelihood principle. Pages 267-281 in B.N. Petrov, and F. Csaki, editors. Second International Symposium on Information Theory. Akademiai Kiado, Budapest.

Aldridge, C.L., and R.M. Brigham. 2001. Nesting and reproductive activities of Greater Sage-Grouse in a declining northern fringe population. Condor 103:537-543.

Andren, H., P. Angelstam, E. Lindstrom, and P. Widen. 1985. Differences in predation pressure in relation to habitat fragmentation –an experiment. Oikos 45:273-277.

Andren, H, and P. Angelstam. 1988. Elevated predation rates as an edge effect in habitat islands: experimental evidence. Ecology 69:544-547.

Andrén, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos* **7**1: 355–366.

Autenrieth, R.E. 1981. Sage Grouse management in Idaho. ID Dept. Fish Game, Boise. Wildlife Bulletin 9. 238pp.

Bailey, R.G. 1980. Description of the ecoregions of the United States. USDA Forest Service. Misc. Publ. 1391. 77p.

Barnett, J.K., and J.A. Crawford. 1994. Pre-laying nutrition of sage grouse hens in Oregon. Journal of Range Management 47:114-118.

Batterson, W.M., and W.B. Morse. 1948. Oregon State Comm. Fauna Ser. 1. 29pp.

Beck, J.L., D.L. Mitchell, and B.D. Maxfield. 2003. Changes in the distribution and status of sage grouse in Utah. Western North American Naturalist 63:203-214.

Beckerton, P.R., and A.L.A. Middleton. 1982. Effects of dietary protein levels on ruffed grouse reproduction. Journal of Wildlife Management 46:569-579.

Benkobi, L., D.W. Uresk, G. Schenbeck, and R.M. King. 2000. Protocol for monitoring standing crop in grasslands using visual obstruction. Journal of Range Management 53:627-633.

Bergerud, A.T. 1988. Population ecology of North American grouse. Pages 578-648 in A.T. Bergerud, and M.W. Gratson, editors. Adaptive strategies and population ecology of northern grouse. University of Minnesota Press, Minneapolis, USA.

Bergerud, A.T., and M.W. Gratson. 1988. Survival and breeding strategies of grouse. Pp. 473-577 in Adaptive strategies and population ecology of northern grouse (A.T. Bergerud and M.W. Gratson, eds.). Univ. of Minnesota Press, Minneapolis.

Bradbury, J.W., R.M. Gibson, and C.E. McCarthy. 1989. Dispersion of displaying Sage Grouse males: II. The role of female dispersion. Behavioral Ecol. Sociobiobl. 24:15-24.

Braun, C.E., T. Britt, and R.O. Wallestad. 1977. Guidelines for maintenance of Sage Grouse habitats. Wildlife Society Bulletin 5:99-106.

Braun, C.E. 1987. Current issues in sage grouse management. Proc. West. Assoc. Fish and Wildlife Agencies 67:134-144.

Braun, C.E. 1998. Sage grouse declines in western North America: what are the problems? Pages 139-156 *in* Proceedings of the Western Association of Fish and Wildlife Agencies 78th Annual Conference in Jackson, WY, USA.

Burnham, K.P., and D.R. Anderson. 1992. Data-based selection of an appropriate biological model: the key to modern data analysis. Pages 16-30 in D. McCullough, and R. Barrett, eds. Wildlife 2001. Populations. Elsevier Sci. Publ. Ltd., New York.

Burnham, K.P., and D.R. Anderson. 1998. Model selection and inference: a practical information-theoretic approach. Springer-Verlag, New York, New York, USA.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach, second edition. Springer-Verlag, New York, New York, USA.

Carey, C. 1996. Female reproductive energetics. Pages 324-374 in C. Carey, editor. Avian energetics and nutritional ecology. Chapman and Hall, New York, New York, USA.

Connelly, J.W., W.L. Wakkinen, A.D. Apa, and K.P. Reese. 1991. Sage grouse use of nest sites in southeastern Idaho. Journal of Wildlife Management 55:521-524.

Connelly, J.W., and C.E. Braun. 1997. Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. Wildlife Biology 3:229-234.

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:1-19.

Connelly, J.W. S.T. Knick, M.A. Schroeder, and S.J. Stiver. 2004. Conservation Assessment of Greater Sage-grouse and Sagebrush Habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming.

Cooch, Evan, and G. White. 2006. Program MARK, "A Gentle Introduction" 4th Edition.

Cottam, G., and J. T. Curtis. 1956. The use of distance measures in phytosociological sampling. Ecology 37(3): 451-460.

Crawford, J.A., R.A. Olson, N.E. West, J.C. Moseley, M.A. Schroeder, T.D. Whitson, R.F. Miller, M.A. Gregg, and C.S. Boyd. 2004. Ecology and management of Sage-Grouse habitat. Journal of Range Management 57:2-19.

Daubenmire, R.F. 1959. A canopy-coverage method of vegetation analysis. Northwest Sci. 33:224-227.

Davies, K.W., J.D. Bates, R.E. Miller. 2006. Vegetation characteristics across part of Wyoming big sagebrush alliance. Rangeland Ecology and Management 59:567-575.

DeLong, A.K., J.A. Crawford, and D.C. DeLong, Jr. 1995. Relationships between vegetational structure and predation of artificial Sage Grouse nests. Journal of Wildlife Management 59:88-92.

Fischer, R.A., A.D. Apa, W.L. Wakkinen, K.P. Reese, and J.W. Connelly. 1993. Nesting-area fidelity of sage grouse in southeastern Idaho. The Condor 95:1038-1041.

Fischer, R.A. 1994. The effects of prescribed fire on the ecology of migratory Sage Grouse in southwestern Idaho (*Centrocercus urophasianus*). PhD Dissertation. Univ. ID.

Gates, J.M. 1962. Breeding biology of the gadwall in northern Utah. Wilson Bulletin 74:43-67.

Giesen, K.M., T.J. Schoenberg, and C.E. Braun. 1982. Methods for trapping Sage Grouse in Colorado. Wildlife Society Bulletin 10:224-231.

Gill, R.B. 1965. Distribution and abundance of a population of Sage Grouse in North Park, Colorado. M.S. Thesis. CO State Univ., Ft. Collins. 187pp.

Girard, G.L. 1937. Life history, habits, and food of the sage grouse, Centrocercus urophasians Bonaparte. University of Wyoming. Publ. 3:1-56.

Grand, J.B., and P.F. Flint. 1996. Renesting ecology of northern pintails on the Yukon-Kushkokwin delta, Alaska. Condor 98:820-824.

Gray, G.M. 1967. An ecological study of Sage Grouse broods with reference to nesting, movements, food habits, and sagebrush strip-spraying in the Medicine Lodge Drainage, Clark County, Idaho. M.S. Thesis. Univ. ID, Moscow. 200pp.

Gregg, M.A. 1991. Use and selection of nesting habitat by Sage Grouse in Oregon. M.S. Thesis. OR State Univ., Corvallis.

Gregg, M.A., J.A. Crawford, M.S. Drut, and A.K. Delong. 1994. Vegetational cover and predation of sage grouse nests in Oregon. Journal of Wildlife Management 58:162-166.

Gregg, M.A., M.R. Dunbar, J.A. Crawford, and M.D. Pope. 2006. Total plasma protein and renesting by Greater Sage-Grouse. Journal of Wildlife Management 70:472-478.

Hanf, J.M., P.A. Schmidt, and E.B. Groshens. 1994. Sage Grouse in the high desert of central Oregon: Results of a study, 1988-1993. BLM. 57 pp.

Hays, H., M. Lecroy. 1971. Field criteria for determining incubation stage in eggs of the common tern. Wilson Bull 83:425-429.

Hausleitner, D. 2003. Population dynamics, habitat use, and movements of Greater Sage-Grouse in Moffat County, Colorado. MS Thesis. Univ. Idaho.

Hosmer, D.W., S. Lemeshow. 2000. Applied Logistic Regression, 2nd Edition. John Wiley & Sons Incorporated Publication, New York, New York, USA.

Jenni, D.A., and J.E. Hartzler. 1978. Attendance at a Sage Grouse lek: implications for spring census. Journal of Wildlife Management 42:46-52.

Kaplan, E., and P. Meier. 1958. Non parametric estimation from incomplete observation. Journal of the American Statistical Association 53:457-481.

Klebenow, D.A. 1969. Sage grouse nesting and brood habitat in Idaho. Journal of Wildlife Management 33:649-662.

Kurki, S., A. Nikula, P. Helle, and H. Linden. 1997. Landscape-dependent breeding success of forest grouse in Fennoscandia. Wildlife Biology 3:295.

Lehman, C.P. 2005. Ecology of Merriam's turkeys in the southern Black Hills, South Dakota. Ph.D. Dissertation, South Dakota State University, Brookings, SD.

McCarthy, J.J., and J.D. Kobriger. 2005. Management plan and conservation strategies for greater sage-grouse in North Dakota. North Dakota Game and Fish Department. 96pp.

Mielke, P.W., Jr., and K.J. Berry. 2001. Permutation Methods: A Distance Function Approach. New York: Springer-Verlag. 352 p.

Moss, R. 1972. Food selection by red grouse (Lagopus lagopus scoticus (Lath.)) in relation to chemical composition. Journal of Animal Ecology 41:411-428.

Musil, D.D., K.P. Reese, and J.W. Connelly. 1994. Nesting and summer habitat use by translocated sage grouse (*Centrocercus urophasianus*) in central Idaho. Great Basin Nat. 54:228-233.

North Dakota Agricultural Weather Network. 2006. Climatology Information, Bowman ND. http://ndawn.ndsu.nodak.edu. (Accessed Sept 5, 2006).

Patterson, R.L. 1952. The sage grouse of Wyoming. Sage Books, Inc. Denver, Colo. 341pp.

Petersen, B.E. 1980. Breeding and nesting ecology of female Sage Grouse in North Park, Colorado. M.S. thesis. Colorado State University, Fort Collins, Colorado.

Pollock, K., S. Winterstein, C. Bunck, and P. Curtis. 1989. Survival analysis in telemetry studies: the staggered entry design. Journal of Wildlife Management 53:7-15.

Pollard, J.H. 1971. On distance estimators of density in randomly distributed forests. Biometrics 27:991-1002.

Robel R.J, J.N. Briggs, A.D. Dayton, and L.C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23:295-297.

Roberts, S.D., J.M. Coffey, and W.F. Porter. 1995. Survival and reproduction of female wild turkeys in New York. Journal of Wildlife Management 59:437-447.

Roberts, S.D., and W.F. Porter. 1998. Relation between weather and survival of wild turkey nests. Journal of Wildlife Management 62:1492-1498.

Rumble, M.A. and R.A. Hodorff. 1993. Nesting ecology of Merriam's turkeys in the Black Hills, South Dakota. Journal of Wildlife Management 57:789-801.

Rumble, M.A., B.F. Wakeling, and L.D. Flake. 2003. Factors affecting survival and recruitment in female Merriam's turkeys. Intermountain Journal of Sciences 9:26-37.

Sargeant, A.B., M.A. Sovada, and R.J. Greenwood. 1998. Interpreting evidence of depredation of duck nests in the prairie pothole region. U.S. Geological Survey, northern Prairie Wildlife Research Center, Jamestown, ND and Ducks Unlimited, Inc., Memphis, TN. 72pp.

Sall, J., L. Creighton, and A. Lehman. 2005. SAS: JMP Start Statistics. Sas, Inc.

Schroeder, M.A. 1997. Unusually high reproductive effort by Sage Grouse in a fragmented habitat in north-central Washington. Condor 99:933-941.

Schroeder, M.A., J.R. Young, and C.E. Bruan. 1999. Sage grouse (*Centrocercus urophasianus*). Pages 1-28 *in* A. Poole and F. Gill, editors. The birds of North America, No. 425. The Birds of North America, Inc., Philadelphia, Pennsylvania, USA.

Schroeder, M.A., C.L. Aldridge, A.D. Apa, J.R. Bohne, C.E. Braun, S.D. Bunnell, J.W. Connelly, P.A. Diebert, S.C. Gardner, M.A. Hilliard, G.D. Kobriger, C.W. McCarthy. 2004. Distribution of Sage Grouse in North America. Condor 106:363-376.

Schlatterer, E.F. 1960. Productivity and movements of a population of Sage-Grouse in southeastern Idaho. M.S. Thesis. Univ. ID, Moscow. 87pp.

Seubert, J.L. 1952. Observations on the renesting behavior of the ring-necked pheasant. Transactions of the North American Wildlife Conference 17:305-327.

Sopuck, L.G., and F.C. Zwickel. 1983. Renesting in adult and yearling blue grouse. Canadian Journal of Zoology 61:289-291.

Storaas, T. 1988. A comparison of losses in artificial and naturally occurring capercailiie nests. Journal of Wildlife Management 52:123-126.

Sveum, C.M. 1995. Habitat selection by Sage Grouse hens during the breeding season in south-central eastern Washington. MS Thesis. OR State Univ.

Sveum, C.M. W.D. Edge, and J.A Crawford. 1998. Nesting habitat selection by sage grouse in southcentral Washington. Journal of Range Management 51:265-269.

Syrotuck, W.A. 1972. Scent and the Scenting Dog. Rome, NY: Aren.

Thomas, V.G., and R. Popko. 1981. Fat and protein reserves of wintering and prebreeding rock ptarmigan from south Hudson Bay. Canadian Journal of Zoology 59:1205-1211.

Thomas, V.G. 1982. Energetic reserves of Hudson Bay willow ptarmigan during winter and spring. Canadian Journal of Zoology 60:1618-1623.

Trueblood, R.W. 1954. The effect of grass reseeding in sagebrush lands on Sage Grouse populations. M.S. Thesis. UT State Univ., Logan. 77pp.

Wakkinen, W.L. 1990. Nest site characteristics and spring-summer movements of migratory Sage Grouse in southeastern Idaho. MS Thesis, Univ. Idaho.

Wakkinen, W.L., K.P. Reese, J.W. Connelly, and R.A. Fischer. 1992. An improved spotlighting technique for capturing Sage Grouse. Wildlife Society Bulletin 20:425-426.

Wallestad, R.O., and D.B. Pyrah. 1974. Movement and nesting of sage grouse hens in central Montana. Journal of Wildlife Management 38:630-633.

Welch, B.L. 2005. BigSagebrush: A sea fragmented into lakes, ponds, and puddles. General Technical Reprot RMRS-GTR-144. U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Wik, P.A. 2002. Ecology of Greater Sage-Grouse in south-central Owyhee County, Idaho. M.S. Thesis. Univ. Idaho, Moscow.

White, G.C., and K.P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study Supplement 46:120-138.

Appendix A. Nest ages of greater sage-grouse nests in southwestern, North Dakota, USA, as determined by egg flotation.

Nest age (d)	Mean age (d)	Description
1-4	2	Egg lying flat on bottom
5	5	Large end of egg beginning to float
5-9	7	Egg standing upright on bottom
10-13	12	Egg about to float (middle of water)
14-18	16	Egg floating, top barely breaking water surface
19-23	21	Egg floating high with top out of water surface
24-28	26	Egg floating with noticeable tilt

Note: Ranges for each incubation stage was adapted from Hays and LeCroy 1971, and compared with my own 36 nesting attempts.

Appendix B. Four-digit code, common name and scientific name of plant species identified at nests and random sites in southwestern North Dakota, USA, 2005-2006.

Variable	**Name**	**Scientific Name**
acmi	western yarrow	*Achillea millefolium*
aggl	false dandelion	*Hypochoeris radicata*
agst	redtop	*Agrostis gigantea*
alfa	alfalfa	*Medicago spp.*
arca	silver sage	*Artemesia cana*
arfi	fringed sagewort	*Artemisia frigida*
arlu	cudweed sagewort	*Artemesia ludoviciana*
artr	big sagebrush	*Artemesia tridentata wyomingensis*
atri	*Atriplex spp.*	*Atriplex spp.*
bear	beards tongue	*Penstemon spp.*
bell	bluebells	*Mertensia spp.*
bogr	blue grama	*Bouteloua gracilis*
brin	smooth brome	*Bromus inermis*
brja	japanese brome	*Bromus japonicus*
bkbr	buckbrush	*Symphoricarpus occidentalis*
blue	little bluestem	*Vulpia octoflora*
buda	buffalo grass	*Buchloe dactyloides*
cafi	threadleaf sedge	*Carex filifolia*
calo	prairie sandreed	*Calamovilfa longifolia*
carr	wild carrot	*Daucus carota*
cele	wild celery	*Apium graveolens*
chea	cheatgrass	*Bromus tectorum*
cone	purple coneflower	*Echinacea* Moench
crew	crested wheatgrass	*Agropyron cristatum*
curd	curly doc	*Rumex crispus*
disp	inland saltgrass	*Distichlis spicata*
dwrf	dwarf alyssum	*Alyssum cuneifolium*
ercs	eastern red cedar	*Juniperus virginiana*
fieb	field bindweed	*Convolvulus arvensis*
gayf	gayfeather	*Liatris spicata*
gold	goldenrod	*Solidago spp.*
gotb	goatsbeard	*Tragopogon dubius*
gpea	golden pea	*Thermopsis rhombifolia*
grra	grayragwart	*Senecio incanus*
gumb	gumbo lily	*Oenothera caespitosa*
gumw	curlycup gumweed	*Grindelia squarrosa*
gusa	broom snakeweed	*Gutierrezia sarothrae*
hoju	foxtail barley	*Hordeum jubatum*

Appendix B. Continued

hors	horseweed	*Conyza spp.*
hory	hairy fleabane	*Conyza bonariensis*
indw	indian wheat	*Plantago patagonica*
intw	intermediate wheatgrass	*Thinopyrum intermedium*
koma	junegrass	*Koeleria macrantha*
long	longleaf wormweed	*Artemisa longifolia*
must	mustard	*Cardaria spp.*
navi	green needle	*Nassella viridula*
nutv	nuttall's violet	*Viola nuttallii*
pars	wild parsley	*Musineon* spp.
pasm	western wheatgrass	*Pascopyrum smithii*
penn	pennycress	*Thlaspi arvense*
pepp	pepperweed	*Lepidium densiflorum*
phho	hood's phlox	*Phlox hoodii*
plan	slender plantain	*Plantago heterophylla*
prpr	prickly pear	*Opuntia spp.*
popr	kentucky bluegrass	*Poa pratensis*
povw	povertyweed	*Iva axillaris* Pursh
psut	pussytoes	*Antennaria spp.*
redg	red goosefoot	*Chenopodium rubrum*
redt	redtop	*Agrostis stolonifera*
ripg	prairie cordgrass	*Spartina pectinata*
rose	wild rose	*Rosa woodsii*
rubb	rubber rabittbrush	*Ericameria nauseosa*
sand	sandbergs bluegrass	*Poa secunda*
scgo	scarlet gaura	*Gaura coccinea*
scur	scurfpea	*Psoralea spp.*
side	sideoats grama	Bouteloua curtipendula
silv	silverbladder pod	*Lesquerella argyraea*
skel	skeletonplant	*Lygodesmia* spp.
spid	spiderwort	*Tradescantia ohiensis*
spco	scarlet globemallow	*Sphaeralcea coccinea*
stic	stickseed	*Hackelia* Opiz
stco	needle and thread	*Stipa comata*
sthy	angelita daisy	*Hymenoxys acaulis*
sunf	sunflower	*Eriophyllum spp.*
swee	sweetclover	*Melilotus spp.*
taof	dandelion	*Taraxacum officinale*
toad	bastard toadflax	*Commandra umbellate*
this	thistle	*Cirsium spp.*
txon	textile onion	*Allium spp.*
vetc	*Astragalus spp.*	*Astragalus spp.*

Appendix B. Continued

vuoc	six weeks-fescue	*Vulpia octoflora*
wewa	western wallflower	*Erysimum asperum*
wint	winter fat	*Krascheninnikovia* spp.
yuca	yucca	*Yucca glauca*

Appendix C. Definition of all acronyms used for vegetative sampling in southwestern North Dakota, USA, 2005-2006.

Acronym	**Definition**
TOCO	Percent total vegetative cover
TOFO	Percent total forb cover
TOSH	Percent total sagebrush cover
TOGR	Percent total grass cover
Litter	Percent total litter cover (ie. residual grasses, rocks, feces)
Bareground	Percent bareground
Effective Grass Hgt	Grass height beneath sagebrush from Robel pole (in)
Max Grass Hgt	Tallest reading of grass species surrounding Robel pole (in)
Sagebrush Hgt	Sagebrush height (cm)
Sagebrush Width	Sagebrush width (minor and major cord averaged) (cm)
Sagebrush density	Sagebrush density/hectare
Site-VOR	Visual obstruction reading for the site
1-m VOR	Average visual obstruction reading for 1-m around site
2-m VOR	Average visual obstruction reading for 2-m around site
3-m VOR	Average visual obstruction reading for 3-m around site
4-m VOR	Average visual obstruction reading for 4-m around site
5-m VOR	Average visual obstruction reading for 5-m around site
10-m VOR	Average visual obstruction reading for 10-m around site
20-m VOR	Average visual obstruction reading for 20-m around site
30-m VOR	Average visual obstruction reading for 30-m around site
40-m VOR	Average visual obstruction reading for 40-m around site
50-m VOR	Average visual obstruction reading for 50-m around site

CHAPTER 3- BROOD SURVIVAL AND HABITAT SELECTION OF GREATER SAGE-GROUSE AT THE EASTERN EDGE OF THEIR HISTORIC DISTRIBUTION

INTRODUCTION

Following the arrival of European settlers in the 1800s, greater sage-grouse (*Centrocercus urophasianus*) habitat has continuously been changing (Girard 1937, Patterson 1952). Greater sage-grouse experienced population declines from 45- 80% across their range by the 1950s (Braun 1998) and during the 10-year period from 1985-1995 sage-grouse populations declined 33% (Connelly and Braun 1997). Historically, sage-grouse range is limited to sagebrush (*Artemisia* spp.) vegetation types in at least 12 states and 3 Canadian provinces, but currently they are resident in only 11 states and 2 Canadian provinces (Schroeder et al. 2004). Sage-grouse are sagebrush obligates and degradation and loss of sagebrush resulted in population declines and constriction of the range (Wisdom et al. 2005, Welch 2005).

Discovery of oil and gas throughout the United States in the 1930s and 1940s impacted wildlife habitats in numerous ways. In Colorado, the initial impacts of oil and gas development caused sage-grouse populations to decline drastically from noise, habitat loss, infrastructure and human activities (Braun 1987). The long-term effects are unknown, but there is no evidence that populations of sage-grouse will recover to pre-disturbance populations, and the length of recovery time for these habitats is estimated to range from 2-30 years (Braun 1998). Grazing by domestic livestock, fire, construction, power lines, fences, roads, and drought also contributed to loss of sagebrush to sagebrush ecosystems (Braun 1998, Schroeder et al. 1999, Welch 2005). Habitat fragmentation is of great concern in southwestern North Dakota with an estimate of 1.45 km of roads/km^2 in approximately 900 km^2 area of sage-grouse habitat. These changes have affected brood-rearing habitats through habitat alteration and habitat loss (Connelly and Braun 1997, Beck et al. 2003, Crawford et al. 2004).

The sage-grouse population in North Dakota is contiguous with populations in Montana and South Dakota (McCarthy and Kobriger 2005). Annual rates of change suggest a long-term population decline of about 2.79% per year from 1965 to 2003 (McCarthy and Kobriger 2005). Current breeding populations are estimated to be 3 to 6 times lower than occurred in the late 1960s to early 1970s (Connelly et al. 2004). Sage-grouse are a Priority Level 1 Species of Special Concern in North Dakota, and it is recommended that immediate research and conservation actions be taken. Thus, population declines may be related to declining habitat quality which may result in decreased survival and productivity, however, the significance of these factors is unknown. The fragmentation of sagebrush habitats could render North Dakota unsuitable as brood-rearing habitat and could contribute to population declines by reducing nest success and overall population productivity.

Estimates of sage-grouse chick (0-10 weeks of age) or juvenile survival (10-40 weeks of age) is limited and is not based on standardized time periods, thereby making comparisons and drawing conclusions difficult (Beck et al. 2006). In Oregon, only 10% of sage-grouse chicks survived until their first season as of March 1st (Crawford et al.

2004). Typically, only 50-60% of sage-grouse chicks survive through autumn (Bergerud 1988).

Sage-grouse use a variety of habitats throughout the year including grasslands and mosaics of sagebrush or aspen (*Populus* spp.); Paige and Ritter 1999). Sage-grouse productivity depends on brood-rearing habitat quality and availability (Crawford et al. 1992). Food availability and structure of the stand are common characteristics associated with habitat selection of hens with broods (Klebenow 1969, Peterson 1970, Wallestad 1971, Autenrieth 1981). Limited food resources slow growth and survival of sage-grouse chicks (Johnson and Boye 1990). Dunn and Braun (1986) discovered that vegetative cover and the extent of habitat interspersion are important factors that influence summer habitat use of sage-grouse hens and broods.

Key features of sage-grouse brood-rearing habitat are influenced by shrub density, plant composition and vegetation height (Klott and Lindzey 1990). Early brood-rearing areas are relatively close to nest-sites (Connelly 1982, Gates 1983) for the first 2-3 weeks post hatch (Connelly et al. 1988). Hens with broods prefer areas of abundant herbaceous growth surrounding nest-sites (Wallestad 1971, Klebenow 1985). Areas used by hens with broods usually have shrub cover between 8-14% and shrubs tend to have shorter than average stature (Klebenow 1969, Martin 1970, Wallestad 1971).

Although brood-rearing habitat selection for early and late summer use of sage-grouse chicks and juveniles has been well documented throughout western North America, knowledge of habitat selection by juvenile sage-grouse at the eastern edge of their range distribution where sagebrush communities are different than in the core of sagebrush distribution has not been quantified. Thus, the objectives of my study were to determine brood-rearing habitat selection of sage-grouse, and estimate brood survival for a population of sage-grouse in southwestern North Dakota. I tested the null hypotheses that there were no differences between vegetation composition and structure found at brood and random points. Knowledge of brood habitat use and selection will provide baseline information to develop management recommendations for use by state and federal wildlife and habitat management agencies to improve habitats for sage-grouse.

METHODS

Data Collection

Capture and Marking of Chicks. - I captured birds at night on or near leks from 31 March – 23 April 2005 and 27 March – 27 April 2006. I used hand-held spotlights to locate birds and approached them while shining the spotlight to confuse them and then used long-handled nets to capture hens (Giesen et al. 1982). I recorded age, sex, weight, and placed leg-bands and 20-gram necklace type radio transmitters with mortality sensors on each bird (Advanced Telemetry Systems, Isanti, Minnesota). Each bird was released at the point of capture. The transmitters were less than 2% of the bird's body weight. Each year, I monitored nest completion to estimate initial brood size from egg shells of successful nests.

Monitoring Radio-collared chicks. -Radio-marked hens with broods were located ≥ 2 times/week with a hand-held or vehicle-mounted yagi antenna and portable receiver. To obtain accurate locations and to monitor number of chicks, I obtained visual observations without flushing either the hen or brood. Once per week the hen and chicks were flushed to obtain an accurate estimate of chick numbers. Each brood location coordinate was recorded in a hand-held GPS unit in Universal Transverse Mercator (UTM).

At 5-6 weeks of age, I captured chicks from each brood by night-spotlighting and a long-handled net. Locating broods was aided by radio-marked hens. Each captured chick was weighed, leg-banded (size 14), and radio-marked with a 10.7-gram necklace type transmitter (Advanced Telemetry Systems, Isanti, Minnesota). Each transmitter was no more than 3% of the bird's body weight, and was fitted with mortality switches.

Chicks were located 2-3 times each week from capture date through August in 2005 and 2006 to determine chick and brood survival and cause-specific mortality. I estimated chick survival from initial number of chicks that hatched from successful nests to the number of chicks that survived 3 weeks post hatch. The initial number of chicks that hatched was estimated by examining condition of egg membranes. Chicks were counted twice each week by searching the area, flushing the radio-marked hen and counting her chicks each week. Chicks > 3 weeks of age were difficult to count accurately, so there is a data gap of survival from 3 weeks of age until chicks could be radio-marked at 5-6 weeks of age to estimate juvenile survival.

Habitat measurements. - I recorded vegetation measurements at brood sites and independent random sites within 10 km of leks from May to August of 2005 and 2006. Coordinates of random sites were entered into a GPS to locate the point in the field, created in ArcGIS 9.1 (ESRI, Inc., ArcGIS 9.1, Redlands, CA). The accuracy of GPS units was usually less than ± 10 m. I recorded slope and aspect for each site using a clinometer and compass, respectively as the downhill direction from each site. At each brood and random site I established two 50-m transects which were centered over the brood site or nearest to the random point. I recorded species, height, length, and width (cm) of sagebrush at each brood site and random site. At each 10-m interval (n = 20) along each transect I recorded the distance to the nearest sagebrush using the point-centered-quarter method (Cottam and Curtis 1956). For every sagebrush encountered, I also recorded the height, length, and width of each sagebrush, and measured grass height with the Robel pole. I estimated visual obstruction and height of grass using a modified Robel pole delineated in 2.54 cm increments (Robel et al. 1970, Higgins and Barker 1982, Benkobi et al. 2000). Herbaceous canopy cover was estimated at the brood or random site, and at additional 10-m intervals along 50-m transects in 0.10 m2 quadrats (see Appendix D for species identification at brood sites and random sites; Daubenmire 1959). I recorded total cover, total shrub, total grass, total forb, litter, bareground and common species of grass and shrubs in each quadrat. I obtained measures of maximum and minimum daily temperature, and daily precipitation from a weather station in Bowman County (North Dakota Agricultural Weather Network, 2006).

Data Analysis

Habitat Selection. - Canopy cover values were recoded to mid-point values of the categories and I summarized these data to an average value for each variable for the site. Estimates of sagebrush density were made from maximum likelihood estimates (Pollard 1971). I then used MRPP (Mielke and Berry 2001) to test the distributions of vegetation characteristics at brood sites and random sites, and used this as a screening process to distinguish important variables for future analysis with a critical value of $\alpha \leq 0.05$. Vegetation characteristics included in MRPP evaluation were percent total vegetative cover, percent grass cover, percent forb cover, percent sagebrush cover, percent bareground, percent litter, sagebrush height, average sagebrush width, site-VOR and VOR increments of 10-m extending from the brood site to 50-m, grass height from the Robel pole beneath the sagebrush, and sagebrush density (Appendix E). I compared variables between brood sites and random sites, and between years using MRPP as initial screening between significant variables at the critical value of α at ≤ 0.05.

I used Information Theoretic approach (Burnham and Anderson 2002) with logistic regression to estimate variables selected for by hens with chicks at brood sites using SAS JMP (2005 SAS Institute Inc). I developed 10 a-priori models for resource selection of brood sites. Only variables for which distributions differed between brood and random sites from MRPP were considered for inclusion in these models. The candidate models included percent total vegetative cover, percent grass cover, percent sagebrush cover, shrub height, site-VOR, every 10-m intervals of VOR extending from the center point, grass height from the Robel pole, and sagebrush density. Year was considered a dummy variable in all candidate models. Thus, any differences among the models in the candidate sets were due to differences in the vegetative variables. Year was not included in the tables for ease of interpretation.

To prevent underfitting or overfitting, Akaike's Information Criterion (AIC) was used as the basis for model selection. Using the log-likelihood values and number of parameters (k) provided in the output file from the 10 models within Program JMP. The models were ranked using the equation: *AIC = -2(log-likelihood) + 2k*. The two components of AIC include; *-2(log-likelihood*), which measures discrepancy of the fit between the data and the model, and (*k*) is a penalty for the number of parameters included in the model to prevent overfitting the models. Unless the sample size is large with respect to the number of parameters estimated, the use of *AICc* is recommended; *AIC + 2K(K + 1)/n – K -1.* The models were ranked using Δ *AICc* (Burnham and Anderson 1992).

I tested the strength of the model to predict brood sites using receiver operating characteristic curve (ROC) used as model fit or discrimination diagnostics (SAS JMP 2005). ROC values between 0.7 and 0.8 were considered acceptable discrimination, and values between 0.8 and 0.9 were considered excellent discrimination (Hosmer and Lemeshow 2000).

Chick Survival. -Chicks were radio-marked and located ≥ 2 times/week from capture date through August in 2005 and 2006 to determine chick and brood survival and cause-specific mortality. Chick survival was estimated using Kaplan-Meier product-limit method (Kaplan and Meier 1958) modified for staggered entry (Pollock et al. 1989) throughout the brood-rearing periods. I designated seasons as summer (June-August), autumn (September-November), winter (December-February), and spring (March-May; Leonard et al. 2000).

Differences in distribution of chick survival and average brood size were tested with MRPP between years at the critical value of $\alpha \leq 0.05$. A brood was considered successful if ≥ 1chicks were observed with a radio-marked hen after 1 August, the approximate date of brood breakup (Dalke et al. 1960, Oakleaf 1971). I defined recruitment as a chick surviving through December 31; the approximate date when the highest percent of mortality has decreased when there is low winter mortality (Robertson 1991, Wik 2002, Hausleitner 2003, Zablan 2003).

RESULTS

Chick Survival. **-** I monitored 7 broods in 2005, with an average of 6.86 ± 0.95 chicks/hen at hatch. In 2005, at 3 weeks post hatch, the average brood size from 7 hens was 2.34 chicks/hen representing 34% apparent survival. In 2006, 6 broods averaged 6.67 ± 1.03 chicks/hen at hatch. At 3 weeks post hatch, the average brood size from 6 hens was 2.83 chicks/hen representing 42% apparent survival. Initial brood size at hatch was similar between years ($P = 0.90$, MRPP).

In 2005, 6 hens had at least 1 chick alive on 1 August, and in 2006 only 3 hens had at least 1 chick alive on 1 August. In 2005, 50% (95% CI: 0.23 to 0.58) of chicks radio-marked at 5-6 weeks of age survived to 1 January (Figure 8). In 2006, 32% (95% CI: 0.14 to 0.49) of chicks survived to 1 January (Figure 9). Assuming no mortality from 3 weeks to 5-6 weeks, a very liberal estimate was 17% of chicks recruited into the population in 2005 and 13% recruited in 2006 (Table 10). The majority of identifiable predation events on radio-marked sage-grouse chicks were from canids. Survival of radio-marked chicks between years was similar ($P = 0.32$, MRPP). Combined yearly survival of radio-marked chicks from 5-6 weeks to 1 January was 39% for both field seasons (Figure 10). Brood success in 2005 was the same as in 2006 (75% in 2005 and 75% in 2006).

Brood Site Selection .- I measured vegetative characteristics at 55 and 75 brood sites in 2005 and 2006, respectively. I also measured 107 random sites during the two years (47 in 2005, 60 in 2006). Distributions of percent total vegetative cover, percent forb, percent grass, percent sagebrush, percent litter, site-VOR, and sagebrush density, differed ($P \leq 0.05$) from random sites. Random sites, however, had more bareground and taller grass ($P \leq 0.05$) than brood sites (Table 11). There were also several variables that differed between years for both brood and random sites (Table 12).

Because annual differences between years were evident, one model strongly was supported with selection of brood sites and percent forbs, percent grass, percent sagebrush, percent bareground, sagebrush height and width (Table 13). Sage-grouse brood sites were positively associated with more canopy cover from forbs, grasses, and sagebrush than were present at random sites, and negatively associated to percent bareground, sagebrush height and width. In the model, increasing forb cover by 10%, increased the probability of the site being used by a hen with a brood by a multiplicative factor of 0.09 ± 0.08 (CI 95%). Increasing grass cover by 10% increased the probability of the site being used by a hen with a brood by a multiplicative factor of 0.61 ± 0.56 (CI 95%), and increasing sagebrush cover by 10% increased the probability of the site being used by a hen with a brood by a multiplicative factor of 1.12 ± 0.95 (CI 95%). Increasing the percent of bareground by 10% decreased the probability of the site being used by a hen with a brood by a multiplicative factor of 240.89 ± 32.89 (CI 95%). Increasing sagebrush height by 5 cm decreased the probability of the site being used by a hen with a brood by a multiplicative factor of 22.18 ± 21.42 (CI 95%), and increasing sagebrush width by 5 cm decreased the probability of the site being used by a hen with a brood by a multiplicative factor of 15.5 ±14.68 (Table 14). Classification accuracy of the model was acceptable with an ROC value = 0.78.

Brood sites consisted of 6-16% forb cover, 29-34 % grass cover, and 5% sagebrush cover, and sagebrush 30-38 cm tall, and 50-53 cm wide. Percent bareground cover at brood sites ranged from 11-25% (Table 15).

DISCUSSION:

Chick/Brood Survival. **-** The low chick survival in southwestern North Dakota is typical of other sage-grouse populations (Schroeder et al. 1997). The period of greatest chick mortality occurred from hatch to 3 weeks of age. Canid predation was the largest direct cause of mortality of radio-marked sage-grouse chicks. Exposure to wet and cold weather can also reduce survival of chicks (Patterson 1952). I found high mortality to chicks exposed to rain and cold weather immediately after hatch. Greater precipitation in 2005 resulting in increased herbaceous cover and delayed plant desiccation may have resulted in higher survival of chicks ≥ 5-6 weeks to 1 January in 2005 (Oakleaf 1971).

It is difficult to draw conclusions regarding survival rates of juvenile sage-grouse among various studies because methods of data collection and analyses varied among studies. Nonetheless, Crawford et al. (2004) estimated 10% survival for sage-grouse chicks from hatch to the following breeding season. My estimate of chick survival through 1 January of 13-17% is half that required to sustain a population, assuming reasonable levels (40-60%) of nest success and nesting rates (Aldridge and Brigham 2001). However, the 13-17% is a liberal estimate because I do not know what the mortality was from 3 to 5-6 weeks of age. Poor recruitment could be the limiting factor of population growth (Johnson and Braun 1999). Chick survival could be limited by availability of mesic habitats that contain higher amounts of forbs during 3-4 weeks post hatch (Aldridge 2000). Studies conducted in the Powder River Basin have documented large-scale modification of sagebrush habitat associated with oil and gas development that could have impacts on habitat use or survival rates of sagebrush obligate species (Walker, unpublished data 26th Meeting of the Western Agencies Sage and Columbian Sharp-tailed Grouse Technical Symposium, abstract). Fragmentation of brood rearing habitats results in additional challenges to brood survival and may create travel barriers separating suitable cover from important mesic feeding areas.

Habitat Selection. - A key factor associated with sage-grouse productivity is brood-rearing habitat (Crawford et al. 1992). Availability of food resources such as forbs and insects can limit sage-grouse populations through decreased recruitment of young (Klebenow 1969, Peterson 1970, Wallestad 1975, Autenrieth 1981). The period from 1-10 days is when chick mortality is highest (Patterson 1952, Autenrieth 1981) and they need insects in close proximity to escape cover. Based on the low chick survival in this study, the present availability of high-quality brood-rearing habitat may be an important factor contributing to low survival rates and ultimately to declining populations of sage-grouse in North Dakota.

Hens with broods selected sites with greater herbaceous cover (forbs and grass), greater sagebrush cover, shorter sagebrush, and less bareground. In the Great Basin, hens with chicks also selected for areas of increased forb cover (Klebenow 1969, Autenrieth 1981, Dunn and Braun 1986). Martin (1970) reported heights of sagebrush ranging from 22-38 cm at brood locations in southwestern Montana, which is similar to sage-grouse brood habitat in my study. The primary food for chicks < 10 days old is insects. Chicks require insects for growth and development (Johnson and Boyce 1990) and insect abundance is greater in areas with greater herbaceous biomass (Healy 1985, Rumble and Anderson 1996). Diet and feeding rates of birds have been shown to increase with abundance of food items (Healy 1985, Miller et al. 1994) which may explain why I found that broods selected areas with greater grass and forb abundance.

Sveum et al. (1998) reported that lack of alternate brood-rearing cover types resulted in low chick survival. Unlike many other sage-grouse populations in the core area of sagebrush habitat, hens in my study had little opportunity to choose alternate brood-rearing habitats. Increased food and cover may reduce brood movements, thereby reducing exposure to potential predators. Reduced movement also would result in lower energetic costs associated with obtaining food and higher foraging efficiency, thereby increasing the nutrients available for growth and development resulting in faster rates (Sveum et al. 1998). It is likely that the limited brood-rearing habitat distribution, the lack of alternative habitats, and the disturbance and fragmentation from oil and gas development has a detrimental affect on chick survival and juvenile recruitment.

MANAGEMENT IMPLICATIONS

Low productivity and chick survival rates should be of great concern to managers charged with maintaining viable populations of great sage-grouse (Connelly and Braun 1997, Crawford et al. 2004). Additional research to achieve a better understanding of juvenile survival and understand factors affecting productivity and recruitment is needed. Additional research to verify the effects of oil and gas development in a highly fragmented landscape is warranted.

My results suggest that conservation and/or restoration of native forb and grass communities within sagebrush shrubsteppe dominated habitats would benefit sage-grouse. Vegetative cover and habitat interspersion are also important factors which influence summer habitat use for grouse. Mosaics of patchy sagebrush with openings of native grasses and forbs will sustain brood-rearing habitat.

Trends in sagebrush vegetation in North Dakota are similar to the rest of the sagebrush range. I recommend that managers develop strategies to preserve the integrity of shrubsteppe in southwestern North Dakota. Herbaceous cover in sagebrush habitats is an important component of brood-rearing habitat for sage-grouse. There is little direct evidence associating livestock grazing practices to sage-grouse populations. However, my results suggest excessive grazing within suitable brood-rearing habitats could have a negative impact by reducing grass and forb cover. Improper grazing facilitates invasion by exotic plant species. Additionally, private landowners should be encouraged to participate in programs that are directed at maintaining and improving sage-grouse habitats on private lands.

Oil and gas development in various grouse habitat types has been increasing in southwestern North Dakota. Even though the timing of the increase in oil and gas development has been coincident with the declining trend of sage-grouse populations in southwestern North Dakota, very little is documented about effects this development has on grouse populations. Massive landscape changes within habitats utilized by broods have rarely been documented. Additional research to determine the effects of oil and gas development in relation to survival of great sage-grouse chick habitat selection is needed.

Table 10. Chick recruitment as of 1 January estimated for chick survival from hatch to 3 weeks-post hatch combined with chick survival at 5-6 weeks through recruitment in southwestern North Dakota, USA, 2005-2006.

Year	**3 Week Survival (Apparent)**	**5-6 Week Survival (Kaplan-Meier)**	**Recruitment**
2005	34%	50%	17%
2006	42%	32%	13%

Table 11. Combined average distributions of vegetation characteristics for brood sites and random sites of sage-grouse in southwestern North Dakota using MRPP, 2005-2006.

Variable	**Brood (*n* = 130)**	**Random (*n* = 107)**	**p-value**
Vegetative cover (%)	74	55	< 0.001*
Grass cover (%)	32	21	< 0.001*
Forb cover (%)	11	9	< 0.001*
Sagebrush cover (%)	5	4	0.041*
Bareground cover (%)	17	32	< 0.001*
Site-VOR (in)	3	2	0.107 *
Sagebrush density/hectare	2,300	1,546	< 0.001*
Sage (%)	5	3	< 0.001*
Vegetation height/site (in)	12	14	0.065*
Grass height beneath the sagebrush (cm)	41	42	0.431
Sagebrush height (cm)	33	33	0.646
Sagebrush width (cm)	48	48	0.298

Asterisks (*) indicates significance. Definition of each variable in Appendix E.

Table 12. Combined average distributions of habitat characteristics for brood sites compared between years and random sites compared between years of sage-grouse in southwestern North Dakota using MRPP, 2005-2006.

Variable	**Brood 2005 (*n* = 55)**	**Brood 2006 (*n* = 75)**	**p-value**	**Random 2005 (*n* = 47)**	**Random 2006 (*n* = 60)**	**p-value**
Vegetative cover (%)	67	79	< 0.001*	57	54	0.429
Forb cover (%)	16	6	< 0.001*	13	6	< 0.001*
Grass cover (%)	29	33	0.145	23	19	0.249
Sagebrush cover (%)	5	5	0.334	5	3	0.016*
Bareground cover (%)	25	10	< 0.001*	34	29	0.113
Site-VOR (cm)	6	1	< 0.001*	3	1	< 0.001*
Sagebrush density/hectare	1,619	2,991	0.001*	1,011	1,966	< 0.103*
Sagebrush (%)	5	5	0.4075	4	3	0.220
Grass hgt beneath the sagebrush (cm)	48	36	< 0.001*	49	37	< 0.001*
Sagebrush hgt (cm)	38	30	< 0.001*	38	29	< 0.001*
Sagebrush width (cm)	51	45	0.011*	53	44	< 0.002*

Asterisks (*) indicates significance. Definition of each variable (Appendix E).

Table 13. Logistic regression models predicting greater sage-grouse brood sites (n = 130) versus random sites (n = 107) using vegetal data collected in North Dakota, USA, 2005-2006. Log-likelihood *(-2 ln [L])*, number of parameters including year indicator variable plus 2 *(intercept + SE) (K)*, Akaike's Information Criterion adjusted for small sample size *(AICc*), difference in AICc *(ΔAICc),* Akaike weights *(wi*). Models with ΔAICc < 2 are highlighted as the best model.

Model	**Log-likelihood**	**K**	**AICc**	**Δ AICc**	**Wi**
Togr (+) + Tofo (+) + Tosh (+) + Bare (-) Shrub hgt (-) + Shrub w (-)	**-135.97149**	**9**	**258.9682**	**0**	**0.890**
Toco (+) + cover (+) + Shrub hgt (-) + Shrub w (-)	-123.91192	7	263.215	4.247	0.106
Toco (+)	-123.78395	8	271.0118	12.044	0.002
Tofo (+) + Togr (+) + Tosh (+) + Bare (-) + Cover (+)	-140.64085	8	271.943	12.975	0.001
Tofo (+) + Togr (+) + Tosh (+)	-145.71992	6	288.5111	29.543	<0.001
Tofo (+) + Togr (+) + Tosh (+) + Bare (-)	-137.64685	7	288.5452	29.577	<0.001
Toco (+) + Cover (+) + Shrub den (+)	-136.44906	6	289.1771	30.209	<0.001
Tofo (+) + Togr (+) + Cover (+) + Shrub den(+)	-137.97987	7	296.9288	37.961	<0.001
Tofo (+) + Togr (+) + Cover (+)	-146.78395	6	304.6572	45.689	<0.001
Tofo (+) + Togr (+) + Tosh (+) + Bare (-) + Height (-) + Shrub w (-)	-134.0313	9	314.2952	55.327	<0.001

a I included the following vegetation variables in my models: total vegetative cover (TOCO), percent forb cover (TOFO), percent grass cover (TOGR), percent sagebrush cover (TOSH), sagebrush height (SHRUB HGT), sagebrush width (SHRUB W), site-VOR (COVER), percent bareground cover (BARE), sagebrush density/hectare (SHRUB DEN), and grass height around the Robel pole (HEIGHT).

b To facilitate interpretation, I excluded year indicator variable from model column.

Table 14. Odds ratio and confidence intervals associated with independent variables that best explain brood sites or random sites in southwestern North Dakota, USA, 2005-2006.

Variable	**Odds Ratio**	**Site**	**Odds Lower CI**	**Odds Upper CI**
TOFO	0.009	Brood	0.001	0.100
TOGR	0.061	Brood	0.005	0.728
TOSH	0.112	Brood	0.017	0.684
Bareground	24.088	Random	3.289	198.941
Sagebrush height	4.435	Random	0.152	133.220
Sagebrush width	3.100	Random	0.165	104.172

Table 15. Average vegetation characteristic of sage-grouse brood and random sites used in the best model to explain brood sites in southwestern North Dakota, USA, 2005-2006.

	Broods 2005	**Randoms 2005**	**Broods 2006**	**Randoms 2006**
Variable	$\bar{x}$	$\bar{x}$	$\bar{x}$	$\bar{x}$
Forb cover (%)	16	13	6	4
Grass cover (%)	29	23	34	19
Sagebrush cover (%)	5	5	5	3
Bareground cover (%)	25	35	11	29
Sagebrush height (cm)	38	38	30	29
Sagebrush width (cm)	53	55	50	47

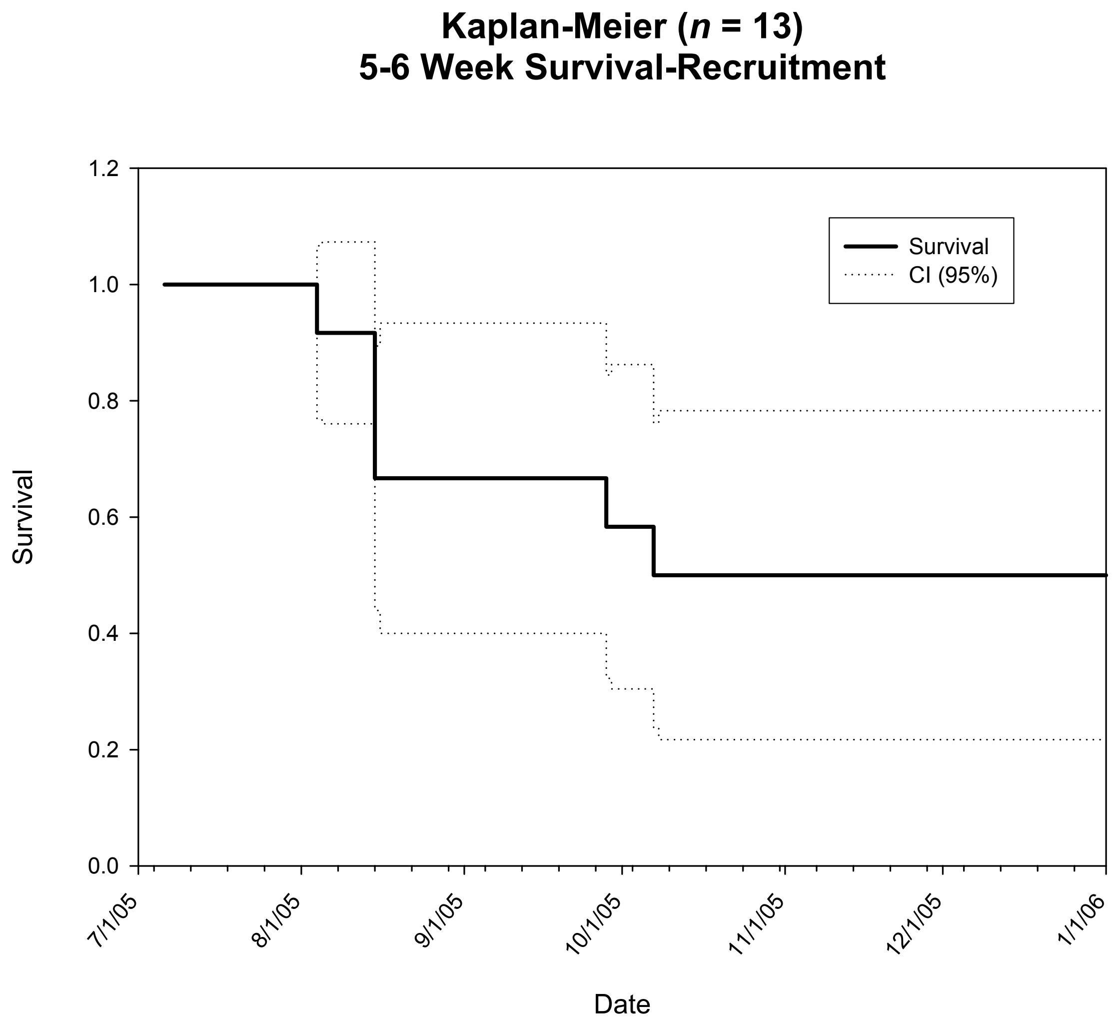


Figure 8. Greater sage-grouse chick survival rate and 95% confidence intervals (dashed lines) of chicks captured at 5-6 weeks of age that recruited into the population as of January 1 2006 in southwestern North Dakota, USA (Kaplan and Meier 1958, Pollock et al. 1989).

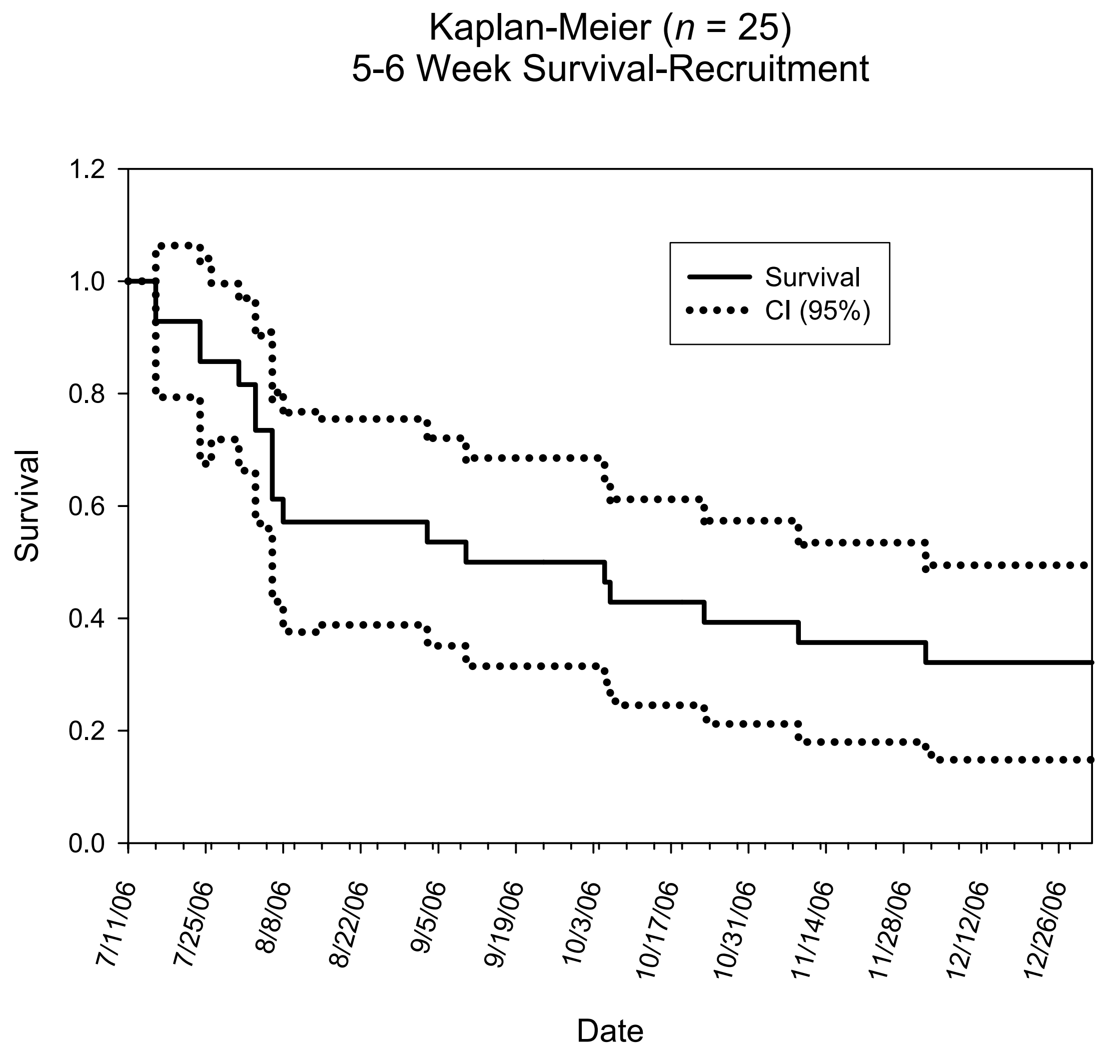


Figure 9. Greater sage-grouse chick survival rate and 95% confidence intervals (dashed lines) of chicks captured at 5-6 weeks of age that recruited into the population as of January 1 2007 in southwestern North Dakota, USA (Kaplan and Meier 1958, Pollock et al. 1989).

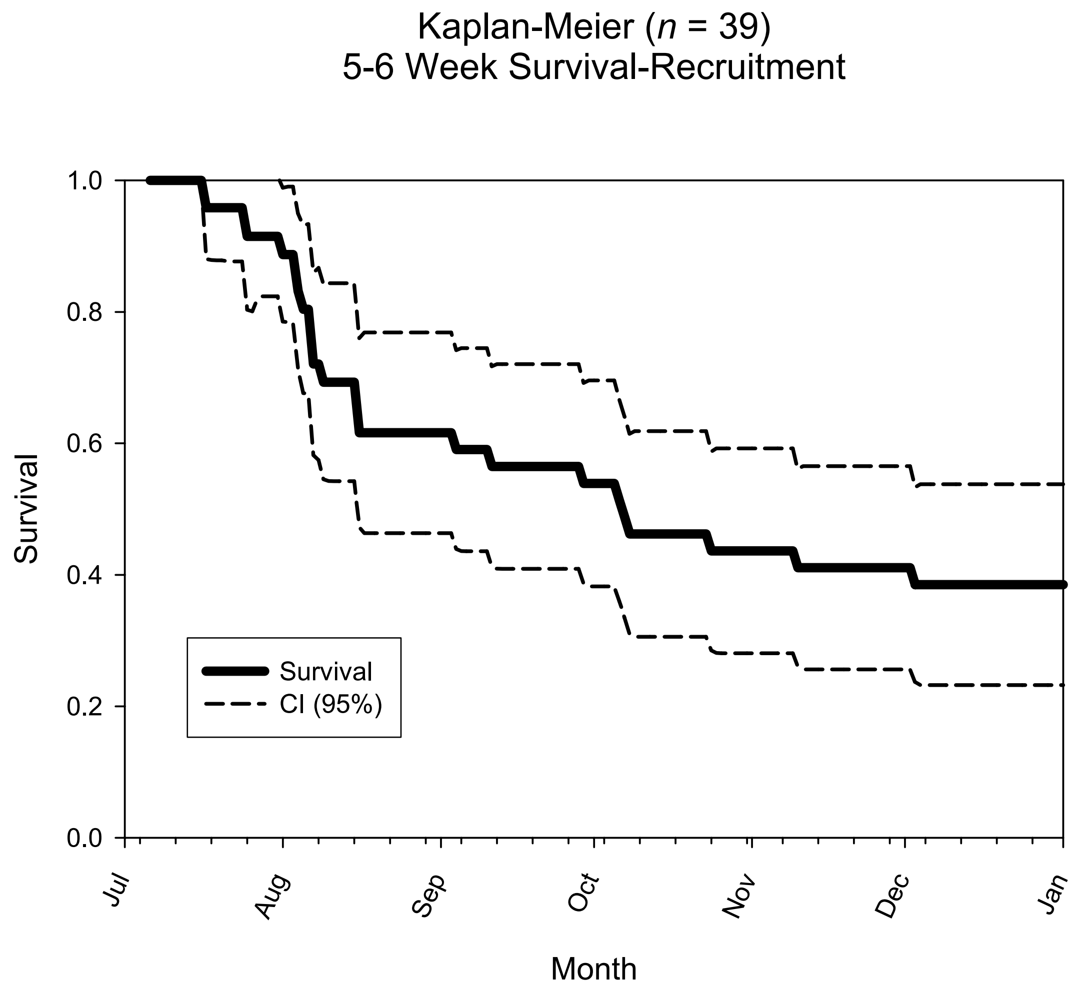


Figure 10. Overall chick survival rate and 95% confidence intervals (dashed lines) of chicks captured at 5-6 weeks of age that recruited into the population as of January 1 for 2005 and 2006 in southwestern North Dakota, USA (Kaplan and Meier 1958, Pollock et al. 1989).

LITERATURE CITED

Aldridge, C.L. 2000. Reproduction and habitat use by Sage Grouse (*Centrocercus urophasianus*) in Canada. Proc. 5th Prairie Conservation and Endangered Species Workshop. Prov. Mus. Alberta Nat. Hist. Occasional Paper 24:197-205.

Aldridge, C.L., and R.M. Brigham. 2001. Nesting and reproductive activities of Greater Sage-Grouse in a declining northern fringe population. Condor 103:537-543.

Autenrieth, R.E. 1981. Sage Grouse management in Idaho. ID Dept. Fish Game, Boise. Wildlife Bulletin 9. 238pp.

Beck, J.L., D.L. Mitchell, and B.D. Maxfield. 2003. Changes in the distribution and status of sage grouse in Utah. Western North American Naturalist 63:203-214.

Beck, J.L., K.P. Reese, J.W. Connelly, and M.B. Lucia. 2006. Movements and Survival of Juvenile Greater Sage-Grouse in Southeastern Idaho. Wildlife Society Bulletin 34(4):1070-1078.

Benkobi, L., D.W. Uresk, G. Schenbeck, and R.M. King. 2000. Protocol for monitoring standing crop in grasslands using visual obstruction. Journal of Range Management 53:627-633.

Bergerud, A.T. 1988. Population ecology of North American grouse. Pages 578-648 in A.T. Bergerud, and M.W. Gratson, editors. Adaptive strategies and population ecology of northern grouse. University of Minnesota Press, Minneapolis, USA.

Braun, C.E. 1987. Current issues in sage grouse management. Proc. West. Assoc. Fish and Wildlife Agencies 67:134-144.

Braun, C.E. 1998. Sage grouse declines in western North America: what are the problems? Pages 139-156 *in* Proceedings of the Western Association of Fish and Wildlife Agencies 78th Annual Conference in Jackson, WY, USA.

Burnham, K.P., and D.R. Anderson. 1992. Data-based selection of an appropriate biological model: the key to modern data analysis. Pages 16-30 in D. McCullough, and R. Barrett, eds. Wildlife 2001. Populations. Elsevier Sci. Publ. Ltd., New York.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach, second edition. Springer-Verlag, New York, New York, USA.

Connelly, J.W. 1982. An ecological study of Sage Grouse in southeastern Idaho. PhD Dissertation. Washington State University.

Connelly, J.W., H.W. Browers, and R.J. Gates. 1988. Seasonal movements of Sage Grouse in southeastern Idaho. Journal of Wildlife Management 52:116-122.

Connelly, J.W., and C.E. Braun. 1997. Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. Wildlife Biology 3:229-234.

Connelly, J.W. S.T. Knick, M.A. Schroeder, and S.J. Stiver. 2004. Conservation Assessment of Greater Sage-grouse and Sagebrush Habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming.

Cottam, G., and J. T. Curtis. 1956. The use of distance measures in phytosociological sampling. Ecology 37(3): 451-460.

Crawford, J.A., M.A. Gregg, M.S. Drut, and A.K. DeLong. 1992. Habitat use by female Sage Grouse during the breeding season in Oregon. Final report, BLM Coop. Res. Unit, Oregon State Univ., Corvallis.

Crawford, J.A., R.A. Olson, N.E. West, J.C. Moseley, M.A. Schroeder, T.D. Whitson, R.F. Miller, M.A. Gregg, and C.S. Boyd. 2004. Ecology and management of Sage-Grouse habitat. Journal of Range Management 57:2-19.

Dalke, P.D., D.B. Pyrah, D.C. Stanton, J.E. Crawford, and E.F. Schlatterer. 1960. Seasonal movements and breeding behavior of Sage Grouse in Idaho. Transactions of Northern American Wildlife Conference 25:396-407.

Daubenmire, R.F. 1959. A canopy-coverage method of vegetation analysis. Northwest Sci. 33:224-227.

Dunn, P.O., and C.E. Braun. 1986. Late summer-spring movements of juvenile Sage Grouse. Wilson Bulletin 98:83-92.

Gates, R.J. 1983. Sage Grouse, Lagomorph, and pronghorn use of sagebrush grassland burn site on the Idaho National Engineering Laboratory. MS Thesis. Montana State University.

Giesen, K.M., T.J. Schoenberg, and C.E. Braun. 1982. Methods for trapping Sage Grouse in Colorado. Wildlife Society Bulletin 10:224-231.

Girard, G.L. 1937. Life history, habits, and food of the sage grouse, *Centrocercus urophasians* Bonaparte. University of Wyoming. Publ. 3:1-56.

Hausleitner, D. 2003. Population dynamics, habitat use, and movements of Greater Sage-Grouse in Moffat County, Colorado. MS Thesis. Univ. Idaho.

Healy, W.M. 1985. Turkey poult feeding activity, invertebrate abundance and vegetation structure. Journal of Wildlife Management 49:466-472.

Hosmer, D.W., S. Lemeshow. 2000. Applied Logistic Regression, 2nd Edition. John Wiley & Sons Incorporated Publication, New York, New York, USA.

Johnson, K.H., and C.E. Braun. 1999. Viability and conservation of an exploited Sage Grouse population. Conservation Biology 13:77-84.

Johnson G.D., and M.S. Boyce. 1990. Feeding trials with insects in the diets of sage grouse chicks. Journal of Wildlife Management 54:89-91.

Kaplan, E., and P. Meier. 1958. Non parametric estimation from incomplete observation. Journal of the American Statistical Association 53:457-481.

Klebenow, D.A. 1969. Sage grouse nesting and brood habitat in Idaho. Journal of Wildlife Management 33:649-662.

Klebenow, D.A. 1985. Habitat management for Sage Grouse in Nevada. World Pheasant Association 10:34-46.

Klott, J.H., and F.G. Lindzey. 1990. Brood habitats of sympatric Sage Grouse and Columbian Sharp-tailed Grouse in Wyoming. Journal of Wildlife Management 54:84-88.

Leonard, K.M., K.P. Reese, and J.W. Connelly. 2000. Distribution, movements, and habitats of Sage Grouse *Centrocercus urophasianus* on the upper Snake River Plain of Idaho.: changes from the 1950's to the 1990's. Wildlife Biology 6:265-270.

Lyon, A.G. 2000. The potential effects of natural gas development on Sage Grouse (*Centrocercus urophasianus*) near Pinedale, Wyoming. MS Thesis. Univ. Wyoming.

Martin, N.S. 1970. Sagebrush control related to habitat and Sage Grouse occurrence. Journal of Wildlife Management 34:313-320.

McCarthy, J.J., and J.D. Kobriger. 2005. Management plan and conservation strategies for greater sage-grouse in North Dakota. North Dakota Game and Fish Department. 96pp.

Mielke, P.W., Jr., and K.J. Berry. 2001. Permutation Methods: A Distance Function Approach. New York: Springer-Verlag. 352 p.

Miller, C.K., R.L. Knight, L.C. McEwen and T.L. George. 1994. Responses of nesting savannah sparrows to fluctuations in grasshopper densities in interior Alaska. Auk 111:962-969.

North Dakota Agricultural Weather Network. 2006. Climatology Information, Bowman ND. http://ndawn.ndsu.nodak.edu. (Accessed Sept 5, 2006).

Oakleaf, R.J. 1971. The relationship of Sage Grouse to upland meadows in Nevada. M.S. Thesis. University of Nevada, Reno, Nevada.

Paige, C., and S.A. Ritter. 1999. Birds in a sagebrush sea: managing sagebrush habitats for bird communities. Boise, Idaho: Partners in Flight Western Working Group.

Patterson, R.L. 1952. The sage grouse of Wyoming. Sage Books, Inc. Denver, Colo. 341pp.

Peterson, J.G. 1970. The food habits and summer distribution of juvenile Sage Grouse in central Wyoming. Journal of Wildlife Management 34:147-155.

Pollard, J.H. 1971. On distance estimators of density in randomly distributed forests. Biometrics 27:991-1002.

Pollock, K., S. Winterstein, C. Bunck, and P. Curtis. 1989. Survival analysis in telemetry studies: the staggered entry design. Journal of Wildlife Management 53:7-15.

Robel R.J, J.N. Briggs, A.D. Dayton, and L.C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23:295-297.

Robertson, M.D. 1991. Winter ecology of migratory Sage Grouse and associated effects of prescribed fire in southeastern Idaho. MS Thesis University of Idaho.

Rumble, M.A., and S.H. Anderson. 1996. Habitat selection of Merriam's turkey (*Meleagris gallopavo merriami*) in the Black Hills, South Dakota, Am. Mid. Nat. 136:157-171.

Sall, J., L. Creighton, and A. Lehman. 2005. SAS: JMP Start Statistics. Sas, Inc.

Schroeder, M.A. 1997. Unusually high reproductive effort by Sage Grouse in a fragmented habitat in north-central Washington. Condor 99:933-941.

Schroeder, M.A., J.R. Young, and C.E. Bruan. 1999. Sage grouse (*Centrocercus urophasianus*). Pages 1-28 *in* A. Poole and F. Gill, editors. The birds of North America, No. 425. The Birds of North America, Inc., Philadelphia, Pennsylvania, USA.

Schroeder, M.A., C.L. Aldridge, A.D. Apa, J.R. Bohne, C.E. Braun, S.D. Bunnell, J.W. Connelly, P.A. Diebert, S.C. Gardner, M.A. Hilliard, G.D. Kobriger, C.W. McCarthy. 2004. Distribution of Sage Grouse in North America. Condor 106:363-376.

Sveum, C.M. W.D. Edge, and J.A Crawford. 1998. Nesting habitat selection by sage grouse in southcentral Washington. Journal of Range Management 51:265-269.

Wallestad, R.O. 1971. Summer movements and habitat use of Sage Grouse broods in central Montana. Journal of Wildlife Management 35:129-136.

Wallestad, R.O., and D.B. Pyrah. 1974. Movement and nesting of sage grouse hens in central Montana. Journal of Wildlife Management 38:630-633.

Wallestad, R. O. 1975. Male Sage Grouse responses to sagebrush treatment. Journal of Wildlife Management 39:482-484.

Welch, B.L. 2005. BigSagebrush: A sea fragmented into lakes, ponds, and puddles. General Technical Reprot RMRS-GTR-144. U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Wik, P.A. 2002. Ecology of Greater Sage-Grouse in south-central Owyhee County, Idaho. M.S. Thesis. Univ. Idaho, Moscow.

Zablan, M.A., C.E. Braun, G.C White. 2003. Estimation of Greater Sage Grouse in North Park Colorado. Journal of Wildlife Management 67(1):144-154.

Appendix D. Four-digit code, common name and scientific name of plant species identified at brood sites and random sites in southwestern North Dakota, USA, 2005-2006.

Variable	**Name**	**Scientific Name**
acmi	Western yarrow	*Achillea millefolium*
aggl	false dandelion	*Hypochoeris radicata*
agst	Redtop	*Agrostis gigantea*
alfa	Alfalfa	*Medicago spp.*
arca	silver sage	*Artemesia cana*
arfi	fringed sagewort	*Artemisia frigida*
arlu	cudweed sagewort	*Artemesia ludoviciana*
artr	big sagebrush	*Artemesia tridentata wyomingensis*
atri	*Atriplex spp.*	*Atriplex spp.*
bear	beards tongue	*Penstemon spp.*
bell	bluebells	*Mertensia spp.*
bogr	blue grama	*Bouteloua gracilis*
brin	Smooth brome	*Bromus inermis*
brja	japanese brome	*Bromus japonicus*
bkbr	buckbrush	*Symphoricarpus occidentalis*
blue	little bluestem	*Vulpia octoflora*
buda	buffalo grass	*Buchloe dactyloides*
cafi	threadleaf sedge	*Carex filifolia*
calo	prairie sandreed	*Calamovilfa longifolia*
carr	wild carrot	*Daucus carota*
cele	wild celery	*Apium graveolens*
chea	cheatgrass	*Bromus tectorum*
cone	purple coneflower	*Echinacea* Moench
crew	Crested wheatgrass	*Agropyron cristatum*
curd	curly doc	*Rumex crispus*
disp	inland saltgrass	*Distichlis spicata*
dwrf	dwarf alyssum	*Alyssum cuneifolium*
ercs	Eastern red cedar	*Juniperus virginiana*
fieb	field bindweed	*Convolvulus arvensis*
gayf	gayfeather	*Liatris spicata*
gold	goldenrod	*Solidago spp.*
gotb	goatsbeard	*Tragopogon dubius*
gpea	golden pea	*Thermopsis rhombifolia*
grra	grayragwart	*Senecio incanus*
gumb	gumbo lily	*Oenothera caespitosa*
gumw	curlycup gumweed	*Grindelia squarrosa*
gusa	broom snakeweed	*Gutierrezia sarothrae*
hoju	foxtail barley	*Hordeum jubatum*

Appendix D. Continued

hors	horseweed	*Conyza spp.*
hory	hairy fleabane	*Conyza bonariensis*
indw	indian wheat	*Plantago patagonica*
intw	intermediate wheatgrass	*Thinopyrum intermedium*
koma	junegrass	*Koeleria macrantha*
long	Longleaf wormweed	*Artemisa longifolia*
must	Mustard	*Cardaria spp.*
navi	green needle	*Nassella viridula*
nutv	nuttall's violet	*Viola nuttallii*
opun	Cactus	*Opuntia spp.*
pars	wild parsley	*Musineon* spp.
pasm	Western wheatgrass	*Pascopyrum smithii*
penn	pennycress	*Thlaspi arvense*
pepp	pepperweed	*Lepidium densiflorum*
phho	hood's phox	*Phlox hoodii*
plan	Slender plantain	*Plantago heterophylla*
prpr	prickly pear	*Opuntia spp.*
popr	kentucky bluegrass	*Poa pratensis*
povw	povertyweed	*Iva axillaris* Pursh
psut	pussytoes	*Antennaria spp.*
redg	red goosefoot	*Chenopodium rubrum*
redt	Redtop	*Agrostis stolonifera*
ripg	prairie cordgrass	*Spartina pectinata*
rose	wild rose	*Rosa woodsii*
rubb	rubber rabittbrush	*Ericameria nauseosa*
sand	sandbergs bluegrass	*Poa secunda*
scgo	scarlet gaura	*Gaura coccinea*
scur	scurfpea	*Psoralea spp.*
side	sideoats grama	Bouteloua curtipendula
silv	silverbladder pod	*Lesquerella argyraea*
skel	skeletonplant	*Lygodesmia* spp.
spid	spiderwort	*Tradescantia ohiensis*
spco	scarlet globemallow	*Sphaeralcea coccinea*
stic	stickseed	*Hackelia* Opiz
stco	needle and thread	*Stipa comata*
sthy	Angelita daisy	*Hymenoxys acaulis*
sunf	sunflower	*Eriophyllum spp.*
swee	sweetclover	*Melilotus spp.*
taof	dandelion	*Taraxacum officinale*
toad	Bastard toadflax	*Commandra umbellate*
this	Thistle	*Cirsium spp.*
txon	textile onion	*Allium spp.*
vetc	*Astragalus spp.*	*Astragalus spp.*

Appendix D. Continued

vuoc	six weeks-fescue	*Vulpia octoflora*
wewa	Western wallflower	*Erysimum asperum*
wint	winter fat	*Krascheninnikovia* spp.
yucca	yucca	*Yucca glauca*

Appendix E. Definition of all acronyms used for vegetative sampling in southwestern North Dakota, USA, 2005-2006.

Acronym	Definition
TOCO	Percent total vegetative cover
TOFO	Percent total forb cover
TOSH	Percent total sagebrush cover
TOGR	Percent total grass cover
Litter	Percent total litter cover (ie. residual grasses, rocks, feces)
Bareground	Percent bareground
Effective Grass Hgt	Grass height beneath sagebrush from Robel pole (in)
Max Grass Hgt	Tallest reading of grass species surrounding Robel pole (in)
Sagebrush Hgt	Sagebrush height (cm)
Sagebrush Width	Sagebrush width (minor and major cords averaged) (cm)
Sagebrush density	Sagebrush density/hectare
Site-VOR	Visual obstruction reading for the site
10-m VOR	Average visual obstruction reading for 10-m around site
20-m VOR	Average visual obstruction reading for 20-m around site
30-m VOR	Average visual obstruction reading for 30-m around site
40-m VOR	Average visual obstruction reading for 40-m around site
50-m VOR	Average visual obstruction reading for 50-m around site

Table 8.10
Suggested Reductions in Cattle Grazing Capacity with Distance from Water.

Distance from Water		
Miles	Km	Percent Reduction in Grazing Capacity[a]
0–1	0–1.6	None
1–2	1.6–3.2	50
2	Over 3.2	100 (consider this area ungrazable)

Source: Holechek (1988).

[a]Supporting literature includes Valentine (1947), Martin and Ward (1973), Sneva et al. (1973), Squires (1973), Beck (1978), Pinchak et al. (1991), and Hart et al. (1993).

Sheep and goats make much better use of rugged terrain than do cattle. Because of smaller size, more surefootedness, and a stronger climbing instinct, they naturally use steep areas much more than do cattle. In most cases, sheep are under the control of a herder and can readily be forced to use the steeper hillsides, minimizing overuse of the valley bottoms. McDaniel and Tiedeman (1981) found that sheep on winter range in New Mexico uniformly used slopes of less than 45 percent. However, utilization was reduced sharply when slopes exceeded 45 percent. Based on their study, slopes greater than 45 percent should be considered unusable by sheep, but little or no adjustment appears necessary for slopes under 45 percent.

Forage Demand by Grazing Animals

Forage demand is a function of the number of animals and the number of days they will occupy a particular range. We believe that the best way to derive daily forage demand (dry-matter basis) of ruminant animals is to multiply their body weight by 2 percent. In Chapter 11 (Table 11.2), we will review a wide range of studies that were consistent in showing that range ruminants consume 2 percent of body weight per day in dry matter when forage availability is not restricted. Intake may go as high as 2.6 percent of body weight for short periods when forage quality is high, and it may drop to 1.5 percent or lower when quality and/or quantity is low. However, the yearly averages given for cattle, sheep, goats, deer, elk, pronghorn, moose, and so on are all about 2 percent. Forage intake by horses and donkeys averages about 50 percent higher than that for ruminants (see Chapter 11). Daily forage intake by various range animals is shown in Table 8.12.

Table 8.11
Suggested Reductions in Cattle Grazing Capacity for Different Percentages of Slope.

Percent Slope	Percent Reduction in Grazing Capacity[a]
0–10	None
11–30	30
31–60	60
Over 60	100 (consider these slopes ungrazable)

Source: Holechek (1988).

[a]Supporting literature includes Glendening (1944), Mueggler (1965), Cook (1966b), Gillen et al. (1984), Ganskopp and Vavra (1987), and Pinchak et al. (1991).

Selection of the Harvest Coefficient

The harvest coefficient is the percentage of total forage produced that is assigned to grazing animals for consumption. Holechek (1988) bases harvest coefficient selection on various stocking-rate studies from different range types. For most arid and semiarid areas, a harvest coefficient of 35 percent would be selected while 50 percent would usually be used for annual grasslands and humid areas if the goal is moderate grazing.

Galt et al. (2000) made detailed evaluations of actual forage use when the Holechek (1988) stocking procedure was applied on several New Mexico rangelands. Consistently, actual measured use was 10 percent to 15 percent higher than the intended use. This was attributed to livestock trampling, wildlife consumption, and weathering. On Chihuahuan Desert rangelands, Paulsen and Ares recommended that stocking levels be set for 35 percent use of perennial grasses. However, they noted that the harvest coefficient must be set at 30 percent to obtain 35 percent use because of trampling, wildlife, and weathering losses. Past and recent research has confirmed this wisdom.

Table 8.12
DAILY DRY-MATTER CONSUMPTION BY VARIOUS RANGE ANIMALS BASED ON THEIR BODY WEIGHT.

Animal	Animal Weight[a] lb	Animal Weight[a] kg	Daily Dry- Matter Intake (Percentage of Body Weight)	Daily Dry- Matter Intake lb	Daily Dry- Matter Intake kg	Animal Unit Equivalents (AU_1)
Cattle (mature)	1,000	455	2	20.0	9.1	1.00
Cattle (yearlings)	750	318	2	15.0	6.8	0.75
Sheep	150	68	2	3.0	1.4	0.15
Goat	100	45	2	2.0	0.9	0.10
Horse	1,200	545	3	36.0	10.9	1.80
Donkey	700	318	3	21.0	6.4	1.05
Bison	1,800	818	2	36.0	16.4	1.80
Elk	700	318	2	14.0	6.4	0.70
Moose	1,200	545	2	24.0	10.9	1.20
Bighorn sheep	180	82	2	3.6	1.6	0.18
Mule deer	150	68	2	3.0	1.4	0.15
White-tailed deer	100	45	2	2.0	0.9	0.10
Pronghorn antelope	120	55	2	2.4	1.1	0.12
Caribou	400	182	2	8.0	3.6	0.40

Source: Holechek (1988).

[a]Average weight of mature male or female animal.

Troxel and White (1989) have developed a simpler, more conservative procedure than Holechek (1988) that allocates 25 percent of current year forage production to livestock and another 25 percent to natural disappearance (insects, wildlife, weathering), with 50 percent left for site protection. The approach developed by Holechek (1988) is based on maximizing forage use by livestock, while that of Troxel and White (1989) works well for range betterment and minimization of risk. On most western ranges, partial or complete destocking would be necessary in only about 3 to 4 years out of 20 with the Troxel and White (1989) procedure.

Recently, other rangeland researchers (Lacey et al., 1994; Johnston et al., 1996; White and McGinty, 1997; Galt et al., 2000) as well as the USDA-Natural Resources Conservation Service (1997) have recommended that a 25 percent harvest coefficient be used when forage is allocated to livestock in stocking-rate decisions. It allows both forage species and livestock to maximize their productivity, allows for error in forage production estimates, greatly reduces problems from buying and selling livestock, reduces the risk of financial ruin during drought years, and promotes multiple-use values.

Variability in precipitation and forage production should play a key role in harvest coefficient selection. Forage production is much more erratic on the desert ranges of the intermountain West than in the central Great Plains (Table 8.13). This necessitates a more-conservative approach to stocking on the desert rangelands.

In the Chihuahuan Desert of New Mexico, Galt et al. (2000) found that ranchers who routinely stocked at capacity based on a 25 percent harvest coefficient would need to liquidate or drylot-feed about one-half their herd in 2 years out of 10 years (Table 8.13). In contrast, the rancher using a 35 percent harvest coefficient would need to completely destock in 2 years out of 10 years and partially destock in another 1 to 2 years. However, Galt et al. (2000) acknowledge that ranchers in the more-humid Great Plains rangelands might do better with a harvest coefficient of 35 percent rather than 25 percent because of less annual variation in forage production.

The real problem is that few ranchers have the skills or time/labor resources to annually quantify forage production (Galt et al., 2000). Unless this is done, use of harvest coefficients higher than 25 percent invariably leads to land degradation and severe financial losses when drought occurs because of rancher reluctance to destock. These losses can quickly eliminate any accumulated benefits of more-efficient forage use. Unused forage in wet years provides a reserve of forage for drought and increases plant vigor and soil water infiltration (Molinar et al., 2001). Rather than a waste, it is an investment in the future.

Table 8.13
TEN-YEAR VARIATION IN FORAGE PRODUCTION ON MODERATELY GRAZED NEW MEXICO CHIHUAHUAN DESERT AND COLORADO MIDGRASS PRAIRIE RANGELANDS.

Chihuahuan Desert New Mexico[a]			Midgrass Prairie Colorado[b]		
Year	Annual Precipitation	Forage Production	Year	Annual Precipitation	Forage Production
	(in.)	(lbs/acre)		(in.)	(lbs/acre)
1989	7.6	189	1957	13.2	1141
1990	10.7	270	1958	17.3	1489
1991	15.1	488	1959	13.5	1095
1992	15.4	750	1960	12.5	1140
1993	9.9	203	1961	17.9	1508
1994	7.0	6	1962	16.4	1314
1995	6.7	59	1963	18.7	1327
1996	7.9	145	1964	9.9	1179
1997	11.6	284	1965	19.4	1197
1998	8.2	173	1966	13.8	1267
Average	10.0	257		15.3	1266
Standard deviation	3.0	207		2.9	137
Coefficient variation	30.2	81		18.9	11

Source: Galt et al. (2000).
[a]*Source*: Holechek et al. (1999b).
[b]*Source*: Sims et al. (1976).

Calculation of Stocking Rate

Once the average forage production and the minimum residue required to maintain the site are determined, the initial stocking rate can be set. It is important to recognize that this rate will often need to be modified as experience is gained for the particular range. The stocking rate is determined by dividing the total usable forage per unit area by the total forage demand of the grazing animals for the grazing period.

We are now ready to solve some hypothetical stocking-rate problems using the procedures developed by Holechek (1988) and validated by Holechek and Pieper (1992). Three cases will be used as examples.

GRAZING CAPACITY PROBLEMS

Location: Northeastern New Mexico
Situation: Short grass Prairie Ranch
Ranch Size: 20,000 acres
Forage Production: × 600 lbs/acre
12,000,000 lbs/forage
× .35 harvest coefficient
4,200,000 lbs. forage usable
÷ 7300 lbs/1000 cow/year
575 Animal Unit Years (AUY)
Total Forage Production: 1996 – 400 lbs/acre
1997 – 700 lbs/acre
1998 – 600 lbs/acre
1999 – 800 lbs/acre
2000 – 500 lbs/acre
Average: 600 lbs/acre

Primary Grasses:
Blue grama
Dropseed
Threeawn
Bluestem
Average: Annual Net Return/All for last 10 years = $80/AU

Questions:

1. What is the grazing capacity of this ranch?

   575 AUYs

2. What is the fair market value of this ranch?

   $80/AU × 575 AUY × 15* = $690,000

GRAZING CAPACITY PROBLEMS (*Continued*)

3. How many 150 lb. sheep will this ranch carry?

$$\frac{1000 \text{ lbs/cows}}{150 \text{ lbs/sheep}} \times 575 = 3{,}833 \text{ sheep}$$

4. What is the average net return acre for this ranch?

$$\$80 \text{ AU} \times 575 \text{ AUs} \div 20{,}000 \text{ AC} = \$2.30$$

*15 is the average multiple investors of stocks have been willing to pay for $1.00 of earnings. This equates to a 6.67 percent return on investment or a 15-year time period for recovery of investment.

CASE 1

You are contemplating buying a ranch on shortgrass prairie range in eastern Colorado. You have determined that the range condition is good. The range is flat and well-watered (no part of the pasture is over 2.4 km from water). Based on information from the Natural Resources Conservation Service and your own ocular estimates, production of key forage species averages about 700 kg/ha of dry matter per year. The ranch is 2,000 ha in size, and you are planning a cow-calf operation.

Question

How many 400-kg cows can you have in your base herd?

Calculation of total usable forage:

Forage production (kg/ha) × percent allowable use
× area(ha)
= total forage (kg) available for grazing

$$700 \times 0.50 \times 2{,}000 = 700{,}000 \text{ kg}$$

Calculation of forage demand:

Weight of cows (kg) × daily dry-matter intake (2% body weight)
× number of days pasture will be grazed (365)
= forage demand per cow per year

$$400 \times 0.02 \times 365 = 2{,}920 \text{ kg of forage/cow/year}$$

Calculation of stocking rate:

Total usable forage (kg) ÷ forage/cow/year
= number of cows pasture will carry

$$700{,}000 \div 2{,}920 = 240 \text{ total cattle}$$

One bull is recommended per 20 cows. Therefore, this range would support a base herd of about 228 cows and 12 bulls.

Question

If sheep (ewes) were substituted for cattle, the number of sheep in the base herd (assume that sheep weigh 65 kg) would be calculated as follows:

$$240 \div \frac{65 \text{ kg (weight per sheep)}}{400 \text{ kg (weight per cow)}} = 1{,}447 \text{ sheep}$$

If this range were used for only 9 months, the total number of cattle would be calculated as follows:

$$\frac{12 \text{ months}}{9 \text{ months}} \times 240 \text{ cattle} = 320 \text{ cattle}$$

At the end of the dormant season (mid-April), 350 kg/ha should remain to protect the site.

CASE 2

You have summer range in the mountains of northeastern Oregon. Condition of the range is poor. Although the terrain is rugged, water is well distributed. You graze this range for 4 months (June through September). Production of key forage species averages about 200 kg/ha/year. The total area is 1,000 ha. Slope on this range is as follows: 40 percent of the range has 0 to 10 percent slope, 20 percent has 11 percent to 30 percent slope, 30 percent has 31 percent to 60 percent slope, and 10 percent has over 60 percent slope.

Question

How many 400-kg cows can you have in your base herd?

Calculation of total usable forage:

Forage production (kg/ha) = × percent allowable use
× area (ha)
= total forage (kg) available for grazing

$$200 \times 0.25 \times 1{,}000 = 50{,}000 \text{ kg}$$

(*continued*)

Calculation of forage demand:

Weight of cows (kg) $\times$ daily dry-matter intake (% of body weight)
$\times$ number of days pasture will be grazed (120 days)
= forage demand per cow per 120 days

$400 \times 0.02 \times 120 = 960$ kg/cow/120 days

Calculation of stocking rate:

Total usable forage (kg) $\div$ forage/cow/120 days
= number of cows pasture will carry (unadjusted for slope)

$50{,}000 \div 960 = 52$ cows (unadjusted for slope)

Adjustment for slope:

[Amount of area with 0 − 10 percent slope (40 percent)
$\times$ adjustment for slope (100 − 0 percent)]
+ [amount of area with 11 − 30 percent slope (20 percent)
$\times$ adjustment for slope (100 − 30 percent)]
+ [amount of area with 31 − 60 percent slope (30 percent)
$\times$ adjustment for slope (100 − 60 percent)]
+ [amount of area with over 60 percent slope (10 percent)
$\times$ adjustment for slope (100 − 100 percent)] $\times$ 52 cows
= grazing capacity of pasture adjusted for slope

$[0.40 \times 1] + [0.2 \times 0.7] + [0.3 \times 0.4] + [0.1 \times 0] \times 52 = 34$ cattle (32 cows + 2 bulls)

CASE 3

You have semidesert grassland range in south-central New Mexico. The condition of your range is highly variable (ranging from poor to excellent) due to poor water distribution. About 60 percent of the range is within 1.6 km of water, 30 percent is between 1.6 km and 3.2 km from water, and 10 percent is over 3.2 km from water. Pasture terrain is generally flat. Based on past experience, you know that production of key forage species averages about 300 kg/ha per year. The total area of the pasture is 4,000 ha. The range is grazed yearlong (365 days).

Question

How many 400-kg cows would this range support without adjustment for water distribution?

Calculation of total usable forage:

Forage production (kg/ha) $\times$ percent allowable use
$\times$ area (ha)
= total forage (kg) available for grazing

$300 \times 0.30 \times 4{,}000 = 360{,}000$ kg

Calculation of forage demand:

Weight of cows (kg) $\times$ daily dry-matter intake (2 percent of body weight)
$\times$ number of days pasture will be grazed (365)
= forage demand per cow per year

$400 \times 0.02 \times 365 = 2{,}920$ kg/cow/year

Calculation of stocking rate:

Total usable forage (kg) $\div$ forage/cow/year
= number of cows the pasture will carry unadjusted for distance from water

$360{,}000 \div 2{,}920 = 123$ cows (unadjusted for distance from water)

Adjustment for water distribution:

[Amount of area within 1.6 km of water (60 percent)
$\times$ adjustment for distance from water (100 − 0 percent)]
+ [amount of area 1.6 − 3.2 km from water (30 percent)
$\times$ adjustment for distance from water (100 − 50 percent)]
+ [amount of area over 3.2 km from water (10 percent)
$\times$ adjustment for distance from water (100 − 100 percent)]
= 123 cows grazing capacity of pasture adjusted for distance to water

$[0.6 \times 1] + [0.3 \times 0.5] + [0.1 \times 0] \times 123 = 92$ cows

We recommend keeping the base herd at 90 percent of grazing capacity on this range to maximize stability during drought. This would result in grazing capacity of 83 total cattle (79 cows + 4 bulls).

(continued)

Question

How many cows and how many yearlings should you have in your herd in an average forage production year if 30 percent of your grazing capacity is used for 275-kg yearlings?

$$92 \text{ cows} \times 0.7(\% \text{ cows in base herd}) = 64 \text{ cows}$$

92 cows in base herd (unadjusted for yearlings)
− 64 cows (adjusted for yearlings)
= 28 cows that can be converted to yearlings

$$28 \text{ cows} \frac{400 \text{ kg (average weight/cow)}}{275 \text{ kg (average weight/yearling)}} = \text{yearlings}$$

In an average year, the base herd would be comprised of 61 cows, 3 bulls, and 41 yearlings.

Question

During a drought year when forage production is only 150 kg/ha, how should cattle numbers be adjusted in mid-October after the growing season?

On this range, 210 kg/ha of residue is required for protection (300 kg/ha forage production in average years × 0.70). Theoretically, based on the current year's forage production, nearly all cattle must be removed to protect this range. However, this could be financially disastrous to the rancher and probably is not necessary to maintain the health of the range. In this situation, empirical judgment on the part of the rancher would be of critical importance. If the drought followed 2 or more years of average or above-average forage production, sufficient carryover residue from previous years should maintain site stability. Grazing on perennial grasses would not become heavy until after the growing season. In some years, winter precipitation in south-central New Mexico results in substantial growth of palatable forbs in late winter and early spring. These forbs take much of the pressure off perennial grasses. Areas long distances from permanent water with large forage supplies can serve as a forage reserve in drought. Utilization is possible by hauling water to these areas.

The best plan would be to sell all yearlings in mid-October and any dry or otherwise undesirable cows. If there was little fall-winter precipitation and forage was showing signs of depletion, the remaining cow herd could be brought into a drylot and fed harvested forage until initiation of forage growth on the range in the spring or summer. Herbel et al. (1984) provide guidelines for feeding confined cattle and marketing calves on desert ranges during drought. Their data show that a part-year confinement of the cow herd (spring), coupled with early weaning of calves in late spring or summer rather than in October, can be economically advantageous over yearlong grazing during periods of drought in south-central New Mexico. Good ranchers plan for drought by having reserves of range forage and/or harvested forage. They cull heavily and reduce herd size after 3 to 4 wet years when the probability of drought becomes high. Consecutive droughts lasting 2 or more years are the ones most damaging to good ranchers and the range. Under these conditions, the most effective strategy financially has been to sell livestock down to levels supportable by range forage resources (Boykin, et al. 1962; Holechek, 1996b).

KEY-PLANT AND KEY-AREA PRINCIPLES

The key-plant and key-area concepts have proven highly useful to managers in evaluating grazing effects on range vegetation (Holechek, 1988). A ***key species*** is defined as "a forage species whose use serves as an indicator to the degree of use of associated species, and because of its importance, must be considered in any management program" (Society for Range Management, 1989a). Key management species are those on which management of grazing on a specific range is based. The key species and key area serve as indicators of management effectiveness. Generally, when the key species and key area are considered properly used, the entire pasture is considered correctly used.

In most cases, one to three plant species are used as key species. These plants should be abundant, productive, and palatable. They should provide the bulk of the forage for grazing animals within the pasture. The "ice-cream" plants (rare but highly profitable plants) are not used because of their scarcity and low resistance to grazing. Key species are usually decreaser plants that are an important part of the climax vegetation. If the range has been grazed heavily, decreasers may be in short supply, but they have the potential to become abundant if grazing pressure is reduced. Conditions do exist in which the climax plants are not the most desirable or a reduction in stocking rate will not restore the climax plants within a reasonable period (5 to 15 years). In these cases, a palatable increaser plant may be selected as a key species. It is important to recognize that key species for one type of animal may be different for another type due to differences in food habits. As an example, bitterbrush (*Purshia tridentata*) is the key species for mule deer on many eastern Oregon ranges, but the key species for cattle on these same ranges is bluebunch wheatgrass (*Agropyron spicatum*). The key species for elk is Idaho fescue (*Festuca idahoensis*) in most of this country.

Under the key-species approach, secondary forage species (e.g., sandberg bluegrass [*Poa sandbergii*] in eastern

Oregon) will receive light use (10 percent to 25 percent), key species (bluebunch wheatgrass) will receive moderate use (30 percent to 40 percent), and the ice-cream plants (arrowleaf, balsamroot [*Balsamorhiza sagittata*]) may be used excessively (over 40 percent).

The ***key area*** is a portion of range that, because of its location, grazing or browsing value, and/or use, serves as an indicative sample of range conditions, trend, or degree of seasonal use (Society for Range Management, 1989a). The key area guides the general management of the entire area of which it is part. Successful range management practices within a pasture are usually judged by the response of the key plant species on the key area.

The key-area concept is based on the premise that no range of appreciable size will be utilized uniformly. Even under light grazing intensities, areas around watering points, salt grounds, valley bottoms, and driveways will often be heavily used. These preferred areas are referred to as sacrifice areas because setting stocking rates for proper use of these areas will result in underuse of the bulk of the pasture. A major objective of specialized grazing systems is to minimize the size of sacrifice areas and provide them with periodic opportunity for recovery. These strategies are discussed in detail in Chapter 9.

When selecting the key area, parts of the pasture remote from water, on steep slopes, or with poor accessibility due to physical barriers should be disregarded. Proper use of these areas will generally result in destructive grazing on most of the pasture. These areas should be omitted when carrying capacity is estimated.

Evaluating Grazing Intensity

A number of qualitative guidelines have been developed for judging intensity of grazing on a range. We have found that a simple categorization into heavy, moderate, and light use is most practical using the following criteria:

- Heavy use. Range has a "clipped" or mowed appearance. Over half of the fair and poor forage-value plants are used. All accessible parts of the range show use, and key areas are cropped closely. They may appear stripped if grazing is very severe (Figure 8.3). There is evidence of livestock trailing to forage.
- Moderate use (proper use). About one-half of the good and fair forage-value plants are used. There is little evidence of livestock trailing. Most of the accessible range shows some use. The range has a patchy appearance (see Figure 8.3).
- Light use. Only choice plants and areas are used. There is no use of poor forage plants. The range appears practically undisturbed.

On key areas, average stubble heights of 30 to 35 cm (12 to 14 in.) for tallgrasses, 15 to 20 cm (6 to 8 in.) for midgrasses, 5 to 8 cm (2 to 3 in.) for shortgrasses, and 2.54 to 3.81 cm (1 to 1.5 in.) for extra shortgrass are recommended minimums for proper use (Holechek and Galt, 2000, 2004).

Figure 8.3
This photo from southeastern Arizona shows the patchy appearance of a moderately grazed pasture on the left and the complete lack of standing forage on the severely grazed pasture on the right.

Guidelines for minimum stubble heights under proper use for selected grass species are provided in Table 8.14. Considerable research exists on minimum stubble height guidelines for grass species such as Kentucky bluegrass, blue grama, and black grama, while for other plants, such as big bluestem and sideoats grama, the main basis for our guidelines is practical experience by range professionals. We freely acknowledge that situations exist where these guidelines may be conservative. However, in nearly all situations, their application should ensure protection of soil and vegetation resources as well as maintenance of livestock performance and wildlife habitat.

In our opinion, it seems reasonable to allow public land ranchers to exceed grazing intensity guidelines (stubble heights and/or residues) on 30 percent of the key areas during any particular year. We believe that these guidelines should be tailored to management objectives for specific allotments. Considerable information is available from various grazing studies that allow development of specific guidelines based on residues, stubble heights, and/or percent use. Generally, management changes are

Table 8.14
Minimum Recommended Stubble Heights for Selected Grass Species Under Proper Grazing Use.

Grass Species	Minimum Stubble Height (in.)[a]	Range Type	Authority
Shortgrasses			
Blue grama	1½–2	Shortgrass	Crafts and Glendening, 1942
Buffalo grass	1–2	Shortgrass	Costello and Turner, 1944
Curly mesquite	1½	Chihuahuan desert	Parker and Glendening, 1942
Black grama	3	Chihuahuan desert	Paulsen and Ares, 1962; Valentine, 1970
Sandberg bluegrass	3–4	Sagebrush-palouse	Practical experience
Mountain muhly	4	Coniferous forest	Johnson, 1953
Kentucky bluegrass	3–5	Mountain meadows	Clary, 1995; Hall and Bryant, 1995
Sedges	3–5	Mountain meadows	Clary, 1995
Midgrasses			
Arizona fescue	6–7	Coniferous forest	Johnson, 1953
Idaho fescue	5–6	Coniferous forest-palouse	Practical experience
Bluebunch wheatgrass	6	Sagebrush-palouse	Anderson, 1969
Little bluestem	6–8	Tallgrass-mixed prairie	Practical experience
Sand dropseed	6–8	Mixed prairie Chihuahuan desert	Practical experience
Sideoats grama	6	Mixed prairie-Chihuahuan desert	Practical experience
Green needlegrass	6	Northern mixed prairie	Practical experience
Western wheatgrass	3–4	Shortgrass-mixed prairie	Holscher and Woolfolk, 1953
Crested wheatgrass	3–4½	Sagebrush	Frischknecht and Harris, 1968
Threeawns	3–5	Mixed prairie Chihuahuan desert	Practical experience
Tallgrasses			
Big bluestem	12–14	Tallgrass	Practical experience
Indiangrass	12–14	Tallgrass	Practical experience
Switchgrass	12–14	Tallgrass	Practical experience
Giant sacaton	12–14	Chihuahuan desert	Practical experience
Basin wildrye	12–14	Sagebrush	Practical experience
Riparian grasses	3–7	Coniferous forest	Clary and Webster, 1990; Clary, 1995; Hall and Bryant, 1995; Clary et al., 1996
Annual grasses	2	California annual grassland	Bentley and Talbot, 1951; Hooper and Heady, 1970

[a]Recommended stubble height minimums should maintain or improve soil, vegetation, and wildlife resources, and provide adequate plant material to meet livestock nutritional needs. We recognize that in some cases, our guidelines may be conservative if the only goal is maintenance of key forage plants.

Table 8.15
GENERAL GRAZING INTENSITY GUIDE FOR CONVERTING STUBBLE HEIGHTS OF SHORTGRASSES, MIDGRASSES, AND TALLGRASSES INTO PERCENTAGE UTILIZATION.

Qualitative Grazing Intensity Category	Stubble Height Guide (Inches)			Percentage of Forage Use by Weight
	Shortgrasses	Midgrasses	Tallgrasses	
Light use to nonuse	2.5+	9+	16+	0–30
Conservative	2.0–2.5	8–9	14–16	31–40
Moderate	1.5–2.0	6–8	12–14	41–50
Heavy	1.0–1.5	4–6	10–12	51–60
Severe	<1.0	<4	<10	<60

Source: Based on Holechek and Galt (2000).

needed if grazing intensity guidelines are exceeded on over 30 percent of the pasture or allotment for 2 consecutive years or in any 2 years out of 5 (Holechek et al., 1998b). If in any year grazing intensity becomes severe (complete lack of stubble height) on one-third or more of the range, management changes should be implemented. An important part of this approach is to encourage ranchers to avoid exceeding residue or stubble height guidelines year after year on the same key areas and to make every effort to keep individual key areas from being severely grazed in any year.

Stubble height is one of the few measurements of range use that is highly repeatable and can be collected quickly. We have found that measurement of 40 randomly or systematically selected plants of each key forage species in key areas usually gives a reliable estimate of grazing use. Long-term studies by Johnson (1953), Paulsen and Ares (1962), and Valentine (1970) have shown grass heights to be well related to grazing intensity and forage productivity. Readers are referred to Clary and Webster (1990), Clary (1995), Hall and Bryant (1995), and Clary and Leininger (2000) for detailed stubble height guidelines on riparian zones. Detailed approaches for evaluating grazing intensity are provided by Anderson and Currier (1973) and Holechek and Galt (2000, 2004b).

Generally, percent forage use as a measure of grazing intensity is more understandable to ranchers and the public, while grass stubble heights are easier to measure and may better reflect grazing severity. Holechek and Galt (2000) developed guides for New Mexico rangelands that permit converting stubble height measurements into percent use. Based on their research and other studies, we have developed a simple guide that should be helpful to managers if its limitations are recognized (Table 8.15). This guide will not apply in all situations (see Holechek and Galt, 2004b), and we encourage managers to develop their own guides for their specific range types.

On some rangelands, shrubs such as common winterfat, fourwing saltbush, and mountain mahogany are the primary forage species. Techniques for evaluating grazing use on shrubs are somewhat different than for herbaceous forage. Holechek and Galt (2000) have developed a guide that relates percentage of leaders (shoot of shrub on tree) browsed to percent use of browse for common New Mexico shrubs (Table 8.16).

Table 8.16
GRAZING INTENSITY GUIDE FOR KEY SHRUB SPECIES.[a]

Qualitative Grazing Intensity Category	Use of Current Year Browse Production by Percentage of Weight	Leaders Browsed
Light use to nonuse	<30	<15%
Conservative	31–50	16–50%
Moderate	51–75	51–80%
Heavy	75–90	81–100%
Severe	<90	All leaders plus old growth used

Source: From Holechek and Galt (2000).
[a]Common winterfat, fourwing saltbush, mountain mahogany.

Figure 8.4 *Moderately browsed antelope bitterbush plants in central Utah.*

Generally, moderate browsing on shrubs involves visible use on 51 percent to 80 percent of the leaders or 51 percent to 75 percent use of current year's growth by weight (Figure 8.4). Other more-quantitative techniques for determining shrub utilization are discussed by Cook and Stubbendieck (1986) and Bonham (1989).

FORAGE ALLOCATION TO MORE THAN ONE ANIMAL SPECIES

Many ranges are grazed by a combination of animals rather than by a single species. The grazing of two or more animals on the same range to obtain more efficient use is referred to as ***common use***. It is well recognized that forage species selection varies considerably among different animal species on the same range. Mule deer on northwestern Colorado ranges heavily use big sagebrush but make little use of needlegrass (*Stipa* sp.) (Hansen et al., 1977). Conversely, on these same ranges, needlegrass is an important component of cattle diets, but cattle will not consume big sagebrush. This range can be used more efficiently by a combination of cattle and deer than by deer or cattle alone. The important questions relate to how much grazing capacity can be increased by the use of both animals, and what amount of the grazing capacity on these ranges should be allocated to deer and to cattle.

In the preceding case, little dietary overlap (less than 5 percent) occurs between the two animals, and, therefore, grazing capacity is additive when both animals are grazing. Because the key species are different for the two animals, no adjustment in cattle or deer numbers is necessary to compensate for forage consumed by the other animal.

On low-elevation winter range in north-central New Mexico, cattle and sheep use the same ranges and have high dietary overlap (over 80 percent) (Holechek et al., 1986). On these ranges, common winterfat (*Ceratoides lanata*) and western wheatgrass (*Agropyron smithii*) are key species for both animals. Here, grazing with cattle and sheep in combination is nonadditive, and animal unit equivalents of one animal can be substituted directly for the other animal. Grazing by both animals in combination gives little improvement in efficiency of use of the forage resource.

However, on many ranges, cattle and sheep have moderate dietary overlaps (30 percent to 60 percent). This is also often true of cattle and elk. Here, allocation of forage is more complicated.

Controversy exists over how grazing capacity should be evaluated when common use is involved. Scarnecchia (1985, 1986) argues that grazing capacity should be based on animal-related factors because dietary overlaps between different animal species vary with terrain, season of use, grazing system, stocking rate, and year-to-year weather fluctuations that affect forage production and species composition.

In contrast, Hobbs and Carpenter (1986) advocate that animal unit equivalents should be weighted

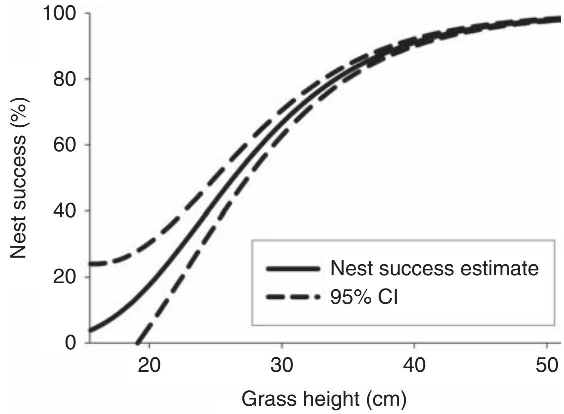


Figure 8.2. Effect of grass height on nest success of Greater Sage-Grouse in northwestern South Dakota, 2006–2007. Nest success estimates were derived from back-transformed beta estimates included in top model. Confidence intervals estimated from the delta method (Seber 1982).

The second-ranked model (AIC_c weight = 0.15) included grass height, litter, daily precipitation, and a 1-day lag of precipitation. Daily precipitation had a positive association with DSR (β = 29.5, SE = 40.4) and the 1-day lag of precipitation was negatively associated with DSR (β = –1.89, SE = 0.77). These variables were only included in supported models when combined with grass height and litter. The third- and fourth-ranked models both included grass height and litter along with the variables daily precipitation and bird age, respectively. Nest success differed between years from 37.7 ± 7.3 SE % in 2006 to 52.5 ± 7.2 SE % in 2007. However, adding a year effect to the top model did not improve model fit.

DISCUSSION

Our study of Greater Sage-Grouse on the easternmost portion of their range in South Dakota identified interesting aspects of sage grouse ecology that have not previously been documented. Female body condition was above average and nesting initiation rates were also high. Similar to other studies, sagebrush cover was an important variable in nest site selection, but at a much lower density than expected. Grass structure, which far exceeded range-wide estimates, played an important role in providing increased cover for successful nests (Connelly et al. 2004). Overall, nest success was within range-wide estimates, suggesting certain features of the habitat condition in South Dakota are productive for sage grouse.

Nesting Parameters

Nest initiation rates for sage grouse are generally low compared to other prairie grouse (Bergerud 1988). However, estimates of nesting initiation based on telemetry are probably underestimated in the literature, as follicular development indicated that at least 98.2% of females laid eggs the previous spring in Idaho (Dalke et al. 1963, Schroeder et al. 1999). Nonetheless, nest initiation rates were high in this study relative to range-wide estimates (Connelly et al. 2004). Females in our study were approximately 63 g (~4%) heavier than the average for 673 individuals in eight other studies (Schroeder et al. 1999). Heavier body mass in female Wild Turkeys (*Meleagris gallopavo*) increased the likelihood of breeding (Porter et al. 1983, Hoffman et al. 1996). Sage grouse exhibit considerable temporal variation in nest initiation rates between years, which may be related to nutrition before and during the breeding season (Hungerford 1964, Barnett and Crawford 1994, Moynahan et al. 2007). High rates of initiation suggest that habitat conditions in our study site were above average.

Renesting rates in sage grouse are highly variable (0–87%), and are linked to environmental effects and habitat quality (Schroeder 1997, Moynahan et al. 2007). Low renesting rates may be related to low primary productivity in the arid and semiarid environments occupied by sage grouse (Schroeder and Robb 2003). For example, Moynahan et al. (2007) found no renesting by sage grouse in dry years with little vegetative growth. In North Dakota, Herman-Brunson et al. (2009) reported 9.5% renesting in sage grouse. The relatively high proportion of renesting females in our study and greater female mass suggest that nesting habitat in South Dakota is of higher quality than elsewhere in sage grouse range. The inverse relationship between length of incubation and renesting propensity suggests that the condition of the female may decline as

POPULATION RESPONSE OF YEARLING GREATER SAGE-GROUSE TO THE INFRASTRUCTURE OF NATURAL GAS FIELDS IN SOUTHWESTERN WYOMING

Completion Report

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Wyoming Cooperative Fish and Wildlife Research Unit
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ABSTRACT

Energy development throughout the western United States has caused habitat changes resulting in local sage-grouse population declines. Sagebrush-dominated habitats in the Green River Basin of southwestern Wyoming have experienced extensive, rapid changes due to the development of natural gas fields. It is unclear whether population declines in natural gas fields are caused by avoidance or demographic impacts, and which age classes are most affected. We investigated habitat selection during the breeding season and demographics of greater sage-grouse to determine if natural gas development has influenced yearling male and yearling female populations in the Upper Green River Basin of southwestern Wyoming. Yearling males avoided leks near the infrastructure of natural gas fields when establishing breeding territories. Additionally, yearling males reared in areas influenced by infrastructure established breeding territories less often, were observed on leks during the breeding period less often, and had lower annual survival rates compared to yearling males reared in areas with no infrastructure. Yearling females avoided nesting within 930 m of the infrastructure of natural gas fields. Additionally, yearling females reared in areas influenced by infrastructure had lower annual survival rates than females reared in areas with no infrastructure. Our results suggest that development of natural gas fields will result in the loss of leks within developed areas and in the functional loss of nesting habitat within 930 m of infrastructure. Because both yearling dispersal from infrastructure and reduced demographics are contributing to abandonment of leks and nesting habitat within natural gas fields, we suggest that peripheral areas be protected from energy development and managed to sustain robust populations to ensure that greater sage-grouse may be available to re-colonize disturbed areas following reclamation.

INTRODUCTION

Populations of greater sage-grouse (*Centrocercus urophasianus*) throughout North America are one-half to one-third the size of those during the late 1960s (Connelly et al. 2004), and the species currently occupies 56% of its pre-European settlement distribution (Schroeder et al. 2004). Throughout Wyoming, greater sage-grouse populations declined an average of 5.2% annually between 1965 and 2003, and the average number of males per lek declined by 49% over that 38-year period (Connelly et al. 2004). Although factors responsible for declines vary regionally, Braun (1998) suggested that declines are primarily a result of human-caused habitat changes. The development of gas and oil fields throughout the western United States (U.S.) has been recognized as one of several anthropogenic changes associated with reduced sage-grouse (*Centrocercus* spp.) populations (U.S. Fish and Wildlife Service 2005).

Approximately 2.7 million ha of land managed by the U.S. Bureau of Land Management (BLM) in the western U.S. are currently in production status for oil, natural gas, or geothermal energy (Knick et al. 2003). A minimum of 25-28% of the total area delineated by a 50-km buffer around the pre-settlement distribution of sage-grouse was influenced by the infrastructure of oil or natural gas developments in 2003 (Connelly et al. 2004). Extraction of oil resources in Wyoming began in the early 1880s (Salt Creek and Dallas Dome oil fields), but industry emphasis has shifted to extraction of natural gas resources since the 1960s (Braun et al. 2002, Connelly et al. 2004; E. T. Rinkes, BLM Lander, Wyoming Field Office; personal communication). Connelly et al (2004) estimated that in 2003, 6 major fields producing oil and gas in the Greater Green River Basin of southwestern Wyoming covered over 8,740 km^2, and active and potential wells numbered approximately 7,890. The infrastructure associated with natural gas developments in the region is expected to increase by 40% by 2015 (Connelly et al. 2004). Existing and proposed oil and gas wells in Wyoming are primarily within landscapes dominated by sagebrush (*Artemisia* spp.; Knick et al. 2003), which are essential for persistence of greater sage-grouse populations.

In southwestern Wyoming, researchers have observed that as the distances between leks and the infrastructure of natural gas fields decrease and as the level of development surrounding leks increase, declines in lek attendance by males approached 100% (Holloran 2005). Walker et al. (2007) reported that only 38% of greater sage-grouse leks active in 1997 or later within coal-bed methane (CBM) fields in the Powder River Basin (PRB) of northeastern Wyoming and southeastern Montana were still active in 2004-2005, compared to 84% of leks outside CBM fields. Active leks in CBM fields had 46% few males per lek than leks outside the fields (Walker et al. 2007). Similarly, Braun et al. (2002) found that the average number of males on leks within 0.4 km of CBM wells was significantly lower than leks greater than 0.4 km from CBM wells. Between 1983 and 1985, 3 lek complexes in southern Canada were disturbed by oil and gas activities within 200 m, and none of these leks have been active since disturbance (Braun et al. 2002, Aldridge and Brigham 2003). In northern Colorado, the overall decline in the number of males on 4 leks near the infrastructure of coal mines was 73% from peak numbers prior to development to approximately 3 years after an increase in mining activity; declines in the number of males were significantly higher than changes witnessed on non-impacted leks (Braun 1986, Remington and Braun 1991).

Impacts of energy developments on sage-grouse can include behavioral avoidance of anthropogenic disturbance and/or increased risk of mortality (Connelly et al. 2004). Lyon and Anderson (2003) observed that female greater sage-grouse nested significantly farther from leks disturbed by roads associated with natural gas fields compared to birds on leks in undisturbed areas in southwestern Wyoming. Significantly fewer females from disturbed leks nested within 3 km of the lek where they were captured compared to birds from undisturbed leks (Lyon and Anderson 2003). Additionally, Holloran (2005) suggested that nesting females avoided areas with high densities of natural gas wells (i.e., 16 ha well spacing). In the PRB, Doherty et al. (2008) concluded that greater sage-grouse avoided CBM wells located in otherwise suitable wintering habitat. At CBM well densities of 12.3 wells/4 km^2 greater sage-grouse were 1.3 times more likely to occupy sagebrush habitats with no CBM wells (Doherty et al. 2008). Greater sage-grouse in Canada avoided nesting in areas with high proportion of non-natural edge habitats, and brood-rearing females avoided areas with high densities of visible wells within 1 km (Aldridge and Boyce 2007). The authors noted that avoidance of human features effectively removed nesting habitat within a 1-km^2 area of these structures (i.e., functional habitat loss).

In Colorado, the probability of detecting Gunnison sage-grouse (*Centrocercus minimus*) declined as sagebrush patches became smaller and were situated closer to roads (Oyler-McCance 1999). Similarly, in southwestern Kansas, lesser prairie-chickens (*Typmanuchus pallidicinctus*) avoided wells and power lines, and the presence of high densities of either type of feature in areas with otherwise suitable habitat precluded use (Hagen 2003). The odds of a power line or road occurring within a monthly-range were 3 times and 11% less likely than in a non-use range. Additionally, lesser prairie-chickens selected nesting sites farther from wellheads, improved roads, buildings (including natural gas compressor stations), and transmission lines than was expected at random (Pitman et al. 2005). Avoidance of anthropogenic features resulted in a functional loss of 58% of the total amount of suitable lesser prairie-chicken nesting habitat (Robel et al. 2004).

Adverse impacts of energy development to demographic parameters have also been noted. Lyon and Anderson (2003) suggested that nesting propensity was significantly lower for females breeding on leks disturbed by roads associated with natural gas fields compared to females in undisturbed areas. The risk of chick mortality among greater sage-grouse increased by a factor of 1.5 for each additional well visible within 1 km of brooding locations (Aldridge and Boyce 2007). Population growth rates of greater sage-grouse and lesser prairie-chickens influenced by energy development were less than growth rates of non-impacted populations (Hagen 2003, Holloran 2005). Both authors suggested that lower population growth rates were primarily due to lower survival and nesting success in the impacted populations.

Research has suggested that energy developments can cause the loss of affected populations. Remington and Braun (1991) suggested that greater sage-grouse population declines in areas near coal mines may have been caused by displacement of yearlings to leks situated away from development. Holloran and Anderson (2004) were able to reproduce observed declines in the number of males occupying 3 natural gas development-impacted leks in southwestern Wyoming by assuming adult male tenacity and minimal yearling male recruitment. A delayed shift in nesting habitat selection away from the infrastructure has been documented in southwestern Wyoming, a pattern consistent with adult females showing nest-site fidelity and yearling females avoiding gas fields (Holloran 2005). Although these studies suggest that the elimination of populations from energy fields may have resulted from the reaction of the yearling cohorts to developments, the response of yearling greater sage-grouse to development of natural gas fields has not been quantified. It is important to determine if yearlings are being primarily displaced or if development negatively influences demographics as these scenarios suggest different mitigation alternatives.

Our objectives were to determine if natural gas development influences habitat selection and demographics of yearling male and yearling female greater sage-grouse in southwestern

Wyoming. We investigated habitat selection and demographics relative to the locations of drilling rigs, producing well pads, and main haul roads. For males, we investigated the location of leks where yearlings established breeding territories, date of territory establishment, breeding-period lek tenacity, and annual and seasonal survival probabilities for both the yearling male cohort overall and for yearlings of known maternity. For females, we investigated nesting habitat selection, nesting propensity, dates-of-nest establishment, nest success, chick productivity, and annual and seasonal survival for both the overall yearling female cohort and for yearlings of known maternity.

STUDY AREA

The study area (42°60′ N, 109°75′ W) encompassed 17 leks primarily within the boundaries of the Pinedale Anticline Project Area (PAPA) and portions of the Jonah II gas field in the upper Green River Basin in southwestern Wyoming (Figure 1; Bureau of Land Management 2000). The study area covered approximately 255,000 ha (2,550 km^2) dominated by Wyoming big sagebrush (*Artemesia tridentata wyomingensis*) shrub-steppe habitats. Elevation ranged from 2,100 to 2,350 m and annual precipitation averaged 27.3 cm (Western Regional Climate Center, Reno, NV). Natural gas development and livestock grazing were the predominant human uses of the area (Bureau of Land Management 2000).

FIELD METHODS

We captured female greater sage-grouse on and near leks from mid-March through April in 2004 and 2005 by spot-lighting and hoop-netting (Giesen et al. 1982, Wakkinen et al. 1992). We secured radio transmitters to females with PVC-covered wire necklaces (Advanced Telemetry Systems Inc. [ATS], Isanti, MN, USA). Transmitters weighed 19.5 g, had a battery life expectancy of 530 days, and were equipped with motion sensors (i.e., radio-transmitter pulse rate increased in response to inactivity).

We used hand-held receivers and 3-element Yagi antennas (ATS) to monitor radio-marked females at least twice weekly through pre-laying (April) and nesting (May-June). We located nests of radio-marked birds by circling the signal source until females could be directly observed. We monitored incubating females after nest identification from a distance of ≥60 m to minimize chances of human-induced nest predation or nest abandonment. We established nest fate (successful or unsuccessful) when radio monitoring indicated that the female had left the area. We considered nests successful if ≥1 egg hatched, indicated by presence of detached eggshell membranes (Wallestad and Pyrah 1974). We monitored unsuccessful females twice weekly for 2 weeks following nest failure to detect re-nests.

We located females that nested successfully 14 days post-hatch. We considered females with ≥1 chick to have been successful through the early brood-rearing stage. We based chick existence on either visual confirmation of chick(s) or the reaction of brooding females to the presence of a potential predator (i.e., the researcher; Schroeder et al. 1999). We relocated females for which no live chicks were detected at 14 days post-hatch 2 to 4 days following the initial location to confirm brood loss.

We monitored females that successfully raised ≥1 chick through the early brood-rearing stage from ≥100 m at least twice weekly through 10 weeks post-hatch. In late summer 2004 and 2005, we captured male and female chicks (e.g., hatch-year birds) that were ≥10 weeks old by spot-lighting radio-equipped brood-rearing females. We captured chicks with the brooding females using hoop-nets (Giesen et al. 1982, Wakkinen et al. 1992). We weighted captured chicks to ensure that radio transmitters could be safely attached (Caccamise and Hedin 1985). We sexed captured chicks based on weights or plumage and aged the birds (to ensure captured grouse were hatch-year birds) based on the shape of the outermost wing primaries (Eng 1955). We collected blood samples by clipping the middle toenail and stored blood on Whatman FTA micro cards (Whatman 2005). We secured 16- or 19.5-g radio transmitters (depending on chick weight) to chicks with PVC-covered wire necklaces (ATS). Transmitters had battery life expectancies of 500 or 530 days, respectively, and were equipped with motion-sensors. We considered radio-equipped male chicks that survived to 1 March and female chicks that survived to 1 April the spring following capture the yearling sample.

Yearling Males

We collected lek visitation data for yearling males using data-logger stations (ATS) situated near 17 leks throughout the study area (Figure 1). Data loggers allowed for constant monitoring of leks during the breeding season. Radio-equipped yearlings visiting a monitored lek were recorded as being on or near that lek at specific dates and times.

Data Loggers.--Data-logger stations consisted of 1 data logger run by 2 deep-cycle recreational vehicle (RV) gel batteries charged by solar panels; all equipment was housed in metal Knaack® boxes. We mounted omni antennas on steel casing pipe such that the top of the antenna was 3 m high. Data loggers were attenuated (i.e., calibration of data logger sensitivity) to detect the entire area utilized by strutting males, and situated to minimize detection of birds using non-strutting habitat surrounding leks. We set data loggers to scan for ATS transmitters (Model A4000) with 35 and 45 pulse per minute (PPM) signals. Due to the possible effects of cold weather on transmitter pulse rates, we allowed a tolerance of 1 (e.g., 35 PPM: 34-36 was recorded; 45 PPM: 44-46 was recorded). We directly accessed stations when leks were not occupied (e.g., non-crepuscular periods) and downloaded data loggers to a laptop computer at least twice during the breeding season. We placed reference transmitters at each data-logger station to verify logging accuracy on all downloads. We monitored leks annually from 1 April to 15 May.

Lek Counts.--Annual lek counts on the 17 monitored leks were conducted by personnel from the Wyoming Cooperative Fish and Wildlife Research Unit (COOP), the Wyoming Game and Fish Department (WGFD), and the Pinedale field office of the BLM. Lek counts were conducted according to standardized methods outlined by the WGFD's Sage-Grouse Technical Committee (Cheyenne, WY, USA; also see Connelly et al. 2003:19-20).

Survival.--We used hand-held telemetry equipment (ATS) to locate yearling males during the breeding season to assess survival. Annual survival for yearling males was assessed from 1 March through the end of February. We assessed survival directly between 1 April and 15 May by locating males weekly. From 15 May through August, we located males from long-range bi-weekly and used transmitter pulse-rates (e.g., motion sensors) to assess survival. Survival from 1 September through March was assessed using fixed-wing aircraft (Mountain Air Research, Driggs, ID, USA; Sky Aviation, Dubois, WY, USA). Flights were conducted at least bi-monthly and we used motion-sensors to evaluate whether individuals were dead or alive.

Yearling Females

Demographics.--We assessed yearling female demographics similarly to those described for the original sample of radio-equipped females. We used hand-held telemetry equipment (ATS) to locate nests by circling the signal source until females could be directly observed. We monitored incubating females from a distance of ≥60 m to minimize abandonment risks. Nest fate (successful or unsuccessful) was established when radio monitoring indicated that the female had left the area; we considered nests successful if ≥1 egg hatched, indicated by presence of detached eggshell membranes (Wallestad and Pyrah 1974). We monitored unsuccessful yearling females twice weekly for 2 weeks following nest failure to assess re-nesting attempts.

We located yearling females that nested successfully weekly from hatch through 35 days post-hatch. We considered females with ≥1 live chick to have been successful through each brooding stage. We based chick existence during the early brooding stage (i.e., hatch through 2 weeks post-hatch) on either visual confirmation of chick(s) or the reaction of brooding females to the presence of a potential predator (i.e., the researcher; Schroeder et al. 1999). During the 2005 late-brooding stages, we obtained fledge estimates (i.e., the number of chicks per brood) by spot-light surveys conducted during trapping. In 2006, we obtained fledge estimates from spot-light surveys conducted 35 days post-hatch (Walker et al. 2006). We relocated females found without live chicks during any of these stages 2 to 4 days following the initial location to confirm brood loss.

Survival.--We assessed annual survival for yearling females from 1 April through March. We located all females twice weekly between 1 April and hatch (approximately 15 June), and brooding females weekly from hatch through August. We assessed survival directly from observations during these periods. We monitored barren females from long-range weekly from nest loss through June, and bi-weekly from July 1 through August; motion sensors were used to evaluate barren female survival during these stages. We assessed survival from 1 September through March for all females from fixed-wing aircraft (Mountain Air Research, Driggs, ID, USA; Sky Aviation, Dubois, WY, USA). Flights were conducted at least bi-monthly and we used the motion sensors to evaluate whether individuals were dead or alive.

STATISTICAL METHODS

Infrastructure of Natural Gas Fields

We mapped features of the infrastructure of natural gas fields within 5 km (Holloran and Anderson 2005) of the 17 monitored leks using ArcGIS 9 (Environmental Systems Research Institute [ESRI], Redlands, CA, USA). We mapped producing well pads, drilling rigs, and main haul roads; state highways, the Paradise Road, and the Green River Road were included as main haul roads (Figure 1). We obtained infrastructure location, drilling activity date, and well producing date information from the Wyoming Oil and Gas Conservation Commission and verified these data using information supplied by Western Ecosystems Technology, Inc. (Cheyenne, WY, USA), Edge Environmental, Inc. (Laramie, WY, USA), individual gas companies (i.e., operators) responsible for specific wells, and through direct ground-truthing using hand-held, 12 channel, Garmin RINO 110 Global Positioning System units (Garmin International, Olathe, KS, USA). Infrastructure data were dynamic and were modified to reflect the conditions encountered seasonally. We considered well pads with multiple producing wells single active locations.

Maternity

We established yearling maternity using microsatellite polymerase chain reaction (PCR) analyses of DNA extracted from blood samples collected during trapping (Taylor et al. 2003, Hawk et al. 2004); 5 primers were used in the analysis (LLSD4, LLSD8, LLST1, SGCA11, and SGCTAT1; Wyoming Game and Fish Laboratory, Laramie, WY, USA). We obtained genotypes following methods described by Frantz et al. (2003). We determined maternity using program Cervus 3.0.3 (Marshall et al. 1998). The simulated population genetic structure was based on 10,000 simulations with 5,000 potential parents, 1% of the candidate parents sampled, and 25% relatedness. Candidate mothers were all females identified by the analysis with ≥80% confidence in parentage assignment. We based final maternal assignment on trap location; if a chick was trapped from the same flock as a candidate mother, maternity was assigned.

We estimated natal areas as the area within 1.9 km of natal nests. We used this distance because 1.9 km represents the mean radius of home ranges during early brood-rearing (Drut et al. 1994) and the upper 95% confidence limit of the mean distance from nest to early brood-rearing locations (Lyon 2000, Slater 2003). We defined natal treatment yearlings as any yearling whose natal area contained >1 producing well pad or >1 km of main haul road; all others were considered natal control yearlings. The inclusion of natal areas with 1 well or a short distance of main haul road in the control population was to guard against including yearlings raised in areas with isolated well pads (e.g., wildcat wells) as treatment birds.

Greater Sage-grouse Yearling Variables

Survival.--We estimated yearling male annual (March-February), yearling female annual (April-March), and monthly survival estimates and standard errors using the staggered entry Kaplan-Meier estimator (Pollock et al. 1989). We censored birds that were not found during any monthly period. We combined monthly survival estimates into sexually distinct seasonal periods: for males, breeding (Mar.-May), summer (June-Aug.) and winter (Sept.-Feb.); and for females nesting (April-June), summer (July-Aug.) and winter (Sept.-Mar.).

Overall Lek Recruitment.--We estimated overall lek recruitment of males annually from lek counts. We estimated the number of males recruited to a lek as the annual change in the maximum number of males minus the number of adult males expected to return to a lek the following year (37%; Zablan et al. 2003).

Yearling Male Demographics.--We based lekking demographics of yearling males on information from data loggers or telemetry. Logged signals consisted of the date, time, transmitter frequency, signal strength, number of pulses recorded in 15 seconds, transmitter pulse-per-minute (PPM) value, and the number of pulse matches (ATS algorithms). The steps taken for distinguishing radio-transmitter detection versus interference included: (1) signals that logged at a PPM outside the range of values set for the data-logger were discounted as interference (e.g., PPM <34, 37-43, >47). (2) Given transmitter pulse rates of either 35 or 45 PPM, the data-loggers accepting pulse rates of 36 and 46 PPM, respectively for these transmitter types, and a 15 second scan time, the number of pulses detected for 35 PPM transmitters had to be ≤ 9 ([36 PPM/60 sec] × 15) and for 45 PPM transmitters ≤ 12 ([46 PPM/60 sec] × 15); if the number of pulses matched was outside these ranges, logged signals were discounted as interference. Logged signals remaining were potential birds. We primarily used pulse match to pulse detected ratios (e.g., the number of matched pulses relative to the number of detected pulses) and the number of logs over a given time period to validate remaining detections as birds. We established the protocol for assessing bird probabilities using pulse match-to-detected ratios and the number of detections by evaluating data from reference collar logs. Reference collar downloads suggested a high pulse match-to-detected ratio, numerous detections, and a recorded pulse count >4 and <30 was a validated detection of a radio-transmitter and not interference. Numerous logs by the same frequency, especially numerous within the same relative time period, with high pulse match-to-detected ratios, had higher potential to be a confirmed bird detection.

We did not consider those frequencies only logged once as bird detections until compared with future data and telemetry locations. We consulted ATS experts for verification of questionable data. We considered confirmed yearling male detections between 0430 and 0730 hours daily lek visits.

The average date that radio-equipped yearling males were first documented on established leks was April 8; thus yearlings were available to be logged for 37 days. Because yearling male daily lek attendance rates in a previous study averaged 19% (Walsh et al. 2004), we considered a bird to have established on a particular lek if it had ≥7 confirmed daily lek visits during the monitoring period. We assessed lek establishment of males not detected on data-logger-monitored leks using telemetry data. A yearling male had to be detected on a lek ≥3 times during the crepuscular daily breeding period between 1 April and 15 May to verify establishment. The date of establishment was estimated as the first day yearling males were documented on the lek where established. Yearling male lek tenacity was estimated as the total number of confirmed daily lek visits on the lek where established. The number of different leks visited by yearling males was estimated as the number of leks with ≥1 confirmed daily lek visit(s), and included leks where established. We only estimated establishment dates, lek tenacity, and number of different leks visited for yearlings that visited leks monitored by data-loggers.

Distance from natal nest-to-established lek was estimated as the straight-line distance from the nest site where a yearling male hatched to the lek where he established the following spring. The probability of establishing a breeding territory on a lek was estimated as the number of yearling males with confirmed lek establishment divided by the total number of available males. Available males survived the breeding season and were those we actively attempted to document establishment leks using telemetry (i.e., those monitored during the breeding season).

Nest Site Designations (Yearling Females).--Females that nested within 930 m of an infrastructure feature of a natural gas field were considered to have been potentially influenced by infrastructure (i.e., nesting treatment females); those nesting outside the 930-m buffer were considered nesting control females (Figure 2). The 930-m buffer represented the upper limit of the 95% confidence interval around mean distances between consecutive year’s nests and, due to nesting area fidelity, represented a female’s life-time nesting area (Holloran and Anderson 2005).

Natal nesting areas were an estimate of the area around the natal nest where a yearling female will usually select a nest location. We used the upper limit of the 95% confidence interval around the mean natal nest-to-yearling nest distances for females raised in areas without the infrastructure of natural gas fields to establish the natal nesting area.

Yearling Female Demographics.--Nesting propensity was estimated as the number of females initiating a nest divided by the total number of yearlings intensively monitored throughout the entire nesting season. We did not include females found for the first time after 15 May annually in nesting propensity estimates (15 May represented the latest date of incubation initiation based on mean latest hatch date and 27 days to incubate a clutch [Schroeder et al. 1999]). The date of nest establishment was the first day females were documented on a nest. Apparent nest success was the number of successfully hatched nests divided by the total number of known nests. Early brood-rearing success was the number of females successfully raising ≥1 chick through 14 days post-hatch divided by the total number of successfully nesting females monitored through the early brood-rearing period. Overall brood-rearing success was the number of females successfully fledging ≥1 chick divided by the total number of successfully nesting females that were monitored throughout the entire brood-rearing period. Natal nest-to-yearling nest distances were estimated as the straight-line distance from the nest site where a yearling female hatched to her first nest the following spring.

Yearling Male Comparisons

We investigated overall male recruitment to monitored leks and radio-equipped yearling male lek establishment relative to the distance of leks to infrastructure of natural gas fields. We also investigated yearling male lek establishment demographics and survival relative to infrastructure impacts to natal areas.

Overall Recruitment.--We used Chi-square tests with continuity corrections (due to sample sizes <25 in certain instances; Dowdy and Wearden 1991) to compare overall recruitment of males among leks. Although we assumed that the number of recruited males was related to lek size, the relationship was probably not 100% correlated. Therefore, we established expected proportions using a scaled allocation of the total recruited population. Leks with ≤50 total males the preceding year were expected to recruit either 4.5 or 5%, leks with >50 and ≤100 males were expected to recruit either 7 or 8.5%, and leks with ≥100 males were expected to recruit either 9.5

or 12.25% of the total recruited population. We used different proportions annually because some of the leks changed size categories between years, and we needed the total proportion of the expected population to sum to 100%. We categorized leks as those recruiting more, less, or equal to the expected number of males. We compared categories by distance to closest active drilling rig, producing well pad, and main haul road using 95% confidence interval overlap.

Lek Establishment.--We generated minimum convex polygons (Kenward 1987) around all producing well pads, and categorized monitored leks as either: contained within the polygon, ≤2 km outside, between 2 and 5 km outside, or >5 km outside the polygon. We used Chi-square tests with continuity corrections (Dowdy and Wearden 1991) to compare the number of radio-equipped yearling males establishing on leks by category (i.e., observed establishment). We assumed equal availability between leks for each yearling male, thus expected proportions were based on the total number of leks within each buffer. We compared dates-of-establishment, lek tenacity, and annual and seasonal survival by buffer using 95% confidence interval overlap.

Natal Areas.--We compared the probability of establishing a breeding territory on a lek between natal treatment and natal control yearling males using Chi-square tests with continuity corrections (Dowdy and Wearden 1991). We determined the expected establishment rate from the control population (e.g., results suggest a difference between natal treatment and natal control groups). We compared the number of different leks visited during the breeding season, the distance from natal nest-to-established lek, dates-of-establishment, lek tenacity, and annual and seasonal survival by natal area category using 95% confidence interval overlap.

Yearling Female Comparisons

General Habitat Selection.--We investigated habitat selection of yearling females relative to infrastructure features of natural gas fields by comparing nesting treatment and nesting control females using Chi-square tests with continuity corrections (Dowdy and Wearden 1991). We estimated the expected number of nests per category as the proportion of the total area within 5 km of trapped leks (Holloran and Anderson 2005) that was within 930 m of an infrastructure variable (Figure 2). We only considered nests located within the 5-km buffer in the comparison.

We assumed suitable nesting habitats were sagebrush and desert shrub-dominated areas within 2 standard deviations of the mean roughness of nest sites located within the 5-km buffer between 2000 and 2006 (Holloran 2005). Jensen (2006) suggested roughness (i.e., the ratio of actual surface area to planimetric area) was the terrain measure best distinguishing greater sage-grouse nests from available locations in southwestern Wyoming. We used Gap Analysis Program (GAP) landcover layers (Wyoming Geographic Information Science Center (WyGISC), University of Wyoming, Laramie, WY, USA) to identify sagebrush and desert shrub-dominated areas, and Hawth's Analysis Tools 3 (Beyer 2004) within ArcView 3 (ESRI, Redlands, CA, USA) to calculate roughness from digital elevation models (DEM; WyGISC). We compared the proportion of suitable nesting habitat within 930 m of infrastructure and outside of the 930-m buffer but within the 5-km buffer to investigate if the proportion of suitable habitat in compared areas differed.

Overall Demographics.--We used nesting or spring locations to categorize all yearling females as treatment (i.e., within 930 m of infrastructure) or control individuals (Figure 2). Differences in nesting propensity, apparent nest success, early brood-rearing success, and overall brood-rearing success were investigated using Chi-square tests with continuity corrections (Dowdy and Wearden 1991). We established expected proportions from the control population (e.g., results suggest a difference between treatments and controls). The date of nest establishment, and annual and seasonal survival were compared between categories using 95% confidence interval overlap.

Natal Areas.--We compared nesting propensity and apparent nest success between natal treatment and control yearling females using Chi-square tests with continuity corrections (Dowdy and Wearden 1991). We determined expected nesting propensity and success rates from the control population. Distances from the natal nest to the yearling's nest, date of nest initiation, and annual and seasonal survival differences between treatment and control populations were compared using 95% confidence interval overlap.

To examine nest site selection of yearling females relative to where they were raised and the existence of infrastructure features of natural gas fields, we compared the proportion of yearlings with infrastructure in the natal nesting area (i.e., the area around the natal nest where a yearling female will usually select a nest location) that nested within and beyond 930 m of infrastructure using Chi-square tests with continuity corrections (Dowdy and Wearden 1991). We used all natal nesting areas with infrastructure present in the analysis. We estimated the expected number of nests per category (i.e., within or beyond 930 m of infrastructure) as the proportion of the total natal nesting area (i.e., all natal nesting areas with gas field infrastructure present combined) within 930 m of infrastructure.

Because of relatively small sample sizes and the possibility that single measures could disproportionately influence results, we identified influential observations and considered those when interpreting results. We performed statistical procedures with MINITAB 13.1 (Minitab Inc., State College, PA, USA). We estimated distance variables (km) using ArcGIS 9 (ESRI).

RESULTS

We radio-tagged 64 male and 76 female chicks (45 males and 39 females during fall 2004; 19 males and 37 females during fall 2005). Between capture and yearling status designation, 41 chicks died, 7 lost the radio-transmitter (based on field sign at retrieved transmitter location), and 6 were never found. Thirty-four male and 52 female radio-equipped chicks were available as yearlings at the beginning of the breeding season monitoring periods. Maternity was confirmed for 16 male and 17 female yearlings, and breeding-season data were collected on 15 males and 16 females with known maternity.

Because of sample size constraints, we chose to use conservative statistical approaches when comparing treatment and control groups of yearlings.

Yearling Male Comparisons

Overall Recruitment.--Leks that recruited fewer than expected males were significantly closer to producing well pads, and tended to be closer to main haul roads compared to leks that recruited the same number of males as expected. Generally, greater sage-grouse leks that recruited significantly less than expected numbers of males were closer to infrastructure features of natural gas fields than those that recruited equal to or significantly more males than expected. Leks that recruited more than expected males were consistently closer to infrastructure than those that recruited the same number of males as expected (Table 1; Figure 3).

Lek Establishment.--The proportion of radio-equipped yearling males that established on leks inside and outside the development boundaries (as designated by minimum convex polygons around producing well pads) of the natural gas field differed significantly from that expected assuming equal establishment probabilities for all leks ($\chi^2_1 = 4.54$; $P = 0.03$; Table 2). Yearling males establishing on leks within the interior (2) were less than expected (7.4), while numbers establishing on leks outside the development boundaries (23) were more than expected (17.6). The number of radio-equipped yearling males that established on leks outside development and categorized by distance to the development boundary did not differ from expected ($\chi^2_2 = 0.12$; $P = 0.94$; Table 2).

Mean date of establishment, lek tenacity, and annual survival of yearling males did not differ inside and outside gas fields (Table 2).

Natal Areas.--Lek tenacity of natal treatment and natal control yearling males did not differ. However, after removing a natal treatment male (e.g., male reared in an area with infrastructure of natural gas fields present) that was documented on a lek 2.5 times as often as any other treatment male, lek tenacity of treatment males (9.3 days) was significantly less than control males (22.8 days; Table 3). Annual survival of natal treatment yearling males (52.5%) was significantly lower than natal control yearling males (100%; Table 3). Additionally, although not significantly different ($\chi^2_1 = 1.53$; $P = 0.22$), the estimated probability of natal treatment yearling males establishing on a lek was half that of natal control yearling males; 7 of 7 control yearling males and 4 of 8 treatment yearling males established breeding territories. The number of different leks visited during the breeding season, distance from natal nest-to-established lek, dates-of-establishment, and seasonal survival probabilities did not differ between natal treatment and control yearling males (Table 3).

Yearling Female Comparisons

General Habitat Selection.--The proportion of radio-equipped yearling females that selected nest locations within 930 m of an infrastructure feature of the natural gas fields and those nesting outside the 930-m buffer differed significantly from that expected assuming spatially proportional selection of nest locations ($\chi^2_1 = 4.10$; $P = 0.04$). The number of yearling female nests located within 930 m of infrastructure (6) was less than expected (11.5), while nest numbers located outside the buffer (19) were more than expected (13.5). The proportions of area assessed to be suitable nesting habitat within (75.1%) and outside (80.9%) the 930-m buffer were similar.

Overall Demographics.--Nesting propensity, apparent nest success, early brood-rearing success, and overall brood-rearing success did not differ between treatment (i.e., nesting within 930 m of gas field infrastructure) and control individuals ($\chi^2_1 < 0.12$; $P > 0.72$; Table 4). Date of nest establishment and annual survival were not related to nest location treatment status (Table 4).

Natal Areas.--Annual survival of natal treatment yearling females (69.4%) was significantly lower than natal control yearling females (100%; Table 5). Nesting propensity and nest success probabilities were not related to natal area ($\chi^2_1 < 0.13$; $P > 0.71$; Table 5). Natal nest-to-yearling nest distances, nest initiation dates, and seasonal survival did not differ between natal treatment and control yearling females (Table 5).

The upper limit of the 95% confidence interval around the mean natal nest-to-yearling nest distances for natal control females suggested that a 4.0-km buffer around natal nesting locations represented the area around the natal nest where a yearling female typically selected a nest location (i.e., natal nesting area; Table 5). There was weak evidence that the proportion of natal yearling females reared near infrastructure that selected nest locations within 930 m of infrastructure and those that nested outside the 930-m buffer differed from that expected assuming spatially proportional selection of nest locations ($\chi^2_1 = 3.49$; $P = 0.06$). The number of yearling female nests located within 930 m of infrastructure (3) was less than expected (6.3), while nest numbers located outside the buffer (7) were more than expected (3.7).

DISCUSSION

Energy development impacts to greater sage-grouse populations typically result from a combination of demographic and behavioral responses (i.e., cumulative effects) affecting different age classes. Our results suggest that avoidance of infrastructure by breeding yearlings, decreased yearling survival, and reduced fecundity of yearling males contribute to abandonment of leks and nesting habitat within natural gas fields.

Greater sage-grouse leks situated near the infrastructure of natural gas fields recruited fewer males than expected. Because of lek tenacity by adult males (Patterson 1952, Wiley 1973, Gibson 1992), a majority of the birds recruited were probably yearling males. There was also a tendency for leks situated on the periphery of the fields to recruit a higher proportion of yearling males than those farther from disturbance, suggesting that yearling males avoid natural gas fields and move to the periphery of the fields when establishing breeding territories. Additionally, yearling males reared in areas with infrastructure features of natural gas fields were less likely to establish a breeding territory, did not occupy leks during the breeding period as tenaciously, and had lower annual survival than males reared in areas with no activities associated with natural gas fields. Dunn and Braun (1985) suggested that leks selected by yearling males were spatially associated to natal areas. Thus, decreased fecundity may be in response to anthropogenic activity encountered either as chicks, or in response to conditions encountered during inaugural breeding seasons. Regardless, natural gas development appeared to influence negatively both the breeding-season distribution and success of the yearling male population.

Greater sage-grouse yearling females generally avoided nesting within 930 m of the infrastructure of natural gas field. Yearling females with natural gas infrastructure present in their natal nesting area also generally avoided nesting within 930 m of infrastructure; this general avoidance results in the functional loss of at least the habitats within 930 m of infrastructure. However, distance from natal-nest to first-year-nest locations did not differ, suggesting that yearling females did not vacate natal areas but simply avoided nesting near infrastructure within natal areas. Holloran (2005) suggested that the eventual response of greater sage-grouse nesting populations will be avoidance of natural gas development, but the avoidance response would be driven by habitat selection of yearling females due to nesting-area fidelity of adult females. Further, Wiens et al. (1986) suggested that site fidelity in breeding birds could delay population response to habitat changes, and that a clear response required that most site-tenacious individuals be dead. Fidelity of adults to nesting areas and fidelity of yearlings to natal areas may delay a population-level avoidance response, and may explain time lags between the development of gas fields and the abandonment of gas fields by greater sage-grouse found in previous studies (Holloran 2005, Walker et al. 2007).

Yearling females reared in areas with natural gas infrastructure had lower annual survival rates than females reared in areas without infrastructure. However, we detected no negative effects of natal-area condition on productivity. These results are similar to analyses investigating population growth differences between anthropogenically disturbed and undisturbed populations that attributed differences in population growth to lower female annual survival in impacted populations (Hagen 2003, Holloran 2005). Natural gas development appeared to influence negatively both the nesting-season distribution and annual survival of the yearling female population.

MANAGEMENT IMPLICATIONS

The results from this study suggest that dispersal of yearling greater sage-grouse from the infrastructure of natural gas fields and demographic impacts are contributing to abandonment of leks and nesting habitat within natural gas fields. This implies that developing a natural gas field reduces the extent of the landscape used by sage-grouse populations. Sage-grouse populations typically inhabit large, unbroken expanses of sagebrush and are characterized as a landscape-scale species (Patterson 1952, Connelly et al. 2004). Thus, preserving sagebrush-dominated areas within an impacted landscape as refugia may be necessary to maintain remnant sage-grouse populations. To ensure that viable populations are conserved, we recommend managers rely on seasonal habitat selection and movement information collected from individual sage-grouse residing in proposed refugia to determine appropriate refugia size and configuration. Additionally, if impacts continue through the gas field production phases as suggested by Aldridge and Brigham (2003) and Walker et al. (2007), refugia will have to be maintained until developed areas are re-occupied by sustainable sage-grouse populations (gas well life-expectancy estimated at 25 to 40 years for the types of formations encountered in the PAPA; Wyoming Oil and Gas Conservation Commission, personal communication 2005).

Dispersal corridors may be needed to ensure the maintenance of the genetic diversity of sage-grouse populations potentially isolated into refugia, and to allow for immigration if a stochastic natural event (i.e., drought, fire, disease outbreak) eliminates a protected population. Sage-grouse can disperse long distances between seasonal ranges (Connelly et al. 2000*b*), and are physically capable of traversing natural gas fields. However, because of strong adult fidelity to breeding sites (Patterson 1952, Wiley 1973, Gibson 1992, Fischer et al. 1993, Schroeder and Robb 2003, Holloran and Anderson 2005) and the propensity of yearling females to nest near natal areas, large-scale movements of individuals does not necessarily equate to the dispersal of genetic material nor the functional immigration of individuals. If genetic diversity is maintained through the dispersal of yearling males, and yearlings tend to establish breeding territories on leks near natal areas, the abandonment of leks situated between distinct population segments may genetically isolate those segments. We recommend research investigating the mechanisms responsible for the dispersal of greater sage-grouse genetic information throughout a landscape.

Sage-grouse survival and fecundity have been linked to sagebrush-steppe habitat quality, and the dependence of the species on sagebrush through all seasonal periods has been well documented (see Connelly et al. 2004 for review). Sagebrush habitat enhancements typically entail manipulation of shrub overstories in an attempt to increase herbaceous understories and improve brood survival (e.g., prescribed fire, herbicide application). However, no research to date has shown a positive response of sage-grouse populations to sagebrush treatment (Wallestad 1975, Martin 1990, Fischer et al. 1996). In fact, large-scale shrub manipulations, particularly in winter, nesting, or year-round habitats may result in population declines (Swenson et al. 1987, Connelly et al. 2000*a*, Nelle et al. 2000). We recommend that land managers exercise extreme caution in applying shrub manipulations (Connelly et al. 2000*b*, Dahlgren et al. 2006), and focus instead on management options that enhance or restore herbaceous understories within sagebrush stands (e.g., via livestock grazing management [Beck and Mitchell 2000]). The establishment of interconnected refugia managed to sustain robust populations will help ensure that greater sage-grouse are present to re-colonize natural gas fields following reclamation.

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LITERATURE CITED

Aldridge, C. L., and M. S. Boyce. 2007. Linking occurrence and fitness to persistence: habitat-based approach for endangered Greater Sage-Grouse. Ecological Applications 117:508-526.

Aldridge, C. L., and R. M. Brigham. 2003. Distribution, abundance, and status of the greater sage-grouse, *Centrocercus urophasianus*, in Canada. Canadian Field-Naturalist 117:25-34.

Beck, J. L., and D. L. Mitchell. 2000. Influences of livestock grazing on sage grouse habitat. Wildlife Society Bulletin 28:993-1002.

Beyer, H. L. 2004. Hawth's Analysis Tools for ArcGIS. Available at http://www.spatialecology.com/htools.

Braun, C. E. 1986. Changes in sage grouse lek counts with advent of surface coal mining. Proceedings: Issues and Technology in the Management of Impacted Western Wildlife, Thorne Ecological Institute 2:227-231.

Braun, C. E. 1998. Sage grouse declines in western North America: what are the problems? Proceedings of the Western Association of State Fish and Wildlife Agencies 78:139-156.

Braun, C. E., O. O. Oedekoven, and C. L. Aldridge. 2002. Oil and gas development in western North America: effects on sagebrush steppe avifauna with particular emphasis on sage grouse. Transactions North American Wildlife and Natural Resources Conference 67:337-349.

Bureau of Land Management. 2000. Record of decision, environmental impact statement for the Pinedale anticline oil and gas exploration and development project Sublette County, Wyoming. United States Bureau of Land Management, Pinedale, WY, USA.

Caccamise, D. F., and R. S. Hedin. 1985. An aerodynamic basis for selecting transmitter loads in birds. Wilson Bulletin 97:306-318.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of Greater Sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, WY, USA.

Connelly, J. W., K. P. Reese, R. A. Fischer, and W. L. Wakkinen. 2000*a*. Response of sage grouse breeding population to fire in southeastern Idaho. Wildlife Society Bulletin 28:90-95.

Connelly, J. W., K. P. Reese, and M. A. Schroeder. 2003. Monitoring of greater sage-grouse habitats and populations. College of Natural Resources Experiment Station publication No. 979, University of Idaho, Moscow, ID, USA.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000*b*. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:1-19.

Dahlgren, D. K., R. Chi, and T. A. Messmer. 2006. Greater sage-grouse response to sagebrush management in Utah. Wildlife Society Bulletin 34:975-985.

Doherty, K. E., D. E. Naugle, B. L. Walker, and J. M. Graham. 2008. Greater sage-grouse winter habitat selection and energy development. Journal Wildlife Management 72:In press.

Dowdy, S. M., and S. Wearden. 1991. Statistics for research second edition. John Wiley & Sons, Inc., New York, NY, USA.

Drut, M. S., J. A. Crawford, and M. A. Gregg. 1994. Brood habitat use by sage grouse in Oregon. Great Basin Naturalist 54:170-176.

Dunn, P.O., and C.E. Braun. 1985. Natal dispersal and lek fidelity of sage grouse. Auk 102:621-627.

Eng, R. L. 1955. A method for obtaining sage grouse age and sex ratios from wings. Journal Wildlife Management 19:267-272.

Fischer, R. A., A. D. Apa, W. L. Wakkinen, and K. P. Reese. 1993. Nesting-area fidelity of sage grouse in southeastern Idaho. Condor 95:1038-1041.

Fischer, R. A., K. P. Reese, and J. W. Connelly. 1996. An investigation of fire effects within xeric sage grouse brood habitat. Journal Range Management 49:194-198.

Frantz, A. C., L. C. Pope, P. J. Carpenter, T. J. Roper, G. J. Wilson, R. J. Delahay, and T. Burke. 2003. Reliable microsatellite genotyping of the Eurasian badger (*Meles meles*) using faecal DNA. Molecular Ecology 12:1649-1661.

Gibson, R. M. 1992. Lek formation in sage grouse: the effect of female choice on male territorial settlement. Animal Behaviour 43:443-450.

Giesen, K. M., T. J. Schoenberg, and C. E. Braun. 1982. Methods for trapping sage grouse in Colorado. Wildlife Society Bulletin 10:224-231.

Hagen, C. A. 2003. A demographic analysis of Lesser Prairie-Chicken populations in southwestern Kansas: survival, population viability, and habitat use. PhD Dissertation, Kansas State University, Manhattan, KS, USA.

Hawk, D., K. Sargeant, and S. Sharpe. 2004. Wyoming Game and Fish laboratory forensic procedure manual and molecular diagnostic protocols for fish health. Wyoming Game and Fish Department, Laramie, WY, USA.

Holloran, M. J. 2005. Greater sage-grouse (*Centrocercus urophasianus*) population response to natural gas field development in western Wyoming. PhD Dissertation, University of Wyoming, Laramie, WY, USA.

Holloran, M. J., and S. H. Anderson. 2004. Greater sage-grouse population response to natural gas development in western Wyoming: are regional populations affected by relatively localized disturbances? Transactions North American Wildlife and Natural Resources Conference 70:In press.

Holloran, M. J., and S. H. Anderson. 2005. Spatial distribution of greater sage-grouse nests in relatively contiguous sagebrush habitats. Condor 107:742-752.

Jensen, B. M. 2006. Migration, transition range and landscape use by greater sage-grouse (*Centrocercus urophasianus*). MS Thesis, University of Wyoming, Laramie, WY, USA.

Kenward, R. 1987. Wildlife Radio Tagging: Equipment, Field Techniques and Data Analysis. Academic Press, London.

Knick, S. T., D. S. Dobkin, J. T. Rotenberry, M. A. Schroeder, W. M. Vander Haegen, and C. Van Riper III. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. Condor 105:611-634.

Lyon, A. G. 2000. The potential effects of natural gas development on sage grouse near Pinedale, Wyoming. MS Thesis, University of Wyoming, Laramie, WY, USA.

Lyon, A. G., and S. H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin 31:486-491.

Marshall, T. C., J. Slate, L. E. B. Kruuk, and J. M. Pemberton. 1998. Statistical confidence for likelihood-based paternity inference in natural populations. Molecular Ecology 7:639-655.

Martin, R. C. 1990. Sage grouse responses to wildfires in spring and summer habitats. MS Thesis, University of Idaho, Moscow, ID, USA.

Nelle, P. J., K. P. Reese, and J. W. Connelly. 2000. Long-term effect of fire on sage grouse nesting and brood-rearing habitats in the Upper Snake River Plain. Journal Range Management 53-586-591.

Oyler-McCance, S. J. 1999. Genetic and habitat factors underlying conservation strategies for Gunnison sage-grouse. PhD Dissertation, Colorado State University, Fort Collins, CO, USA.

Patterson, R. L. 1952. The Sage grouse in Wyoming. Sage Books, Inc., Denver, CO, USA.

Pitman, J. C., C. A. Hagen, R. J. Robel, T. M. Loughin, R. D. Applegate. 2005. Location and success of lesser prairie-chicken nests in relation to vegetation and human disturbance. Journal of Wildlife Management 69:1259-1269.

Pollock, K. H., S. R. Winterstein, C. M. Bunck, and P. D. Curtis. 1989. Survival analysis in telemetry studies: The staggered entry design. Journal of Wildlife Management 53:7-15.

Remington, T. E. and C. E. Braun. 1991. How surface coal mining affects sage grouse, North Park, Colorado. Proceedings, Issues and Technology in Management of Impacted Western Wildlife. Thorne Ecological Institute 5:128-132.

Robel, R. J., J. A. Harrington Jr., C. A. Hagen, J. C. Pitman, and R. R. Reker. 2004. Effect of energy development and human activity on the use of sand sagebrush habitat by lesser prairie-chickens in southwestern Kansas. Transactions of the North American Natural Resources Conference 69:251-266.

Schroeder, M. A., C. L. Aldridge, A. D. Apa, J. R. Bohne, C. E. Braun, S. D. Bunnell, J. W. Connelly, P. A. Deibert, S. C. Gardner, M. A. Hilliard, G. D. Kobriger, S. M. McAdam, C. W. McCarthy, J. J. McCarthy, D. L. Mitchell, E. V. Rickerson, and S. J. Stiver. 2004. Distribution of Sage-grouse in North America. Condor 106:363-376.

Schroeder, M. A., and L. A. Robb. 2003. Fidelity of Greater Sage-Grouse *Centrocercus urophasianus* to breeding areas in a fragmented landscape. Wildlife Biology 9:291-299.

Schroeder, M. A., J. R. Young, and C. E. Braun. 1999. Sage grouse (*Centrocercus urophasianus*). Pages 1-28 *in* A. Pool and F. Gill [eds.], The Birds of North America, No. 425. The Birds of North America, Inc. Philadelphia, PA, USA.

Slater, S. J. 2003. Sage-grouse (*Centrocercus urophasianus*) use of different-aged burns and the effects of coyote control in southwestern Wyoming. MS Thesis, University of Wyoming, Laramie, WY, USA.

Swenson, J. E., C. A. Simmons, and C. D. Eustace. 1987. Decrease of sage grouse Centrocercus urophasianus after ploughing of sagebrush steppe. Biological Conservation 41:125-132.

Taylor, S. E., S. J. Oyler-McCance, and T. W. Quinn. 2003. Isolation and characterization of microsatellite loci in greater sage-grouse (*Centrocercus urophasianus*). Molecular Ecology Notes 3:262-264.

U.S. Fish and Wildlife Service. 2005. Endangered and threatened wildlife and plants; 12-month finding for petitions to list the greater sage-grouse as threatened or endangered; proposed rule. Federal Register 70:2244-2282.

Wakkinen, W. L., K. P. Reese, J. W. Connelly, and R. A. Fischer. 1992. An improved spotlighting technique for capturing sage grouse. Wildlife Society Bulletin 20:425-426.

Walker, B. L., K. E. Doherty, and D. E. Naugle. 2006 (*abstract only*). Spotlight counts: A new method for assessing chick survival in greater sage-grouse. University of Montana, Missoula, MT, USA.

Walker, B. L., D. E. Naugle, and K. E. Doherty. 2007. Greater sage-grouse habitat response to energy development and habitat loss. Journal Wildlife Management 51:In press.

Wallestad, R. O. 1975. Life history and habitat requirements of sage grouse in central Montana. Montana Fish and Game Department Technical Bulletin, Helena, MT, USA.

Wallestad, R. O., and D. B. Pyrah. 1974. Movement and nesting of sage grouse hens in central Montana. Journal Wildlife Management 38:630-633.

Walsh, D. P., G. C. White, T. E. Remington, and D. C. Bowden. 2004. Evaluation of the lek-count index for greater sage-grouse. Wildlife Society Bulletin 32:56-68.

Whatman. 2005. Available at www.whatman.com/products/?pageID=7.31.31.

Wiens, J. A., J. T. Rotenberry, and B. Van Horne. 1986. A lesson in the limitations of field experiments: shrubsteppe birds and habitat alteration. Ecology 67:365-376.

Wiley, R. H., Jr. 1973. Territoriality and non-random mating in sage grouse, *Centrocercus urophasianus*. Animal Behavior Monographs 6:85-169.

Zablan, M. A., C. E. Braun, and G. C. White. 2003. Estimation of greater sage-grouse survival in North Park, Colorado. Journal Wildlife Management 67:144-154.

Table 1. Mean (95% confidence interval [95% CI]) distance (km) from greater sage-grouse leks to natural gas field infrastructure in southwestern Wyoming, 2005-06. Leks were categorized as recruiting significantly less than, equal to, or more than expected numbers of males based on Chi-squared analyses of annual changes in the maximum number of males documented on leks during lek count procedures. Notice that leks recruiting fewer than expected males were those relatively close to gas field infrastructure and that leks recruiting more than expected males tended to be closer to development than those recruiting the same number of males as expected (suggesting yearling dispersal to the periphery of developing energy fields).

Relative Number of Males Recruited	n[a]	Distance Drill Rig		Distance Well Pad		Distance Haul Road	
		mean	95% CI	mean	95% CI	mean	95% CI
Less than expected	11	3.6	(2.4, 4.8)	1.7	(0.6, 2.7)	2.2	(1.0, 3.4)
Equal to expected	10	6.1	(4.0, 8.2)	5.0	(2.9, 7.1)	4.0	(3.2, 4.8)
More than expected	9	5.9	(3.8, 8.0)	4.0	(2.0, 5.9)	3.6	(2.0, 5.1)

[a] Total number of lek years.

Table 2. Establishment locations and breeding season demographics (means and 95% confidence intervals [95% CI]) of yearling male greater sage-grouse establishing breeding territories on leks categorized by lek-to-natural gas field development distances in southwestern Wyoming, 2005-06. Notice that leks situated within the development boundaries of the natural gas fields recruited fewer yearling males than expected.

Lek-to-Development Distance Catagories[a]	n[b]	Number of Males		Date of Establishment[e]		Lek Tenacity[f]		Annual Survival[g]	
		Established[c]	Expected[d]	mean	95% CI	mean	95% CI	mean	95% CI
Within Development	10	2	7.4	4/1	N/A[h]	37.5	(24.8, 50.2)		
Between 0 and 2 km of development	10	11	7.4	4/9	(4/3, 4/16)	21.9	(15.1, 28.7)	83.3	(64.8, 101.8)
Between 2 and 5 km of development	4	3	2.9	4/11	(3/23, 4/30)	27.3	(14.9, 39.7)		
More than 5 km from development	10	9	7.4	4/8	(4/2, 4/14)	19.6	(13.5, 25.6)	100	N/A[h]

[a] Development represents the area within a minimum convex polygon (Kenward 1987) around all producing well pads.
[b] Total number of lek years within buffer distance.
[c] Number of yearling males documented on a lek for at least 7 days.
[d] Number of yearling males expected on leks with the buffer based on the total number of lek years (i.e., leks equally available for establishment by yearling males).
[e] First date established yearling males documented on lek.
[f] Total number of days established yearling males documented on lek.
[g] Annual survival estimated using Kaplan-Meier estimator (Pollock et al. 1989); because of sample sizes, annual survival was not estimated for males establishing within the buffer, and males establishing on leks more than 2 km from development were combined.
[h] Standard error = 0.

Table 3. Mean (95% confidence interval [95% CI]) of breeding season demographics of yearling male greater sage-grouse reared within 1.9 km of natural gas field infrastructure (natal treatment males) compared to yearling males reared in areas with limited natural gas field infrastructure (natal control males) in southwestern Wyoming, 2005-06. Notice that lek tenacity and annual survival were lower for natal treatment yearling males.

	Natal Treatment Males			Natal control Males		
Male Demographic	n	mean	95% CI	n	mean	95% CI
Leks visited[a]	7	1.86	(1.3, 2.4)	7	1.57	(1.2, 2.0)
Natal nest-to-lek distance[b]	4	4.76	(1.2, 8.3)	7	7.38	(1.5, 13.3)
Natal nest-to-lek distance_2[c]	4	4.76	(1.2, 8.3)	6	5.02	(1.5, 8.5)
Date of establishment[d]	4	4/5	(3/28, 4/12)	6	4/11	(4/2, 4/19)
Lek tenacity[e]	4	14.5	(4.2, 24.8)	6	22.8	(15.1, 30.6)
Lek tenacity_2[f]	3	9.3	(6.5, 12.2)	6	22.8	(15.1, 30.6)
Annual survival[g]	8	52.5	(27.4, 77.6)	7	100	N/A[h]

[a] Total number of leks yearling males documented visiting.

[b] Straight line distance from natal nest to lek where yearling males established.

[c] One natal control male established on a lek 2.0 times as far from the natal nest than any other male; confidence intervals were re-computed after removing that observation.

[d] First date established yearling males documented on lek.

[e] Total number of days established yearling males documented on lek.

[f] One natal treatment male was documented on a lek 2.5 times as often as any other treatment male; confidence intervals were re-computed after removing that observation.

[g] Annual survival estimated using Kaplan-Meier estimator (Pollock et al. 1989).

[h] Standard error = 0.

Table 4. Breeding demographic probabilities and means (95% confidence intervals [95% CI]) of yearling female greater sage-grouse nesting within 930 m of natural gas field infrastructure (nesting treatment females) or nesting beyond 930 m of development (nesting control females) in southwestern Wyoming, 2005-06. Notice no differences in demographic probabilities.

	Nesting Treatment Females			Nesting Control Females		
Female Demographic	Available[a]	Documented[b]	95% CI	Available[a]	Documented[b]	95% CI
Nesting propensity[c]	12	8		31	22	
Nesting success[d]	8	4		21	10	
Early brood success[e]	4	3		9	8	
Overall brood success[f]	4	1		8	4	
Nest establishment date[g]	8	5/6	(5/1, 5/12)	21	5/7	(5/4, 5/9)
Annual survival (%)[h]	8	80.0	(55.2, 104.8)	21	61.8	(45.5, 78.1)

[a] Total number of yearling females available for the demographic (e.g., the denominator for estimating demographic probability).
[b] Total number of yearling females documented successful (e.g., the numerator).
[c] Number of females documented nesting versus the number monitored during the nesting season.
[d] Number of females hatching at least 1 egg versus the total number initiating a nest
[e] Number of successfully nesting females with at least 1 chick to 2 weeks post-hatch.
[f] Number of successfully nesting females with at least 1 chick 35 days or 10 weeks post-hatch (see methods).
[g] Date females first documented on nest.
[h] Annual survival estimated using Kaplan-Meier estimator (Pollock et al. 1989).

Table 5. Breeding demographic probabilities and means (95% confidence intervals [95% CI]) of yearling female greater sage-grouse reared within 1.9 km of natural gas field infrastructure (natal treatment females) compared to yearling females reared in areas with limited natural gas field infrastructure (natal control females) in southwestern Wyoming, 2005-06. Notice that annual survival of natal treatment yearling females was lower than natal control yearlings.

	Natal Treatment Females			Natal Control Females		
Female Demographic	Available[a]	Documented[b]	95% CI	Available[a]	Documented[b]	95% CI
Nesting propensity[c]	9	5		7	5	
Nesting success[d]	4	1		6	2	
Natal nest-to-yearling nest distance (km)[e]	5	3.33	(1.1, 5.6)	6	2.83	(1.6, 4.0)
Nest establishment date[f]	5	5/6	(5/1, 5/10)	6	5/8	(5/1, 5/16)
Annual survival (%)[g]	9	69.4	(44.4, 94.5)	7	100	N/A[h]

[a] Total number of yearling females available for the demographic (e.g., the denominator for estimating demographic probability).
[b] Total number of yearling females documented successful (e.g., the numerator).
[c] Number of females documented nesting versus the number monitored during the nesting season.
[d] Number of females hatching at least 1 egg versus the total number initiating a nest
[e] Straight line distance from natal nest to yearling female nest.
[f] Date females first documented on nest.
[g] Annual survival estimated using Kaplan-Meier estimator (Pollock et al. 1989).
[h] Standard error = 0.

Figure 1. Yearling greater sage-grouse study location in southwestern Wyoming, 2005-06. The figure illustrates producing well pads and main haul roads present during the breeding seasons of 2005 and 2006; well pads within 5 km of trapped leks are included.

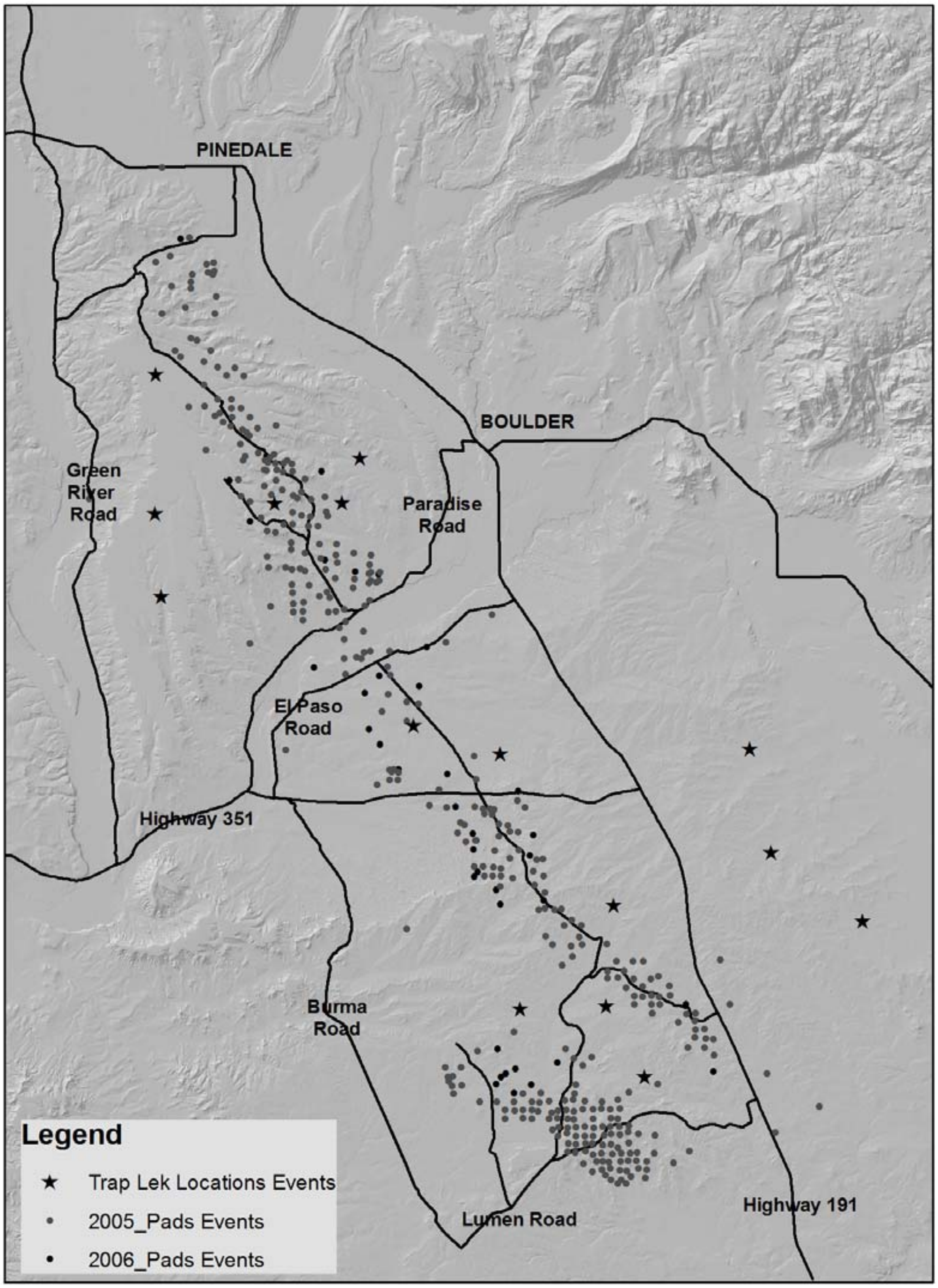


Figure 2. Yearling greater sage-grouse study location in southwestern Wyoming, 2005-06. The figure illustrates producing well pads and main haul roads present during the breeding seasons of 2005 and 2006; well pads within 5 km of trapped leks are included. Natural gas field infrastructure were buffered by 930 m (hatched areas) to determine areas of potential influence to nesting yearling females within the area of interest (i.e., within 5 km of trapped leks).

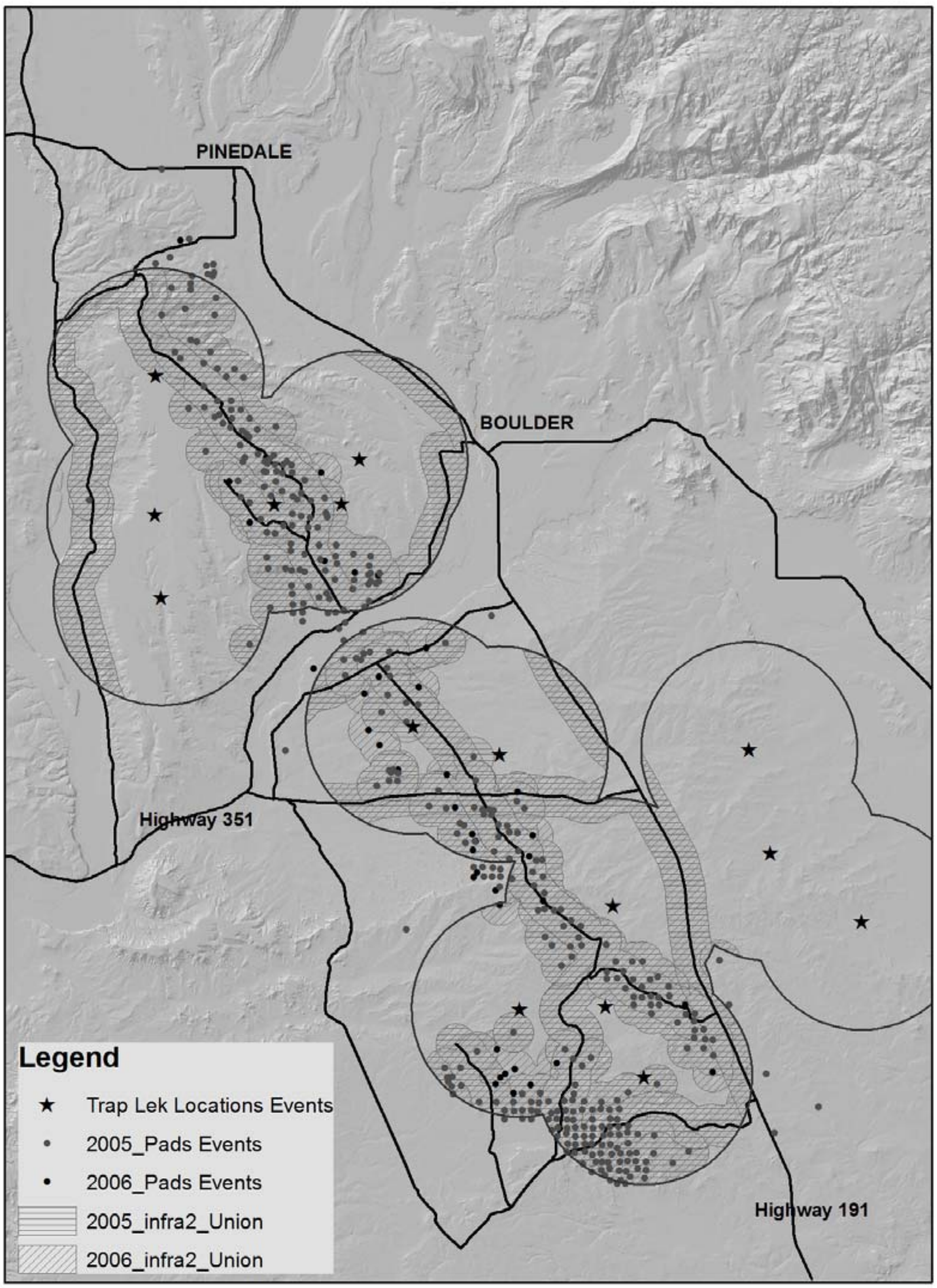
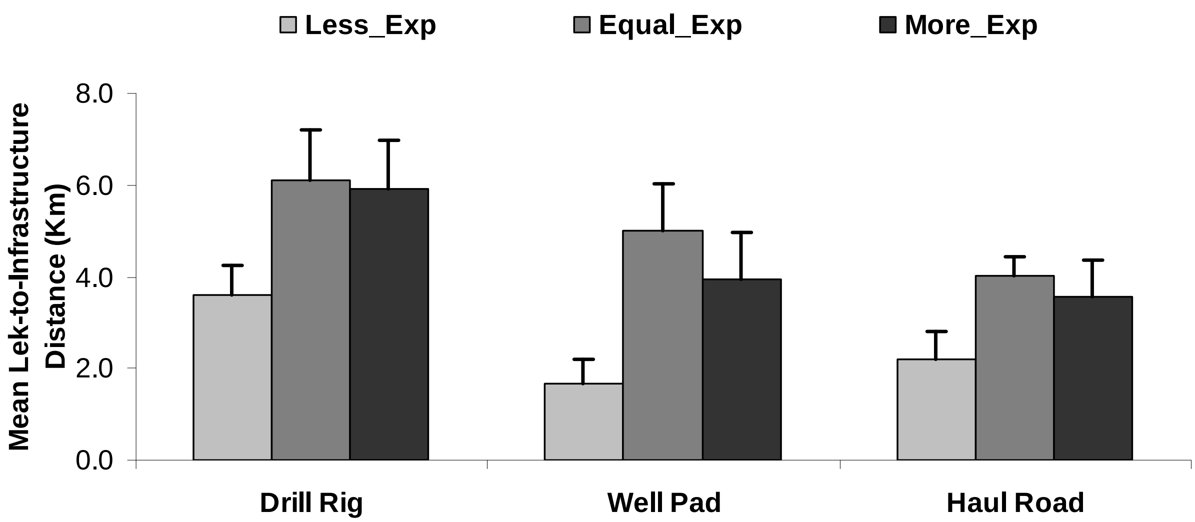


Figure 3. Mean (standard error) distances (km) from greater sage-grouse leks to natural gas field infrastructure in southwestern Wyoming, 2005-06. Leks were categorized as recruiting significantly less than, equal to, or more than expected numbers of males based on Chi-squared analyses of annual changes in the maximum number of males documented on leks during lek count procedures.



NESTING AND BROOD-REARING SUCCESS AND RESOURCE SELECTION OF GREATER SAGE-GROUSE IN NORTHWESTERN SOUTH DAKOTA

BY

NICHOLAS W. KACZOR

A thesis submitted in partial fulfillment of the requirements for the

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Wildlife and Fisheries Sciences

South Dakota State University

2008

NESTING AND BROOD-REARING SUCCESS AND RESOURCE SELECTION OF GREATER SAGE-GROUSE IN NORTHWESTERN SOUTH DAKOTA

This thesis is approved as a creditable and independent investigation by a candidate for the Master of Science degree and is acceptable for meeting the thesis requirements for this degree. Acceptance of this thesis does not imply that the conclusions reached by the candidate are necessarily the conclusions of the major department.

14 April 2008

Dr. Kent C. Jensen Date
Major Advisor

4/14/08

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I am forever indebted to Thomas Berdan for being a stand up technician both years of the research; his determination and willingness to capture chicks at night after tracking birds all day will not be forgotten. I also want to thank my other technicians, Sarah Nelson-Harrelson, Tom Juntti, and Ben Hauser for helping with the tedious habitat sampling. Other volunteers were Ryan Williamson, Heidi Jacobsen, Dawn Gardner, Ashly Steinke, Adam Giegle, and Rocky Ruzicka. Thank you to Bill Eastman, Rudy King, Tom Kirschenmann, Jerry Kobriger, Stan Kohn, Kent Undlin, Greg Wolbrink, John Wrede, and Tim Zachmeier for either lending an ATV, capturing chicks, helping with statistical analyses or any other aspect of the project. In addition, I want to thank Tony Apa for showing me my first sage-grouse, and the techniques to capture and radio-mark them. Two individuals who helped prepare me for this research were Bill Vodehnal and Greg Schenbeck. Thank you for taking a chance on me and providing me with my first work experience in the wildlife field.

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ABSTRACT

NESTING AND BROOD-REARING SUCCESS AND RESOURCE SELECTION OF GREATER SAGE-GROUSE IN NORTHWESTERN SOUTH DAKOTA

Nicholas W. Kaczor

May 2008

Understanding population dynamics and resource selection is crucial in developing wildlife resource management plans, particularly for sensitive species. Greater sage-grouse (*Centrocercus urophasianus*) populations have declined range-wide at a rate of 2% per year from 1965 to 2003. In South Dakota, populations have generally declined. Reasons for the decline are mostly attributed to human-induced factors such as sagebrush degradation and removal, improper range management practices, oil and gas exploration, and West Nile virus infection. Sage-grouse occupy habitats at the eastern edge of their range in western South Dakota. We conducted a 2-year study to investigate the nesting and brood-rearing ecology of sage-grouse in northwestern South Dakota.

Female sage-grouse were captured and radio-marked (n = 53) on traditional display grounds. Radio-marked hens were tracked to estimate nesting effort, nest success, and associated habitats. Nest initiation was 95.9%, with an overall nest success of 45.6 ± 5.3%. Hens selected habitats with greater sagebrush canopy cover and nest bowl visual obstruction compared to random sites. Nest success models developed in Program MARK indicated taller grass structures increased nest success.

Chick survivorship to seven weeks post hatch ranged from 31 to 43% over the two year period and recruitment of chicks into the breeding population (1 March) was estimated to be between 5 and 10%. Between 12 July and 31 September, West Nile virus accounted for 7 to 21% of the mortality incurred by chicks, however WNv reduced recruitment by 2 to 4%. Sage-grouse selected brood-rearing habitats that provided increased visual obstruction and bluegrass (*Poa spp.*) cover. More herbaceous vegetation at these sites may provide increased invertebrate abundance, which is necessary in the diets of sage-grouse chicks.

Management of sage-grouse nesting habitat on the eastern edge of their range should focus on increasing levels of sagebrush density and canopy cover while maintaining cover and height of grasses. We recommend that land managers maintain maximum grass heights of 26 cm. For brood-rearing sites, managers should maintain high vegetation biomass (visual obstruction) for protective cover and increased invertebrate abundance. We recommended that land managers strive to attain >10% chick recruitment into the breeding season.

TABLE OF CONTENTS

LIST OF TABLES

Table Page

LIST OF FIGURES

LIST OF APPENDICES

GENERAL INTRODUCTION

Greater sage-grouse (*Centrocercus urophasianus*) populations have declined range-wide at a rate of 2% per year from 1965 to 2003 (Connelly et al. 2004). These declines have been attributed to many factors, mostly human-induced (Connelly and Braun 1997). Factors for decline include, but are not limited to: sagebrush (*Artemisia spp.*) degradation and removal (Knick et al. 2003, Wisdom et al. 2005), livestock grazing (Beck and Mitchell 2000), fire (Baker 2006), construction of highways, fences, and power lines, (Braun 1998, Schroeder et al. 1999, Aldridge and Brigham 2001) oil and gas development (Lyon and Anderson 2003), and increased mortality due to West Nile virus infections (Naugle et al. 2005).

Further declines in sage-grouse populations are a concern to many stakeholders in the western United States landscape, as several petitions have been filed for sage-grouse to be listed under the Endangered Species Act (ESA) of 1973 (Connelly et al. 2004). Currently, Federal land management agencies are responsible for approximately 66% of the sagebrush landscape in the United States (Connelly et al. 2004). Federal agencies such as the Bureau of Land Management (BLM) and U.S. Forest Service (USFS) are directed by administrative policy to manage public lands for sustained multiple use under the Federal Land Policy and Management Act (1976), and Public Rangelands Improvement Act (1978). In addition, sage-grouse are a considered a sensitive species for the BLM and USFS. Listing of sage-grouse under the ESA could have major ramifications on the use and management of public lands in of the western United States (Knick et al. 2003).

It has been widely documented that sage-grouse are sagebrush obligates during winter and depend heavily upon it throughout their annual life cycle (Patterson 1952, Connelly and Braun 1997, Schroeder et al. 1999, Connelly et al. 2004, Moynahan et al. 2007). Sagebrush provides food resources, nesting cover, and protection from predators (Schroeder et al. 1999). Since the arrival of European settlers, sagebrush habitats have undergone numerous alterations and degradations (Patterson 1952). Sagebrush has been lost to tillage agricultural (Swenson et al. 1987), energy development (Braun 1998, Walker et al. 2007, Doherty et al. 2008), and urban expansion, reservoirs, and roads (Braun 1998, Aldridge and Brigham 2001). Furthermore, degradation and fragmentation of sagebrush has occurred from chemical and mechanical treatments of sagebrush, livestock grazing (Knick et al. 2003, Wisdom et al. 2005), construction of fences and powerlines (Braun 1998), and the introduction of invasive species (Knick et al. 2003).

Current guidelines for sage-grouse management (Connelly et al. 2000) are based on extensive studies in core sage-grouse range (e.g., Wyoming and Montana). These studies typically focused on varying aspects of sage-grouse ecology; particularly nesting and brood-rearing ecology. However, little research has been conducted on the eastern limit of sage-grouse distribution. Western South Dakota forms a transitional zone between the northern wheatgrass-needlegrass prairie that dominates most of the Dakotas and the big sagebrush plains of Wyoming (Johnson and Larson 1999). In South Dakota, sage-grouse are imperiled because of rarity or some factor(s) making them very vulnerable to extinction within the state (South Dakota Department of Game, Fish, and Parks 2006). Smith et al. (2004) reported steady declines in South Dakota sage-grouse populations since 1972 that were possibly the result of sagebrush removal through cultivation and herbicides (Smith et al. 2005). No study has been conducted in western South Dakota investigating sage-grouse nesting and brood-rearing success and associated habitats.

The objectives of this study were to (1) determine and quantify nesting and brood-rearing resource selection of radio-marked sage-grouse, (2) estimate nest success and evaluate cause and timing of nest failures, and (3) estimate chick survival and recruitment. This study will complement previous and concurrent research conducted on sage-grouse in the Dakotas, thus providing regional land managers with baseline ecology of sage-grouse. Furthermore, management recommendations produced from this research will aid in resource management plans and coordination efforts to enhance sage-grouse habitats.

This thesis is designed as two chapters dealing with the nesting and brood-rearing aspects of sage-grouse in western South Dakota. It is the intent to publish these papers in the Journal of Wildlife Management (JWM) or a similar type of peer-reviewed journal. Therefore, publication style will follow JWM guidelines unless otherwise noted. This research was a team approach, including multiple authors on publications so I have substituted the pronoun "I" for "We". Data will be archived at the U.S. Forest Service Rocky Mountain Research Station, Fort Collins, CO.

STUDY AREA

The study was conducted within a 3,500-km^2 area in Butte and Harding counties, South Dakota; Crook County, Wyoming; and Carter County, Montana (44°44'N to 45°20'N, 103°15'W to 104°21'W; Figure 1). Approximately 75% of the area was privately owned and we conducted research on 40 private ranches. The remaining 25% of the study area was managed by the United States Bureau of Land Management (BLM), and State of South Dakota School and Public Lands Division (SDSPL). The area is predominately used for grazing purposes although small grain production is evident. Open-pit mining for bentonite occurs at the south end of the study site on Pierre soils (Charles Berdan, BLM, Belle Fourche, South Dakota, personal communication).

Vegetation consists of short shrubs, mostly Wyoming big sagebrush (*Artemisia tridentata spp.*) and plains silver sagebrush (*A. cana spp.*). Other shrubs include broom snakeweed (*Gutierrezia sarothrae*), greasewood (*Sarcobatus vermiculatus*), and saltbushes (*Atriplex spp.*) (Johnson and Larson 1999). Common grasses include western wheatgrass (*Pascopyrum smithii*), Junegrass (*Koeleria macrantha*), bluegrass species (*Poa spp.*), green needle-grass (*Nassella viridula*), and Japanese brome (*Bromus japonicus*). Common forbs include western yarrow (*Achillea millefolium*), common dandelion (*Taraxacum officinale*), pepperweed (*Lepidium densiflorum*), and pennycress (*Thlaspi arvense*) (Johnson and Larson 1999).

Temperatures in summer (May-August) average 20.1° C but can reach up to 43.3°C (South Dakota State Climate Office 2007). Mean annual precipitation is 35.3 cm, with a majority occurring during the months of April through July (South Dakota State

Climate Office 2007). Elevation ranges from 840 – 1225 m above sea level with nearly level to moderately steep clayey soils over clay shale (Johnson 1976).

Common predators included red fox (*Vulpes vulpes*), coyote (*Canis latrans*), bobcat (*Lynx rufus*), badger (*Taxidea taxus*), raccoon (*Procyon lotor*), golden eagle (*Aquila chrysaetos*), ferruginous hawk (*Buteo regalis*), American crow (*Corvus brachyrhynchos*), long-tailed weasel (*Mustela frenata*), and red-tailed hawks (*Buteo jamaicensis*).

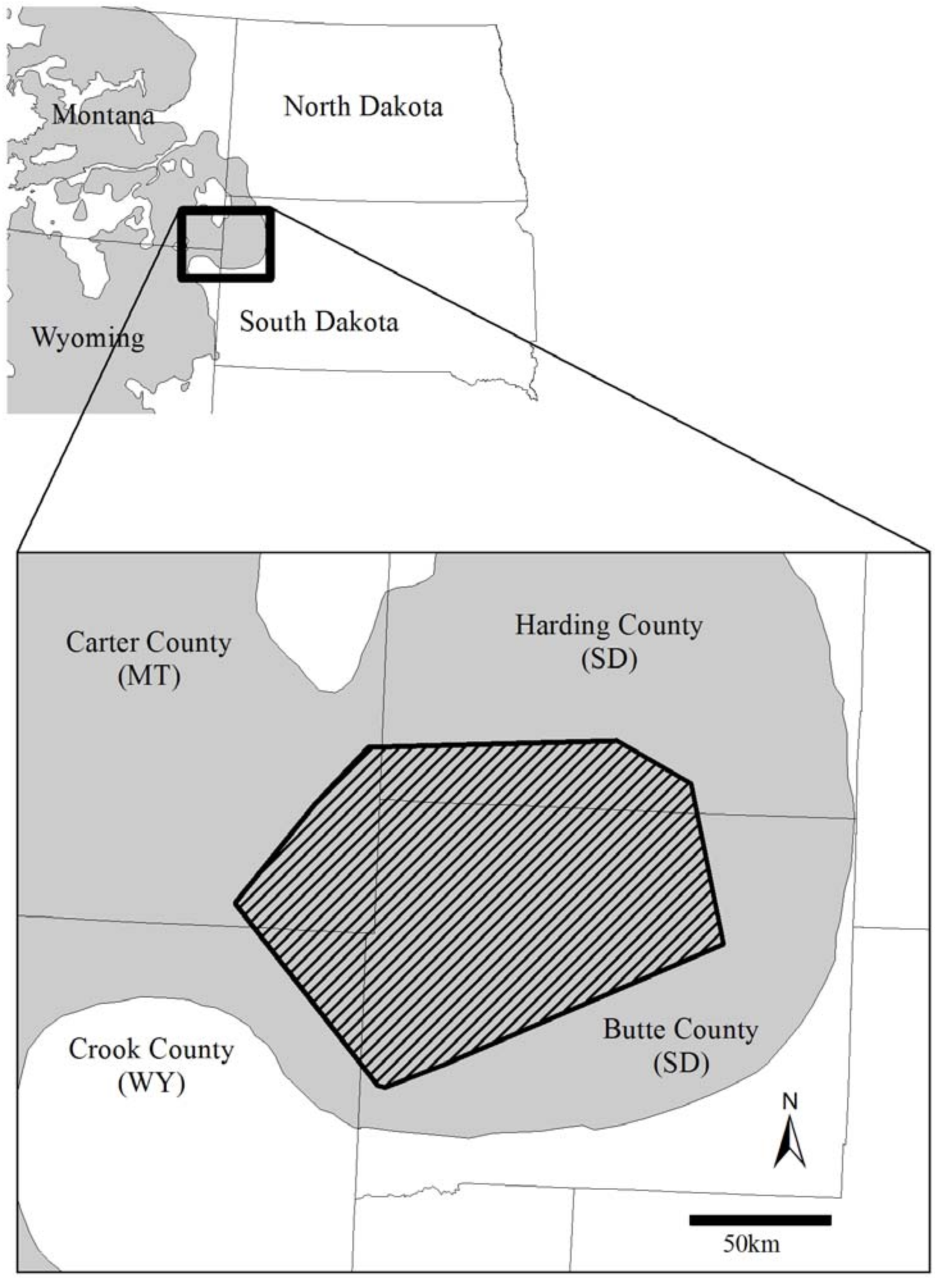

Figure 1. Study area of Butte, Carter, Crook, and Harding counties where we researched greater sage-grouse during 2006-2007. The dashed area encompasses all locations and the grayed area is current sage-grouse range (Schroeder et al. 2004).

LITERATURE CITED

Aldridge, C. L. and R. M. Brigham. 2001. Nesting and reproductive activities of greater sage-grouse in a declining northern fringe population. Condor 103:537-543.

Baker, W. L. 2006. Fire and restoration of sagebrush ecosystems. Wildlife Society Bulletin 34:177-185.

Beck, J. L., and D. L. Mitchell. 2000. Influences of livestock grazing on sage-grouse habitat. Wildlife Society Bulletin 28:993-1002.

Braun, C. E. 1998. Sage grouse declines in western North America: what are the problems? Proceedings of the Western Association State Fish and Wildlife Agencies 78:139-156.

Connelly, J. W., and C. E. Braun. 1997. Long-term changes in sage grouse *Centrocercus urophasianus* in western North America. Wildlife Biology 3:229-234.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Doherty, K. E., D. E. Naugle, B. L. Walker, and J. M. Graham. 2008. Greater sage-grouse winter habitat selection and energy development. Journal of Wildlife Management 72:187-195.

Johnson, J. R., and G. E. Larson. 1999. Grassland plants of South Dakota ant the northern Great Plains. South Dakota State University. Brookings, South Dakota, USA.

Johnson, P. R., 1976. Soil survey of Butte County, South Dakota. U.S. Department of Agriculture, South Dakota Agricultural Experiment Station, Brookings, South Dakota, USA.

Knick, S. T., D. S. Dobkin, J. T. Rotenberry, M. A. Schroeder, W. M. Vander Haegen, and C.Van Riper III. 2003. Teetering of the edge or too late: conservation and research issues for avifauna of sagebrush habitats. Condor 105:611-634.

Lyon, A. G., and S. H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin 31:486-491.

Moynahan, B. J., M. S. Lindberg, J. J. Rotella, and J. W. Thomas. 2007. Factors affecting nest survival of Greater sage-grouse in northcentral Montana. Journal of Wildlife Management 71:1773-1783.

Naugle, D. E., C. L. Aldridge, B. L. Walker, K. E. Doherty, M. R. Matchett, J. McIntosh, T. E. Cornish, and M. S. Boyce. 2005. West Nile virus and sage-grouse: What more have we learned? Wildlife Society Bulletin 33:616-623.

Patterson, R. L. 1952. The sage grouse in Wyoming. Sage Books, Inc., Denver, CO.

Schroeder, M. A., C. L. Aldridge, A. D. Apa, J. R. Bohne, C. E. Braun, S. D. Bunnell, J. W. Connelly, P. A. Deibert, S. C. Gardner, M. A. Hilliard, G. D. Kobriger, S. M. McAdam, C. W. McCarthy, J. J. McCarthy, D. L. Mitchell, E. V. Rickerson, and S. J. Stiver. 2004. Distribution of sage-grouse in North America. Condor 106:363-376.

Schroeder, M. A., J. R. Young, and C. E. Braun. 1999. Sage grouse (*Centrocercus urophasianus*). Account No. 425 *in* A. Poole. And F. Gill, editors. The Academy of Natural Sciences, Philadelphia, Pennsylvania, and The American Ornithologists' Union, Washington, D.C., USA.

Smith, J. T., L. D. Flake, K. F. Higgins, and G. D. Kobriger. 2004. History of greater sage-grouse in the Dakotas: Distribution and population trends. Prairie Naturalist 36:213-230.

Smith, J. T., L. D. Flake, K. F. Higgins, G. D. Kobriger, and C. G. Homer. 2005. Evaluating lek occupancy of greater sage-grouse in relation to landscape cultivation in the Dakotas. Western North American Naturalist 65:310-320.

South Dakota Department of Game, Fish, and Parks. 2006. South Dakota comprehensive wildlife conservation plan. South Dakota Dept. of Game, Fish, and Parks, Pierre, Wildlife Division Report 2006-08.

South Dakota State Climate Office. 2007. Office of the State Climatologist. http://climate.sdstate.edu. Accessed on 12 October 2007.

Swenson, J. E., C. A. Simmons, and C. D. Eustace. 1987. Decrease of sage grouse, *Centrocercus urophasianus*, after ploughing of sagebrush steppe. Biological Conservation 41:125-132.

Walker, B. L., D. E. Naugle, K. E. Doherty. 2007. Greater sage-grouse population response to energy development and habitat loss. Journal of Wildlife Management 71:2644-2654.

Wisdom, M. J., M. M. Rowland, and L. H. Suring, editors. 2005. Habitat threats in the sagebrush ecosystem: methods of regional assessment and applications in the Great Basin. Alliance Communications Group, Lawrence, Kansas, USA.

CHAPTER 1 – NESTING SUCCESS AND RESOURCE SELECTION OF GREATER SAGE-GROUSE IN NORTHWESTERN SOUTH DAKOTA.

INTRODUCTION

Greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse) were once distributed in parts of at least 12 states and 3 provinces, but have been extirpated from Nebraska and British Columbia (Schroeder et al. 2004). Furthermore, sage-grouse currently inhabit only 56% of their pre-settlement potential habitat (Schroeder et al. 2004) and populations have declined at an estimated rate of 2.0% per year from 1965 to 2003 (Connelly et al. 2004). Greater sage-grouse have become a sensitive species due to decreases in populations, (Aldridge and Brigham 2001, Connelly et al. 2004) and degradation of quality nesting habitat (Braun 1998, Connelly et al. 2004). Populations in South Dakota declined steadily from 1973 to 1997, and then recovered from 1997 to 2002 (Smith 2003, Connelly et al. 2004). However, in South Dakota, population indices from lek-counts were inconsistent over these time periods and meaningful assessments are lacking (Connelly et al. 2004). Nest fate and what factors determine nest success are of particular interest to biologists as it has been shown that nest success has the potential to limit population growth of sage-grouse (Schroeder 1997, Braun 1998, Schroeder et al. 1999, Dinsmore and Johnson 2005). Yet, information is lacking on the ecological requirements of nesting sage-grouse in western South Dakota. The objectives of this study were to develop an understanding on the nesting ecology, success, and resource selection of sage-grouse on the eastern edge of their range.

METHODS

Data Collection

Female Capture – We identified six active sage-grouse leks for which we had landowner cooperation for trapping. We captured female sage-grouse with large nets by spotlighting them from all-terrain vehicles between March 2006-2007 and mid-April 2006-2007 (Giesen et al. 1982). Females were weighed and equipped with a 22-g necklace-style transmitter, which were ~1.4% of mean female sage-grouse body mass and a life-expectancy of 434 days. Transmitters could be detected from approximately 2.0 to 5.0 km from the ground and were equipped with an 8-hour mortality switch. Females were classified as adults (≥2 yr old) or yearlings (<1 yr old) based upon primary wing feather characteristics (Eng 1955, Crunden 1963). The South Dakota State University Institutional Animal Care and Use Committee approved trapping and handling techniques, and study design (Approval #07-A032).

Locating and Monitoring Nests – We located radio-marked female sage-grouse twice each week during the breeding, laying, and incubation periods. In the event we could not locate an individual(s) from the ground, we searched the study-area from a fixed-wing aircraft to obtain an approximate location. Once a hen was believed to be incubating, we marked four coordinates approximately 15 m away in the four cardinal directions with a Global Positioning System (GPS) receiver (Garmin Ltd., Olathe, KS). We confirmed nest presence/absence during the subsequent visit. If a hen was present on the second visit, we flushed her to determine clutch size. This method did not cause nest abandonment as only 1 of 80 (1.3%) females abandoned their nests. Nests were checked approximately twice each week until nest fate was determined. Nests were considered successful if ≥1 egg hatched. We documented evidence (e.g., nest bowl disturbance, eggshell remains, etc.) at the nest site to estimate predator type (i.e., mammalian or avian) (Sargeant et al. 1998). Nest distances from nearest active display ground, renests, and prior nests were calculated by Hawth's Analysis Tool (Beyer 2004) in ArcMap 9.1 (ESRI, Inc., Redlands, CA.).

Habitat Measurements – We characterized vegetation at nest sites after the fate was determined. Four, 50-m transects were established radiating in the 4 cardinal directions from the nest bowl. A modified Robel pole (Robel et al. 1970, Benkobi et al. 2000) was used to estimate visual obstruction readings (VOR) and maximum grass height at 1-m intervals from 0 m to 5m (n = 21), and at 10-m intervals out to 50 m (n = 20). We estimated sagebrush (*A. tridentata spp.* and *A. cana spp.*) density and height at 10 m intervals (n = 80) using the point-centered-quarter method (Cottam and Curtis 1956). We added four, 5-m transects, radiating in the 4 ordinal directions from the nest bowl for vegetation cover measurements. Vegetation cover was estimated using a 0.10 m^2 quadrat (Daubenmire 1959) at 1-m intervals to 5 m (n = 44) and then alternating out to 30 m (n = 52). We recorded total cover, grass cover, forb cover, shrub cover, litter cover, bare ground, and individual shrub and grass species canopy cover. In addition, we measured an equal number of random sites within a 3 km buffer of capture leks to estimate resource selection. We entered the coordinates of the random sites into a GPS and navigated to the location, then located the center over the nearest sagebrush to the coordinate.

Data Analyses

Nesting Parameters – We used the multi-response permutation procedure (MRPP; Mielke and Berry 2001) to test the null hypothesis that there were no differences among weights, clutch size, nest initiation dates, nest site fidelity, and distances to display grounds between years and between ages of females. Chi-square goodness of fit test was used to test differences of nest initiation rates between years and between ages of females. For these analyses, results were considered significant at a critical value of $\alpha \leq 0.05$.

Habitat Measurements – Maximum grass height and VOR were summarized for each of the intervals and then averages were calculated for 0 to 5 m, 1 to 5 m, 10 to 50 m, and the site level (0 to 50 m). Sagebrush density and height was estimated from a maximum likelihood estimate (Pollard 1971) and summarized for the site. Canopy coverage values were recorded to mid-point values of categories for each species, or category. These were then summarized to an average for 0 to 5 m, 1 to 5 m, 6 to 30 m, and to the site (0 to 30 m). With over 100 variables in the data set, we then screened all variables using MRPP (Mielke and Berry 2001) to identify important variables between nest and random sites and between successful and failed nests (Boyce et al. 2002). A relaxed critical value of $\alpha \leq 0.15$ was used in the screening process to reduce the risk of excluding a potentially important variable.

Resource Selection – We identified 10 habitat variables (Table 1) from the screened variables along with a year effect to investigate sage-grouse nesting habitat preferences. Variables selected included: total cover, grass cover, sagebrush cover, litter cover, mean sagebrush height, maximum grass height, and visual obstruction all at the site level. In addition, grass height 0-5 m away from the nest bowl, visual obstruction at the nest bowl, and visual obstruction 1 m away from nest bowl were included in the data set. Year was considered a design variable in all candidate models. We used an information theoretic approach (Burnham and Anderson 2002) with nominal logistic regression to estimate the importance of various *a priori* and *post-hoc* exploratory models in SAS JMP (2005 SAS Institute Inc.). Due to a small sample size with respect to the number of parameters estimated, AIC*c* (Akaike's Information Criterion) was used being derived from our log-likelihood estimate (Burnham and Anderson 2002). Model strength was estimated using a receiver operation characteristic curve (ROC) with values between 0.7 and 0.8 considered as acceptable discrimination and values higher than 0.8 were considered excellent discrimination (Hosmer and Lemeshow 2000).

Nest Success – We used the nest survival module in program MARK (White and Burnham 1999, Dinsmore et al. 2002) to evaluate environmental and biological factors that might influence nest success. We standardized nesting dates among years by using the earliest location date for any year as the first day of the nesting season. We monitored nests over a 59-day period beginning 23 April and ending 20 June, which comprised 58 daily intervals of observations to be used in estimating daily survival rate (DSR) for the 27 day incubation period. We identified four variables from the screen process as having a potential impact on nest success which included: grass height at the site level, visual obstruction at the site level, litter cover at the site level, and 0 m forb cover (Table 2). These variables were combined with daily precipitation, daily minimum temperature, bird age, and year. We did not model nesting attempt because of a small number of renests (n = 10), or days into incubation because we could not accurately measure them. Daily weather variables were obtained from the nearest daily weather station located at Nisland, South Dakota, approximately 50 km from the center of the study area (South Dakota State Climate Office 2007).

We used an information theoretic approach (Burnham and Anderson 2002) to evaluate support for models of DSR and variables. We began by developing base models which included bird age, year, and constant survival. From these base models we further explored the degree to which habitat and weather variables improved model fit. We used back-transformed estimates of DSR (Dinsmore et al. 2002) to determine effect of variables on nesting success for the best supported model. We plotted DSR versus simulated values of variables to determine the effect of variables independently from one another. We estimated standard error of DSR using the delta method (Seber 1982).

RESULTS

Nesting Parameters

Trapping and Monitoring – We captured 53 female sage-grouse (25 adults and 28 yearlings) and fitted them with transmitters during the study, 29 individuals were included both years. Adults weighed (1664 g, range: 1492 – 1912 g) more (P <0.01) than yearlings (1524 g, range: 1332 – 1734 g), but there were no differences between years (P = 0.20). We found 80 nests (41 in 2006, and 39 in 2007) and 73 were included in nest survival analyses. Seven nests were excluded because either we did not collect vegetative measurements (n = 5), we felt we caused nest abandonment (n = 1), or were denied access to private land (n = 1).

Nest Initiation – Nest initiation rates (proportion of individuals initiating ≥1 nest) for all nests was 95.9% (Table 3) and did not differ between years (P = 0.09) or bird age (P = 0.89). Renest initiation rate was 28.6% (10/35) and did not differ between years (P = 0.67) or bird age (P = 0.24). Females were more likely to renest (P = 0.02) if their first nest was lost early into incubation with the number of first nest observation days being 7.9 ± 1.3 days for females that renested and 14.6 ± 1.8 days for females that did not renest.

Average date of nest initiation for first nests was 24 April ± 1.6 days (Table 4), with adults (≥2 years) initiating egg laying approximately 6.7 days earlier than yearlings (P = 0.02). No differences of nest initiation dates were detected between years for first nests (P = 0.27). Average hatch date for first nests was 31 May ± 1.5 days. Average renest initiation was approximately 15 days later (9 May ± 2.6 days) than first nests, with hatch date occurring 14 June ± 2.0 days. Clutch size varied between nesting attempts (first nests: 8.3 ± 0.2, renests: 6.4 ± 0.6, P < 0.01) (Table 4), but not between nest success (P = 0.83), bird age (P = 0.98), or year (P = 0.10).

Nest Location in Relation to Leks – Female sage-grouse visited multiple leks during the breeding season. One adult female in 2007 nested approximately 30.3 km from lek of capture. In 2006, successful nests were significantly closer to an active lek (P = 0.04) than failed nests (1.5 ± 0.3 km vs. 2.9 ± 0.5 km) (Figure 2), however there was no difference in 2007 (2.5 ± 0.5 km vs. 3.2 ± 0.7 km, P = 0.70), or when both years were combined (2.1 ± 0.3 km vs. 3.0 ± 0.4 km, P = 0.13). The distance that adults and yearlings nested to the nearest active lek did not differ significantly (2.2 ± 0.3 km vs.

3.3 ± 0.5 km, $P = 0.08$). Sixty-eight percent of nests were within 3 km of a documented active lek, and 97% of nests were within 7 km (Figure 3).

Nest site Fidelity – Mean distance between an individuals' nest in 2006 to its subsequent nest in 2007 was 1.08 ± 0.40 km ($n = 21$), but was highly variable (range: 0.07 km to 6.62 km). However, 76% of nests were within 0.70 km from a previous year's nest. There was no difference ($P = 0.65$) of nest site fidelity between adults and yearlings, or between nests that either failed or were successful the first year ($P = 0.47$). Mean distance between a failed first nest and subsequent renest was 1.85 ± 0.55 km ($n = 10$, range: 0.22 km – 5.12 km). Successful renests (0.95 ± 0.36 km, $n = 5$) were not significantly closer ($P = 0.17$) to first nests than failed renests (2.03 ± 0.91 km, $n = 5$).

Precipitation – During the months of March through June 2006, the study area received approximately 14 cm of precipitation (Figure 4). This was 33% less than the 58-year mean of 21 cm of precipitation. However, in 2007 the study area received approximately 22 cm, or 5% more precipitation than the 58-year mean for the same time period.

Resource Selection

Distributions of total cover, grass cover, grass height, visual obstruction and sagebrush height differed ($P < 0.05$) between nest sites in 2006 and 2007 (Table 1). There were also some year effects that were evident in the data for random sites, thus all logistic models included the design variable year (Table 5).

The best-approximating model (AIC*c* weight = 0.39) predicting nest sites from random sites included sagebrush canopy coverage at the site level and visual obstruction at the nest (Table 5). Both variables positively influenced the site selected for a nest (Table 6). Increasing sagebrush cover by 5% increased the odds of use 6.1 (95% CI: 5.5 – 6.9) times. Increasing visual obstruction at the nest by 2.54 cm increased the odds of use 3.2 (95% CI: 3.0 – 3.4) times (Table 6). A second model including sagebrush canopy coverage, visual obstruction at the nest, and average grass height within 5 m was also strongly supported (AIC*c* weight = 0.35). Model discrimination (ROC values) for the top two models was excellent at 0.93 for both models. Sagebrush canopy coverage and visual obstruction at the nest had the highest summed AIC*c* weights, both achieving values of 1.0. Although the combination of sagebrush canopy coverage and visual obstruction at the nest was the strongest model, there was little evidence for a model involving them individually; visual obstruction at the nest and sagebrush canopy coverage were 11.26 and 74.54 AIC*c* units higher, respectively.

Nest Success

Most nests were located under Wyoming big sagebrush (90%) or silver sagebrush (7%). One nest was located under the side of a large boulder, and another was in a dense stand of prairie cordgrass (*Spartina pectinata*). Breeding success rates (proportion of females hatching $\geq$ 1 egg in a season) averaged 47.9%. Egg hatchability (proportion of eggs hatching from successful clutches) averaged 78.3%. Most of the eggs that did not hatch were infertile.

Constant nest survival rates (similar to Mayfield 1975) were 45.6 ± 5.3%, but constant survival was a poor model. Four models were within 2 AIC*c* units of the top model. The best model with an AIC*c* weight of 0.23, included grass height and litter cover (Table 7) with a predicted nest success of 51.6 ± 6.3%. Grass height had a positive impact ($\beta = 0.15$ SE = 0.03) on nest success (Figures 5 & 6) and was present in all of the models considered. In contrast, litter cover negatively ($\beta = -0.08$ SE = 0.03) influenced nest success (Figures 6 & 7), but was also present in all of models considered.

The second-ranked model (AIC*c* weight = 0.15) included grass height, litter, daily precipitation, and a 1-day lag effect of precipitation. Although, daily precipitation had a positive influence on nest success ($\beta = 29.45$ SE = 40.35), and the 1-day lag effect negatively influenced nest success ($\beta = -1.89$ SE = 0.77), neither variable improved the top model and were only present due to being combined with grass height and litter. The third and fourth ranked models included daily precipitation, and bird age, respectively, but they were also combined with grass height and litter. Nest success varied 14.8% between years (37.7 ± 7.3% in 2006 compared to 52.5 ± 7.2% in 2007). However, adding a year affect to the top model did not improve model fit.

DISCUSSION

Nesting Parameters

Nest Initiation – Nest initiation rates for sage-grouse are generally believed to be lower compared to other prairie grouse species (Bergerud 1988). However, Schroeder et al. (1999) suggested that nesting attempts from telemetry based studies are probably under-represented in the literature, as follicular development indicated that at least 90.4% of females laid eggs the prior spring in three different studies. Our estimates of nest initiation in 2006 were probably influenced by a snow storm in late April (Figure 4) that hampered our tracking efforts during which we might have missed some nests. After the storm we observed several "dumped" eggs suggesting that during the storm some individual females were unable to locate their nests and expelled those eggs. Nonetheless, nest initiation rates were high in this study relative to range-wide estimates (Connelly et al. 2004).

Females in our study were approximately 125 g greater than the average for 8 other studies (i.e., adults – 1525 g, yearlings – 1413 g, Schroeder et al. 1999). Heavier eastern wild turkey females (*Meleagris gallopavo silvestris*) were more likely to breed than lighter females (Porter et al. 1983), as were yearling Merriam's turkeys (*M. g. merriami*) (Hoffman et al. 1996). Sage-grouse exhibit considerable temporal variation in nest initiation rates (Moynahan et al. 2007) which may be related to nutrition during the breeding season (Hungerford 1964, Barnett and Crawford 1994).

Renest rates in sage-grouse are highly variable from 0 to 87% and are likely linked to environmental effects and habitat quality (Schroeder 1997, Moynahan et al. 2007). Low renesting rates may also be related to the relatively low productivity in these arid and semiarid environments as habitat productivity/quality has been suggested to regulate nesting and renesting in wild turkeys (Rumble and Hodorff 1993, Hoffman et al. 1996, Rumble et al. 2003). Moynahan et al. (2007) found no renest initiation for sage-grouse in dry years with little vegetative growth. Only 9.5% of hens renested in a population in North Dakota (Herman-Brunson 2007). Our observations suggest that hens that incubated nests for shorter periods were more likely to renest than hens that incubated longer. Other populations of sage-grouse on the edge of the range also showed an inverse relation between length of incubation and renesting (Aldridge and Brigham 2001, Herman-Brunson 2007).

It has been suggested that sage-grouse nest later in more northern latitudes (Peterson 1980). South Dakota is further south than Washington and North Dakota, but had later hatch dates (Schroeder 1997, Herman-Brunson 2007), suggesting other variables (e.g., habitat, weather) may influence sage-grouse nesting chronology. Furthermore, hatch dates in South Dakota were comparable to what was reported for a northern sage-grouse population in Alberta (Aldridge and Brigham 2001)

We predicted age-specific variations in clutch size (Wallestad and Pyrah 1974, Peterson 1980, Moynahan et al. 2007) as adult females were significantly heavier than yearlings entering the breeding season. However, that was not observed in this study, or by Schroeder (1997), and Herman-Brunson (2007). Clutch size was lower for renests which was expected as female grouse expend substantial endogenous body reserves during the initial nesting attempt (Naylor and Bendell 1989).

Nest Location in Relation to Leks – Leks are the focal points of breeding and nesting conservation for non-migratory populations of sage-grouse (Connelly et al. 2000). Populations in South Dakota are believed to be non-migratory and contiguous with North Dakota and Montana populations (McCarthy and Kobriger 2005). It has been suggested that in areas with uniformly distributed habitats around leks, habitat conservation be implemented within a 3.2 km buffer (Connelly et al. 2000). However, Herman-Brunson et al. (*in review*) recommended a 5 km buffer to limit energy development and grazing activities during the nesting period. A 5 km buffer would encompass 82% of nests in our study.

Nest site Fidelity – Sage-grouse, along with other grouse species, demonstrate fidelity in nesting areas from year to year (Fischer et al. 1993, Schroeder and Robb 2003). However, sage-grouse typically do not exhibit as strong of fidelity as other grouse, but usually 84% of nests are <3 km from a previous year's (Schroeder and Robb 2003). Seventy-six percent of nests in our study were within 0.70 km of the prior year's nest. Our results illustrate that sage-grouse in South Dakota may show more fidelity to nesting areas compared to other edge populations, which may be related to the availability of suitable nest areas around leks.

Fidelity to nesting areas may be advantageous as hens are able to maximize use of productive habitats and minimize the risk of predation (Greenwood and Harvey 1982). However, fidelity may lead to decreased productivity if sage-grouse hens occupy sink habitats (Aldridge and Boyce 2007), or it may indicate that the appropriate habitat is limited and clumped in distribution. Predators can key in on high densities of nests, increasing predation rates (e.g., Lariviére and Messier 1998). If predators are able to recognize high densities of sage-grouse nest locations due to fidelity, increased predation could occur.

Resource Selection

Sage-grouse in South Dakota selected nest sites with higher sagebrush cover and placed their nests beneath sagebrush plants with greater horizontal cover (VOR) than random sites. In North Dakota, shrub density and nest-bowl VOR were also important predictors of sage-grouse nests (Herman-Brunson 2007).

Connelly et al. (2000) recommended 15-25% sagebrush canopy coverage for nesting sage-grouse. Meta-analysis (Hagen et al. 2007) confirmed mean sagebrush canopy coverage at sage-grouse nest sites was 21.51%. In South Dakota, sage-grouse selected the best of what was available, but that was less than the optimum. In contrast to sagebrush, grass structure in South Dakota exceeds both management recommendations (Connelly et al. 2000) and range-wide averages (Hagen et al. 2007). Western South Dakota forms a transition zone between the northern wheatgrass-needlegrass prairie that dominates most of the Dakotas and the big sagebrush plains of Wyoming (Johnson and Larson 1999). Thus, while South Dakota may have sub-optimal sagebrush cover for sage-grouse, the grass structure may be compensating the sagebrush component. However, grass structure is highly correlated with annual precipitation, and in periods of drought may not provide the necessary protection for sage-grouse nests. Poor rangeland management practices such as overgrazing will reduce grass structure which could have detrimental affects on sage-grouse populations.

Nest Success

Sage-grouse nest success varies widely across the range (Gregg 1991, Chi 2004), and is generally believed to be related to habitat conditions (Wallestad and Pyrah 1974, Connelly et al. 1991, Aldridge and Brigham 2002, Hagen et al. 2007). Our estimate of nest success was typical of other sage-grouse studies (48%, Connelly et al. 2004), despite the fact that available sagebrush canopy coverage was less than other areas. Grass height in our study had a substantial impact on nest success (Figure 5) and probably provides the structural component necessary for nests. Successful nests in our study had taller grass structures than both failed nests and random sites, with failed nests being more comparable to random sites; this was also documented in Oregon (Gregg et al. 1994). Taller live and residual grass surrounding nests also increased nest success in Alberta (Aldridge and Brigham 2002), and was suggested to provide ample nest concealment in both sagebrush and non-sagebrush overstories in Washington (Sveum et al. 1998). Although litter cover entered our models as being an important predictive variable for nest success, the impact litter actually has on nest success is unknown. Litter could be considered as a measure of the prior year's herbaceous growth by being lower following less productive seasons, but it could also be lower after intensive grazing pressure (Hart et al. 1988, Naeth et al. 1991).

MANAGEMENT IMPLICATIONS

If sage-grouse populations continue to decrease and/or maintain sensitive status, sagebrush conservation and enhancement should be top priority for land management agencies to enable sage-grouse persistence in western South Dakota. Management for greater grass cover and height, reduced conversion to tillage agricultural, and minimizing habitat fragmentation such as energy development should be encouraged. Little information is known about the direct impacts livestock grazing has on sage-grouse habitats (Beck and Mitchell 2000) but it may be the least expensive practice to restore degraded sagebrush steppe (Braun 2006, Woodward 2006). Grazing by domestic sheep

(*Ovis aries*) has effectively controlled sagebrush (Baker et al. 1976) which could reduce sagebrush cover further in South Dakota.

Range management practices that could increase sagebrush and grass cover and height might include: rest-rotation grazing, where the rested pasture in not grazed until early July to allow for undisturbed nesting, or reduced grazing intensities and/or season of use to reduce impact on sagebrush and grass growth (Adams et al. 2004). Land managers should attempt to leave or maintain maximum grass heights ≥ 26 cm, the inflection point for 50% nest success. In addition, annual grazing utilization should not exceed 35% in order to improve rangeland conditions, particularly sagebrush cover (Holechek et al. 1999). Construction of new fences should be avoided as fences provide predator corridors, raptor perches, and pose a risk for collisions (Braun 1998). We agree with Braun (2006) and Woodward (2006) that larger pastures with fewer fences are better. Wyoming big sagebrush typically recovers from a fire in 50-120 years (Baker 2006), and because the restricted distribution and limited cover of sagebrush in South Dakota, we recommend no use of prescribed fire in areas with sagebrush.

With 75% of the study area in private ownership and the patchy network of public land; sage-grouse conservation and persistence lies in hands of private landowners. To increase sage-grouse habitats, long-term (>20 yrs) partnerships and incentives with ranchers will be imperative. This will require cooperation from state wildlife agencies, federal land management agencies, local natural resource conservation districts, and committed landowners. Forming a South Dakota sage-grouse working group may be in order to accomplish this goal as many landowners were interested in sage-grouse conservation.

Table 1. Mean vegetation characteristics of nest sites and random sites between years for greater sage-grouse used in logistic regression models in northwestern South Dakota, USA, using MRPP (Mielke and Berry 2001), 2006-2007.

	Nest			Random			Both Years		
Variable	**2006 (*n* = 34)**	**2007 (*n* = 39)**	**P-value**	**2006 (*n* = 35)**	**2007 (*n* = 39)**	**P-value**	**Nest (*n* = 73)**	**Random (*n* = 74)**	**P-value**
Total Cover (%)	61.1	75.1	<0.01	55.8	66.1	<0.01	68.6	61.2	<0.01
Litter Cover (%)	7.6	7.1	0.79	6.5	6.1	0.88	7.4	6.3	0.04
Grass Cover (%)	24.2	31.4	0.01	21.1	25.8	0.21	28.1	23.6	0.01
Max Grass Hgt. (cm)	23.4	29.5	<0.01	20.4	25.0	<0.01	26.7	22.8	<0.01
Max Grass Hgt. 0-5m (cm)	25.7	30.9	0.02	20.3	24.3	0.01	28.5	22.4	<0.01
Visual Obstruction (cm)	5.5	11.1	<0.01	3.7	5.1	0.14	8.5	4.4	<0.01
Visual Obstruction 0m (cm)	20.8	29.4	<0.01	10.5	8.9	0.13	25.4	9.6	<0.01
Visual Obstruction 1m (cm)	7.3	13.7	<0.01	3.7	4.1	0.05	10.7	3.9	<0.01
Sagebrush Cover (%)	10.3	10.1	0.75	6.3	6.3	0.98	10.2	6.2	<0.01
Sagebrush Hgt. (cm)	25.8	29.7	0.04	23.8	24.0	0.97	27.9	23.9	<0.01

Table 2. Observed mean values for habitat variables between greater sage-grouse successful and failed nests used in nest success models in northwestern South Dakota, USA, using MRPP (Mielke and Berry 2001) 2006-2007.

	Successful (*n* = 33)		**Failed (*n* = 40)**		
Variable	**Mean**	**SE**	**Mean**	**SE**	**P-value**
Max Grass Hgt. (cm)	30.64	1.6	23.4	1.0	<0.01
Litter Cover (%)	6.4	0.5	8.1	0.8	0.07
Forb Cover 0 m (%)	5.3	0.8	3.9	0.6	0.09
Visual Obstruction (cm)	10.2	1.1	7.2	0.8	0.02

Table 3. Nest initiation rates of radio-marked adult and yearling greater sage-grouse in northwestern South Dakota, USA, 2006-2007.

	Ad			**Yearlings**			**Total**		
Yr	**Estimate**	**SE**	***n***	**Estimate**	**SE**	***n***	**Estimate**	**SE**	***n***
2006	90.5%	6.6	21	94.1%	5.9	17	92.1%	4.4	38
2007	100.0%	0.0	25	100.0%	0.0	10	100.0%	0.0	35
Total	95.7%	3.0	46	96.3%	3.7	27	95.9%	2.3	73

Table 4. Average clutch size and average hatch dates for first nests and renests of greater sage-grouse in northwestern South Dakota, USA, 2006-2007.

	First Nest			**Renest**		
Yr	**Initiation Date[ab]**	**Hatch Date[a]**	**Clutch Size**	**Initiation Date[ab]**	**Hatch Date[a]**	**Clutch Size**
2006	26 April ± 2.8 *n* = 13	3 June ± 2.6 *n* = 13	7.9 ± 0.3 *n* = 26	10 May ± 1.5 *n* = 2	16 June ± 1.5 *n* = 2	7.3 ± 0.5 *n* = 4
2007	21 April ± 1.7 *n* = 17	29 May ± 1.5 *n* = 17	8.5 ± 0.2 *n* = 30	9 May ± 4.7 *n* = 3	12 June ± 3.2 *n* = 3	5.5 ± 0.9 *n* = 4
Avg.	24 April ± 1.6 *n* = 30	31 May ± 1.5 *n* = 30	8.3 ± 0.2 *n* = 56	9 May ± 2.6 *n* = 5	14 June ± 2.0 *n* = 5	6.4 ± 0.6 *n* = 8

[a] Estimated only for successful nests.
[b] Estimated date of first egg laid.

Table 5. Results from logistic regression models predicting greater sage-grouse nest sites (n = 73) versus random sites (n = 74) in northwestern South Dakota, USA, 2006-2007.

Model[a]	**K**[b]	**AIC*c***	**Δ AIC*c***[c]	***wi***[d]
Sagebrush Cover + Visual Obstruction 0m	5	112.02	0.00	0.39
Sagebrush Cover + Visual Obstruction 0m + Max Grass Hgt. 0-5m	6	112.23	0.22	0.35
Sagebrush Cover+ Visual Obstruction 0m + Visual Obstruction 1m	6	113.96	1.94	0.15
Sagebrush Cover + Visual Obstruction 0m + Visual Obstruction 1m + Max Grass Hgt. 0-5m	7	114.40	2.39	0.12

[a] For ease of interpretation, year variable was excluded from model column. See Appendix 1 for full model results
[b] Number of habitat parameters plus intercept, SE, and year.
[c] Change in AIC*c* value
[d] Model weight

Table 6. Parameter Estimates, odds ratios, and corresponding confidence intervals for the best-approximating model of greater sage-grouse nests sites versus random sites in northwestern South Dakota, 2006-2007.

	Parameter			**Odds Ratio**		
Variable	**Estimate**	**Lower 95%CI**	**Upper 95%CI**	**Ratio**	**Lower 95%CI**	**Upper 95%CI**
Sagebrush Cover	0.195	0.086	0.325	1.215	1.090	1.384
Visual Obstruction 0 m	0.220	0.155	0.300	1.246	1.168	1.350

Table 7. Summary of model selection results for nest survival between year and age of greater sage-grouse in northwestern South Dakota, USA, 2006-2007.

Model [a]	**K** [b]	**AIC*c***	**Δ AIC*c*** [c]	***wi*** [d]
Max Grass Hgt. + Litter	3	225.79	0.00	0.23
Max Grass Hgt. + Litter + Daily Precip + Precip Lag	5	226.75	0.96	0.15
Max Grass Hgt. + Litter + Daily Precip	4	227.39	1.60	0.11
Max Grass Hgt. + Litter + Bird Age	4	227.77	1.98	0.09

[a] See appendix 2 for full model results
[b] Number of variables
[c] Change in AIC*c* value
[d] Model weight

Distance from Nearest Lek

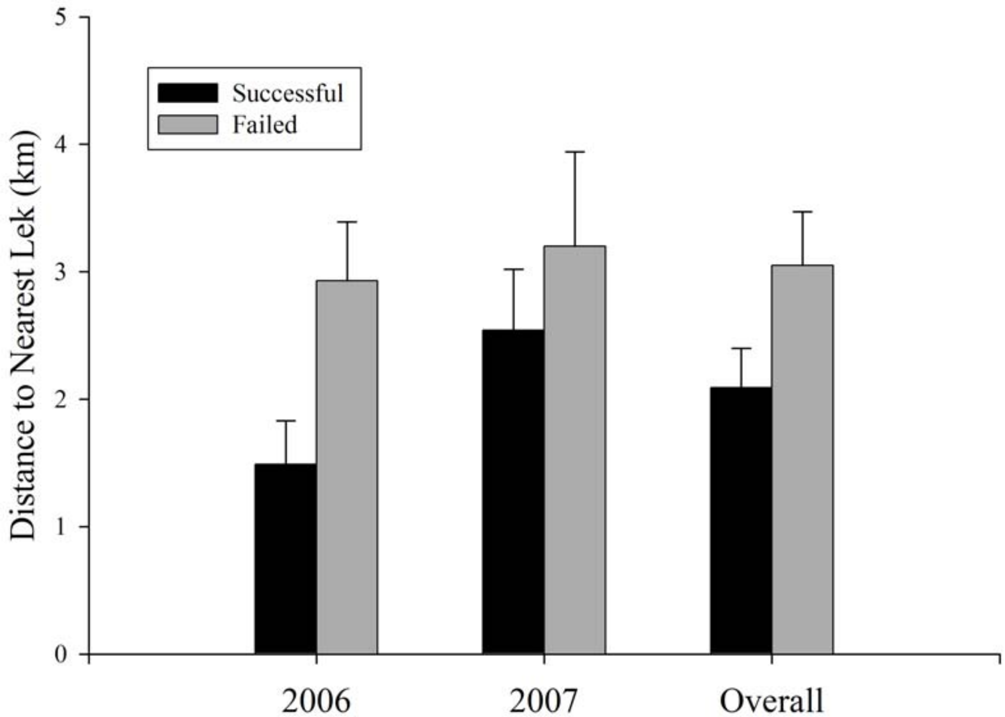

Figure 2. Mean distances plus one standard error (SE) of successful and failed greater sage-grouse nests to nearest documented active lek in northwestern South Dakota, USA, 2006-2007.

Number of Nests Within Particular Lek Buffers

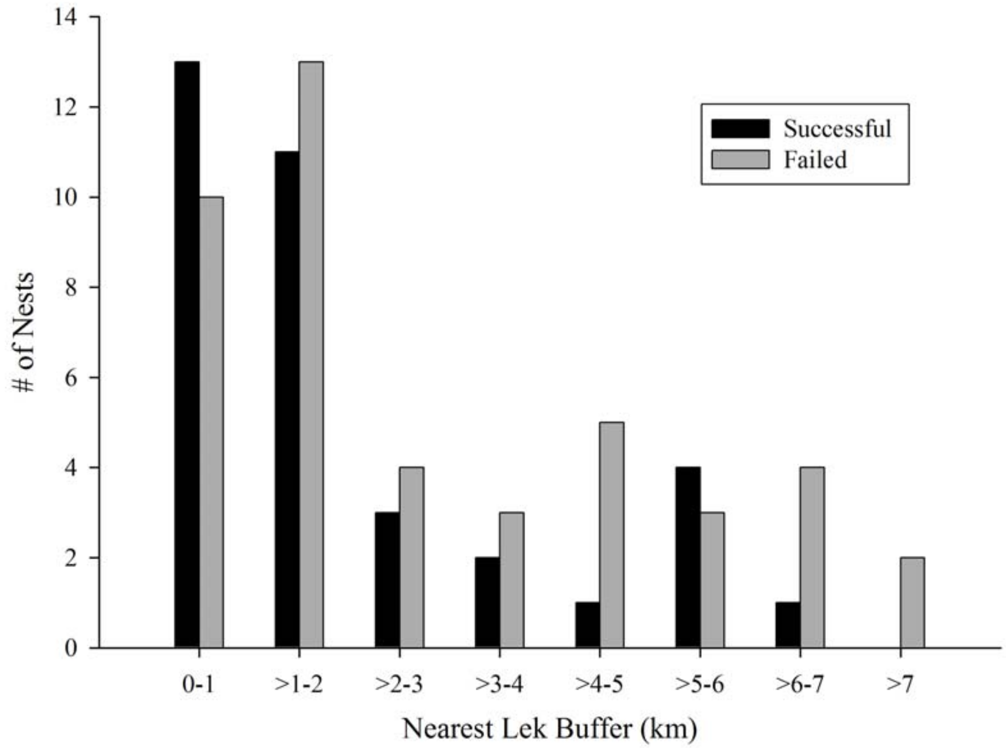

Figure 3. Distribution of successful and failed nests to nearest documented lek distances for greater sage-grouse in northwestern South Dakota, USA, 2006-2007.

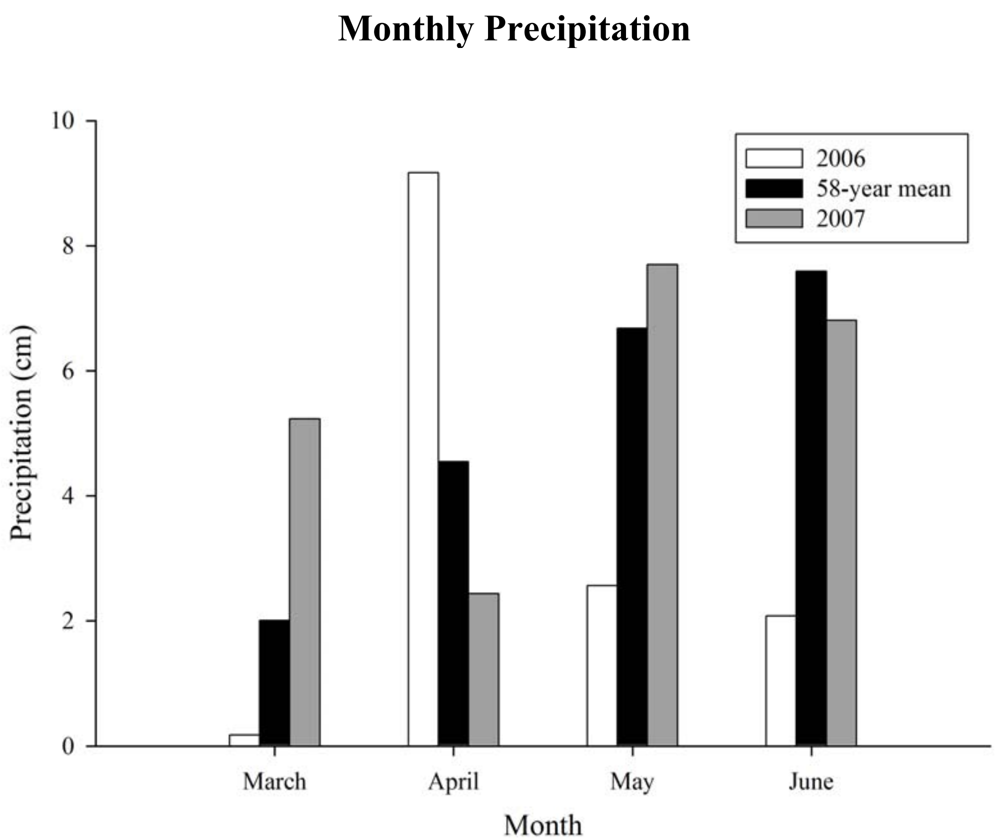

Figure 4. Monthly precipitation received during the breeding and nesting periods in 2006 – 2007 compared to the 58-year mean from the nearest daily weather station (Nisland, SD).

Effect of Grass Height on Nest Success

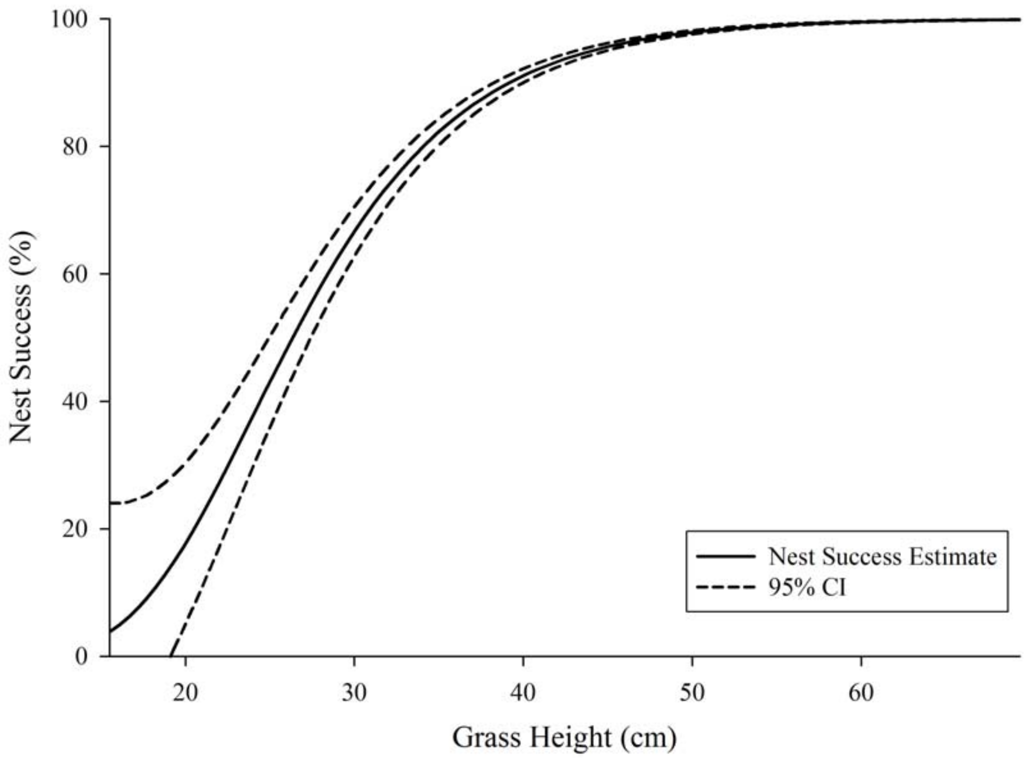

Figure 5. Effect of grass height on greater sage-grouse nest success in northwestern South Dakota, USA, 2006-2007. Nest success estimate derived from back-transformed beta estimates included in top model. Confidence intervals estimated from the delta method (Seber 1982).

Effect of Grass Height and Litter on Nest Success

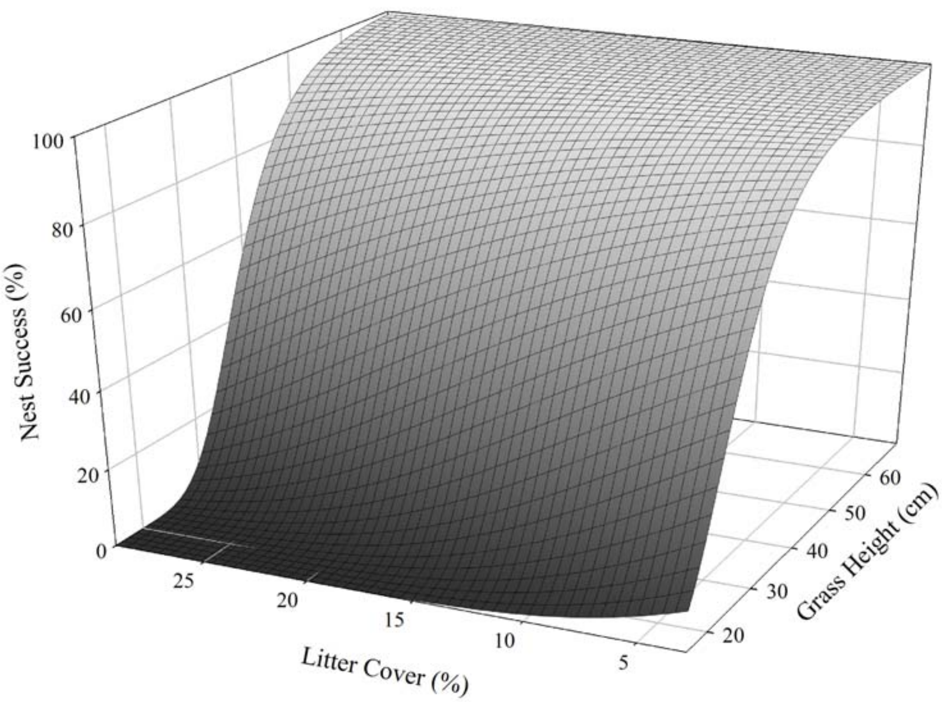

Figure 6. Effect of grass height and litter canopy coverage on greater sage-grouse nest success in northwestern South Dakota, USA, 2006-2007. Nest success estimate derived from back-transformed beta estimates included in top model.

Effect of Litter Canopy Coverage on Nest Success

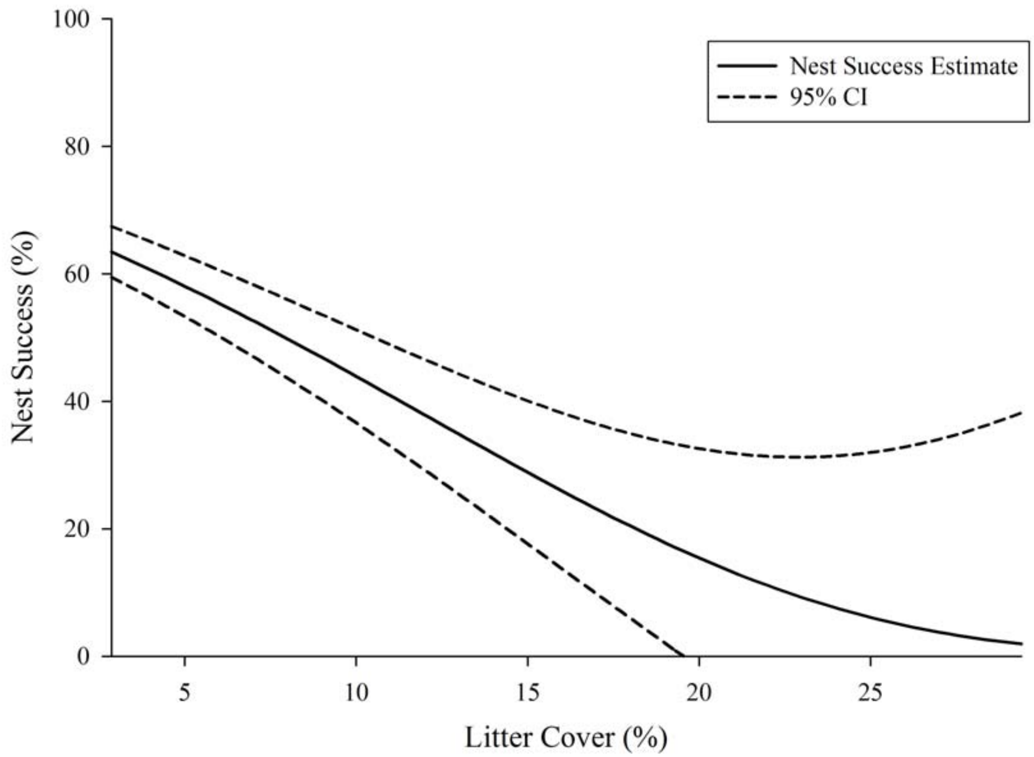

Figure 7. Effect of litter canopy coverage on greater sage-grouse nest success in northwestern South Dakota, USA, 2006-2007. Nest success estimate derived from back-transformed beta estimates included in top model. Confidence intervals estimated from the delta method (Seber 1982).

LITERTURE CITED

Adams, B. W., J. Carlson, D. Milner, T. Hood, B. Cairns, and P. Herzog. 2004. Beneficial grazing management practices for Sage-grouse (*Centrocercus urophasianus*) and ecology of silver sagebrush (*Artemisia cana*) in southeastern Alberta. Technical Report, Public Lands and Forests Division, Alberta Sustainable Resource Development. Pub. No. T/049. 60pp.

Aldridge, C. L. and M. S. Boyce. 2007. Linking occurrence and fitness to persistence: habitat based approach for endangered greater sage-grouse. Ecological Applications 17:508-526.

Aldridge, C. L. and R. M. Brigham. 2001. Nesting and reproductive activities of greater sage-grouse in a declining northern fringe population. Condor 103:537-543.

Aldridge, C. L. and R. M. Brigham. 2002. Sage-grouse nesting and brood-rearing habitat use in southern Canada. Journal of Wildlife Management 66:433-444.

Baker, M. F., R. L. Eng. J. S. Gashwiler, M. H. Schroeder, and C. E. Braun. 1976. Conservation committee report on the effects of alteration of sagebrush communities on the associated avifauna. Wilson Bulletin 88:165-170.

Baker, W. L. 2006. Fire and restoration of sagebrush ecosystems. Wildlife Society Bulletin 34:177-185.

Barnett, J. K., and J. A. Crawford. 1994. Pre-laying nutrition of sage-grouse hens in Oregon. Journal of Range Management 47:114-118.

Beck, J. L., and D. L. Mitchell. 2000. Influences of livestock grazing on sage-grouse habitat. Wildlife Society Bulletin 28:993-1002.

Benkobi, L., D. W. Uresk, G. Schenbeck, and R. M. King. 2000. Protocol for monitoring standing crop in grasslands using visual obstruction. Journal of Range Management 53:627-633.

Bergerud, A. T. 1988. Population ecology of North American grouse. Pages 578-685 *in* A. T. Bergerud, and M. W. Gratson, editors. Adaptive Strategies and Population Ecology of Northern Grouse. Wildlife Management Institute, Minneapolis, Minnesota, USA.

Beyer, H. L. 2004. Hawth's Analysis Tools for ArcGIS. Available at http://www.spatialecology.com/htools.

Boyce, M. S., P. R. Vernier, S. E. Nielson, and F. K. A. Schmiegelow. 2002. Evaluating resource selection functions. Ecological Modeling 157:281-300.

Braun, C. E. 1998. Sage grouse declines in western North America: what are the problems? Proceedings of the Western Association of Fish and Wildlife Agencies 78:139-156.

Braun, C. E. 2006. A blueprint for sage-grouse conservation and recovery [unpublished]. Grouse Inc. Tucson, AZ.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multi-model inference: a practical information theoretic approach. Second edition. Springer-Verlag, New York, New York, USA.

Chi, R. Y. 2004. Greater sage-grouse on Parker Mountain, Utah. Thesis, Utah State University, Logan, USA.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Connelly, J. W., W. L. Wakkinen, A. D. Apa, and K. P. Reese. 1991. Sage-grouse use of nest sites in southeastern Idaho. Journal of Wildlife Management 55:521-524.

Cottam, G., and J. T. Curtis. 1956. The use of distance measures in phytosociological sampling. Ecology 37:451-460.

Crunden, C. W. 1963. Age and sex of sage grouse from wings. Journal of Wildlife Management 27:846-849.

Daubenmire, R. F. 1959. A canopy-coverage method of vegetation analysis, Northwest Science 33:43-64.

Dinsmore, S. J., D. H. Johnson. 2005. Population analysis in wildlife biology. Pages 154-184 *in* C. E. Braun, editor. Techniques for wildlife investigations and management. Sixth edition. The Wildlife Society, Bethesda, Maryland, USA.

Dinsmore, S. J., G. C. White, and F. L. Knopf. 2002. Advanced techniques for modeling avian nest survival. Ecology 83:3476-3488.

Eng, R. L. 1955. A method for obtaining sage grouse age and sex ratios from wings. Journal of Wildlife Management 19:267-272.

Fischer, R. A., A. D. Apa, W. L. Wakkinen, K. P. Reese, and J. W. Connelly. 1993. Nesting-area fidelity of sage-grouse in southeastern Idaho. Condor 95:1038-1041.

Giesen, K. M., T. J. Schoenberg, and C. E. Braun. 1982. Methods for trapping sage grouse in Colorado. Wildlife Society Bulletin 10:224-231.

Greenwood, P. J., and P. H. Harvey. 1982. The natal and breeding dispersal of birds. Annual Review of Ecology and Systematics 13:1-21.

Gregg, M. A. 1991. Use and selection of nesting habitat by sage-grouse in Oregon. M.S. Thesis, Oregon State University, Corvallis, USA.

Gregg, M. A., J. A. Crawford, M. S. Drut, and A. K. DeLong. 1994. Vegetational cover and predation of sage-grouse nests in Oregon. Journal of Wildlife Management 58:162-166.

Hagen, C. A., J. W. Connelly, and M. A. Schroeder. 2007. A meta-analysis of greater sage-grouse *Centrocercus urophasianus* nesting and brood-rearing habitats. Wildlife Biology Supplement 13:42-50.

Hart, R. H., M. J. Samuel, P. S. Test, and M. A. Smith. 1988. Cattle, vegetation, and economic responses to grazing systems and grazing pressure. Journal of Range Management 41:282-286.

Herman-Brunson, K. H. 2007. Nesting and brood-rearing habitat selection of greater sage-grouse and associated survival of hens and broods at the edge of their historic distribution. Thesis, South Dakota State University, Brookings, USA.

Herman-Brunson, K. H., N. W. Kaczor, C. C. Swanson, K. C. Jensen, M. A. Rumble, R. W. Klaver. *Submitted.* Nesting ecology of greater sage-grouse at the eastern edge of their distribution. Journal of Wildlife Management.

Hoffman, R. W., M. P. Luttrell, and W. R. Davidson. 1996. Reproductive performance of Merriam's Wild Turkeys with suspected *Mycoplasma* infection. Proceedings of National Wild Turkey Symposium. 7:145-151.

Holechek, J. L., H. Gomez, F. Molinar, and D. Galt. 1999. Grazing studies: what we've learned. Rangelands 21:12-16.

Hosmer, D. W., and S. Lemeshow. 2000. Applied Logistic Regression, 2nd Edition. John Wiley and Sons Inc. Publication, New York, NY, USA.

Hungerford, C. R. 1964. Vitamin A and productivity in Gambel's quail. Journal of Wildlife Management 28:141-147.

Johnson, J. R., and G. E. Larson. 1999. Grassland plants of South Dakota ant the northern Great Plains. South Dakota State University. Brookings, South Dakota, USA.

Lariviére, S., and Messier, F. 1998. Effect of density and nearest neighbours on simulated waterfowl nests: can predators recognize high-density nesting patches? Oikos 83:12-20.

Mayfield, H. F. 1975. Suggestions for calculating nest success. Wilson Bulletin 87:456-466.

McCarthy, J. J., and J. D. Kobriger. 2005. Management plan and conservation strategies for greater sage-grouse in North Dakota. North Dakota Game and Fish Department. 96pp.

Mielke, P. W., and K. J. Berry. 2001. Permutation Methods: A Distance Function Approach. New York: Springer-Verlag.

Moynahan, B. J., M. S. Lindberg, J. J. Rotella, and J. W. Thomas. 2007. Factors affecting nest survival of Greater sage-grouse in northcentral Montana. Journal of Wildlife Management 71:1773-1783.

Naeth, M. A., A. W. Bailey, D. J. Pluth, D. S. Chanasyk, and R. T. Hardon. 1991. Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland ecosystems in Alberta. Journal of Range Management 44:7-12.

Naylor, B. J., and J. F. Bendell. 1989. Clutch size and egg size of spruce grouse in relation to spring diet, food supply, and endogenous reserves. Canadian Journal of Zoology 67:969-980.

Peterson, B. E. 1980. Breeding and nesting ecology of female sage grouse in North Park, Colorado. Thesis, Colorado State University, Fort Collins, Colorado.

Pollard, J. H. 1971. On distance estimators of density in randomly distributed forests. Biometrics 27:991-1002.

Porter, W. F., G. C. Nelson, and K. Mattson. 1983. Effects of winter conditions on reproduction in a northern wild turkey population. Journal of Wildlife Management 47:281-290.

Robel, R. J., J. N. Briggs, A. D. Dayton, and L. C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23:295-298.

Rumble, M. A., and R. A. Hodorff. 1993. Nesting ecology of Merriam's turkeys in the Black Hills, South Dakota. Journal of Wildlife Management 57:789-801.

Rumble, M. A., B. F. Wakeling, and L. D. Flake. 2003. Factors affecting survival and recruitment in female Merriam's turkeys. Intermountain Journal of Sciences 9:26-37.

Sargeant, A. B., M. A. Sovada, and R. J. Greenwood. 1998. Interpreting evidence of depredation of duck nests in the prairie pothole region. U.S. Geological Survey, Northern Prairie Wildlife Research Center, Jamestown, ND and Ducks Unlimited, Inc., Memphis, TN, 72 pp.

Schroeder, M. A. 1997. Unusually high reproductive effort by sage grouse in a fragmented habitat in north-central Washington. Condor 99:933-941.

Schroeder, M. A., and L. A. Robb. 2003. Fidelity of greater sage-grouse *Centrocercus urophasianus* to breeding areas in a fragmented landscape. Wildlife Biology 9:291-299.

Schroeder, M. A., C. L. Aldridge, A. D. Apa, J. R. Bohne, C. E. Braun, S. D. Bunnell, J. W. Connelly, P. A. Deibert, S. C. Gardner, M. A. Hilliard, G. D. Kobriger, S. M. McAdam, C. W. McCarthy, J. J. McCarthy, D. L. Mitchell, E. V. Rickerson, and S. J. Stiver. 2004. Distribution of sage-grouse in North America. Condor 106:363-376.

Schroeder, M. A., J. R. Young, and C. E. Braun. 1999. Sage grouse (*Centrocercus urophasianus*). *In* The Birds of North America, No. 425 (A. Poole. and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Seber, G. A. F. 1982. The estimation of animal abundance and related parameters. Second edition. Macmillan Book, New York, New York, USA.

Smith, J. 2003. Greater sage grouse on the edge of their range: leks and surrounding landscapes in the Dakotas. Thesis, South Dakota State University, Brookings, USA.

South Dakota State Climate Office. 2007. Office of the State Climatologist. http://climate.sdstate.edu Accessed on 15 December 2007.

Sveum, C. M., W. D. Edge, and J. A. Crawford. 1998. Nesting habitat selection by sage-grouse in south-central Washington. Journal of Range Management 51:265-269.

Wallestad, R. O., and D. B. Pyrah. 1974. Movement and nesting of sage-grouse hens in central Montana. Journal of Wildlife Management 38:630-633.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46(Supplement):120-138.

Woodward, J. K. 2006. Greater sage-grouse (*Centrocercus urophasianus*) habitat in central Montana. Thesis, Montana State University, Bozeman, USA.

Appendix 1. Complete results from logistic regression models predicting greater sage-grouse nest sites (n = 73) versus random sites (n = 74) in northwestern South Dakota, USA, 2006-2007.

Model[a]	**K**[b]	**AIC*c***	**Δ AIC*c***[c]	***wi***[d]
Sagebrush Cover + Visual Obstruction 0m	5	112.02	0.00	0.39
Sagebrush Cover + Visual Obstruction 0m + Max Grass Hgt. 0-5m	6	112.23	0.22	0.35
Sagebrush Cover + Visual Obstruction 0m + Visual Obstruction 1m	6	113.96	1.94	0.15
Sagebrush Cover + Visual Obstruction 0m + Visual Obstruction 1m + Max Grass Hgt. 0-5m	7	114.40	2.39	0.12
Visual Obstruction 0m	4	123.27	11.26	0.00
Visual Obstruction 0m + Max Grass Hgt. 0-5m	5	123.36	11.35	0.00
Visual Obstruction 0m + Total Cover	5	124.14	12.12	0.00
Visual Obstruction 0m + Visual Obstruction 1m	5	124.45	12.44	0.00
Visual Obstruction 0m + Max Grass Hgt.+ Sagebrush Hgt.	6	125.91	13.90	0.00
Total Cover + Max Grass Hgt. + Visual Obstruction 0m	6	125.93	13.91	0.00
Total Cover + Max Grass Hgt. + Sagebrush Hgt. + Visual Obstruction 0m	7	127.34	15.32	0.00
Visual Obstruction 1m + Sagebrush Cover	5	146.97	34.96	0.00
Visual Obstruction 1m	4	157.93	45.91	0.00
Visual Obstruction 1m + Max Grass Hgt. 0-5m	5	158.56	46.54	0.00
Sagebrush Cover + Visual Obstruction	5	162.19	50.17	0.00
Sagebrush Cover + Max Grass Hgt. 0-5m	5	166.21	54.20	0.00
Sagebrush Cover + Grass Cover	5	173.65	61.63	0.00
Sagebrush Cover + Total Cover	5	175.41	63.39	0.00
Visual Obstruction	4	176.55	64.53	0.00
Max Grass Hgt. + Sagebrush Cover	5	177.19	65.18	0.00
Total Cover + Visual Obstruction	5	178.69	66.68	0.00
Litter + Sagebrush Cover	5	180.14	68.12	0.00
Litter + Max Grass Hgt. 0-5m + Sagebrush Hgt.	6	181.63	69.62	0.00
Max Grass Hgt. 0-5m + Sagebrush Hgt.	5	182.11	70.10	0.00
Sagebrush Cover	4	186.55	74.54	0.00
Max Grass Hgt. 0-5m + Litter	5	187.00	74.99	0.00
Max Grass Hgt. 0-5m	4	187.20	75.18	0.00
Litter + Max Grass Hgt. + Sagebrush Hgt.	6	191.89	79.87	0.00
Max Grass Hgt. + Sagebrush Hgt.	5	193.07	81.06	0.00
Max Grass Hgt. + Sagebrush Hgt. + Total Cover	6	193.81	81.79	0.00
Litter + Max Grass Hgt.	5	199.64	87.63	0.00
Litter + Sagebrush Hgt.	5	199.82	87.80	0.00
Max Grass Hgt.	4	200.24	88.22	0.00
Sagebrush Hgt.	4	201.82	89.80	0.00
Total Cover	4	201.92	89.90	0.00
Grass Cover	4	206.70	94.68	0.00
Litter	4	208.96	96.94	0.00

[a] For ease of interpretation, year variable was excluded from model column.
[b] Number of habitat parameters plus intercept, SE, and year.
[c] Change in AIC*c* value
[d] Model weight

Appendix 2. Complete summary of model selection results for nest survival between year and age of greater sage-grouse in northwestern South Dakota, USA, 2006-2007.

Model	**K** [a]	**AIC*c***	**Δ AIC*c*** [c]	***wi*** [d]
Max Grass Hgt. + Litter	3	225.79	0.00	0.23
Max Grass Hgt. + Litter + Daily Precip + Precip Lag	5	226.75	0.96	0.15
Max Grass Hgt. + Litter + Daily Precip	4	227.39	1.60	0.11
Max Grass Hgt. + Litter + Bird Age	4	227.77	1.98	0.09
Max Grass Hgt. + Litter + Forb 0m	4	227.80	2.01	0.09
Year*Max Grass Hgt. + Litter	6	228.64	2.85	0.06
Max Grass Hgt.	2	228.85	3.06	0.05
Max Grass Hgt. + Litter + Forb 0m + Daily Precip	5	229.41	3.62	0.04
Max Grass Hgt. + Litter + Forb 0m+ Bird Age	5	229.79	3.99	0.03
Max Grass Hgt. + DailyPrecip + Precip Lag	4	229.96	4.17	0.03
Year + Max Grass Hgt.	3	230.15	4.36	0.03
Max Grass Hgt. + DailyPrecip	3	230.38	4.59	0.02
Max Grass Hgt. + Forb 0m	3	230.65	4.86	0.02
Max Grass Hgt. + Bird Age	3	230.78	4.99	0.02
Year*Max Grass Hgt.	4	231.18	5.39	0.02
Max Grass Hgt. + Litter + Forb 0m + DailyPrecip + MinTemp	6	231.35	5.56	0.01
Bird Age*Max Grass Hgt.	4	232.46	6.66	0.01
Year*Bird Age + Max Grass Hgt.	5	233.81	8.02	0.00
Year*Visual Obstruction + Litter	6	240.37	14.58	0.00
Year*Visual Obstruction + Litter + Forb 0m	8	240.82	15.03	0.00
Visual Obstruction + Litter	3	243.27	17.47	0.00
Visual Obstruction + Litter + Forb 0m	4	245.01	19.21	0.00
Visual Obstruction + Litter + Bird Age	4	245.11	19.32	0.00
DailyPrecip + Visual Obstruction + Litter + Forb 0m	5	246.05	20.26	0.00
Year*Visual Obstruction	4	246.35	20.56	0.00
Visual Obstruction + Litter + Forb 0m+ Bird Age	5	246.88	21.08	0.00
Daily Precip + Min Temp + Visual Obstruction + Litter + Forb 0m	6	247.27	21.48	0.00
Visual Obstruction	2	248.05	22.26	0.00
Litter	2	249.97	24.17	0.00
Year + Visual Obstruction	3	250.04	24.25	0.00
Visual Obstruction + Forb 0m	3	250.06	24.27	0.00
Visual Obstruction + Bird Age	3	250.06	24.27	0.00
Year + Litter	3	250.46	24.66	0.00
Litter + Bird Age	3	251.23	25.44	0.00
Litter + Forb 0m	3	251.49	25.70	0.00
Daily Precip + Litter + Forb 0m	4	251.91	26.12	0.00
Visual Obstruction + Forb 0m+ Bird Age	4	252.07	26.28	0.00
Year*Litter	4	252.47	26.67	0.00
Constant	1	252.71	26.92	0.00
Daily Precip	2	252.99	27.20	0.00
Year	2	253.01	27.22	0.00
Min Temp	2	253.04	27.25	0.00
Year*Forb 0m	4	253.33	27.54	0.00
Daily Precip + Precip Lag	3	253.70	27.91	0.00
Min Temp + Temp Lag	3	254.05	28.26	0.00
Year*Litter + Forb 0m	6	254.14	28.35	0.00
Daily Precip + Precip Lag + Min Temp	4	254.28	28.49	0.00
Forb 0m	2	254.36	28.57	0.00

Appendix 2. continued.

Bird Age	2	254.52	28.73	0.00
Daily Precip + Forb 0m	3	254.73	28.94	0.00
Year + Forb 0m	3	255.00	29.21	0.00
Daily Precip + Precip Lag + Min Temp + Temp Lag	5	255.06	29.27	0.00
Forb 0m + Bird Age	3	256.22	30.42	0.00
Year*Bird Age	4	256.87	31.08	0.00

[a] Number of variables
[b] Change in AIC*c* value
[c] Model weight

Appendix 3. Demographic information for all greater sage-grouse captured in northwestern South Dakota, USA, 2006-2007.

Band #	Capture Date	X[a]	Y[a]	Nearest Lek	Sex[b]	Age[c]	Weight (g)	Radio Freq.
1001	28-Mar-06	583058	4972413	Crago	F	A	1654	150.064
1002	31-Mar-06	583874	4972344	Crago	F	A	1552	150.073
1003	1-Apr-06	605131	4983015	Two Top	F	A	1618	150.083
1004	1-Apr-06	604838	4982844	Two Top	F	Y	1612	150.094
1005	1-Apr-06	604840	4983075	Two Top	F	A	1602	150.103
1006	1-Apr-06	605197	4983537	Two Top	F	A	1732	150.114
1007	1-Apr-06	605399	4982814	Two Top	F	A	1648	151.074
1008	3-Apr-06	594044	4989246	Widdoss	F	A	1586	150.133
1009	3-Apr-06	595437	4988647	Widdoss	F	Y	1734	150.145
1010	3-Apr-06	595437	4988647	Widdoss	F	Y	1464	150.155
1011	3-Apr-06	595437	4988647	Widdoss	F	Y	1482	151.085
1012	3-Apr-06	595594	4988735	Widdoss	F	A	1594	150.173
1013	3-Apr-06	595758	4988629	Widdoss	F	Y	1482	150.183
1014	3-Apr-06	595619	4988954	Widdoss	F	Y	1520	150.193
1015	4-Apr-06	623696	4994653	McFarland	F	A	1758	150.204
1016	4-Apr-06	623922	4994453	McFarland	F	Y	1556	150.214
1017	5-Apr-06	583265	4972042	Crago	F	A	1650	150.353
1018	5-Apr-06	581965	4969635	Rumph	F	Y	1520	150.363
1019	7-Apr-06	606987	5006247	County Line	F	Y	1610	150.373
1020	7-Apr-06	606596	5006738	County Line	F	A	1704	150.383
1021	7-Apr-06	606596	5006738	County Line	F	A	1626	151.014
1022	7-Apr-06	606490	5006922	County Line	F	A	1610	151.022
1023	7-Apr-06	606616	5007299	County Line	F	A	1806	151.033
1024	7-Apr-06	606053	5006751	County Line	F	A	1590	150.503
1025	7-Apr-06	605932	5006832	County Line	F	A	1642	150.703
1026	7-Apr-06	605849	5006714	County Line	F	A	1634	150.714
1027	8-Apr-06	623462	4994283	McFarland	F	A	1756	150.732
1028	8-Apr-06	623243	4995268	McFarland	F	A	1738	150.973
1029	8-Apr-06	623243	4995268	McFarland	F	Y	1470	150.764
1030	8-Apr-06	623494	4994808	McFarland	F	A	1606	150.772
1031	9-Apr-06	583034	4972327	Crago	F	Y	1472	150.785
1032	9-Apr-06	581219	4969831	Rumph	F	Y	1628	150.804
1033	9-Apr-06	581315	4969863	Rumph	F	Y	1613	150.812
1034	9-Apr-06	581512	4969966	Rumph	F	A	1636	151.333
1035	9-Apr-06	581403	4970033	Rumph	F	A	1782	151.343
1036	9-Apr-06	583487	4972092	Crago	F	Y	1544	151.353
1037	9-Apr-06	594466	4990149	Widdoss	F	A	1690	151.362
1038	10-Apr-06	605130	4983164	Two Top	F	Y	1658	151.375
1039	10-Apr-06	604967	4983102	Two Top	F	Y	1594	151.382
1040	10-Apr-06	604946	4983024	Two Top	F	Y	1480	151.393
1041	17-Jul-06	626931	4986394	Quad 7	unk	C	558	150.024
1042	17-Jul-06	626931	4986394	Quad 7	unk	C	422	151.553
1043	17-Jul-06	626931	4986394	Quad 7	unk	C	468	151.533
1044	17-Jul-06	617726	4993470	McFarland	unk	C	466	150.993
1045	17-Jul-06	617726	4993470	McFarland	unk	C	664	151.442
1046	17-Jul-06	617726	4993470	McFarland	unk	C	476	151.422
1047	18-Jul-06	602067	4986019	Widdoss	unk	C	490	150.573
1048	18-Jul-06	600432	4986227	Widdoss	unk	C	576	150.654

Appendix 3. cont.

1049	18-Jul-06	600432	4986227	Widdoss	unk	C	698	151.503
1050	18-Jul-06	600512	4987086	Widdoss	unk	C	338	151.151
1051	18-Jul-06	600512	4987086	Widdoss	unk	C	432	151.524
1052	18-Jul-06	600512	4987086	Widdoss	unk	C	600	151.245
1053	18-Jul-06	600512	4987086	Widdoss	unk	C	466	151.524
1054	18-Jul-06	596981	4987357	Widdoss	unk	C	646	151.562
1055	18-Jul-06	596981	4987357	Widdoss	unk	C	838	151.483
1056	17-Jul-06	617726	4993470	McFarland	F	A	1362	151.413
1057	18-Jul-06	596981	4987357	Widdoss	unk	C	812	151.543
1058	18-Jul-06	596981	4987357	Widdoss	unk	C	816	151.094
1059	18-Jul-06	596981	4987357	Widdoss	unk	C	644	151.533
1060	19-Jul-06	606966	4983857	Two Top	unk	C	642	151.713
1061	19-Jul-06	606966	4983857	Two Top	unk	C	628	151.453
1062	20-Jul-06	600796	4987123	Widdoss	unk	C	552	151.733
1063	31-Jul-06	599438	4991214	Widdoss	unk	C	430	150.284
1064	31-Jul-06	599438	4991214	Widdoss	unk	C	396	150.303
1065	2-Aug-06	606586	5004830	County Line	unk	C	566	151.043
1066	10-Aug-06	600069	5012561	Split Lek	unk	C	602	150.443
1067	10-Aug-06	600069	5012561	Split Lek	unk	C	494	150.524
1069	19-Jul-07	600206	4986435	Two Top	M	C	612	151.942
1070	19-Jul-07	600206	4986435	Two Top	unk	C	486	151.803
1071	19-Jul-07	600206	4986435	Two Top	unk	C	552	151.755
1072	19-Jul-07	600206	4986435	Two Top	unk	C	656	151.763
1073	19-Jul-07	600206	4986435	Two Top	unk	C	510	151.783
1074	19-Jul-07	600206	4986435	Two Top	M	C	552	151.934
1077	19-Jul-06	569728	4980943	State Line	unk	C	630	150.402
1078	19-Jul-06	569728	4980943	State Line	unk	C	500	150.127
1079	19-Jul-06	569728	4980943	State Line	unk	C	662	150.022
1080	31-Jul-06	570999	4978754	State Line	unk	C	420	150.163
1081	31-Jul-06	570999	4978754	State Line	unk	C	460	150.742
1082	20-Jul-06	600777	4987058	Widdoss	unk	C	632	N/A
1083	20-Jul-06	600777	4987058	Widdoss	unk	C	520	N/A
1084	20-Jul-06	600777	4987058	Widdoss	unk	C	584	N/A
1085	20-Jul-06	600234	4986337	Widdoss	unk	C	568	N/A
1086	20-Jul-06	600234	4986337	Widdoss	unk	C	626	N/A
1087	20-Jul-06	600234	4986337	Widdoss	unk	C	642	N/A
1088	20-Jul-06	600234	4986337	Widdoss	unk	C	640	N/A
1090	22-Aug-06	603221	4985402	Widdoss	unk	C	N/A	N/A
1092	22-Aug-06	603221	4985402	Widdoss	unk	C	N/A	N/A
1093	22-Aug-06	603221	4985402	Widdoss	unk	C	N/A	N/A
1094	22-Aug-06	603221	4985402	Widdoss	F	Y	N/A	N/A
1095	22-Aug-06	603221	4985402	Widdoss	F	C	N/A	151.123
1096	22-Aug-06	603221	4985402	Widdoss	unk	C	N/A	N/A
1097	20-Mar-07	624299	4994777	McFarland	F	Y	1566	150.984
1098	21-Mar-07	585688	4972089	Crago	F	Y	1474	150.954
1099	20-Mar-07	628371	4995961	Quad 7	F	A	N/A	N/A
1100	21-Mar-07	624274	4994608	McFarland	F	A	N/A	N/A
1101	22-Mar-07	603438	5007080	County Line	F	Y	1492	151.002
1102	22-Mar-07	585462	4970879	Crago	F	A	N/A	N/A
1103	26-Mar-07	594427	4989883	Widdoss	F	Y	1396	151.053
1104	26-Mar-07	594408	4989863	Widdoss	F	A	1684	151.064
1105	1-Apr-07	unk	unk	unk	F	unk	unk	N/A

Appendix 3. cont.

1106	1-Apr-07	unk	unk	unk	F	unk	unk	N/A
1107	1-Apr-07	unk	unk	unk	F	unk	unk	N/A
1108	1-Apr-07	unk	unk	unk	F	unk	unk	N/A
1109	23-Mar-07	605528	4982812	Two Top	F	A	N/A	N/A
1110	26-Mar-07	594255	5990427	Widdoss	F	Y	1498	151.103
1111	26-Mar-07	593709	4990683	Widdoss	F	A	1634	151.115
1112	26-Mar-07	593709	4990683	Widdoss	F	Y	1552	151.133
1119	19-Jul-07	603730	4988165	Two Top	unk	C	560	151.133
1120	19-Jul-07	603730	4988165	Two Top	unk	C	380	150.624
1121	19-Jul-07	603730	4988165	Two Top	unk	C	422	150.064
1122	19-Jul-07	606678	4984369	Two Top	unk	C	798	150.643
1123	19-Jul-07	606678	4984369	Two Top	unk	C	774	150.673
1124	19-Jul-07	606678	4984369	Two Top	unk	C	772	150.683
1125	19-Jul-07	606678	4984369	Two Top	unk	C	812	151.824
1126	23-Jul-07	580091	4970734	South Owl	unk	C	590	150.722
1127	23-Jul-07	589059	4991119	Widdoss	unk	C	532	150.793
1128	23-Jul-07	589059	4991119	Widdoss	unk	C	506	150.824
1129	23-Jul-07	589059	4991119	Widdoss	unk	C	682	150.833
1130	23-Jul-07	589059	4991119	Widdoss	unk	C	562	150.764
1131	24-Jul-07	606022	5009500	County Line	unk	C	602	150.373
1132	24-Jul-07	592056	4990220	Widdoss	unk	C	914	151.895
1133	24-Jul-07	600496	4985607	Two Top	unk	C	874	150.873
1134	2-Aug-07	608346	5002699	County Line	unk	C	966	150.883
1135	2-Aug-07	606150	5009419	County Line	unk	C	554	150.914
1136	7-Aug-07	594637	4987901	Widdoss	unk	C	566	150.923
1151	24-Oct-07	605829	5006655	County Line	M	C	2252	151.583
1152	24-Oct-07	595309	4988513	Widdoss	F	A	1500	151.393
1153	24-Oct-07	595420	4988559	Widdoss	F	A	1544	150.094
1154	24-Oct-07	605921	5006498	County Line	F	A	1496	151.363
1155	24-Oct-07	605844	5006720	County Line	F	A	1476	150.973
1501	31-Mar-06	583997	4972302	Crago	M	A	3040	151.036
1502	4-Apr-06	623572	4994708	McFarland	M	A	2920	151.194
1503	10-Apr-06	604849	4982804	Two Top	M	A	3320	151.574
1504	10-Apr-06	604701	4983175	Two Top	M	A	3216	151.585
1505	10-Apr-06	604879	4982796	Two Top	M	A	3304	151.594
1506	4-May-06	606663	5006951	County Line	M	A	3058	151.604
1507	4-May-06	606476	5006526	County Line	M	A	3048	151.614
1508	4-May-06	606663	5006951	McFarland	M	A	3022	151.962
1509	4-May-06	624042	4994699	McFarland	M	A	3094	151.973
1510	4-May-06	606508	5007060	County Line	M	A	2962	151.645
1511	5-May-06	583496	4972516	Crago	M	A	3040	151.655
1512	5-May-06	583783	4972382	Crago	M	A	3254	151.664
1513	5-May-06	581257	4969846	Rumph	M	A	2954	151.675
1514	5-May-06	594613	4989913	Widdoss	M	A	3078	151.983
1515	5-May-06	594548	4989957	Widdoss	M	A	3206	151.994
1516	5-May-06	594573	4989618	Widdoss	M	A	3044	151.036
1517	5-May-06	594437	4989670	Widdoss	M	A	3066	N/A
1518	5-May-06	594393	4989788	Widdoss	M	A	3010	N/A
1519	5-May-06	594605	4989797	Widdoss	M	A	3030	N/A
1520	20-Mar-07	624060	4994448	McFarland	M	A	3344	151.982
1522	26-Mar-07	594402	4989990	Widdoss	M	A	3140	151.803
1523	26-Mar-07	593674	4989252	Widdoss	M	Y	2378	151.813

Appendix 3. cont.

1524	26-Mar-07	594499	4989909	Widdoss	M	A	3124	151.824
1525	26-Mar-07	594409	4989727	Widdoss	M	A	3206	151.834
1526	8-May-07	606576	5006401	County Line	M	A	2932	151.843
1527	8-May-07	606581	5006401	County Line	M	Y	2302	151.854
1528	8-May-07	606648	5006757	County Line	M	A	2762	151.883
1529	8-May-07	606649	5006756	County Line	M	Y	2174	151.903
1530	10-Apr-07	583326	4972901	Crago	M	A	3234	151.914
1531	10-Apr-07	583278	4972599	Crago	M	Y	2752	151.923
1532	10-Apr-07	583280	4972594	Crago	M	Y	2550	151.934
1533	6-Apr-07	623766	4994869	McFarland	M	A	3138	151.942
1534	6-Apr-07	623813	4994912	McFarland	M	A	3046	151.956
1535	10-Apr-07	583324	4972905	Crago	M	A	2958	151.895
1536	8-May-07	632577	5029924	Squaw Creek	M	A	3230	N/A
1537	8-May-07	632419	5029864	Squaw Creek	M	A	2804	N/A
1538	8-May-07	632427	5029824	Squaw Creek	M	A	3146	N/A
1539	8-May-07	632308	5029856	Squaw Creek	M	A	3051	N/A
1540	8-May-07	632283	5029860	Squaw Creek	M	A	3190	N/A
1541	8-May-07	632251	5029908	Squaw Creek	M	A	2962	N/A
1542	8-May-07	632296	5029969	Squaw Creek	M	A	2500	N/A
1543	8-May-07	632281	5029958	Squaw Creek	M	A	2900	N/A
1544	8-May-07	632356	5029936	Squaw Creek	M	A	3190	N/A
1545	8-May-07	632099	5029946	Squaw Creek	M	A	2806	N/A
1546	8-May-07	594446	4989880	Widdoss	M	Y	2316	151.175
1547	9-May-07	605043	4982559	Two Top	M	A	2926	151.824
1548	9-May-07	583447	4972548	Crago	M	A	2828	151.895
1549	9-May-07	583149	4972598	Crago	M	Y	2310	151.914
1550	9-May-07	583115	4972531	Crago	M	A	3134	151.923
1601	16-May-06	586803	5042787	Valley Creek	M	Y	2352	N/A
1604	16-May-06	586476	5042810	Valley Creek	M	A	2874	N/A
1606	16-May-06	586717	5042928	Valley Creek	M	Y	2414	N/A
1607	16-May-06	586319	5042651	Valley Creek	M	A	2868	N/A
1608	16-May-06	586522	5042693	Valley Creek	M	A	3170	N/A
1609	16-May-06	586685	5042726	Valley Creek	M	A	3002	N/A
1610	16-May-06	586528	5042756	Valley Creek	M	A	2922	N/A
1611	16-May-06	586794	5042842	Valley Creek	M	Y	2298	N/A
1612	16-May-06	586799	5042754	Valley Creek	M	A	2864	N/A
1613	16-May-06	586671	5042868	Valley Creek	M	A	2918	N/A
1614	16-May-06	586660	5042780	Valley Creek	M	A	2738	N/A
1615	16-May-06	586597	5042715	Valley Creek	M	A	2852	N/A
1616	16-May-06	586509	5042708	Valley Creek	M	A	2990	N/A
1617	16-May-06	586433	5042659	Valley Creek	M	A	2920	N/A
1618	16-May-06	586317	5042837	Valley Creek	M	A	3034	N/A
1619	16-May-06	586459	5042861	Valley Creek	M	A	2896	N/A

[a] UTM coordinates in NAD 27, zone 13.
[b] Sex classification are: F-female, M-male, and unk-unknown.
[c] Age classification are: A-adult, Y-yearling, and C-hatch year chick.

CHAPTER 2 – BROOD-REARING SUCCESS AND RESOURCE SELECTION OF GREATER SAGE-GROUSE IN NORTHWESTERN SOUTH DAKOTA

INTRODUCTION

Knowledge of seasonal habitat selection and associated survival is important in developing management strategies for sensitive wildlife species. Concerns that greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse) populations may be declining, date back > 90 years (Hornaday 1916). In the past decade, at least seven petitions have been filed to list sage-grouse under the Endangered Species Act (ESA) of 1973 (Connelly et al. 2004). More recently, data suggest that sage-grouse populations have declined range-wide at a rate of 2.0% per year since 1965 (Connelly et al. 2004). Sage-grouse population estimates in South Dakota declined steadily from 1973 to 1997, but appeared to recover some from 1997 to 2002 (Smith 2003, Connelly et al. 2004). However, the data in South Dakota were inconsistent and firm conclusions could not be made (Connelly et al. 2004). In addition, information is lacking on the ecological requirements of sage-grouse in western South Dakota.

Initial sage-grouse brood-rearing sites are typically in close proximity of nest sites and must provide high invertebrate abundance and diversity. Invertebrates are necessary for growth, development and survival of sage-grouse chicks (Johnson and Boyce 1990). Invertebrates continue to be important in the development and survival of sage-grouse chicks >3 weeks of age (Johnson and Boyce 1990), as chicks include greater amounts of forbs in their diet after 3 weeks (Klebenow and Gray 1968). Chicks that fed in forb-rich habitats gained more weight than when they fed in forb-poor habitats (Huwer 2004) and areas with greater forb cover may attract higher numbers of invertebrates (Jamison et al. 2002). Greater invertebrate abundance may explain why sage-grouse tend to select areas with higher forb cover (Drut et al. 1994*a*, Apa 1998, Sveum et al. 1998, Holloran 1999).

Estimates of sage-grouse chick survival are limited, and have not been based on standardized time periods, thus making comparisons among studies difficult (Beck et al. 2006). Chick survival during the first 50 days post-hatch is generally low ranging from 18 – 33% (Schroeder 1997, Aldridge and Brigham 2001). Juvenile sage-grouse survival is greater ranging from 64% to 86% for chicks 10 weeks old to about 40 weeks (Beck et al. 2006). Combined, survival from hatch to first breeding season is estimated to be about 10% (Crawford et al. 2004). To our knowledge, no study has attempted, or been able to follow sage-grouse chicks from hatch to recruitment of 1 March.

Sage-grouse in northwestern South Dakota occupy transitional habitats between the northern wheatgrass-needlegrass prairie that dominates most of the Dakotas and the big sagebrush plains of Wyoming (Johnson and Larson 1999). In South Dakota, sage-grouse are imperiled because of rarity or some factor(s) making them very vulnerable to extinction within the state (South Dakota Department of Game, Fish, and Parks 2006). The objectives of this study were to develop an understanding of brood-rearing survival, home range, and resource selection of sage-grouse in northwestern South Dakota. This information will be useful in developing conservation and management plans for sage-grouse in South Dakota and other eastern fringe populations.

METHODS

Data Collection

Female Capture – We identified six active sage-grouse leks for which we had landowner cooperation for trapping. We captured female sage-grouse with large nets by spotlighting from all-terrain vehicles between March 2006-2007 and mid-April 2006-2007 (Giesen et al. 1982). Females were weighed and equipped with a 22-g necklace-style transmitter, which were ~1.4% of mean female sage-grouse body mass and a life-expectancy of 434 days. Transmitters could be detected from approximately 2.0 to 5.0 km from the ground and were equipped with an 8-hour mortality switch. Females were classified as adults (≥2 yr old) or yearlings (≤1 yr old) based upon primary wing feather characteristics (Eng 1955, Crunden 1963). The South Dakota State University Institutional Animal Care and Use Committee approved trapping and handling techniques, and study design (Approval #07-A032).

Monitoring and Chick Capture – We located radio-marked female sage-grouse twice each week throughout the nesting season. For hens that successfully nested, we located these hens and broods twice each week. Broods were approached cautiously to minimize the possibility of flushing or scattering the brood, with most locations being acquired within 20 m of actual locations. When chicks reached approximately 3 and 5 weeks of age we flushed the brood and searched the area to obtain estimates of brood size. We recorded the site as brood failure if no chicks were present with a hen, and subsequent locations of the hen for 2 weeks showed no evidence of chicks.

At 7 weeks of age, we attempted to capture and radio-mark as many chicks in each remaining brood as possible. Aided by radio-telemetry of the female, chicks were captured at night by a 3-5 person crew using a spotlight. We counted chicks that flew off during chick capture to estimate survival to 7 weeks of age. Chicks were weighed and equipped with a 10.7 g necklace style transmitter with mortality indicator which weighed <3% of mean chick body mass at the time of capture. These transmitters had a guaranteed life-expectancy of 150 days. The South Dakota State University Institutional Animal Care and Use Committee approved all trapping and handling techniques and study design (Approval #07-A032).

We located radio-marked chicks twice each week to obtain survival estimates. Field necropsies were conducted to identify primary predators. Dead birds that yielded testable carcasses (i.e., brain, wing or leg bones, internal organs, or spinal column present) were tested for West Nile virus (WNv) infections using real-time polymerase chain reaction (Shi 2001) and immunohistochemistry (Kiupel et al. 2003).

Habitat Measurements - We characterized vegetation at sites used by females with broods about 12.6 ± 0.6 days after the location. Two 50 m transects were established in the north-south cardinal directions. A modified Robel pole (Robel et al. 1970, Benkobi et al. 2000) was used to quantify visual obstruction readings (VOR) and maximum grass height at 10 m intervals (n = 11). We estimated sagebrush (*Artemisia. tridentata spp.* and *A. cana spp.*) density and height at 10 m intervals (n = 11) using the point-centered-quarter method (Cottam and Curtis 1956). Canopy coverage was estimated using a 0.10 m^2 quadrat (Daubenmire 1959) at each 10 m interval. Four

Daubenmire frames were placed at the interval in an H-shape with each leg 1 m long, resulting in 44 quadrats per site. We recorded total cover, grass cover, forb cover, shrub cover, litter cover, bare ground, shrub species, grass species, and forb species cover. In addition, we measured an equal number of random sites during the same period. Random points were generated within a 10 km buffer of capture leks in a Geographic Information System (GIS) (ESRI, Inc. ArcMap 9.1, Redlands, CA.). Random points were not sampled if they were on a road, in a road ditch, or on private land we did not have access.

Data Analyses

Survival – We estimated apparent survival for chicks at 3, 5, and 7 weeks of age. Mean hatch date of first nests (31 May) was used as the starting point for chick survival. Broods <7 weeks old were censored from the analysis if we witnessed brood-mixing (>1 female present), or chick-adoption (more chicks present than hatched). If the female died before chicks reached 7 weeks of age, we assumed complete brood loss. For chicks that were radio-marked at 7 weeks, we used a Kaplan-Meier product-limit method (Kaplan and Meier 1958) modified for staggered entry (Pollock et al. 1989) starting at the 7-week apparent survival rate. We monitored chicks at least once each week until they were recruited into the population (1 March). We used Program CONTRAST (Hines and Sauer 1989) to test for differences between years, with a critical value of $\alpha \leq 0.05$.

Because some carcasses of chicks were not suitable for testing for WNv infections, we estimated a minimum and maximum WNv mortality rate during the peak WNv transmission period of 12 July through 31 September for chicks (Walker et al. 2007). Minimum mortality rates were based on confirmed WNv mortalities, while maximum mortality rates were based on total mortalities minus negative cases and included mortalities where the carcass was not testable, no carcass was recovered and inconclusive tests (Walker et al. 2007).

Brood Home Range – We used the home range extension (Rodgers et al. 2007) in a Geographic Information System (GIS) (ESRI, Inc. ArcMap 9.1, Redlands, CA.) to calculate 50% and 95% adaptive kernel brood-rearing home ranges. Home ranges were estimated for broods with at least 18 locations between hatch and 31 August. If a female was monitored both years, only the home range with the most points was used to reduce dependency in our data set.

Resource Selection – All measurements were summarized to a value for the site. Sagebrush density and height was estimated from a maximum likelihood estimate (Pollard 1971). Canopy coverage values were to mid-point values of categories and summarized to an average value for the site. To reduce biologically insignificant variables, we screened canopy coverage variables and excluded any variables with canopy coverage less than 2% on sites which they were present. We then conducted a principal components analysis to distinguish important variables that captured the variation among sites. We could not discriminate between early (<5 weeks of age) and late brood sites (5 to 11 weeks of age), thus we combined early and late brood-rearing sites to test for overall habitat selection.

We identified 8 variables (Table 8) with a year effect to investigate sage-grouse brood habitat resource selection. These included: sagebrush density, visual obstruction, maximum grass height, total cover, grass cover, sagebrush cover, bluegrass (*Poa spp.*)

cover, and Japanese brome (*Bromus japanicus*) cover. Year was considered a design variable in all candidate models. We used an information theoretic approach (Burnham and Anderson 2002) with nominal logistic regression to estimate the importance of various *a priori* and *post-hoc* exploratory models in SAS JMP (2005 SAS Institute Inc.). Due to a small sample size with respect to the number of parameters estimated, AIC*c* (Akaike's Information Criterion) was used. Model predictive strength was estimated using a receiver operation characteristic curve (ROC) with values between 0.7 and 0.8 considered as acceptable discrimination and values higher than 0.8 were considered excellent discrimination (Hosmer and Lemeshow 2000).

RESULTS

Chick Survival

We monitored 10 and 14 broods in 2006 and 2007, respectively. Survival at 3 weeks post hatch was similar between years at 52%. Apparent chick survival to 7 weeks post-hatch, ranged between years from 31% in 2007 to 43% in 2006 (Table 9). Recruitment was estimated to be 9.5% (95% CI: 2.8 to 16.1%, *n* =31) in 2006 (Figure 8) and 5.1% (95% CI: 0 to 10.1%, *n* =24) in 2007 (Figure 9). There was no statistical difference between years (x^2 = 1.09, df = 1, P = 0.30), and combined recruitment for both years was 6.3% (95% CI: 2.7 – 9.9%, *n* = 55). Mortalities were attributed to WNv infections and predation by red foxes (*Vulpes vulpes*), coyotes (*Canis latrans*), bobcats (*Lynx rufus*), long-tailed weasels (*Mustela frenata*), and red-tailed hawks (*Buteo jamaicensis*).

Between 12 July and 31 September, WNv infection was attributed ≥ 6.5% (95% CI: 0 – 15.1%, n =31) of chick mortalities in 2006, but may have caused up to 71.0% (95% CI: 55.0 – 86.9%, n =31) of mortalities (Table 10). In 2007 the minimum WNv mortality rate was 20.8% (95% CI: 4.6 – 37.1%, n =24) which did not differ from 2006 (x^2 = 2.32, df = 1, P = 0.13). Maximum WNv mortality rate for 2007 was 62.5% (95% CI: 43.1 – 8.19%, n =21), which also did not differ from 2006 (x^2 = 0.42, df = 1, P = 0.52).

Brood-rearing Home Range

We estimated home ranges for 15 broods. Mean 50% adaptive kernel home range was 7.59 ± 2.35 km^2 and did not vary between years (x^2 = 1.498, df = 1, P = 0.221). Mean 95% adaptive kernel home range was 51.81 ± 16.31 km^2 and did not vary between years (x^2 = 1.279, df = 1, P = 0.258). The largest estimated 50 and 95% adaptive kernel home ranges were 31.39 km^2 and 201.76 km^2 (n = 21), respectively, while the smallest home ranges were 0.22 km^2 (n = 22) and 1.48 km^2, respectively.

Resource Selection

We sampled 59 and 60 brood sites and 56 and 60 random sites in mid June through August 2006 and 2007, respectively. All variables were significantly different between years for either brood or random sites, thus we applied a design variable, year, to all logistic models (Table 11). Brood-rearing sites had higher visual obstruction, taller grass heights, greater total cover, grass cover, sagebrush cover, Japanese brome cover, and bluegrass cover than random sites (Table 8). In contrast, sagebrush density was higher at random sites. The best approximating model (AIC*c* weight = 0.23) indicated visual obstruction and bluegrass cover to be the best habitat predictors for brood-rearing sites (Table 11). The addition of other non-correlated habitat variables to the top model (sagebrush cover, sagebrush density, or Japanese brome), did not increase model fit. Model discrimination was acceptable with a ROC value of 0.73.

Both visual obstruction and bluegrass cover positively influenced brood-rearing site selection as parameter estimates were positive (Table 12), with visual obstruction having a slightly larger impact (Figure 10). Broods were 3.06 times (95% CI: 2.84– 3.34) more likely to select an area if visual obstruction increased by 2.54 cm, and 5.61 times (95% CI: 5.15 – 6.13) more likely to select an area if bluegrass cover increased by 5% canopy cover.

DISCUSSION

Survival

Survival of sage-grouse chicks to 3 to 4 weeks of age is generally low, ranging from 22 to 50% (Burkepile et al. 2002, Aldridge 2005, Gregg et al. 2007, Herman-Brunson 2007). We did not attach transmitters to sage-grouse chicks <1 week, but our estimated survival rate to 3 weeks (52%) was among the highest reported. Sage-grouse chick survival to 7 weeks (34%) in our study was higher than reported for a declining population in Alberta (Aldridge and Brigham 2001, Aldridge 2005), but similar to a stable population in Washington (Schroeder 1997). Our estimate to 7 weeks is conservative, as flush counts may underestimate chick survival (Aldridge and Brigham 2001). We feel that our 7 week survival estimate is fairly accurate as it was conducted at night when broods tend to group together, and the count was always conducted by at least

3 people. Furthermore, survival rates between flush counts and telemetry estimates for sage-grouse chicks at approximately 8 weeks of age have been documented to be similar (Aldridge 2005). Aldridge (2005) suggested that accuracy of flush counts increase as chicks become larger in size, making them easier to locate and flush.

Survival of sage-grouse chicks from 10 weeks through the following March, ranges from 64 to 86% (Beck et al. 2006). Sage-grouse chick survival to 1 January in North Dakota was 13 to 17% (Herman-Brunson 2007). However, our data suggest that chick survival to recruitment would be half that. Although seemingly low, our recruitment rate of 6% suggests that the index of recruitment by Crawford et al. (2004) was realistic. However, West Nile virus infections in 2006 decreased chick recruitment the next spring by about 2%. In 2007, WNv decreased chick recruitment by approximately 4%.

Using our estimates of nest initiation (95.9%), breeding success (47.9%), clutch size (8.0), egg hatchability (78.3%), 1:1 sex ratio, and recruitment rates of 5.1 and 9.5%, annual survival of adult hens would need to be 93 to 86% to maintain a stable population, respectively. If recruitment increased to 15 or 20%, hen survival necessary for a stable population would be lower at 78 and 71%, respectively. The latter estimate may be more reasonable for sage-grouse populations as annual female survival varies from 37 to 78% (Connelly et al. 2004). However, fluctuations of nesting parameters and recruitment could substantially alter these estimates, but chick recruitment of >10% should help maintain stable populations even in years with poor nesting success or extreme WNv infections.

Brood-rearing Home Range

Few studies have attempted to quantify brood-rearing home ranges for sage-grouse (Wallestad 1971, Connelly and Markham 1983, Drut et al. 1994*a*). However, home range estimates have ranged widely from 0.51 km^2 (Wallestad 1971) to 51.00 km^2, Drut et al. 1994*a*). Differences in home range size have been suggested to be related to forb availability with home ranges being both smaller and larger in areas with increased forb abundance (Drut et al. 1994*a*, Connelly and Markham 1983). However, forbs did not appear to be an important predictor variable in our analyses, suggesting other variables (e.g., visual obstruction, sagebrush distribution) may better explain why home range estimates in South Dakota were rather large.

Resource Selection

Visual obstruction and bluegrass cover were identified to be the best variables at predicting brood-rearing sites for sage-grouse in South Dakota. Increased visual obstruction provides protection from predators, and perhaps more importantly, greater herbaceous biomass which is correlated with greater invertebrate abundance (Healy 1985, Rumble and Anderson 1996). Invertebrates are an important component of sage-grouse chicks' diets (Johnson and Boyce 1990, Drut et al. 1994*b*). Female sage-grouse tend to move their broods from upland, nesting-type areas, to more mesic, greener areas later in the summer (Peterson 1970, Dunn and Braun 1986, Sveum et al. 1998). Adapted to a broad range of soils, bluegrass is common on sites with abundant soil moisture in South Dakota (Stubbendieck et al. 1997). Although we were not able to differentiate between early and late brood-rearing habitats, broods may be selecting areas with greater bluegrass cover for the increased invertebrate abundance that greener areas tend to provide.

Sage-grouse brood-rearing habitats are generally linked to forb abundance (Drut et al. 1994a, Apa 1998, Sveum et al. 1998, Holloran 1999). Forbs not only provide direct food resources (Drut et al. 1994*b*), but increased invertebrate abundance (Jamison et al. 2002). We did not note a difference in forb cover between brood (7.6%) and random sites (7.1%), and it was not an important predictor in our analysis, while other studies have shown sage-grouse broods to use areas with forb cover up to 41.3% (Schoenberg 1982). In contrast, females with broods in South Dakota selected areas with higher grass cover that was greater than typically reported in the literature (Klott and Lindzey 1990, Drut et al. 1994*b*, Sveum et al. 1998, Thompson et al. 2006). Western South Dakota forms a transition zone between the northern wheatgrass-needlegrass prairie that dominates most of the Dakotas and the big sagebrush plains of Wyoming (Johnson and Larson 1999), and possesses a greater grass component compared to the shrub-steppe region (Lewis 2004). Grass structure is highly correlated with visual obstruction, which, provides increased protection from predators and invertebrate abundance. Therefore, forbs may be more important to sage-grouse brood-rearing habitat in core sagebrush areas (e.g., Columbia Basin) where there is more bareground, while grass structure may be more important for broods on the eastern edge of their range (e.g., South Dakota). In Alberta, another edge-type habitat, key brood habitat in moist areas and drainages was suggested to be limiting sage-grouse productivity (Aldridge and Brigham 2002).

MANAGEMENT IMPLICATIONS

With possible listing under the Endangered Species Act, sage-grouse conservation and preservation will be a priority for many western land management agencies. For sage-grouse brood-rearing habitat in western South Dakota and other eastern edge populations, management strategies should focus on maintaining or increasing grass structure (cover and height) which provides high visual obstruction for sage-grouse broods. In addition, managers should promote and protect greener areas during mid to late summer. These areas typically have higher production and invertebrate abundance. This may include government programs that defer or eliminate grazing and haying operations in these areas.

Domestic livestock grazing by cattle (*Bos taurus*) and sheep (*Ovis aries*) has been shown to have both positive and negative impacts on rangeland condition and health in the sagebrush ecosystem (Holechek et al. 2001) and sage-grouse habitats (Beck and Mitchell 2000). Grazing by sheep can be an effective way of reducing sagebrush (Baker et al. 1976) which could negatively affect sage-grouse productivity in South Dakota, particularly during the nesting period. High intensity cattle grazing of the herbaceous understory (grasses and forbs), may allow for greater forb and sagebrush growth (Paige and Ritter 1999) but that may also negatively influence sage-grouse productivity by decreasing plant biomass and protective cover and consequently, reduce insect abundance. However, light or moderate grazing in dense, grassy meadows increased sage-grouse use (Klebenow 1982) but overgrazing of these areas reduced sage-grouse habitat (Klebenow 1985, Oakleaf 1971) and were avoided by sage-grouse (Klebenow 1982).

WNv was an important factor for sage-grouse chick survival. Management practices to mitigate its affect on sage-grouse chick survival appear to be minimal and tied to anthropogenic water sources, particularly coal-bed natural gas ponds (Walker et al. 2007). Unless sage-grouse develop stronger immunity to this disease, their future looks uncertain. However, small increases in chick recruitment, either through increased nesting success or increased chick survival should have positive effects on sage-grouse populations.

With 75% of the study area in private ownership and the patchy network of public land; sage-grouse conservation and persistence lies in hands of private landowners. To increase sage-grouse habitats, long-term (>20 yrs) partnerships and incentives with ranchers will be imperative. This will require cooperation from state wildlife agencies, federal land management agencies, local natural resource conservation districts, and committed landowners. Forming a South Dakota sage-grouse working group may be in order to accomplish this goal, as many landowners were interested in sage-grouse conservation.

Table 8. Observed mean values for habitat variables between greater sage-grouse brood-rearing and random sites, and between years used in logistic regression in northwestern South Dakota, USA, using MRPP (Mielke and Berry 2001) 2006-2007.

	Brood			**Random**			**Both Years**		
Variable	**2006 (*n*=59)**	**2007 (*n*=60)**	**P-value**	**2006 (*n*=56)**	**2007 (*n*=60)**	**P-value**	**Brood (*n*=119)**	**Random (*n*=116)**	**P-value**
Sagebrush Density (plants/m^2)	0.3	0.5	<0.01	0.7	0.4	<0.01	0.4	0.5	0.08
Sagebrush Cover (%)	4.6	4.7	0.94	4.5	2.8	0.03	4.6	3.6	0.04
Visual Obstruction (cm)	5.4	7.1	0.12	2.3	4.7	<0.01	6.2	3.5	<0.01
Grass Height (cm)	23.3	37.5	<0.01	19.2	31.9	<0.01	30.5	25.7	<0.01
Total Cover (%)	61.3	55.6	<0.01	51.0	51.0	1.00	58.4	51.0	<0.01
Grass Cover (%)	34.4	28.3	<0.01	28.6	24.8	0.26	31.3	26.6	<0.01
Japanese Brome Cover (%)	10.4	9.9	0.66	4.9	11.4	<0.01	10.1	8.3	0.04
Bluegrass Cover (%)	5.9	2.3	<0.01	3.8	2.2	<0.01	4.0	3.0	0.08

Table 9. Apparent greater sage-grouse chick survival to 7 weeks post hatch, and recruitment as of 1 March using a Kaplan-Meier product-limit method (Kaplan and Meier 1958) modified for staggered entry (Pollock et al. 1989) in northwestern South Dakota, USA, 2006-2008. Estimated survival rates given as mean (95% CI).

Year	**3 Week Survival (Apparent)**	**5 Week Survival (Apparent)**	**7 Week Survival (Apparent)**	**Recruitment (Apparent + Kaplan-Meier)**
2006	52.4% (*n* = 42)	45.2% (*n* = 42)	42.9% (*n* = 42)	9.5% (2.8 – 16.1%, *n* = 31)
2007	52.2% (*n* = 115)	41.7% (*n* = 115)	31.3% (*n* = 115)	5.1% (0 – 10.1%, *n* = 24)
Combined	52.2% (*n* = 157)	42.7% (*n* = 157)	34.3% (*n* = 157)	6.3% (2.7 – 9.9%, *n* = 55)

Table 10. West Nile virus (WNv) mortality rates and testing for greater sage-grouse chicks during the peak WNv transmission period (12 July – 31 September) in northwestern South Dakota, USA, 2006-2007. Estimated minimum and maximum mortality given as mean (95% CI) after Walker et al. (2007).

Year	**No. Monitored**	**No. Mortalities**	**No. Tested**	**No. Positive**	**No. Negative**	**No. Inconclusive**	**Minimum WNv mortality rate**	**Maximum WNv mortality rate**
2006	31	22	10	2 (23 July - 22 Aug.)	0	8	6.5% (0 – 15.1%)	71.0% (55.0 – 86.9%)
2007	24	18	10	5 (8 Aug. – 14 Sept.)	3	2	20.8% (4.6 – 37.1%)	62.5% (43.1 – 81.9%)

Table 11. Results from logistic regression models predicting greater sage-grouse brood-rearing sites (n = 119) versus random sites (n = 116) in northwestern South Dakota, USA, 2006-2007.

Model[a]	**K**[b]	**AIC*c***	**Δ AIC*c***[c]	***wi***[d]
Visual Obstruction + Bluegrass Cover	5	303.547	0.000	0.231
Visual Obstruction + Bluegrass Cover + Sagebrush Cover	6	304.275	0.728	0.160
Visual Obstruction + Bluegrass Cover + Sage Density	6	304.455	0.908	0.146
Visual Obstruction + Bluegrass Cover + Japanese Brome Cover	6	304.798	1.251	0.123
Visual Obstruction + Bluegrass Cover + Japanese Brome Cover + Sage Density	7	305.459	1.911	0.089
Herbaceous Cover + Bluegrass Cover + Grass Height.	6	305.503	1.956	0.087

[a] For ease of interpretation, year variable was excluded from model column. See Appendix 3 for full model results
[b] Number of habitat parameters plus intercept, SE, and year.
[c] Change in AIC*c* value
[d] Model weight

Table 12. Parameter Estimates, odds ratios, and corresponding confidence intervals for the best-approximating model of greater sage-grouse brood-rearing sites versus random sites in northwestern South Dakota, 2006-2007.

	Parameter			**Odds**		
Variable	**Estimate**	**Lower 95%CI**	**Upper 95%CI**	**Ratio**	**Lower 95%CI**	**Upper 95%CI**
Visual Obstruction	0.186	0.110	0.272	1.204	1.116	1.313
Bluegrass	0.114	0.029	0.204	1.121	1.029	1.226

2006 Chick Survival
Apparent & Kaplan-Meier

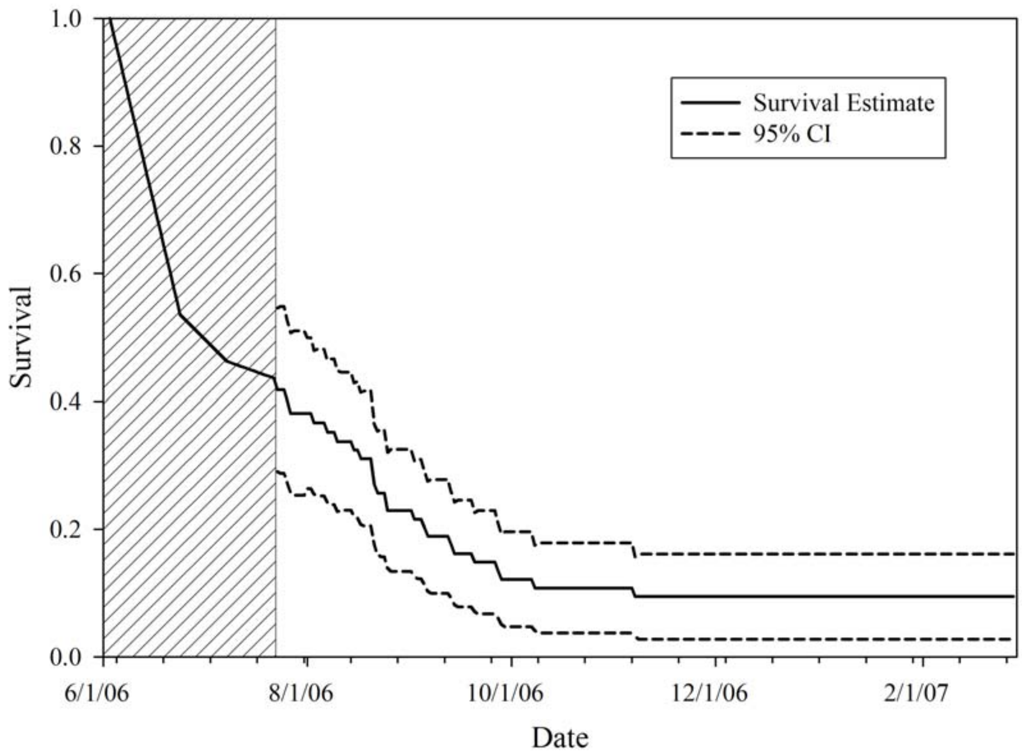

Figure 8. Greater sage-grouse apparent chick survival to 7 weeks post hatch (dashed area), and recruitment as of 1 March 2007 using a Kaplan-Meier product-limit method (Kaplan and Meier 1958) modified for staggered entry (Pollock et al. 1989) in northwestern South Dakota, USA, 2006-2007. A sample size of $n = 31$, was used in the Kaplan-Meier analysis.

2007 Chick Survival
Apparent & Kaplan-Meier

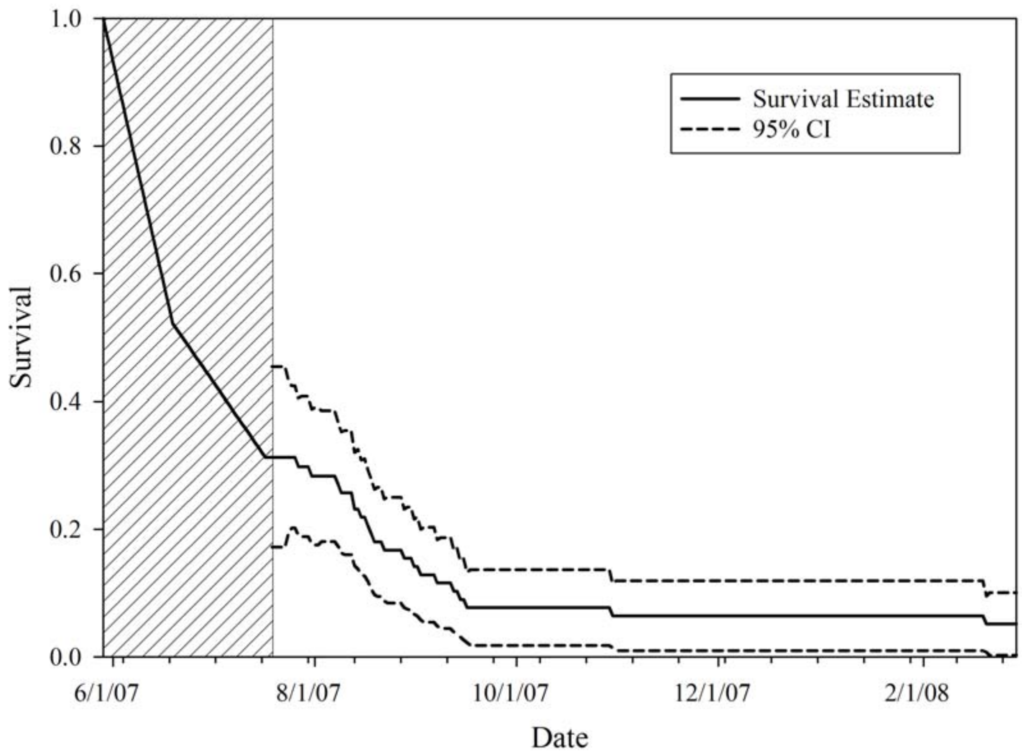

Figure 9. Greater sage-grouse apparent chick survival to 7 weeks post hatch (dashed area), and recruitment as of 1 March 2008 using a Kaplan-Meier product-limit method (Kaplan and Meier 1958) modified for staggered entry (Pollock et al. 1989) in northwestern South Dakota, USA, 2007-2008. A sample size of *n* = 24, was used in the Kaplan-Meier analysis.

Effect of Visual Obstruction and Bluegrass Cover On Brood-rearing Habitat Selection

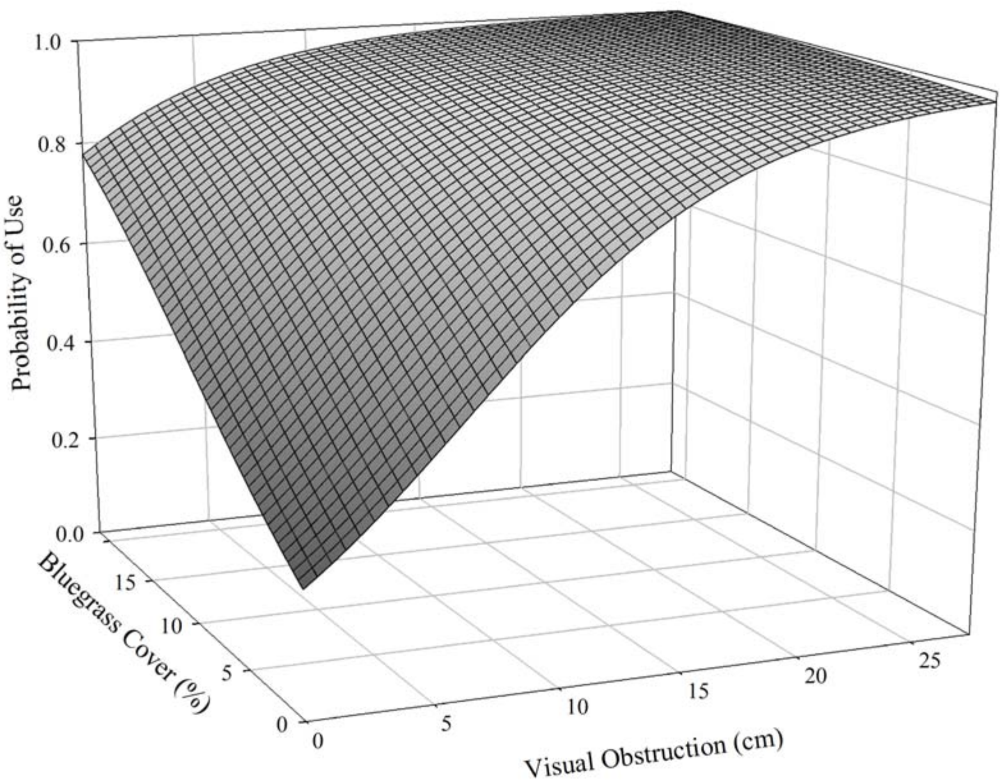

Figure 10. Effect of visual obstruction and bluegrass cover on greater sage-grouse brood-rearing habitat selection in northwestern South Dakota, USA, 2006-2007. Probability of use derived from parameter estimates in best approximated model (visual obstruction + bluegrass cover).

LITERATURE CITED

Aldridge, C. L. 2005. Identifying habitats for persistence of greater sage-grouse (*Centrocercus urophasianus*) in Alberta, Canada. Dissertation, University of Alberta, Edmonton, Canada.

Aldridge, C. L. and R. M. Brigham. 2001. Nesting and reproductive activities of greater sage-grouse in a declining northern fringe population. Condor 103:537-543.

Aldridge, C. L. and R. M. Brigham. 2002. Sage-grouse nesting and brood habitat use in southern Canada. Journal of Wildlife Management 66:433-444.

Apa, A. D. 1998. Habitat use and movements of sympatric sage and Columbian sharp-tailed grouse in southeastern Idaho. Dissertation, University of Idaho, Moscow, USA.

Baker, M. F., R. L. Eng. J. S. Gashwiler, M. H. Schroeder, and C. E. Braun. 1976. Conservation committee report on the effects of alteration of sagebrush communities on the associated avifauna. Wilson Bulletin 88:165-170.

Beck, J. L., and D. L. Mitchell. 2000. Influences of livestock grazing on sage-grouse habitat. Wildlife Society Bulletin 28:993-1002.

Beck, J. L., K. P. Reese, J. W. Connelly, and M. B. Lucia. 2006. Movements and survival of juvenile greater sage-grouse in southeastern Idaho. Wildlife Society Bulletin 34:1070-1078.

Benkobi, L., D. W. Uresk, G. Schenbeck, and R. M. King. 2000. Protocol for monitoring standing crop in grasslands using visual obstruction. Journal of Range Management 53:627-633.

Burkepile, N. A., J. W. Connelly, D. W. Stanley, and K. P. Reese. 2002. Attachment of radiotransmitters to one-day-old sage-grouse chicks. Wildlife Society Bulletin 30:93-96.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multi-model inference: a practical information theoretic approach. Second edition. Springer-Verlag, New York, New York, USA.

Connelly, J. W., and O. D. Markham. 1983. Movements and radionuclide concentrations of sage-grouse in southeastern Idaho. Journal of Wildlife Management 47:169-177.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming.

Cottam, G., and J. T. Curtis. 1956. The use of distance measures in phytosociological sampling. Ecology 37:451-460.

Crawford, J. A., R. A. Olson, N. E. West, J. C. Mosley, M. A. Schroeder, T. D. Whitson, R. F. Miller, M. A. Gregg, and C. S. Boyd. 2004. Ecology and management of sage-grouse and sage-grouse habitat. Journal of Range Management 57:2-19.

Crunden, C. W. 1963. Age and sex of sage grouse from wings. Journal of Wildlife Management 27:846-849.

Daubenmire, R. F. 1959. A canopy-coverage method of vegetation analysis, Northwest Science 33:43-64.

Drut, M. S., J. A. Crawford, and M. A. Gregg. 1994*a*. Brood habitat use by sage-grouse in Oregon. Great Basin Naturalist 54:170-176.

Drut, M. S., W. H. Pyle, and J. A. Crawford. 1994*b*. Diets and food selection of sage-grouse chicks in Oregon. Journal of Range Management 47:90-93.

Dunn, P. O., and C. E. Bruan. 1986. Summer habitat use by adult female and juvenile sage-grouse. Journal of Wildlife Management 50:228-235.

Eng, R. L. 1955. A method for obtaining sage grouse age and sex ratios from wings. Journal of Wildlife Management 19:267-272.

Giesen, K. M., T. J. Schoenberg, and C. E. Braun. 1982. Methods for trapping sage grouse in Colorado. Wildlife Society Bulletin 10:224-231.

Gregg, M. A., M. R. Dunbar, and J. A. Crawford. 2007. Use of implanted radiotransmitters to estimate survival of greater sage-grouse chicks. Journal of Wildlife Management 71:646-651.

Healy, W. M., 1985. Turkey poult feeding activity, invertebrate abundance and vegetation structure. Journal of Wildlife Management 49:466-472.

Herman-Brunson, K. H. 2007. Nesting and brood-rearing habitat selection of Greater sage-grouse and associated survival of hens and broods at the edge of their historic distribution. Thesis, South Dakota State University, Brookings, USA.

Hines, J. E., and J. R. Sauer. 1989. Program CONTRAST - a general program for the analysis of several survival or recovery rate estimates. Fish and Wildlife Technical Report 24:1-7.

Holechek, J. L., R. D. Pieper, and C. H. Herbel. 2001. Range management principles and practices. Fourth edition. Prentice Hall, Upper Saddle River, New Jersey, USA.

Holloran, M. J. 1999. Sage-grouse (*Centrocercus urophasianus*) seasonal habitat use near Casper, Wyoming. M.S. Thesis, University of Wyoming, Laramie, USA.

Hornaday, W. T. 1916. Save the sage-grouse from extinction, a demand from civilization to the western states. New York Zoological Park Bulletin 5:179-219.

Hosmer, D. W., and S. Lemeshow. 2000. Applied Logistic Regression, Second Edition. John Wiley and Sons Inc. Publication, New York, NY, USA.

Huwer, S. L. 2004. Evaluating greater sage-grouse brood habitat use using human-imprinted chicks. M.S. Thesis, Colorado State University, Fort Collins, USA.

Jamison, B. E., R. J. Robel, J. S. Pontius, and R. D. Applegate. 2002. Invertebrate biomass: associations with Lesser Prairie Chicken habitat use and sand sagebrush density in southwestern Kansas. Wildlife Society Bulletin. 30:517-526.

Johnson, G. D., and M. S. Boyce. 1990. Feeding trials with insects in the diet of sage-grouse chicks. Journal of Wildlife Management 54:89-91.

Johnson, J. R., and G. E. Larson. 1999. Grassland plants of South Dakota ant the northern Great Plains. South Dakota State University. Brookings, South Dakota, USA.

Kaplan, E., and P. Meier. 1958. Non parametric estimation from incomplete observation. Journal of the American Statistical Association 53:457-481.

Kiupel, M. H., H. A. Simmons, S. D. Fitzgerald, A. Wise, J. G. Sikarskie, T. M. Cooley, S. R. Hollamby, and R. Maes. 2003. West Nile virus infection in eastern fox squirrels (*Sciurus niger*). Veterinary Pathology 40:703-707.

Klebenow, D. A. 1982. Livestock grazing interactions with sage-grouse. Pages 113-123 *in* J. M. Peek and P. D. Dalke, editors. Proceedings of the Wildlife-Livestock Relationships Symposium, 20-22 April 1981, Coeur d'Alene, Idaho. Proceeding 10, University of Idaho Forestry, Wildlife, and Range Experiment Station, Moscow, USA.

Klebenow, D. A. 1985. Habitat management of sage-grouse in Nevada. World Pheasant Association Journal 10:34-46.

Klebenow, D. A., and G. M. Gray. 1968. Food habits of juvenile sage-grouse. Journal of Range Management 12:80-83.

Klott, J. H., and F. G. Lindzey. 1990. Brood habitats of sympatric sage-grouse and Columbian sharp-tailed grouse in Wyoming. Journal of Wildlife Management 54:84-88.

Lewis, A. R. 2004. Sagebrush steppe habitats and their associated bird species in South Dakota, North Dakota, and Wyoming: Life on the edge of the sagebrush ecosystem. Dissertation, South Dakota State University, Brookings, South Dakota, USA.

Oakleaf, R. J. 1971. The relationship of sage-grouse to upland meadows in Nevada. Thesis, University of Nevada, Reno, USA.

Paige, C. and S. A. Ritter. 1999. Birds in a sagebrush sea: managing sagebrush habitats for bird communities. Partners in Flight Western Working Group, Boise, ID.

Peterson, J. G., 1970. The food habits and summer distribution of juvenile sage-grouse in central Montana. Journal of Wildlife Management 34:147-155.

Pollard, J. H. 1971. On distance estimators of density in randomly distributed forests. Biometrics 27:991-1002.

Pollock, K., S. Winterstein, C. Bunck, and P. Curtis. 1989. Survival analysis in telemetry studies: the staggered entry design. Journal of Wildlife Management 53:7-15.

Robel, R. J., J. N. Briggs, A. D. Dayton, and L. C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23:295-298.

Rodgers, A.R., A.P. Carr, H.L. Beyer, L. Smith, and J.G. Kie. 2007. HRT: Home Range Tools for ArcGIS. Version 1.1. Ontario Ministry of Natural Resources, Centre for Northern Forest Ecosystem Research, Thunder Bay, Ontario, Canada.

Rumble, M. A., and S. H. Anderson. 1996. Habitat selection of Merriam's turkey (*Meleagris gallopavo merriami*) in the Black Hills, South Dakota, American Midland Naturalist 136:157-171.

Schoenberg, T. J., 1982. Sage-grouse movements and habitat selection in North Park, Colorado. Thesis, Colorado State University, Fort Collins, USA.

Schroeder, M. A. 1997. Unusually high reproductive effort by sage grouse in a fragmented habitat in north-central Washington. Condor 99:933-941.

Shi. P. 2001. High-throughput detection of West Nile virus RNA. Journal of Clinical Microbiology 39:1264-1271.

Smith, J. 2003. Greater sage grouse on the edge of their range: leks and surrounding landscapes in the Dakotas. M.S. Thesis, South Dakota State University, Brookings, USA.

South Dakota Department of Game, Fish, and Parks. 2006. South Dakota comprehensive wildlife conservation plan. South Dakota Dept. of Game, Fish, and Parks, Pierre, Wildlife Division Report 2006-08.

Stubbendieck, J., S. L. Hatch, and C. H. Butterfield. 1997. North American Range Plants. Fifth edition. University of Nebraska Press, Lincoln, Nebraska, USA.

Sveum, C. M., J. A. Crawford, and W. D. Edge. 1998. Use and selection of brood-rearing habitat by sage-grouse in south-central Washinton. Great Basin Naturalist 58:344-351.

Thompson, K. M., M. J. Holloran, S. J. Slater, J. L. Kuipers, and S. H. Anderson. 2006. Early brood-rearing habitat use and productivity of greater sage-grouse in Wyoming. Western North American Naturalist 66:332-342.

Walker, B. L., D. E. Naugle, K. E. Doherty, and T. E. Cornish. 2007. West Nile virus and greater sage-grouse: Estimating infection rate in a wild bird population. Avian Diseases 51:691-696.

Wallestad, R. O. 1971. Summer movement and habitat use by sage-grouse broods in Montana. Journal of Wildlife Management 35:129-136.

Appendix 4. Complete results from logistic regression models predicting greater sage-grouse brood-rearing sites (n = 119) versus random sites (n = 116) in northwestern South Dakota, USA, 2006-2007.

Model[a]	**K**[b]	**AIC*c***	**Δ AIC*c***[c]	***wi***[d]
Visual Obstruction + Bluegrass	5	303.547	0.000	0.231
Visual Obstruction + Bluegrass + Sagebrush Cover	6	304.275	0.728	0.160
Visual Obstruction + Bluegrass + Sage Density	6	304.455	0.908	0.146
Visual Obstruction + Bluegrass + Jap. Brome	6	304.798	1.251	0.123
Visual Obstruction + Bluegrass + Jap. Brome + Sage Density	7	305.459	1.911	0.089
Total Cover + Bluegrass + Grass Hgt.	6	305.503	1.956	0.087
Grass Hgt. + Total Cover	5	307.403	3.856	0.034
Visual Obstruction + Sagebrush Cover	5	307.961	4.414	0.025
Visual Obstruction	4	308.259	4.712	0.022
Grass Hgt. + Sage Density + Bluegrass	6	308.829	5.281	0.016
Grass Hgt. + Total Cover + Sage Density	6	309.376	5.829	0.013
Visual Obstruction + Jap. Brome	5	309.416	5.869	0.012
Grass Hgt. + Bluegrass	5	309.893	6.346	0.010
Grass Hgt. + Bluegrass + Sagebrush Cover	6	310.219	6.671	0.008
Visual Obstruction + Sage Density	5	310.330	6.783	0.008
Bluegrass + Sage Density + Grass Hgt. + Jap. Brome	7	310.395	6.848	0.008
Grass Hgt. + Sagebrush Cover	5	312.905	9.358	0.002
Grass Hgt. + Grass Cover	5	313.128	9.581	0.002
Grass Hgt.	4	313.669	10.122	0.001
Sagebrush + Grass Hgt. + Jap. Brome	6	314.112	10.565	0.001
Grass Hgt. + Sagebrush Density	5	314.348	10.800	0.001
Grass Hgt. + Jap. Brome	5	315.110	11.563	0.001
Sagebrush + Total Cover	5	318.870	15.323	0.000
Total Cover + Bluegrass	5	320.013	16.465	0.000
Total Cover	4	320.699	17.152	0.000
Grass Cover + Sagebrush Cover	5	321.890	18.343	0.000
Sage Density + Total Cover	5	322.539	18.992	0.000
Grass Cover + Bluegrass	5	324.656	21.109	0.000
Grass Cover	4	326.626	23.078	0.000
Bluegrass + Sage Density	5	326.866	23.319	0.000
Bluegrass + Jap. Brome + Sage Density	6	327.142	23.595	0.000
Bluegrass + Jap. Brome	5	328.135	24.588	0.000
Sage Density + Grass Cover	5	328.447	24.900	0.000
Bluegrass	4	328.972	25.425	0.000
Sagebrush Cover + Bluegrass	5	329.056	25.509	0.000
Sagebrush Cover + Jap. Brome	5	330.167	26.620	0.000
Sagebrush Cover	4	330.739	27.191	0.000
Sage Density	4	331.620	28.073	0.000
Jap. Brome	4	331.657	28.110	0.000
Sage Density + Jap. Brome	5	332.235	28.688	0.000

[a] For ease of interpretation, year variable was excluded from model column.
[b] Number of habitat parameters plus intercept, SE, and year.
[c] Change in AIC*c* value
[d] Model weight

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Contents

Table

Conversion Factors

Inch/Pound to SI

Multiply	By	To obtain
	Length	
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
yard (yd)	0.9144	meter (m)
	Area	
acre	4,047	square meter (m^2)
acre	0.4047	hectare (ha)
acre	0.004047	square kilometer (km^2)
section (640 acres or 1 square mile)	259.0	square hectometer (hm^2)
square mile (mi^2)	259.0	hectare (ha)
square mile (mi^2)	2.590	square kilometer (km^2)

SI to Inch/Pound

Multiply	By	To obtain
	Length	
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
meter (m)	1.094	yard (yd)
	Area	
square meter (m^2)	0.0002471	acre
hectare (ha)	2.471	acre
square kilometer (km^2)	247.1	acre
square hectometer (hm^2)	0.003861	section (640 acres or 1 square mile)
hectare (ha)	0.003861	square mile (mi^2)
square kilometer (km^2)	0.3861	square mile (mi^2)

Conservation Buffer Distance Estimates for Greater Sage-Grouse—A Review

By Daniel J. Manier, Zachary H. Bowen, Matthew L. Brooks, Michael L. Casazza, Peter S. Coates, Patricia A. Deibert, Steven E. Hanser, and Douglas H. Johnson

Introduction

This report was prepared at the request of the U.S. Department of the Interior and is a compilation and summary of published scientific studies that evaluate the influence of anthropogenic activities and infrastructure on Greater Sage-Grouse (*Centrocercus urophasianus*; hereafter, sage-grouse) populations. The purpose of this report is to provide a convenient reference for land managers and others who are working to develop biologically relevant and socioeconomically practical buffer distances around sage-grouse habitats. The framework for this summary includes (1) addressing the potential effects of anthropogenic land use and disturbances on sage-grouse populations, (2) providing ecologically based interpretations of evidence from the scientific literature, and (3) informing implementation of conservation buffers around sage-grouse communal breeding locations—known as leks.

We do not make specific management recommendations but instead provide summarized information, citations, and interpretation of findings available in scientific literature. We also recognize that because of variation in populations, habitats, development patterns, social context, and other factors, for a particular disturbance type, there is no single distance that is an appropriate buffer for all populations and habitats across the sage-grouse range. Thus, we report values for distances upon which protective, conservation buffers might be based, in conjunction with other considerations (table 1). We present this information for six categories of land use or disturbance typically found in land-use plans which are representative of the level of definition available in the scientific literature: surface disturbance (multiple causes; immediate and cumulative influences); linear features (roads); energy development (oil, gas, wind, and solar); tall structures (electrical, communication, and meteorological); low structures (fences and buildings); and activities (noise and related disruptions). Minimum and maximum distances for observed effects found in the scientific literature, as well as a distance range for possible conservation buffers based on interpretation of multiple sources, expert knowledge of the authors regarding affected areas, and the distribution of birds around leks are provided for each of the six categories (table 1). These interpreted values for buffer distances are an attempt to balance the extent of protected areas with multiple land-use requirements using estimates of the distribution of sage-grouse habitat. Conservation efforts may then focus on the overlap between potential effect zone and important habitats. We provide a brief discussion of some of the most relevant literature for each category. References associated with the minimum and maximum values in table 1 are identified in the References Cited section with corresponding symbols.

Distances in this report reflect radii around lek locations because these locations are typically (although not universally) known, and management plans often refer to these locations. Lek sites are most representative of breeding habitats, but their locations are focal points within populations, and as such, protective buffers around lek sites can offer a useful solution for identifying and conserving seasonal habitats required by sage-grouse *throughout* their life cycle. However, knowledge of local and regional patterns of seasonal habitat use may improve conservation of those important areas, especially regarding the distribution and utilization of nonbreeding season habitats (which may be underrepresented in lek-based designations).

Analytical Realities and Additional Background

Understanding the effects of multiple human land uses on sage-grouse and their habitats is complicated by the combination of environmental, ecological, and socioeconomic conditions across the species range, which includes parts of 11 U.S. States and 2 Canadian Provinces in western North America. Responses of individual birds and populations, coupled with variability in land-use patterns and habitat conditions, add variation in research results. This variability presents a challenge for land managers and planners seeking to use research results to guide management and plan for sage-grouse conservation measures.

Variability between sage-grouse populations and their responses to different types of infrastructure can be substantial across the species' range. Our interpretations attempt to encompass variability in populations (for example, migratory versus nonmigratory) and rangewide response patterns of sage-grouse to various human activities. Logical and scientifically justifiable departures from the "typical response," based on local data and other factors, may be warranted when implementing buffer protections or density limits in parts of the species' range.

Natural movement behaviors of sage-grouse have been documented by multiple studies that provide direct evidence of inter- and intraseasonal movements from a few kilometers (km) (nonmigratory populations; Berry and Eng, 1985; Connelly and others, 2004) to 20–30 km or more (Connelly and others, 2004; Fedy and others, 2012; Tack and others, 2012). An influential, telemetry-based, tracking project in central Montana indicated more than 90 percent of *breeding season* movements by male grouse were within 1.3 km (0.8 mi) of a lek and 76 percent were within 1 km of a lek (0.6 mi; Wallestad and Schladweiler, 1974). The 1-km (0.6-mi) buffer used in many management efforts was based upon this research. More recent analyses have indicated that 90–95 percent of habitat use at the population level was focused within approximately 8 km (5 miles [mi]) of several California and Nevada lek sites (Coates and others, 2013), and 95 percent of all nests were located within approximately 5 km (3.1 mi) of leks. Holloran and Anderson (2005) found that 64 percent of nests in Wyoming occurred within 5 km (3.1 mi) of leks, suggesting considerable protection of sage-grouse within these proximate habitats. In contrast, home ranges as large as 2,975 km^2 (1,149 mi^2) have been documented (Connelly and others, 2000, 2004) in some portions of the species' range. These larger distances suggest that for some populations, the minimum distance inferred here (5 km [3.1 mi]) from leks may be insufficient to protect nesting and other seasonal habitats. Based on the collective information reviewed for this study, conservation practices that address habitats falling within the interpreted distances may be expected to protect as much as 75 percent (Doherty and others, 2010) to 95 percent (Coates and others, 2013) of local population's habitat utilization.

Habitat condition, composition, structure, and distribution are important potential modifiers of the effect of human infrastructure and activities on sage-grouse populations (Dinkins and others, 2014; Walters and others, 2014). The distribution of sagebrush (*Artemisia* spp.) is a well-known biological and statistical predictor of sage-grouse response to their environment (for example, Connelly and others, 2004; Aldridge and Boyce, 2007; Hagen and others, 2007; National Technical Team, Sage Grouse, 2011; Wisdom and others, 2011; Kirol and others, 2012; Beck and others, 2014; Smith and others, 2014). Differences among sagebrush communities within a population range may also affect the impact of infrastructure. For example, primary productivity of sites is typically greater in mountain big sagebrush (*A. tridendata* ssp. *vaseyana*) communities than Wyoming big sagebrush (*A. t.* ssp. *wyomingensis*) communities (Davies and Bates, 2010).

Sage-grouse depend on sagebrush, so buffer protections may be most effective when focused on avoidance of disturbance to sagebrush that provides the keystone to sage-grouse habitat. Important sage-grouse habitats include those with >40 percent sagebrush landcover (within 5 km [3.1 mi] radial assessment area; Knick and others, 2013), sagebrush patch sizes greater than 1 km^2 (0.4 mi^2) (Aldridge and Boyce, 2007), and plot-level composition of approximately 10–30 percent sagebrush cover and >15 percent grasses and forbs (Connelly and others, 2004; Stiver and others, 2006). Avoidance of activities that increase distance between sagebrush patches or that impose barriers to dispersal could also help maintain populations (Wisdom and others, 2011; Knick and Hanser, 2011).

Various protection measures have been developed and implemented, including complete closure of important habitats, distance buffers that restrict disturbing activities within designated distances, and development-disturbance density limits within habitats (for examples see, “Policy and Rules for Development” at *http://utahcbcp.org/htm/tall-structure-info*). Timing restrictions have also commonly been employed at lek sites, primarily to reduce disturbance to breeding sage-grouse. Although specific details and implementation of these different approaches have varied, each approach has the ability (alone or in concert with others) to protect important habitats, sustain populations, and support multiple-use demands for public lands. As such, local and regional differences in design and implementation of conservation plans should be assessed with explicit attention to the details and cumulative impact of a suite of actions, including but not limited to the buffer distances, which are the focus of this report.

Surface Disturbance

Surface disturbance represents a combination of human activities that alter or remove the natural vegetation community on a site. Isolating the potential effects of human land-use patterns on sage-grouse is challenging because causal factors are frequently interrelated and interactive (for example roads and distribution lines or roads and well pads) making a general discussion of “development effects” necessary. In cases where better discrimination is available, those specific types of surface disturbances are addressed in the following sections. The values in this section reflect a nondiscriminatory understanding of the independent and interactive and cumulative effects of activities that remove sagebrush cover and other natural vegetation, and often include continual and (or) intermittent activities, such as running motors and pumps, vehicle visits, and equipment servicing. The collective influence of human activity on the landscape, often referred to as the human footprint (Leu and others, 2008), has been associated with negative trends in sage-grouse lek counts (Johnson and others, 2011) and population persistence (Aldridge and others, 2008; Wisdom and others, 2011). A multiscale assessment of factors associated with lek abandonment between 1965 and 2007 found that the level of the human footprint within 5 km (3.1 mi) of the lek was negatively associated with lek persistence (Knick and Hanser, 2011). Agricultural activities, including tilling, seeding, and other highly managed activities, are a component of the human footprint and clearly fall into the category of surface disturbance (removal of native vegetation); however, agriculture is a special case because, although agriculture occupies large areas with transformed conditions, these lands are typically privately owned and the habitat value of agricultural areas is not zero because these lands can provide cover and forage for some populations in some seasons (Fischer and others, 1996). For example, sage-grouse have been known to use agricultural lands in late summer and early spring (Fischer and others, 1996). Though we found no direct evidence for spacing recommendations between agricultural lands and leks or other sage-grouse habitat, the conversion of sagebrush to agriculture within a landscape has been shown to lead to decreased abundance of sage-grouse in many portions of their range (Swenson and others, 1987; Smith and others, 2005; Aldridge and Boyce, 2007; Aldridge and others, 2008). A potential mechanism for this decrease in abundances, besides the direct loss of habitat, is the association of generalist predators (Common Raven [*Corvus corax*] and Black-billed Magpie [*Pica hudsonia*]) with agricultural infrastructure (Vander Haegen and others, 2002) and subsequent predation on sage-grouse (Connelly and others, 2004; Coates and Delehanty, 2010).

Estimated distance effects were translated to a 5- to 8-km (3.1- to 5-mi) radius around each lek to describe a possible conservation buffer area (interpreted range) based on interpretation of two principal factors: the potential effect area and the potential distribution of habitat use within affected areas. The need for protection of populations that are not well understood requires some generalization, and this distance range is proposed because research suggests that a majority of sage-grouse distributions and movements (within and between seasons) occur within this range (for example, Berry and Eng, 1985; Lyon and Anderson, 2003; Holloran and Anderson, 2005; Walker and others, 2007; Aldridge and others, 2008; Knick and others, 2011; Naugle and others, 2011; Coates and others, 2013). Importantly, due to variability among individuals and populations, some individuals in most populations (migratory and nonmigratory) may move greater distances than those included in the buffer, but specific protections cannot, practically, be determined for all individuals and all behavioral patterns. Although leks are generally recognized as the center of breeding and nesting habitats, recent utilization distribution analyses have helped to refine understanding of sage-grouse habitat-use patterns throughout the year. Based on this approach, Coates and others (2013) suggested that an 8-km (5-mi) protection area centered on an active lek location should encompass the seasonal movements and habitat use of 90–95 percent of sage-grouse associated with the lek. Longer distance movements are not always explicitly protected in this context, and habitats associated with previously unidentified leks may not be protected. However, final settling locations for more mobile individuals may be associated with quality habitats protected by buffers around adjacent lek sites. Furthermore, buffer distances beyond 8 km (5 mi) result in a decreasing benefit (cost-benefit trade-off) of increasing protection in areas that are less commonly used by sage-grouse. Without population-specific information regarding the location of habitats and movement of birds, which may be utilized when available (for an example see, Colorado Greater Sage-grouse Steering Committee, 2008), this generalized protection area (circular buffer around active leks with radius of 8 km [5mi]) offers a practical tool for determining important habitat areas. (Note: the Colorado Plan [Colorado Greater Sage-grouse Steering Committee, 2008] recommended a 6.4-km [4-mi] circular buffer, which may be well suited for those populations and falls within the range identified here.)

Importantly, similar results and interpretations to those derived from California and Nevada populations (Coates and others, 2013) were attained from the eastern portion of sage-grouse range; namely, Holloran and Anderson (2005) reported 64 percent of monitored nests fell within 5 km (3.1 mi) of a lek, and response to industrial development (decreased nesting rates and success rates) was observable to distances between 5 and 10 km (3.1–6.2 mi) from a lek suggesting that similar buffer distances are as relevant in Wyoming as in the Great Basin. In Utah, approximately 90 percent of nests (not all movements) were located within 5 km (3 mi) of a lek and threshold distance increased with greater contiguity of habitats. The smallest effect distance (3.2 km [2 mi] from a lek) described by Naugle and others (2011) was previously described and tested in field research by Holloran and Anderson (2005) and Walker and others (2007); these studies were designed to evaluate the effectiveness of existing stipulations. However, recent evaluation of different effect areas (Gregory and Beck, 2014) suggested significant immediate effects on lek attendance with one well pad within 2 km (1.2 mi) of a lek and time-lagged effects due to industrial development within 10 km (6.2 mi) of a lek indicating a habitat within the 8 km (5 mi) identified here may still experience an influence of development on some landscapes. Although considerable protections would be afforded by using a greater buffer distance from leks, research has indicated population effects are variable, and the cumulative effect of development may extend across the landscape many kilometers (>10 km [6 mi]) beyond the immediately affected areas. Diminishing gain analysis (Coates and others, 2013) suggested that sustained gains from habitat protection (based on percent of *highly used areas* protected versus *total* area protected) diminished after 8 km (5 mi)(radius) from leks, which helped to establish a ceiling on interpretations for habitat buffers seeking to maximize conservation benefits and minimize impacts on land uses.

Linear Features

Roads, especially active roads such as collectors, major haul, and service roads, as well as county, State, and Federal highways, create many of the same "aversion" factors described previously that are related to traffic noise on roadways and interactions with infrastructure associated with corridors (such as fences, poles, and towers). One potential mechanism behind road-aversion behavior by sage-grouse could be the intermittent noise produced by passing traffic. Blickley and others (2012) discovered that noise-disturbance simulations that mimicked intermittent sources (road noise), or separately, drilling noises (continuous), generated a significant reduction in lek attendance of sage-grouse (73-percent reduction with road noise, 29 percent with drilling noise).

Most planning related to linear features applies to new construction, that is, avoidance of placing new roads or transmission lines in important habitats, but existing roads might also be addressed by considering seasonal closures, or removal, of roads within protective buffer areas. Fragmentation of habitats related to the network of roads and other linear features (potential for cumulative effects) may have negative effects on sage-grouse populations by reducing and fragmenting sagebrush habitat. When compared to extirpated leks, occupied leks have twice the cover of sagebrush (46 percent versus 24 percent) and ten times larger average sagebrush patches (4,173 hectares [ha] [10,310 acres] versus 481 ha [1,190 acres]) (Wisdom and others, 2011). However, it is important to recognize that previous assessments of relations between sage-grouse distributions and roads include a combination of positive and negative relations (Johnson and others, 2011), and local effects may be restricted to visible (or audible) range. Correlations between the distribution of roads with the distribution of quality sagebrush habitats (due to moderate topographic relief), interactions between influence of roads and infrastructure with topography and habitat conditions (visibility and audibility), and differences in traffic volumes may all contribute to population effects on sage-grouse; not all roads have the same effect (Carpenter and others, 2010; Dinkins and others, 2014). Because roads and other linear features can have different effects on sage-grouse behavior, regional models of distributions and population dynamics have attempted to capture some differences; for example, roads closer to lek locations and other seasonal habitats may have greater effects than those occurring farther from important habitats (Hanser and others, 2011). Effects of pipelines and powerline corridors were tested but were not found to have clear, rangewide effects on lek trends (Johnson and others, 2011). However, it has become evident that interactions and co-location of linear features (for example, power distribution lines along roads and railroads) can make separation of effects difficult (Walters and others, 2014); power lines are addressed in a following section (Tall Structures).

Because of general concerns about habitat fragmentation and loss due to transportation networks, rangewide assessment of the effects of distributed human features, including road proximity (distance) and density, on trends in sage-grouse populations (based on lek counts), were conducted (Johnson and others, 2011). Incremental effects of accumulating length of roads in proximity to leks were apparent rangewide, although limited to major roads (State and Federal highways and interstates). This effect was demonstrated by decreasing lek counts when there were more than 5 km (3.1 mi) of Federal or State highway within 5 km (3.1 mi) of leks and when more than 20 km (12.4 mi) of highway occurs within an 18-km (11.2-mi) window (Johnson and others, 2011). Regional assessments (sage-grouse management zones, MZs; see Stiver and others, 2006) indicated downward trends in northern Great Basin (MZ4 and a portion of MZ5) populations when road density within 5-km (3.1-mi) radius of lek exceeded 30 km (18.6 mi). In Great Plains populations (MZ1), lek trends declined within a 10 km (6.2 mi) radius of a major road. It is important to note that many of the regional assessments did not indicate decreasing lek trends associated with the various size-classes of roads that were assessed (Johnson and others, 2011). In separate analyses in Wyoming, probability of sage-grouse habitat use (based on pellet-count surveys) declined around major roads (State and Federal highways and interstates) when assessed using a 1-km (0.6-mi) exponential decay function ($\exp^{(\text{distance}/-1\text{km})}$; Hanser and others, 2011). Assessment of lek trends in proximity to a large, interstate highway (I-80) indicated that all formerly recorded lek sites within 2 km (1.25 mi) of the highway were unoccupied, and leks within 7.5 km (4.7 mi) of the highway had declining attendance (Connelly and others, 2004).

Radio-telemetry (Very High Frequency, VHF) studies are often used to help track and document animal movements and habitat use, and some have reflected affinity of sage-grouse to roads (for example, Carpenter and others, 2010; Dinkens and others, 2014). However, this pattern may be due to search patterns employed by road-bound investigators (Fedy and others, 2014) or the distribution of roads across quality habitats in flat and lower elevation terrain (Carpenter and others, 2010; Dinkins and others, 2014) as opposed to selection of roads as preferred habitats. Seasonal, Statewide habitat models in Wyoming indicated a difference in seasonal sensitivity to density of paved roads, suggesting a decaying effects function approaching zero as distance approaches 3.2 km (2 mi) of leks (negative exponential) during the nesting and summer seasons, and a decay function approaching zero as distance approaches 1.5 km (0.9 mi) of leks during winter (Fedy and others, 2014). However, Dinkins and others (2014) found decreased risk of death for hens with *increasing* road density, but they also noted that the co-location of road distribution and quality habitat may have influenced this result. Although noise has been clearly demonstrated to influence sage-grouse (Blickley and others, 2012), the influence of individual roads or networks of roads on sage-grouse habitat use and demographic parameters remains a research need. This is a good example of the challenge associated with making clear interpretations of the effect area (and therefore, a definitive buffer distance) for these types of infrastructure.

Energy Development

Research and applications addressing surface disturbances in sagebrush ecosystems have been commonly conducted in relation to energy development activities. Lands affected by these activities have been the focus of many studies investigating the effects of anthropogenic activities on sage-grouse behavior and population dynamics, so the previous section (Surface Disturbance) contains much of the information relevant here.

Direct impacts of energy development on sage-grouse habitats and populations, such as loss of sagebrush canopy or nest failure, have been estimated to occur within a 1.2-ha (3-acre) area of leks (radius: 62 m [68 yards]); indirect influences, such as habitat degradation or utilization displacement, have been estimated to extend out to 19 km (11.8 mi) from leks (Naugle and others, 2011). Regional analyses of well-density and distance effects (Johnson and others, 2011) suggested negative trends in populations (lek counts) when distance was less than 4 km (2.5 mi) to the nearest producing well; whereas density effects were evident rangewide based on decreasing population trends when greater than eight active wells occurred within 5 km (3.1 mi) of leks, or when more than 200 active wells occurred within 18 km (11 mi)of leks. In Wyoming, significant negative relations between use of seasonal habitats and well densities have been demonstrated. Fedy and others (2014) found a significant negative relation between well density and probability of sage-grouse habitat selection during nesting (3.2-km [2-mi] radius) and winter (6.44-km [4-mi] radius) seasons. In the Powder River Basin, wintering sage-grouse were negatively associated with increasing coalbed natural gas well densities within a 2-km × 2-km (1.24-mi × 1.24-mi) window (Doherty and others 2008). Also, Gregory and Beck (2014) documented lek attendance decline when energy development averaged 0.7 well pads/km^2 (1.81 well pads/mi^2; using a 10-km × 10-km [6.2-mi × 6.2-mi] assessment window) across multiple populations and different development patterns.

A key consideration, besides the impacts of the development footprint on habitat condition and predation potential, is the effect of intermittent noise on behavior (avoidance) as evident from work by Blickley and others (2012) who found decreased lek activity due to mimicked drilling and road noise produced at close range (volume level equivalent to a road or well 400 m [1300 ft] away). A precise distance for noise effects has not been determined, but this value likely varies depending on the source (equipment, vehicles) and the terrain.

Less information is available about the effects of renewable energy development, such as wind-turbine arrays, on sage-grouse. LeBeau and others (2014) monitored effects during breeding season (95 nests and 31 broods) and found a linear decline of 7.1 percent in nest failure and 38 percent in brood failure with each 1-km (0.6-mi) increase in distance from wind energy infrastructure (less effect with greater distance). Changes in mortality were not attributed to direct collisions but to increased predation. It is notable that one study on prairie chickens (a related galliform, *Tympanuchus cupido*) found *increased* nest success rates adjacent to recent wind-energy facilities (Winder and others, 2014).

Suggestions that sage-grouse instinctively avoid wind turbines (tall structures) to avoid predators are debated because of the difficulty in directly connecting predation risk to infrastructure, which often includes a combination of features (Walters and others, 2014). A further discussion of this topic is contained in the Tall Structures section below. It is notable that use of wind turbines as perches has not been documented.

Tall Structures

It is important to recognize that the effect of tall structures remains debated, and this category contains a wide array of infrastructure including poles that support lights, telephone and electrical distribution, communication towers, meteorological towers, and high-tension transmission towers. Determining effects of these structures has remained difficult due to limited research and confounding effects (for example, towers and transmission lines are typically associated with other development infrastructure; Messmer and others, 2013; Walters and others, 2014). Lacking precise information regarding the influence of tall structures on the foraging behavior of corvids and raptors, management plans have adopted similar buffer distances to other infrastructure, for example a 1-km (0.6-mi) buffer of avoidance around lek sites. The general assumption is that these structures offer opportunities for increased predator use and thereby generate aversion behaviors among prey species (that is, sage-grouse); however, other effects, such as electro-magnetic radiation, have not been eliminated, and effects on predation rates have not been confirmed (Messmer and others, 2013). Habitat alteration, akin to other linear features (see previous section), may also be considered an important component of interactions between powerline corridors and sage-grouse populations. The 1-km (0.6-mi) buffer indicated here (table 1) was based upon Wallestad and Schladweiler (1974) who observed that more than 90 percent of breeding season movements by male grouse were within 1.3 km (0.8 mi) of a lek (76 percent of movements occurred within 1 km [0.6 mi]). Subsequently, Connelly and others (2000, p. 977) suggested, "avoid building powerlines and other tall structures that provide perch sites for raptors within 3 km of seasonal habitats... lines should be buried or posts modified to prevent use as perches..." Recent research has added important information to previous speculations and estimations, specifying concentrated foraging behaviors by common ravens (a common predator of sage-grouse nests) at 2.2 km (1.4 mi) from electrical transmission towers with the observed foraging area extending out to 11 km (6.8 mi; Coates, and others, 2014a). According to estimates, the greatest potential impact on sage-grouse nests occurs within 570 m (0.35 mi) of structures (Howe and others, 2014). Negative trends in lek counts were associated with increasing number of communication towers within 18km of leks range wide (Johnson and others 2011). Johnson and others (2011) also documented negative trends in lek counts for Great Plains populations within 20 km (12.4 mi) of a power transmission line or when the linear density of powerlines within 5 km (3.1 mi) of leks was greater than 10 km (6.2 mi)—notably, affected areas may be greater in these habitats (compared to other intermountain communities) because visibility is often greater in gentle terrain.

Although considerable attention has been paid to the influence of tall structures (both anthropogenic and trees) on the quality of sage-grouse habitat (for example, Connelly and others, 2000; Connelly and others, 2004; Stiver and others, 2006; National Technical Team, Sage-Grouse, 2011; Manier and others, 2013), solid evidence that sage-grouse instinctively avoid tall structures to avoid predators remains debated because of the difficulty in connecting predation risk to various combinations of infrastructure (Walters and others, 2014). However some evidence exists; in Wyoming the risk of death for sage-grouse hens was greater near potential raptor perches (Dinkins and others, 2014), and in Idaho common raven abundance was greater near energy infrastructure (2.2 km [1.4 mi]; Coates and others 2014a,b). Coates and others (2014b) found different effects of infrastructure on three species of raptor (*Buteo* spp.) and common ravens, with clear increases in raven abundance with infrastructure but less consistent results with raptors. Also, in Wyoming, common raven habitat use was greatest within 3 km (1.8 mi) of human activity centers, and raven occupancy was correlated with nest failure (Bui and others, 2010). These studies suggest a potential increase in predators of sage-grouse, in particular ravens, which may influence predation pressure more than raptors.

Low Structures

Collisions of flying sage-grouse with fences have been associated with mortality (Beck and others, 2006; Stevens and others, 2012a,b). Incidents were focused within 1.6–3.2 km (1–2 mi) of leks on flat to rolling terrain and fences with wide spacing of poles and (or) less visible 't-posts' (as opposed to wooden posts) (Stevens and others 2012a,b). Importantly, the effect of fences was apparently less in rougher terrain, presumably due to differences in flight behaviors in the birds. Marking fences helps flying grouse avoid these collisions; therefore, marking or removal of fences within 2 km (1.2 mi) of leks on flat or rolling terrain can reduce sage-grouse mortality associated with collisions. In a review of previous research, including theses and reports, Connelly and others (2004, p. 4–2) described findings of Rogers (1964) who stated that only 5 percent of leks were found within 200 m (656 ft) of a building, which suggests structures, even without regular activity and (or) noise, may have produced aversion behavior in historic sage-grouse populations. Recent research provides evidence that ravens forage at distances as far as 5.1 km (xx mi) from buildings in sagebrush environments (Coates and others, 2014a) suggesting that a wide distribution of infrastructure that can supply nesting or resting sites for ravens could have negative effects on sage-grouse populations.

Activities (Without Habitat Loss)

Tests using recorded noises and wild sage-grouse populations (Blickley and others, 2012) suggest that loud noises transmitted at decibels (70 dB at 0 m; 40 dB at 100 m [328 ft]) to approximate a noise source 400 m (1300 ft) from leks caused decreased activity on leks. Though they did not test the range of potential noise volumes or activities (different noises) associated with recreation or other (nonindustrial) activities, this research is our best evidence of the effect of noise (independent from infrastructure) on sage-grouse behavior. The upper limit (4.8 km [3 mi]) is the value being used by the State of Nevada for reducing noise effects on sage-grouse due to locations of geothermal energy facilities (Nevada Governor's Sage-Grouse Conservation Team, 2010). Better understanding of the type, frequency, and volume of noise effects on sage-grouse behavior will enhance our ability to define effect areas.

References Cited

Symbols in this section refer to citations in Table 1.

Aldridge, C.L., and Boyce, M.S., 2007, Linking occurrence and fitness to persistence—Habitat-based approach for endangered Greater Sage-Grouse: Ecological Applications, v. 17, p. 508–526.

Aldridge, C.L., Nielsen, S.E., Beyer, H.L., Boyce, M.S., Connelly, J.W., Knick, S.T., and Schroeder, M.A., 2008, Range-wide patterns of Greater Sage-Grouse persistence: Diversity and Distributions, v. 14, p. 983–94.

Beck, J.L., Booth, D.T., and Kennedy, C.L., 2014, Assessing Greater Sage-Grouse breeding habitat with aerial and ground imagery: Rangeland Ecology and Management, v. 67, p. 328–332.

Beck, J.L., Reese, K.P., Connelly, J.W., and Lucia, M.B., 2006, Movements and survival of juvenile Greater Sage-Grouse in southeastern Idaho: Wildlife Society Bulletin v. 34, p. 1070–1078.

Berry, J.D., and Eng, R.L., 1985, Interseasonal movements and fidelity to seasonal use areas by female sage-grouse: Journal of Wildlife Management, v. 49, p. 237–40.

‡ Blickley, J.L., Blackwood, D., Patricelli, G.L., 2012, Experimental evidence for the effects of chronic anthropogenic noise on abundance of greater sage-grouse at leks: Conservation Biology, v. 26, p. 461–471.

Bui, T.V.D., Marzluff, J.M., Bedrosian, B., 2010, Common raven activity in relation to land use in western Wyoming: implications for greater-sage grouse reproductive success: Condor, v. 112, p. 65–78.

Carpenter, J., Aldridge, C.L., and Boyce, M.S., 2010, Sage-grouse habitat selection during winter in Alberta: Journal of Wildlife Management, v. 74, p. 1806–1814.

Coates, P.S. and Delehanty, D.J., 2010, Nest predation of Greater Sage-Grouse in relation to microhabitat factors and predators: Journal of Wildlife Management, v. 74, p. 240–248.

Coates, P.S., Casazza, M.L., Blomberg, E.J., Gardner, S.C., Espinosa, S.P., Yee, J.L., Wiechman, L., and Halstead. B.J., 2013, Evaluating Greater Sage-Grouse seasonal space use relative to leks: Implications for surface use designations in sagebrush ecosystems: Journal of Wildlife Management, v. 77, p. 1598–1609.

Coates, P.S., Howe, K.B., Casazza M.L., and Delehanty, D.J., 2014a, Common raven occurrence in relation to energy transmission line corridors transiting human-altered sagebrush steppe: Journal of Arid Environments, v. 111, p.68–78.

Coates, P.S., Howe, K.B., Casazza, M.L., and Delehanty D.J., 2014b, Landscape alterations influence differential habitat use of nesting buteos and ravens within sagebrush ecosystem: Implications for transmission line development: Condor, v. 116, p. 341–356.

Connelly, J. W., Schroeder, M. A., Sands, A. R., and Braun, C. E., 2000, Guidelines to manage sage grouse populations and their habitats: Wildlife Society Bulletin, v. 28, p. 967–985.

§ Connelly, J. W., Knick, S.T., Schroeder, M. A. and Stiver, S. J., 2004, Conservation assessment of greater sage-grouse and sagebrush habitats: Western Association of Fish and Wildlife Agencies (WAFWA), 600 p.

Colorado Greater Sage-grouse Steering Committee, 2008, Colorado Greater Sage-Grouse Conservation Plan. Colorado Division of Wildlife, Denver, Colorado, USA.

Davies, K.W. and Bates, J.D., 2010, Vegetation Characteristics of Mountain and Wyoming Big Sagebrush Plant Communities in the

Northern Great Basin: Rangeland Ecology and Management, v. 63, p. 461–466.

Dinkins, J.B., Conover, M.R. Kirol, C.P. Beck, J. L. and Frey S.N,. 2014, Greater sage-grouse hen survival: effects of raptors, anthropogenic and landscape features, and hen behavior: Canadian Journal of Zoology, v. 92, p. 319–330.

Doherty, K.E., Naugle, D.E., Walker, B.L., and Graham J.M., 2008, Greater Sage-Grouse Winter Habitat Selection and Energy Development: Journal of Wildlife Management, v. 72, p. 187–95.

Doherty, K.E., Naugle, D.E., and Walker B.L., 2010, Greater Sage-Grouse nesting habitat: the importance of managing at multiple scales: Journal of Wildlife Management, v. 74, p. 1544–1553.

Fedy, B.C., Aldridge, C.L., Doherty, K.E., O'Donnell, M., Beck, J.L., Bedrosian, B., Holloran, M.J., Johnson, G. D., Kaczor, N.W., Kirol, C.P., Mandich, C.A., Marshall, D., McKee, G., Olson, C., Swanson, C.C. and Walker, B.L., 2012, Interseasonal Movements of Greater Sage-Grouse, Migratory Behavior, and an Assessment of the Core Regions Concept in Wyoming: Journal of Wildlife Management, v. 76, p. 1062–1071.

Fedy, B.C., Doherty, K.E., Aldridge, C.L., O'Donnell, M., Beck, J.L., Bedrosian, B., Gummer, D., Holloran, M.J., Johnson, G.D., Kaczor, N.W., Kirol, C.P., Mandich, D.A., Marshall, D., McKee, G., Olson, C., Pratt, A.C., Swanson, C.C.,Walker, B.L., 2014, Habitat prioritization across large landscapes, multiple seasons, and novel areas: An example using Greater Sage-Grouse in Wyoming: Wildlife Monographs, v. 190, p. 1–39.

Fischer, R.A., Reese, K.P., and Connelly, J.W., 1996, Influence of vegetal moisture content and nest fate on timing of female sage grouse migration: Condor, v. 98, p.868–872.

Gregory, A.J., and Beck, J.L., 2014, Spatial heterogeneity in response of male greater sage-grouse lek attendance to energy development: PLoS ONE, v. 9, e97132.

Hagen, C.A., Connelly, J.W., Schroeder, M.A., 2007, A meta-analysis of greater sage-grouse Centrocercus urophasianus nesting and brood-rearing habitats: Wildlife Biology, v. 13, p. 42–50

Hanser, S.E., Aldridge, C.L., Leu, M., Rowland, M.M., Nielsen, S.E., and Knick S.T., 2011, Greater Sage-Grouse: General Use and Roost Site Occurrence with Pellet Counts as a Measure of Relative Abundance, p. 112–140 *in* S. E. Hanser, M. Leu, S. T. Knick, and C. L. Aldridge, eds., Sagebrush ecosystem conservation and management: ecoregional assessment tools and models for the Wyoming Basins: Allen Press, Lawrence, KS.

* Holloran, M.J., and Anderson. S.H., 2005, Spatial Distribution of Greater Sage-Grouse Nests in Relatively Contiguous Sagebrush Habitats: Condor, v. 107, p. 742–52.

° Howe, K.B., Coates, P.S., and Delehanty, D.J., 2014, Selection of anthropogenic features and vegetation characteristics by nesting Common Ravens in the sagebrush ecosystem: Condor, v. 116, p. 25–49.

◊ Johnson, D. H., Holloran, M. J. Connelly, J. W. Hanser, S. E. Amundson, C. L. andKnick S. T., 2011, Influences of Environmental and Anthropogenic Features on Greater Sage-Grouse Populations, 1997–2007, Chapter 17 *In* S. T. Knick and J. W. Connelly, eds., Greater Sage-Grouse: Ecology of a Landscape Species and Its Habitats, Studies in Avian Biology No. 38: University of California Press: Berkeley, CA, p. 407–450.

Kirol, C.P., Beck, J.L., Dinkins, J.B., and Conover, M.R., 2012, Microhabitat selection for nesting and brood-rearing by the Greater Sage-Grouse in xeric big sagebrush: Condor, v. 114, p.75–89.

Knick, S.T., 2011, Historical development, principal federal legislation, and current management of sagebrush habitats: implications for conservation. *in* S. T. Knick and J. W. Connelly, eds. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology No. 38, University of California Press, Berkeley, CA. p. 13–31.

Knick, S.T., and Hanser, S.E., 2011, Connecting pattern and process in Greater Sage-Grouse populations and sagebrush landscapes, *in* S. T. Knick and J. W. Connelly, eds., Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology No. 38: University of California Press, Berkeley, CA. p. 383–405.

Knick, S.T., Hanser, S.E., and Preston, K.L,. 2013, Modeling ecological minimum requirements for distribution of greater sage-grouse leks: implications for population connectivity across their western range, U.S.A.: Ecology and Evolution, v. 3, p.1539–1551.

Knick, S.T., Hanser, S.E. Miller, R.F., Pyke, D.A., Wisdom, M.J., Finn, S. P., Rinkes, E.T. and Henny C.J., 2011, Ecological Influence and Pathways of Land Use in Sagebrush, *In* S. T. Knick and J. W. Connelly, eds., Greater Sage-Grouse: Ecology of a Landscape Species and Its Habitats, Studies in Avian Biology No.38: University of California Press: Berkeley, CA, p. 203–252.

LeBeau, C.W., Beck, J.L., Johnson G.D. and Holloran, M.J., 2014, Short-term impacts of wind energy development on Greater Sage-Grouse fitness: Journal of Wildlife Management, v. 78, p. 522–530.

Leu, M., Hanser, S.E., and Knick S.T., 2008, The human footprint in the West: a large-scale analysis of anthropogenic impacts, Ecological Applications, v. 18, p. 119–1139.

Lyon, A.G., and Anderson S.H,. 2003, Potential Gas Development Impacts on Sage Grouse Nest Initiation and Movement. Wildlife Society Bulletin, v. 31, p. 486–91.

Manier, D.J., Wood, D.J.A., Bowen, Z.H., Donovan, R.M., Holloran, M.J., Juliusson, L.M., Mayne, K.S., Oyler-McCance, S.J., Quamen, F.R., Saher, D.J., and Titolo, A.J., 2013, Summary of science, activities, programs, and policies that influence the rangewide conservation of Greater Sage-Grouse (*Centrocercus urophasianus*): U.S. Geological Survey Open–File Report 2013–1098, 331 p.

Messmer, T.A., Hasenyager, R. Burruss, J. and Liguori S., 2013, Stakeholder contemporary knowledge needs regarding the potential effects of tall structures on sage-grouse. Human-Wildlife Interactions, v. 7, p. 273–298.

National Technical Team, Sage-Grouse, 2011, A report on national Greater Sage-Grouse conservation measures. Bureau of Land Management: Technical Report. Washington, D.C. and Denver, Colorado. 74p.

ǂ Naugle, D. E., Doherty, K.E. Walker, B.L. Holloran, M.J. and Copeland H.E., 2011, Energy Development and Greater Sage-Grouse. In Greater Sage-Grouse: Ecology of a Landscape Species and Its Habitats, edited by S.T. Knick and Connelly J.W. Studies in Avian Biology No. 38, University of California Press: Berkeley, CA, p. 489–504.

ψ Nevada Governor's Sage-Grouse Conservation Team, 2010, Nevada energy and infrastructure development standards to conserve Greater Sage-Grouse populations and their habitats. State of Nevada, Reno, 58 p.

Rogers, G.E., 1964, Sage-grouse investigations in Colorado. Colorado Game and Fish Department, Technical Publication No. 16, Fort Collins, Colo.

Smith, J.T., Flake, L.D., Higgins, K F., Kobriger, G.D., and Homer C.G., 2005, Evaluating lek occupancy of Greater Sage-Grouse in relation to landscape cultivation in the Dakotas: Western North American Naturalist, v. 65, p. 310–320.

Smith, K.T., Kirol, C.P., Beck, J.L, .and Blomquist, F.C., 2014, Prioritizing winter habitat quality for greater sage-grouse in a landscape influenced by energy development: Ecosphere, vol. 5, article 15

Stevens, B.S., Reese, K.P. Connelly, J.W., and Musil D.D., 2012a, Greater sage-grouse and fences: Does marking reduce collisions? Wildlife Society Bulletin, v. 36, p. 297–303.

« Stevens, B.S., Connelly, J.W. and Reese K.P., 2012b, Multi-scale assessment of greater sage-grouse fence collision as a function of site and broad scale factors.: Journal of Wildlife Management, v. 76, p. 1370–1380.

Stiver, S.J., Apa, A.D., Bohne, J.R., Bunnell, S.D., Deibert, P.A., Gardner, S.C., Hilliard, M.A., McCarthy C.W., and Schroeder M.A., 2006, Greater Sage-grouse Comprehensive Conservation Strategy. Western Association of Fish and Wildlife Agencies. WAFWA Report, Cheyenne, Wyoming. Available online: *http://www.wafwa.org/documents/pdf/GreaterSage-grouseConservationStrategy2006.pdf*

Swenson, J. E., Simmons, C.A. and Eustace C.D., 1987, Decrease of Sage Grouse Centrocercus urophasianus after ploughing of sagebrush steppe: Biological Conservation, v. 41, p. 125–132.

Tack, J.D., Naugle D.E., Carlson J.C. and Fargey P.J., 2012,. Greater sage-grouse (Centrocercus urophasianus) migration links the USA and Canada: a biological basis for international prairie conservation: Oryx, vol. 46, p. 64–68.

Vander Haegen, W.M., Schroeder M.A. and DeGraaf R.M.. 2002. Predation on real and artificial nests in shrub steppe landscapes fragmented by agriculture: Condor, v. 104, p. 496–506.

* Walker, B. L., Naugle, D. E. and Doherty K. E.. 2007. Greater Sage-Grouse population response to energy development and habitat loss: Journal of Wildlife Management, v. 71, p. 2644–2654.

° Wallestad, R. O., and Schladweiler, P, 1974. Breeding season movements and habitat selection of male sage grouse: Journal of Wildlife Management, v.38, p. 634–637.

Walters, K., Kosciuch K., and Jones, J., 2014, Can the effect of tall structures on birds be isolated from other aspects of development: Wildlife Society Bulletin, vol. 38, p. 250–256.

Winder, V.L., McNew, L.B., Gregory, A.J., Hunt, L.M., Wisely, S.M., and Sandercock, B.K., 2014, Effects of wind energy development on survival of female greater prairie- chickens: Journal of Applied Ecology, v. 51, p. 395–405.

Wisdom, M. J., Meinke, C. W. Knick, S. T. and Schroeder, M. A., 2011, Factors associated with extirpation of sage-grouse. *In* Knick, S. T. and Connelly, J. W., eds. Greater Sage-Grouse: ecology of a landscape species and its habitats. Studies in Avian Biology No. 38, University of California Press: Berkeley, California, p. 451–472**.**

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Table 1. Lek buffer-distance estimates for six categories of anthropogenic land use and activity. Literature minimum and maximum values are distances for observed effects found in the scientific literature. Interpreted ranges indicate potential conservation buffer distances based on multiple sources. [Citations for literature minimum and maximum values are denoted using corresponding symbols in the References Cited section.]

Category	Literature minimum	Interpreted range (lower)	Interpreted range (upper)	Literature maximum
Surface disturbance	3.2km (2mi) *	5km (3.1mi)	8km (5mi)	20km (12.4mi) ◊
Linear features	400m (0.25mi) ‡	5km (3.1mi)	8km (5mi)	18km (11.2mi) ◊
Energy development	3.2km (2mi) ǂ	5km (3.1mi)	8km (5mi)	20km (12.4mi) ◊
Tall structures	1km (0.6mi) °	3.3km (2mi)	8km (5mi)	18km (11.2mi) ◊
Low structures	200 m (0.12 mi) §	2 km (1.2mi)	5.1 km (3.2mi)	5.1 km (3.2mi) «
Activities	400 m (0.12 mi) ‡	400 m (0.12 mi)	4.8 km (3mi)	4.8 km (3mi) Ψ

A Report on National Greater Sage-Grouse Conservation Measures

Produced by:

Sage-grouse National Technical Team

December 21, 2011

Table of Contents

Introduction

Sagebrush landscapes have changed dramatically over the last two centuries. The vast expanses of sagebrush crossed by early European settlers and used by sage-grouse have been lost, fragmented, or altered due to invasive plants, changes in fire regimes, and impact of land uses (Knick et al. 2003, Knick and Connelly 2011a). As a consequence, sage-grouse and many other wildlife species that depend on sagebrush have undergone long-term range-wide population declines. Sage-grouse populations now occupy approximately one-half of their pre-European settlement distribution (Schroeder et al. 2004). Anthropogenic habitat impacts and lack of regulatory mechanisms to protect against further losses provided the basis for warranting listing under the Endangered Species Act (ESA) in 2010 (75 FR 13910). The need to address higher priority species and limited funding precluded immediate listing action. However, a litigation settlement requires that a listing decision be made by the U.S. Fish and Wildlife Service (USFWS) by September, 2015.

The Bureau of Land Management (BLM) manages approximately 50% of the sagebrush habitats used by sage-grouse (Knick 2011). Therefore, management actions by BLM in concert with other state and federal agencies, and private land owners play a critical role in the future trends of sage-grouse populations. To ensure BLM management actions are effective and based on the best available science, the National Policy Team created a National Technical Team (NTT) in August of 2011. The BLM's objective for chartering this planning strategy effort was to develop new or revised regulatory mechanisms, through Resource Management Plans (RMPs), to conserve and restore the greater sage-grouse and its habitat on BLM-administered lands on a range-wide basis over the long term. The National Greater Sage-Grouse Planning Strategy Charter charged the NTT to serve as a scientific and technical forum to:

- Understand current scientific knowledge related to the greater sage-grouse.
- Provide specialized sources of expertise not otherwise available.
- Provide innovative scientific perspectives concerning management approaches for the greater sage-grouse.
- Provide assurance that relevant science is considered, reasonably interpreted, and accurately presented; and that uncertainties and risks are acknowledged and documented.
- Provide science and technical assistance to the Regional Management Team (RMT) and Regional Interdisciplinary Team (RIDT), on request.
- Articulate conservation objectives for the greater sage-grouse in measurable terms to guide overall planning.

- Identify science-based management considerations for the greater sage-grouse (e.g., conservation measures) that are necessary to promote sustainable sage-grouse populations, and which focus on the threats (75 FR 13910) in each of the management zones.[i]

The National Technical Team (NTT) met from August 28 through September 2, 2011, in Denver, Colorado, and a subset of the team met December 5-8 in Phoenix, Arizona, to further articulate the scientific basis for the conservation measures. Members of the team included resource specialists and scientists from the BLM, State Fish and Wildlife Agencies, USFWS, Natural Resources Conservation Service (NRCS) and U.S. Geological Survey (USGS).

This document provides the latest science and best biological judgment to assist in making management decisions. Fortunately, recent emphasis on sage-grouse conservation has resulted in a substantial number of publications dealing with a variety of aspects of sage-grouse ecology and management, summarized in the 2010 listing petition (75 FR 13910), as well as Knick and Connelly (2011b). Habitat requirements and other life history aspects of sage-grouse, excerpted from the USFWS listing decision (75 FR 13910), are summarized in Appendix A to provide context for the proposed conservation measures. We have attempted to describe the scientific basis for the conservation measures proposed within each program area. Perspectives on the nature and interpretation of the available science are in Appendix B.

The conservation measures described in this report are not an end point but, rather, a starting point to be used in the BLM's planning processes. Due to time constraints, they are focused primarily on priority sage-grouse habitat areas. General habitat conservation areas were not thoroughly discussed or vetted through the NTT, and the concept of connectivity between priority sage-grouse habitat areas will need more development through the BLM planning process.

[i] Identified in the Western Association of Fish and Wildlife Agencies (WAFWA) Conservation Strategy (Stiver et al. 2006).

Goals and Objectives

The BLM, along with a host of other state and federal agencies who participated in development of the Greater Sage-grouse Comprehensive Conservation Strategy (Stiver et al. 2006), endorsed the goal of that document which was "to maintain and enhance populations and distribution of sage-grouse by protecting and improving sagebrush habitats and ecosystems that sustain these populations". Although it was understood that at least in the short term this goal of maintaining sage-grouse population size and distribution as based on trends from 1965 - 2003, or enhancing above these levels was aspirational, the NTT supports it as a guiding philosophy against which management actions and policies of BLM should be weighed. Therefore, the conservation measures and strategies that follow assume the goal and objectives below.

Goal

Maintain and/or increase sage-grouse abundance and distribution by conserving, enhancing or restoring the sagebrush ecosystem upon which populations depend in cooperation with other conservation partners.

Until such time as more specific conservation objectives relative to sage-grouse distribution or abundance by sage-grouse management zone, state, or population are developed, BLM will strive to maintain or increase current distribution and abundance of sage-grouse on BLM administered lands in support of the range-wide goals. BLM will specifically address threats identified by the Fish and Wildlife Service in their 2010 listing decision (75 FR 13910).

Sage-grouse populations have the greatest chance of persisting when landscapes are dominated by sagebrush and natural or human disturbances are minimal (Aldridge et al. 2008, Knick and Hanser 2011, Wisdom et al. 2011). Within priority habitat, a minimum range of 50-70% of the acreage in sagebrush cover is required for long-term sage-grouse persistence (Aldridge et al. 2008, Doherty et al. 2010, Wisdom et al. 2011). Fire and invasion by exotic grasses are widespread causes for habitat loss, particularly in the western part of the sage-grouse range (Miller et al. 2011). Human land use, including tillage agriculture, historic grazing management, energy development, roads and power line infrastructure, and even recreation have contributed both individually and cumulatively to lower numbers of sage-grouse across the range (75 FR 13910, Knick et al. 2011).

New Paradigm

Through the establishment of the National Sage-grouse Planning Strategy, the Bureau of Land Management has committed to a new paradigm in managing the sagebrush landscape. That new paradigm will require collaborative conservation efforts among private, state, tribal, and other federal partners to conserve sage-grouse. Land uses, habitat treatments, and anthropogenic disturbances will need to be managed below thresholds necessary to conserve not only local sage-grouse populations, but sagebrush communities and landscapes as well. Management priorities will need to be shifted and balanced to maximize benefits to sage-grouse habitats and populations in priority habitats. Adequacy of management adjustments will be measured by science-based effectiveness monitoring of the biological response of sagebrush landscapes and sage-grouse populations. Ultimately, success will be measured by the maintenance and enhancement of sage-grouse populations well into the future.

Objectives

The overall objective is to protect priority sage-grouse habitats from anthropogenic disturbances that will reduce distribution or abundance of sage-grouse. Priority sage-grouse habitats are areas that have the highest conservation value to maintaining or increasing sage-grouse populations. These areas would include breeding, late brood-rearing, winter concentration areas, and where known, migration or connectivity corridors. These areas have been, or will be identified by state fish and wildlife agencies in coordination with respective BLM offices. Priority habitat designations must reflect the vision, goals and objectives of this overall plan if the conservation measures are to be effective. Additionally, there is an opportunity for synergy and collaboration with WAFWA in order to identify a consistent way to designate priority sage-grouse habitat areas and develop a range-wide priority habitat area map. This collaborative and overarching approach could help ensure activities immediately outside the priority areas do not impact priority habitat.

To reach this objective, it will be necessary to achieve the following sub-objectives for priority habitat:

- Designate priority sage-grouse habitats for each WAFWA management zone (Stiver et al. 2006) across the current geographic range of sage-grouse that are large enough to stabilize populations in the short term and enhance populations over the long term.
- To maintain or increase current populations, manage or restore priority areas so that at least 70% of the land cover provides adequate sagebrush habitat to meet sage-grouse needs.
- Develop quantifiable habitat and population objectives with WAFWA and other conservation partners at the management zone and/or other appropriate scales. Develop a monitoring and adaptive management strategy to track whether these objectives are being met, and allow for revisions to management approaches if they are not.[ii]
- Manage priority sage-grouse habitats so that discrete anthropogenic disturbances cover less than 3% of the total sage-grouse habitat regardless of ownership. Anthropogenic features include but are not limited to paved highways, graded gravel roads, transmission lines, substations, wind

[ii] As population trends within each Management Zone respond, long-term success can be judged based on comparisons with data from the 1965-2003 period for that specific Management Zone (Stiver et al., 2006).

[iii] Professional judgment as derived from Holloran 2005, Walker et al. 2007, Doherty et al. 2008, Doherty et al. 2011, Naugle et al. 2011a,b.

turbines, oil and gas wells, geothermal wells and associated facilities, pipelines, landfills, homes, and mines. [iii]

- In priority habitats where the 3% disturbance threshold is already exceeded from any source, no further anthropogenic disturbances will be permitted by BLM until enough habitat has been restored to maintain the area under this threshold (subject to valid existing rights).

- In this instance, an additional objective will be designated for the priority area to prioritize and reclaim/restore anthropogenic disturbances so that 3% or less of the total priority habitat area is disturbed within 10 years.

Note to add context to above objective: Disturbance can be described within categories as discrete (having a distinct measureable impact in space and time) or diffuse (pressure is exerted over broad spatial or temporal scales) (Turner and Gardner 1991). Most anthropogenic disturbance (roads, power lines, oil/gas wells, tall structures) are discrete disturbances. Livestock grazing is a diffuse disturbance. Fire can be either discrete or diffuse depending on its characteristics and the scales at which it is measured. Sage-grouse are extremely sensitive to discrete disturbance (Johnson et al. 2011, Naugle et al. 2011a,b) although diffuse disturbance over broad spatial and temporal scales can have similar, but less visible effects.

Spatial and temporal scales are important components in measuring and interpreting the effects of disturbance (Johnson and St-Laurent 2011). A discrete event might be significant to individuals or local communities but have little effect on the larger population or region (See Figure 2 in Appendix B). Therefore, defining the spatial extent (the region bounding the analysis), spatial and temporal scale (the dimension of the event), and the resolution (the precision of the measurement) are fundamental inputs into any assessment of disturbance (Wheatley and Johnson 2009).

Two spatial extents for measuring anthropogenic disturbance will be used: 1) the area contained within individual priority areas and 2) each one-mile section within the priority area. This hierarchical arrangement allows concentrated anthropogenic disturbance to exceed recommended thresholds within a smaller area, yet still maintain an overall level at the scale to which sage-grouse respond within priority areas.

(1) Large-scale disturbances that impact sage grouse distribution and abundance at any level will not be permitted within priority areas (subject to valid existing rights). Other, smaller scale proposed anthropogenic disturbances will not disturb more than a total of 3% of the acreage within each priority area.

[iii] Professional judgment as derived from Holloran 2005, Walker et al. 2007, Doherty et al. 2008, Doherty et al. 2011, Naugle et al. 2011a,b.

(2) Proposed anthropogenic surface disturbances within an individual priority area will be encouraged to occur in areas of existing development, or areas of non-suitable habitats. Suitable buffers, depending on the occurrence of adjacent seasonal habitats and local information (e.g. migratory vs. non-migratory populations; [Connelly et al. 2000]) may be applied in siting a proposed anthropogenic surface disturbance to protect surrounding suitable, undisturbed habitats.

(3) Concentrating or clustering disturbances locally while maintaining total disturbance below 3% at the priority habitat scale may cause some one-mile^2 analysis sections to exceed the 3% anthropogenic disturbance goal. For example, a sand and gravel mine can result in intensive development of 40 acres, effectively rendering that area unsuitable for sage-grouse. The actual 40-acre disturbance may not push total anthropogenic disturbance to more than 3% for the entire priority area, but obviously has a significant local impact. In these situations, 40 acres of off-site mitigation will be necessary to offset this loss of habitat. The priority is to implement off-site mitigation within the priority sage-grouse habitat, followed by general sage-grouse habitat.

If a project proponent agrees to site proposed anthropogenic surface disturbance within areas of existing development or areas of non-suitable habitat in a priority area, and the resulting localized total surface disturbance exceeds 3% (but the anthropogenic surface disturbance of the entire priority area does not exceed 3%), the need for off-site mitigation should be evaluated on a case-by-case basis.

Additionally, there are sub-objectives that must be met in general sage-grouse habitat. General sage-grouse habitat is occupied (seasonal or year-round) habitat outside of priority habitat. These areas have been, or will be identified by state fish and wildlife agencies in coordination with respective BLM offices.

It will be necessary to achieve the following sub-objectives for general habitat:

- Quantify and delineate general habitat for capability to provide connectivity among priority areas (Knick and Hanser 2011).
- Conserve, enhance or restore sage-grouse habitat and connectivity (Knick and Hanser 2011) to promote movement and genetic diversity, with emphasis on those habitats occupied by sage-grouse.
- Assess general sage-grouse habitats to determine potential to replace lost priority habitat caused by perturbations and/or disturbances and provide connectivity (Knick and Hanser 2011) between priority areas.
    - These habitats should be given some priority over other general sage-grouse habitats that provide marginal or substandard sage-grouse habitat.

- Restore historical habitat functionality to support sage-grouse populations guided by objectives to maintain or enhance connectivity. Total area and locations will be determined at the Land Use Plan level.

- Enhance general sage-grouse habitat such that population declines in one area are replaced elsewhere within the habitat.

Conservation Measures

The following conservation measures are designed to achieve population and habitat objectives stated in this report. They are organized by resource programs.

Travel and Transportation

The Travel and Transportation program is principally focused on road networks within the sage-grouse range. Roads can range from state or interstate highways to gravel and two-track roads. Within the sage-grouse range, 95% of the mapped sagebrush habitats are within 2.5 km (1.55 miles) of a mapped road; density of secondary roads exceeds 5 km/km^2 (3.1 miles/247 acres) in some regions (Knick et al. 2011).

Roads have multiple impacts on wildlife in terrestrial ecosystems, including:

1) Increased mortality from collision with vehicles;
2) Changes in behavior;
3) Loss, fragmentation, and alteration of habitat;
4) Spread of exotic species; and
5) Increased human access, resulting in facilitation of additional alteration and use of habitats by humans (Formann and Alexander 1998, Jackson 2000, Trombulak and Frissel 2000).

The effect of roads can be expressed directly through changes in habitat and sage-grouse populations and indirectly through avoidance behavior because of noise created by vehicle traffic (Lyon and Anderson 2003, 75 FR 13910).

<u>Priority sage-grouse habitat areas</u>

- Limit motorized travel to designated roads, primitive roads, and trails at a minimum.
- Travel management should evaluate the need for permanent or seasonal road or area closures.
- Complete activity level plans within five years of the record of decision. During activity level planning, where appropriate, designate routes with current administrative/agency purpose or need to administrative access only.
- Limit route construction to realignments of existing designated routes if that realignment has a minimal impact on sage-grouse habitat, eliminates the need to construct a new road, or is necessary for motorist safety
- Use existing roads, or realignments as described above to access valid existing rights that are not yet developed. If valid existing rights cannot be accessed via existing roads, then build any new road constructed to the absolute minimum standard necessary, and add the surface disturbance to the total disturbance in the priority area. If that disturbance exceeds 3 % for that area, then make additional, effective mitigation necessary to offset the resulting loss of sage-grouse habitat (see Objectives).

- Allow no upgrading of existing routes that would change route category (road, primitive road, or trail) or capacity unless the upgrading would have minimal impact on sage-grouse habitat, is necessary for motorist safety, or eliminates the need to construct a new road.
- Conduct restoration of roads, primitive roads and trails not designated in travel management plans. This also includes primitive route/roads that were not designated in Wilderness Study Areas and within lands with wilderness characteristics that have been selected for protection.
- When reseeding roads, primitive roads and trails, use appropriate seed mixes and consider the use of transplanted sagebrush.

Recreation

Recreational activities in sagebrush habitats range from hiking, camping and hunting to lek viewing, and off-highway vehicle (OHV) use. Many of these activities are benign uses in sagebrush habitats. However, excessive use, such as repeated disturbance to leks for viewing that disrupts sage-grouse breeding activities, can have negative effects (75 FR 13910). Off-trail recreation by OHV users can fragment habitat and create corridors for spread of exotic plant species (Knick et al. 2011).

Special Recreation Permits (SRP)

- Only allow SRPs that have neutral or beneficial affects to priority habitat areas.

Lands/Realty

The Lands and Realty program primarily influences rights-of-way (ROWs), land tenure adjustments, and proposed land withdrawals. Existing and proposed developments for ROWs (such as powerlines, pipelines, and renewable energy projects) and access to various mineral claims or energy development locations have the potential to cause habitat loss and fragmentation that decreases habitat and population connectivity. Roads also create corridors that facilitate spread of exotic plant species (Gelbard and Belnap 2003). In addition, roads and infrastructure networks can increase sage-grouse mortality from increased predation and collisions with vehicles. Sage-grouse may avoid areas because of noise from vehicle traffic (Lyon and Anderson 2003). Adjustments for land tenure and strategically-located land withdrawals can be used to increase connectivity within sage-grouse populations and sagebrush habitats (Knick and Hanser 2011). In addition, land acquisitions and withdrawals may be important conservation strategies because increased development on private lands, which is not subject to mitigation, will focus greater needs for conservation of sage-grouse and sagebrush on public lands (Knick et al. 2011).

Rights of Way

<u>Priority sage-grouse habitat areas</u>

- Make priority sage-grouse habitat areas exclusion areas for new ROWs permits. Consider the following exceptions:

- Within designated ROW corridors encumbered by existing ROW authorizations: new ROWs may be co-located only if the entire footprint of the proposed project (including construction and staging), can be completed within the existing disturbance associated with the authorized ROWs.
- Subject to valid, existing rights: where new ROWs associated with valid existing rights are required, co-locate new ROWs within existing ROWs or where it best minimizes sage-grouse impacts. Use existing roads, or realignments as described above, to access valid existing rights that are not yet developed. If valid existing rights cannot be accessed via existing roads, then build any new road constructed to the absolute minimum standard necessary, and add the surface disturbance to the total disturbance in the priority area. If that disturbance exceeds 3% for that area, then make additional effective mitigation necessary to offset the resulting loss of sage-grouse.

- Evaluate and take advantage of opportunities to remove, bury, or modify existing power lines within priority sage-grouse habitat areas. Sage-grouse may avoid powerlines because of increased predation risk (Steenhof et al. 1993, Lammers and Collopy 2007). Powerlines effectively influence (direct physical area plus estimated area of effect due to predator movements) at least 39% of the sage-grouse range (Knick et al. 2011). Deaths resulting from collisions with powerlines were an important source of mortality for sage-grouse in southeastern Idaho (Beck et al. 2006, 75 FR 13910)
- Where existing leases or ROWs have had some level of development (road, fence, well, etc.) and are no longer in use, reclaim the site by removing these features and restoring the habitat.

Planning Direction Note: While engaged in this sage-grouse EIS planning process, relocate existing designated ROW corridors crossing priority sage-grouse habitat void of any authorized ROWs, outside of the priority habitat area. If relocation is not possible, undesignate that entire corridor during the planning process.

General sage-grouse habitat areas

- Make general sage-grouse habitat areas “avoidance areas” for new ROWs.
- Where new ROWs are necessary, co-locate new ROWs within existing ROWs where possible.

Land Tenure Adjustment

Priority sage-grouse habitat areas

- Retain public ownership of priority sage-grouse habitat. Consider exceptions where:
  - There is mixed ownership, and land exchanges would allow for additional or more contiguous federal ownership patterns within the priority sage-grouse habitat area.
  - Under priority sage-grouse habitat areas with minority federal ownership, include an additional, effective mitigation agreement for any disposal of federal land. As a final preservation measure consideration should be given to pursuing a permanent conservation easement.

- Where suitable conservation actions cannot be achieved, seek to acquire state and private lands with intact subsurface mineral estate by donation, purchase or exchange in order to best conserve, enhance or restore sage-grouse habitat.

Proposed Land Withdrawals

Priority sage-grouse habitat areas

- Propose lands within priority sage-grouse habitat areas for mineral withdrawal.
- Do not approve withdrawal proposals not associated with mineral activity unless the land management is consistent with sage-grouse conservation measures. (For example; in a proposed withdrawal for a military training range buffer area, manage the buffer area with sage-grouse conservation measures.)

Range Management

Potential impacts of herbivory on sage-grouse and their habitat include:

1) Long-term effects of historic overgrazing on sagebrush habitat;

2) Sage-grouse habitat changes due to herbivory;

3) Direct effects of herbivores on sage-grouse, such as trampling of nests and eggs;

4) Altered sage-grouse behavior due to presence of herbivores; and

5) Impacts to sage-grouse and sage-grouse behavior from structures associated with grazing management (Beck and Mitchell 2000).

Managing livestock grazing to maintain residual cover of herbaceous vegetation so as to reduce predation during nesting may be the most beneficial for sage-grouse populations (Beck and Mitchell 2000, Aldridge and Brigham 2003). Other management objectives that control livestock movements and grazing intensities can be achieved broadly through rotational grazing patterns or locally through water and salt placements (Beck and Mitchell 2000). Treatments used to manipulate vegetation ultimately may have far greater effect on sage-grouse through long-term habitat changes rather than direct impacts of grazing itself (Freilich et al. 2003, Knick et al. 2011). An important objective in managing livestock grazing is to maintain residual cover of herbaceous vegetation to reduce predation during nesting (Beck and Mitchell 2000) and to maintain the integrity of riparian vegetation and other wetlands (Crawford et al. 2004). Proper livestock management (timing, location, and intensity) can assist in meeting sage-grouse habitat objectives and reduce fuels (Briske et al. 2011).

- Within priority sage-grouse habitat, incorporate sage-grouse habitat objectives and management considerations into all BLM grazing allotments through AMPs or permit renewals.

- Work cooperatively on integrated ranch planning within sage-grouse habitat so operations with deeded/BLM allotments can be planned as single units.
- Prioritize completion of land health assessments and processing grazing permits within priority sage-grouse habitat areas. Focus this process on allotments that have the best opportunities for conserving, enhancing or restoring habitat for sage-grouse. Utilize Ecological Site Descriptions (ESDs) to conduct land health assessments to determine if standards of range-land health are being met.
- Conduct land health assessments that include (at a minimum) indicators and measurements of structure/condition/composition of vegetation specific to achieving sage-grouse habitat objectives (Doherty et al. 2011). If local/state seasonal habitat objectives are not available, use sage-grouse habitat recommendations from Connelly et al. 2000b and Hagen et al. 2007.

Implementing Management Actions after Land Health and Habitat Evaluations

- Develop specific objectives to conserve, enhance or restore priority sage-grouse habitat based on ESDs and assessments (including within wetlands and riparian areas). If an effective grazing system that meets sage-grouse habitat requirements is not already in place, analyze at least one alternative that conserves, restores or enhances sage-grouse habitat in the NEPA document prepared for the permit renewal (Doherty et al. 2011b, Williams et al. 2011).
- Manage for vegetation composition and structure consistent with ecological site potential and within the reference state to achieve sage-grouse seasonal habitat objectives.
- Implement management actions (grazing decisions, AMP/Conservation Plan development, or other agreements) to modify grazing management to meet seasonal sage-grouse habitat requirements (Connelly et al. 2011c). Consider singly, or in combination, changes in:

    1) Season or timing of use;

    2) Numbers of livestock (includes temporary non-use or livestock removal);

    3) Distribution of livestock use;

    4) Intensity of use; and

    5) Type of livestock (e.g., cattle, sheep, horses, llamas, alpacas and goats) (Briske et al. 2011).
- During drought periods, prioritize evaluating effects of the drought in priority sage-grouse habitat areas relative to their needs for food and cover. Since there is a lag in vegetation recovery following drought (Thurow and Taylor 1999, Cagney et al. 2010), ensure that post-drought management allows for vegetation recovery that meets sage-grouse needs in priority sage-grouse habitat areas.

Riparian Areas and Wet Meadows

- Manage riparian areas and wet meadows for proper functioning condition within priority sage-grouse habitats.
    - o Within priority and general sage-grouse habitats, manage wet meadows to maintain a component of perennial forbs with diverse species richness relative to site potential (e.g., reference state) to facilitate brood rearing. Also conserve or enhance these wet meadow complexes to maintain or increase amount of edge and cover within that edge to minimize elevated mortality during the late brood rearing period (Hagen et al. 2007, Kolada et al. 2009, Atamian et al. 2010).
- Where riparian areas and wet meadows meet proper functioning condition, strive to attain reference state vegetation relative to the ecological site description.
    - o For example: Within priority sage-grouse habitat, reduce hot season grazing on riparian and meadow complexes to promote recovery or maintenance of appropriate vegetation and water quality. Utilize fencing/herding techniques or seasonal use or livestock distribution changes to reduce pressure on riparian or wet meadow vegetation used by sage-grouse in the hot season (summer) (Aldridge and Brigham 2002, Crawford et al. 2004, Hagen et al. 2007).
- Authorize new water development for diversion from spring or seep source only when priority sage-grouse habitat would benefit from the development. This includes developing new water sources for livestock as part of an AMP/conservation plan to improve sage-grouse habitat.
- Analyze springs, seeps and associated pipelines to determine if modifications are necessary to maintain the continuity of the predevelopment riparian area within priority sage-grouse habitats. Make modifications where necessary, considering impacts to other water uses when such considerations are neutral or beneficial to sage-grouse.

Treatments to Increase Forage for Livestock/Wild Ungulates

<u>Priority sage-grouse habitat areas</u>

- Only allow treatments that conserve, enhance or restore sage-grouse habitat (this includes treatments that benefit livestock as part of an AMP/Conservation Plan to improve sage-grouse habitat.[iv]
- Evaluate the role of existing seedings that are currently composed of primarily introduced perennial grasses in and adjacent to priority sage-grouse habitats to determine if they should be restored to sagebrush or habitat of higher quality for sage-grouse. If these seedings are part of an AMP/

[iv] Conserve or enhance means to allow no degradation and can mean that the improvement or livestock supplement is part of a grazing/AMP/Conservation Plan that facilitates meeting sage-grouse habitat objectives within a pasture or allotment.

Conservation Plan or if they provide value in conserving or enhancing the rest of the priority habitats, then no restoration would be necessary. Assess the compatibility of these seedings for sage-grouse habitat or as a component of a grazing system during the land health assessments (Davies et al. 2011).

- For example: Some introduced grass seedings are an integral part of a livestock management plan and reduce grazing pressure in important sagebrush habitats or serve as a strategic fuels management area.

Structural Range Improvements and Livestock Management Tools

Priority sage-grouse habitat areas

- Design any new structural range improvements and location of supplements (salt or protein blocks) to conserve, enhance, or restore sage-grouse habitat through an improved grazing management system relative to sage-grouse objectives. Structural range improvements, in this context, include but are not limited to: cattleguards, fences, exclosures, corrals or other livestock handling structures; pipelines, troughs, storage tanks (including moveable tanks used in livestock water hauling), windmills, ponds/reservoirs, solar panels and spring developments. Potential for invasive species establishment or increase following construction must be considered in the project planning process and monitored and treated post-construction.
- When developing or modifying water developments, use best management practices (BMPs, see Appendix C) to mitigate potential impacts from West Nile virus (Clark et al. 2006, Doherty 2007, Walker et al. 2007b, Walker and Naugle 2011).
- Evaluate existing structural range improvements and location of supplements (salt or protein blocks) to make sure they conserve, enhance or restore sage-grouse habitat.
  - To reduce outright sage-grouse strikes and mortality, remove, modify or mark fences in high risk areas within priority sage-grouse habitat based on proximity to lek, lek size, and topography (Christiansen 2009, Stevens 2011).
  - Monitor for, and treat invasive species associated with existing range improvements (Gelbard and Belnap 2003 and Bergquist et al. 2007).

Retirement of Grazing Privileges

- Maintain retirement of grazing privileges as an option in priority sage-grouse areas when base property is transferred or the current permittee is willing to retire grazing on all or part of an allotment. Analyze the adverse impacts of no livestock use on wildfire and invasive species threats (Crawford et al. 2004) in evaluating retirement proposals.

  Planning direction Note: Each planning effort will identify the specific allotment(s) where permanent retirement of grazing privileges is potentially beneficial.

Wild Horse and Burro Management

Wild horses and burros have the potential to impact habitats used by sage-grouse by reducing grass, shrub, and forb cover and increasing unpalatable forbs and exotic plants including cheatgrass (Beever and Aldridge 2011). Effects of wild equids on habitats may be especially pronounced during periods of drought or vegetation stress. Wild equids have different grazing patterns than domestic livestock, thus increasing the magnitude of grazing across the entire landscape (Beever and Aldridge 2011).

Ongoing Authorizations/Activities

- Manage wild horse and burro population levels within established Appropriate Management Levels (AML).
- Prioritize gathers in priority sage-grouse habitat, unless removals are necessary in other areas to prevent catastrophic environmental issues, including herd health impacts.

Proposed Authorization/Activities

- Within priority sage-grouse habitat, develop or amend herd management area plans (HMAPs) to incorporate sage-grouse habitat objectives and management considerations for all BLM herd management areas (HMAs).
    - For all HMAs within priority sage-grouse habitat, prioritize the evaluation of all AMLs based on indicators that address structure/condition/composition of vegetation and measurements specific to achieving sage-grouse habitat objectives.
- Coordinate with other resources (Range, Wildlife, and Riparian) to conduct land health assessments to determine existing structure/condition/composition of vegetation within all BLM HMAs.
- When conducting NEPA analysis for wild horse and burro management activities, water developments or other rangeland improvements for wild horses in priority sage-grouse habitat, address the direct and indirect effects to sage-grouse populations and habitat. Implement any water developments or rangeland improvements using the criteria identified for domestic livestock identified above in priority habitats.

Minerals

The primary potential risks to sage-grouse from energy and mineral development are:

1) Direct disturbance, displacement, or mortality of grouse;
2) Direct loss of habitat, or loss of effective habitat through fragmentation and reduced habitat patch size and quality; and
3) Cumulative landscape-level impacts (Bergquist et al. 2007, Walston et al. 2009, Naugle et al. 2011).

There is strong evidence from the literature to support that surface-disturbing energy or mineral development within priority sage-grouse habitats is not consistent with a goal to maintain or increase populations or distribution. None of the published science reports a positive influence of development on sage-grouse populations or habitats. Breeding populations are severely reduced at well pad densities commonly permitted (Holloran 2005, Walker et al. 2007a). Magnitude of losses varies from one field to another, but findings suggest that impacts are universally negative and typically severe.

Mechanisms that lead to avoidance and decreased fitness have not been empirically tested but rather suggested from multiple correlative and observational studies. For example, abandonment may increase if leks are repeatedly disturbed by raptors perching on power lines near leks (Ellis 1984), by vehicle traffic on nearby roads (Lyon and Anderson 2003), or by noise and human activity associated with energy development during the breeding season (Remington and Braun 1991, Holloran 2005, Kaiser 2006, Blickley and Patricelli *In review*). One recently completed research study in Wyoming (Blickley et al. *In press*), experimentally validates noise from natural gas drilling and roads resulted in a decline of 29% and 73% respectively in male peak attendance at leks relative to paired controls; declines were immediate and sustained throughout the experiment with low statistical support for a cumulative effect of noise over time. Collisions with nearby power lines and vehicles and increased predation by raptors may also increase mortality of birds at leks (Connelly et al. 2000). Alternatively, roads and power lines may indirectly affect lek persistence by altering productivity of local populations or survival at other times of the year. For example, sage-grouse mortality associated with power lines and roads occurs year-round (Beck et al. 2006, Aldridge and Boyce 2007), and ponds created by coal bed natural gas development may increase the risk of West Nile virus mortality in late summer (Walker et al. 2004, Zou et al. 2006, Walker et al. 2007b). Loss and degradation of sagebrush habitat can also reduce carrying capacity of local breeding populations (Swenson et al. 1987, Braun 1998, Connelly et al. 2000, 2000b, Crawford et al. 2004). Birds may avoid otherwise suitable habitat as the density of roads, power lines, or energy development increases (Lyon and Anderson 2003, Holloran 2005, Kaiser 2006, Doherty et al. 2008, Carpenter et al. 2010).

Negative responses of sage-grouse to energy development were consistent among studies regardless of whether they examined lek dynamics or demographic rates of specific cohorts within populations. Sage-grouse populations decline when birds avoid infrastructure in one or more seasons (Doherty et al. 2008, Carpenter et al. 2010) and when cumulative impacts of development negatively affect reproduction or survival (Aldridge and Boyce 2007), or both demographic rates (Lyon and Anderson 2003, Holloran 2005, Holloran et al. 2010). Avoidance of energy development at the scale of entire oil and gas fields should not be considered a simple shift in habitat use but rather a reduction in the distribution of sage-grouse (Walker et al. 2007). Avoidance is likely to result in true population declines if density dependence, competition, or displacement of birds into poorer-quality adjacent habitats lowers survival or reproduction (Holloran and Anderson 2005, Aldridge and Boyce 2007, Holloran et al. 2010). High site fidelity in sage-grouse also suggests that unfamiliarity with new habitats may also reduce survival, as in other grouse species (Yoder et al. 2004). Sage-grouse in the Powder River Basin were 1.3 times more likely to occupy winter habitats that had not been developed for energy (12 wells per 4 square kilometers or 12 wells per 1.5 square miles), and avoidance of developed areas was most pronounced when it occurred in high-quality winter habitat with abundant sagebrush (Doherty et al. 2008). In a similar study in Alberta, avoidance of otherwise suitable wintering habitats within a 1.9-kilometer (1.2 mile) radius of energy development resulted in substantial loss of functional habitat surrounding wells (Carpenter et al. 2010).

Long-term studies in the Pinedale Anticline Project Area in southwest Wyoming present the most complete picture of cumulative impacts and provide a mechanistic explanation for declines in populations. Early in development, nest sites were farther from disturbed than undisturbed leks, the rate of nest initiation from disturbed leks was 24 percent lower than for birds breeding on undisturbed leks, and 26 percent fewer females from disturbed leks initiated nests in consecutive years (Lyon and Anderson 2003). As development progressed, adult females remained in traditional nesting areas regardless of increasing levels of development, but yearlings that had not yet imprinted on habitats inside the gas field avoided development by nesting farther from roads (Holloran 2005). The most recent study confirmed that yearling females avoided infrastructure when selecting nest sites, and yearling males avoided leks inside of development and were displaced to the periphery of the gas field (Holloran et al. 2010). Recruitment of males to leks also declined as distance within the external limit of development increased, indicating a high likelihood of lek loss near the center of developed oil and gas fields (Kaiser 2006). The most important finding from studies in Pinedale was that sage-grouse declines are explained in part by lower annual survival of female sage-grouse and that the impact on survival resulted in a population-level decline (Holloran 2005). High site fidelity but low survival of adult sage-grouse combined with lek avoidance by younger birds (Holloran et al. 2010) resulted in a time lag of 3–4 years between the onset of development activities and lek loss (Holloran 2005). The time lag observed by Holloran (2005) in the Anticline matched that for leks that became inactive 3–4 years after natural gas development in the Powder River Basin (Walker et al. 2007a). Analysis of seven oil and gas fields across Wyoming showed time lags of 2–10 years between activities associated with energy development and its measurable effects on sage-grouse populations (Harju et al. 2010).

Impacts as measured by the number of males attending leks are most severe near the lek, remain discernible out to >4 miles (Holloran 2005, Walker et al. 2007, Tack 2009, Johnson et al. 2011), and often result in lek extirpations (Holloran 2005, Walker et al. 2007). Negative effects of well surface occupancy were apparent out to 3.1 miles, the largest radius investigated, in 2 of 7 study areas in Wyoming (Harju et al. 2010). Curvilinear relationships show that lek counts decreased with distance to the nearest active drilling rig, producing well, or main haul road and that development within 3 to 4 miles of leks decrease counts of displaying males (Holloran 2005). All well-supported models in Walker et al. (2007) indicate a strong negative effect, estimated as proportion of development within either 0.5 miles or 2 miles, on lek persistence. A model with development at 4 miles had less support, but the regression coefficient indicated that negative impacts within 4 miles were still apparent. Two additional studies reported negative impacts apparent out to 8 miles on large lek occurrence (>25 males; Tack 2009) and out to 11.7 miles on lek trends (Johnson et al. 2011), the largest scales evaluated.

Past BLM conservation measures have focused on 0.25 mile No Surface Occupancy (NSO) buffers around leks, and timing stipulations applied to 0.6 mile buffers around leks to protect both breeding and nesting activities. Given impacts of large scale disturbances described above that occur across seasons and impact all demographic rates, applying NSO or other buffers around leks at any distance is unlikely to be effective. Even if this approach were to be continued, it should be noted that protecting even 75 to >80% of nesting hens would require a 4-mile radius buffer (Table 1). Even a 4-mile NSO buffer would not be large enough to offset all the impacts reviewed above. A 4-mile NSO likely would not be practical given most leases are not large enough to accommodate a buffer of this size, and lek spacing within priority habitats is such that lek-based buffers may overlap and preclude all development.

We do not include timing restrictions on construction and drilling during the breeding season because they do not prevent impacts of infrastructure (e.g., avoidance, mortality) at other times of the year, during the production phase, or in other seasonal habitats that are crucial for population persistence (e.g., winter; Walker et al. 2007). Seasonal timing restrictions may be effective during the exploration phase. Instead, we recommend excluding mineral development and other large scale disturbances from priority habitats where possible, and where it is not limit disturbance as much as possible.

For these reasons, we believe the conservation strategy most likely to meet the objective of maintaining or increasing sage-grouse distribution and abundance is to exclude energy development and other large scale disturbances from priority habitats, and where valid existing rights exist, minimize those impacts by keeping disturbances to 1 per section with direct surface disturbance impacts held to 3% of the area or less.

Table 1. Distance Of Greater Sage-Grouse Nests From Lek Of Capture[1]

% Nests within 2-mi. radius	% Nests Within 4-mi. radius	Location	Study
46.4 (n = 13/28)	85.7 (n = 24/28)	North Park, CO	Peterson (1980)
59.5 (n = 182/306)	85 (n = 260/306)	Idaho	Autenrieth (1981)
71.8 (n = 51/71)	90.1 (n = 64/71)	North Park, CO	Giesen (1995)
49.5 (n = 192/388)	77.1 (n = 299/388)	Moffat County, CO	Thompson et al. 2005, Thompson 2006
48.4 (n = 15/31)	96.8 (n = 30/31)	Eagle and South Routt Counties, CO	Graham and McConnell 2004, Graham and Jones 2005
44.7 (n = 152/340)	74.4 (n = 243/340)	Wyoming	Holloran and Anderson (2005)
35.5 (n = 86/238)	61 (n = 145/238) @ 3 miles (data unavailable at this time for 4 miles)	Montana	Moynahan and Lindberg (2006)
35.5 (n = 27/76)	76.3 (n = 58/76)	Montana	Tack (2009)
50 (n = 495)	>80 (n = 495)	Oregon	Hagen (2011)

[1]Data obtained from Colorado Greater Sage-grouse Conservation Plan and additional recent studies/plans.

Fluid Minerals

Unleased Federal Fluid Mineral Estate

Alternative A

- Close priority sage-grouse habitat areas to fluid mineral leasing. Upon expiration or termination of existing leases, do not accept nominations/expressions of interest for parcels within priority areas.
- Allow geophysical exploration within priority sage-grouse habitat areas to obtain exploratory information for areas outside of and adjacent to priority sage-grouse habitat areas. Allow geophysical operations only by helicopter-portable drilling methods and in accordance with seasonal timing restrictions and/or other restrictions that may apply.

Alternative B

- Close priority sage-grouse habitat areas to fluid mineral leasing. Consider an exception:
    - o When there is an opportunity for the BLM to influence conservation measures where surface and/or mineral ownership is not entirely federally owned (i.e., checkerboard ownership). In this case, a plan amendment may be developed that opens the priority area for new leasing. The plan must demonstrate long-term population increases in the priority area through mitigation (prior to issuing the lease) including lease stipulations, off-site mitigation, etc., and avoid short-term losses that put the sage-grouse population at risk from stochastic events leading to extirpation.
- Allow geophysical exploration within priority sage-grouse habitat areas to obtain exploratory information for areas outside of and adjacent to priority sage-grouse habitat areas. Only allow geophysical operations by helicopter-portable drilling methods and in accordance with seasonal timing restrictions and/or other restrictions that may apply.

Leased Federal Fluid Mineral Estate

<u>Priority sage-grouse habitat areas </u>(with varying levels of exploration & development)

Apply the following conservation measures through Resource Management Plan (RMP) implementation decisions (e.g., approval of an Application for Permit to Drill, Sundry Notice, etc.) and upon completion of the environmental record of review (43 CFR 3162.5), including appropriate documentation of compliance with NEPA. In this process evaluate, among other things:

1. Whether the conservation measure is "reasonable" (43 CFR 3101.1-2) with the valid existing rights; and
2. Whether the action is in conformance with the approved RMP.[v]

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[v] Plan conformance means, "a resource management action shall be specifically provided for in the plan, or if not specifically mentioned, shall be clearly consistent with the terms, conditions, and decisions of the approved plan or amendment." 43 CFR 1601.0-5(b).

Provide the following conservation measures as terms and conditions of the approved RMP:

- Do not allow new surface occupancy on federal leases within priority habitats, this includes winter concentration areas (Doherty et al. 2008, Carpenter et al. 2010) during any time of the year. Consider an exception:
    - If the lease is entirely within priority habitats, apply a 4-mile NSO around the lek, and limit permitted disturbances to 1 per section with no more than 3% surface disturbance in that section.
    - If the entire lease is within the 4-mile lek perimeter, limit permitted disturbances to 1 per section with no more than 3% surface disturbance in that section. Require any development to be placed at the most distal part of the lease from the lek, or, depending on topography and other habitat aspects, in an area that is less demonstrably harmful to sage-grouse.
- Apply a seasonal restriction on exploratory drilling that prohibits surface-disturbing activities during the nesting and early brood-rearing season in all priority sage-grouse habitat during this period.
- Do not use Categorical Exclusions (CXs) including under the Energy Policy Act of 2005, Section 390 in priority sage-grouse habitats due to resource conflicts.
- Complete Master Development Plans in lieu of Application for Permit to Drill (APD)-by-APD processing for all but wildcat wells.
- When permitting APDs on existing leases that are not yet developed, the proposed surface disturbance cannot exceed 3% for that area. Consider an exception if:
    - Additional, effective mitigation is demonstrated to offset the resulting loss of sage-grouse (see Objectives).
        - When necessary, conduct additional, effective mitigation in 1) priority sage-grouse habitat areas or – less preferably – 2) general sage-grouse habitat (dependent upon the area-specific ability to increase sage-grouse populations).
        - Conduct additional, effective mitigation first within the same population area where the impact is realized, and if not possible then conduct mitigation within the same Management Zone as the impact, per 2006 WAFWA Strategy – pg 2-17.
- Require unitization when deemed necessary for proper development and operation of an area (with strong oversight and monitoring) to minimize adverse impacts to sage-grouse according to the Federal Lease Form, 3100-11, Sections 4 and 6.
- Identify areas where acquisitions (including subsurface mineral rights) or conservation easements, would benefit sage-grouse habitat.
- Require a full reclamation bond specific to the site. Insure bonds are sufficient for costs relative to reclamation (Connelly et al. 2000, Hagen et al. 2007) that would result in full restoration. Base the reclamation costs on the assumption that contractors for the BLM will perform the work.

- Make applicable Best Management Practices (BMPs, see Appendix D) mandatory as Conditions of Approval within priority sage-grouse habitat.

Solid Minerals

Coal

Priority sage-grouse habitat areas

- *Surface mines*: Find unsuitable all surface mining of coal under the criteria set forth in 43 CFR 3461.5.
- *Sub-surface mines:* Grant no new mining leases unless all surface disturbances (appurtenant facilities) are placed outside of the priority sage-grouse habitat area.
- For coal mining operations on existing leases:
    - *Sub-surface mining:* in priority sage-grouse habitat areas, place any new appurtenant facilities outside of priority areas. Where new appurtenant facilities associated with the existing lease cannot be located outside the priority sage-grouse habitat area, co-locate new facilities within existing disturbed areas. If this is not possible, then build any new appurtenant facilities to the absolute minimum standard necessary.

General sage-grouse habitat

- Apply minimization of surface-disturbing or disrupting activities (including operations and maintenance) where needed to reduce the impacts of human activities on important seasonal sage-grouse habitats. Apply these measures during activity level planning.
    - Use additional, effective mitigation to offset impacts as appropriate (determined by local options/needs).

Locatable Minerals

Priority sage-grouse habitat areas

- Propose withdrawal from mineral entry based on risk to the sage-grouse and its habitat from conflicting locatable mineral potential and development.
    - Make any existing claims within the withdrawal area subject to validity patent exams or buy out. Include claims that have been subsequently determined to be null and void in the proposed withdrawal.
    - In plans of operations required prior to any proposed surface disturbing activities, include the following:
        - Additional, effective mitigation in perpetuity for conservation (In accordance with existing policy, WO IM 2008-204). Example: purchase private land and mineral rights or severed subsurface mineral rights within the priority area and deed to US Government).

    - Consider seasonal restrictions if deemed effective.
- Make applicable Best Management Practices (see Appendix E) mandatory as Conditions of Approval within priority sage-grouse habitat.

Non-energy Leasable Minerals (i.e. sodium, potash)

Priority sage-grouse habitat areas

- Close priority habitat to non-energy leasable mineral leasing. This includes not permitting any new leases to expand an existing mine.
- For existing non-energy leasable mineral leases, in addition to the solid minerals BMPs (Appendix E), follow the same BMPs applied to Fluid Minerals (Appendix D), when wells are used for solution mining.

Saleable Mineral Materials

Priority sage-grouse habitat areas

- Close priority habitat to mineral material sales.
- Restore saleable mineral pits no longer in use to meet sage-grouse habitat conservation objectives.

Mineral Split Estate

Priority sage-grouse habitat areas

- Where the federal government owns the mineral estate, and the surface is in non-federal ownership, apply the conservation measures applied on public lands.
- Where the federal government owns the surface, and the mineral estate is in non-federal ownership, apply appropriate Fluid Mineral BMPs (see Appendix D) to surface development.

Wildfire Suppression, Fuels Management and Fire Rehabilitation

These programs address the threats resulting from wildfires and post-wildfire effects along with a program (fuels management) designed to try to reduce these impacts. Together these programs provide a significant opportunity to influence sagebrush habitats that benefit sage-grouse. Wildfire, particularly in low elevation Wyoming big sagebrush systems, has resulted in significant habitat loss primarily because of subsequent invasion by cheatgrass and other exotic plant species (Miller et al. 2011). The number of fires and total acreage burned has increased throughout the sage-grouse range (Miller et al. 2011). Long-term monitoring following prescribed fire is important because treatments may not increase either yield or nutritional quality of forbs eaten by sage-grouse, and also may decrease abundance of insects that are important for growth of sage-grouse chicks (Beck et al. 2009, Rhodes et al. 2010). Therefore, it is critical not only to conduct management actions that reduce the long-term loss of sagebrush but also to restore and recover burned areas to habitats that will be used by sage-grouse (Pyke 2011). Prescribed fire is a tool that can assist in the recovery of sagebrush habitat in some vegetation types (Davies et al. 2011).

Fuels Management

Priority sage-grouse habitat areas

- Design and implement fuels treatments with an emphasis on protecting existing sagebrush ecosystems.
    - Do not reduce sagebrush canopy cover to less than 15% (Connelly et al. 2000, Hagen et al. 2007) unless a fuels management objective requires additional reduction in sagebrush cover to meet strategic protection of priority sage-grouse habitat and conserve habitat quality for the species. Closely evaluate the benefits of the fuel break against the additional loss of sagebrush cover in the EA process.
    - Apply appropriate seasonal restrictions for implementing fuels management treatments according to the type of seasonal habitats present in a priority area.
    - Allow no treatments in known winter range unless the treatments are designed to strategically reduce wildfire risk around or in the winter range and will maintain winter range habitat quality.
    - Do not use fire to treat sagebrush in less than 12-inch precipitation zones (e.g., Wyoming big sagebrush or other xeric sagebrush species; Connelly et al. 2000, Hagen et al. 2007, Beck et al. 2009). However, if as a last resort and after all other treatment opportunities have been explored and site specific variables allow, the use of prescribed fire for fuel breaks that would disrupt the fuel continuity across the landscape could be considered, in stands where cheatgrass is a very minor component in the understory (Brown 1982).
    - Monitor and control invasive vegetation post-treatment.
    - Rest treated areas from grazing for two full growing seasons unless vegetation recovery dictates otherwise (WGFD 2011).
    - Require use of native seeds for fuels management treatment based on availability, adaptation (site potential), and probability of success (Richards et al. 1998). Where probability of success or native seed availability is low, non-native seeds may be used as long as they meet sage-grouse habitat objectives (Pyke 2011).
    - Design post fuels management projects to ensure long term persistence of seeded or pre-treatment native plants. This may require temporary or long-term changes in livestock grazing management, wild horse and burro management, travel management, or other activities to achieve and maintain the desired condition of the fuels management project (Eiswerth and Shonkwiler 2006).

- Design fuels management projects in priority sage-grouse habitat to strategically and effectively reduce wildfire threats in the greatest area. This may require fuels treatments implemented in a more linear versus block design (Launchbaugh et al. 2007).

During fuels management project design, consider the utility of using livestock to strategically reduce fine fuels (Diamond et al. 2009), and implement grazing management that will accomplish this objective Davies et al. 2011 and Launchbaugh et al. 2007). Consult with ecologists to minimize impacts to native perennial grasses.

Fire operations

- In priority sage-grouse habitat areas, prioritize suppression, immediately after life and property, to conserve the habitat.
- In general sage-grouse habitat, prioritize suppression where wildfires threaten priority sage-grouse habitat.
- Follow Best Management Practices (WO IM 2011-138, see appendix E.)

Emergency Stabilization and Rehabilitation (ES&R)

- Prioritize native seed allocation for use in sage-grouse habitat in years when preferred native seed is in short supply. This may require reallocation of native seed from ES&R projects outside of priority sage-grouse habitat to those inside it. Use of native plant seeds for ES&R seedings is required based on availability, adaptation (site potential), and probability of success Richards et al. 1998). Where probability of success or native seed availability is low, non-native seeds may be used as long as they meet sage-grouse habitat conservation objectives (Pyke 2011). Re-establishment of appropriate sagebrush species/subspecies and important understory plants, relative to site potential, shall be the highest priority for rehabilitation efforts.
- Design post ES&R management to ensure long term persistence of seeded or pre-burn native plants. This may require temporary or long-term changes in livestock grazing, wild horse and burro, and travel management, etc., to achieve and maintain the desired condition of ES&R projects to benefit sage-grouse (Eiswerth and Shonkwiler 2006).
- Consider potential changes in climate (Miller at al. 2011) when proposing post-fire seedings using native plants. Consider seed collections from the warmer component within a species' current range for selection of native seed. (Kramer and Havens 2009).

Habitat Restoration

Habitat restoration cross-cuts all programs. It is an important tool to create and/or maintain a landscape that benefits sage-grouse.

- Prioritize implementation of restoration projects based on environmental variables that improve chances for project success in areas most likely to benefit sage-grouse (Meinke et al. 2009).
  - Prioritize restoration in seasonal habitats that are thought to be limiting sage-grouse distribution and/or abundance.
- Include sage-grouse habitat parameters as defined by Connelly et al. (2000), Hagen et al. (2007) or if available, State Sage-Grouse Conservation plans and appropriate local information in habitat restoration objectives. Make meeting these objectives within priority sage-grouse habitat areas the highest restoration priority.
- Require use of native seeds for restoration based on availability, adaptation (ecological site potential), and probability of success (Richards et al. 1998). Where probability of success or adapted seed availability is low, non-native seeds may be used as long as they support sage-grouse habitat objectives (Pyke 2011).
- Design post restoration management to ensure long term persistence. This could include changes in livestock grazing management, wild horse and burro management and travel management, etc., to achieve and maintain the desired condition of the restoration effort that benefits sage-grouse (Eiswerth and Shonkwiler 2006).
- Consider potential changes in climate (Miller et al. 2011) when proposing restoration seedings when using native plants. Consider collection from the warmer component of the species current range when selecting native species (Kramer and Havens 2009).
- Restore native (or desirable) plants and create landscape patterns which most benefit sage-grouse.
- Make re-establishment of sagebrush cover and desirable understory plants (relative to ecological site potential) the highest priority for restoration efforts.
- In fire prone areas where sagebrush seed is required for sage-grouse habitat restoration, consider establishing seed harvest areas that are managed for seed production (Armstrong 2007) and are a priority for protection from outside disturbances.

Monitoring of Sage-grouse and Sagebrush Habitats

Given the degree of uncertainty associated with managing natural resources, adaptive management approaches that include rigorous monitoring protocols to support them are essential if conservation goals are to be realized (Walters 1986, Burgman et al. 2005, Stankey et al. 2005, Turner 2005, Lyons et al. 2008). Recent efforts to develop range-wide policy and conservation measures for sage-grouse have emphasized the importance of improving monitoring efforts on both sage-grouse distribution and population trends, and the habitat they depend on (Wambolt et al. 2002, Stiver et al. 2006, Reese and Boyer 2007, Connelly et al. 2011a).

Monitoring is necessary to provide an objective appraisal of the effects of potentially positive conservation actions, and to assess the relative negative effects of management actions to sage-grouse populations and their habitats. Adaptive management planning also reveals substantial gaps in knowledge about key processes and functional relationships (Walters 1987), and therefore helps to identify and prioritize research needs. Ideally, monitoring attributes of sage-grouse habitat and sage-grouse populations will allow linking real or potential habitat changes from natural events and management actions to vital rates of sage-grouse populations (Stiver et al. 2006, Naugle and Walker 2007). Population monitoring led by State wildlife agencies and consistent long-term habitat monitoring among all jurisdictions will enable managers to identify indicators associated with population change across large landscapes and to ameliorate negative effects with appropriate conservation actions (Burgman et al. 2005, Turner 2005).

Sage-grouse select habitats at multiple scales across large landscapes (Connelly et al. 2003, Stiver et al. 2006), which monitoring strategies for sage-grouse habitats must reflect. At landscape levels (RMP level), monitoring should track percent of sagebrush and cover and maturity of stands, preservation of key seasonal habitat components, and the degree of connectivity among populations, seasonal habitats and stands. At the project level, a truly effective monitoring strategy will include measures as to how plant communities respond, how that relates to structural and other sage-grouse habitat requirements, and how sage-grouse populations respond demographically. Quantitative data for habitat measurements should be collected that are sensitive to the land use change being proposed (Stiver et al 2006). Monitoring must occur over the proper time frames to evaluate temporal variation of important components of sage-grouse habitats (Stiver et al. 2006).

Recognizing the importance of monitoring both sage-grouse habitat and populations, BLM in November 2004, completed the National Sage-Grouse Habitat Conservation Strategy (USDI BLM 2004) to address conservation and management of sage-grouse. The overarching goal was to "provide a consistent and scientifically based approach for collection and use of monitoring data for sagebrush habitats, sage-grouse and other components of the sagebrush community." Four action items were identified to accomplish this goal: 1) Develop, cooperatively with our partners, appropriate monitoring strategies and protocols at the appropriate scale for sage-grouse habitat in conjunction with the development of the range-wide conservation action plan; 2) Develop, cooperatively with our partners, a sage-grouse habitat assessment methodology in conjunction with development of the range-wide conservation action plan; 3) Incorporate the sage-grouse habitat assessment framework into the land health assessment process for evaluating indicators of healthy rangelands; and 4) In conjunction with the development of the range-wide conservation action plan, issue guidance for collecting fine-scale monitoring and assessment information and incorporating requirements into implementation projects and plans.

To date, BLM has completed portions of the above action items. In August 2010, the Sage-Grouse Habitat Assessment Framework: Multi-scale Habitat Assessment Tool was completed (Stiver et al. 2010). The assessment framework provides policy makers, resource managers, and natural resource specialists a comprehensive framework for landscape conservation in sagebrush ecosystems with an emphasis on sage-grouse. Implementation policy directing consistent use of the assessment still needs to be completed by BLM in addition to other guidance identified in the strategy.

BLM has recently completed the agency's Assessment, Inventory, and Monitoring (AIM) Strategy (Toevs 2011). The AIM strategy identifies "core indicators" for reporting landscape level attributes. The AIM strategy has resulted in BLM adopting the Natural Resource Conservation Service's National Resource Inventory (NRI) methodology as part of BLM's Landscape Monitoring Project. The NRI protocols provide BLM a statistical framework for evaluating management actions, and programs and policies at a landscape or regional level. Initial NRI data collection occurred on all lands managed by BLM during the summer of 2011. During the summer of 2012 additional NRI monitoring sites are being incorporated to evaluate sagebrush habitats that contain approximately two-thirds of the sage-grouse populations west wide. At this time, the remaining sage-grouse populations have not been identified for long-term habitat monitoring due to funding short falls. In addition to prioritizing funding to fully achieve this objective, habitat monitoring protocols at a fine scale to evaluate impacts at a project level remain to be developed.

Estimates of sage-grouse population size are not available for any population, rather trends in population size are estimated through a lek count index. Exact estimates of sage grouse abundance, while desirable, are probably less important than trends and particularly how sage grouse respond to management actions.

Counts of males attending leks in the spring have been used by wildlife agencies as the primary index to population trends since Patterson suggested that this method might be useful in 1952 (Patterson 1952). Use of convenience sampling to monitor bird populations has been criticized (Ellingson and Lukacs 2003), and lek counts in particular have been challenged as inconsistently conducted, inherently biased and without any known relationship to population size (Beck and Braun 1980, Walsh et al. 2004, Sedinger 2007). Despite limitations of the method, lek counts remain the best available information on population trends over time, and pragmatic strategies to improve population estimation remain elusive (Reese and Bowyer 2007).

It is beyond the scope of this report to develop methodology to better estimate sage-grouse distribution and abundance, but rather to emphasize that WAFWA should convene a technical group for this purpose, and that this group should consider ways to:

1. Standardize, at least within management zones, lek count methodology.
2. Develop and implement methodology to estimate the number of leks in an unbiased manner (Walsh et al. 2004, Sedinger 2007), and determine the location of new or previously unknown leks (particularly important since priority habitat designations are based in large part on locations of leks).
3. Develop and implement methodology to estimate the proportion of males detected while attending leks, and explore degree and nature of variability.
4. Develop and explore methodology to estimate sex ratios within sage-grouse populations.
5. Use Geographic Information System (GIS) mapping technology and analytical tools to track changes in distribution over time, connectivity among populations and population segments, and explore spatially explicit models that link sage-grouse population performance with ecological indicators (Naugle and Walker 2007).

The standardization of monitoring methods and implementation of a defensible monitoring approach is vital if BLM and other conservation partners are to use the resulting information to guide implementation of conservation activities (Naugle and Walker 2007). Monitoring strategies for sage-grouse habitat and populations must be collaborative, as habitat occurs across varied land ownership (52% BLM, 8% USFS, 31% private 5% state, 4% BIA and other Federal; 75 FR 13910), and state fish and wildlife agencies have primary responsibility for population level management of wildlife, including monitoring.

Acronyms

AML	Appropriate Management Level
AMP	Allotment Management Plan
APD	Application of Permit to Drill
BLM	Bureau of Land Management
BMPs	Best Management Practices
CX	Categorical Exclusion
ERMA	Extensive Recreation Management Areas
ESA	Endangered Species Act
ESD	Ecological Site Description
ES&R	Emergency Stabilization and Rehabilitation
IM	Instruction Memorandum
MOU	Memorandum of Understanding
NEPA	National Environmental Policy Act
NGO	non-governmental organization
NMAC	National Multi-Agency Coordination Group
NRCS	Natural Resources Conservation Service
NPT	National Policy Team
NTT	National Technical Team
RIDT	Regional Interdisciplinary Team
RMP	Resource Management Plan
RMT	Regional Management Team
ROW	Right-of-Way
SRMA	Special Recreation Management Area
SRP	Special Recreation Permit
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
WAFWA	Western Association of Fish and Wildlife Agencies

Glossary

2008 WAFWA Sage-grouse MOU: A memorandum of understanding (MOU) among Western Association of Fish and Wildlife Agencies, U.S. Department of Agriculture, Forest Service, U.S. Department of the Interior, Bureau of Land Management, U.S. Department of the Interior, Fish and Wildlife Service, U.S. Department of the Interior, Geological Survey, U.S. Department of Agriculture, Natural Resources Conservation Service, and the U.S. Department of Agriculture, Farm Service Agency. The purpose of the MOU is to provide for cooperation among the participating state and federal land, wildlife management and science agencies in the conservation and management of sage-grouse (*Centrocercus urophasianus*) sagebrush (*Artemisia* spp.) habitats and other sagebrush-dependent wildlife throughout the western United States and Canada and a commitment of all agencies to implement the 2006 WAFWA Conservation Strategy.

2011 Partnership MOU: A partnership agreement among the United States Department of Agriculture Natural Resource Conservation Service, Forest Service, United State Department of the Interior, Bureau of Land Management, and Fish and Wildlife Service. 2011. This MOU is for range management – to implement NRCS practices on adjacent federal properties.

Administrative Access: A term used to describe access for resource management and administrative purposes such as fire suppression, cadastral surveys, permit compliance, law enforcement and military in the performance of their official duty, or other access needed to administer BLM-managed lands or uses.

Avoidance Areas: Areas to be avoided but that may be available for location of ROWs with special stipulations.

Best Management Practices (BMPs): A suite of techniques that guide or may be applied to management actions to aide in achieving desired outcomes. BMPs are often developed in conjunction with land use plans, but they are not considered a planning decision unless the plans specify that they are mandatory.

Casual Use: Casual use means activities ordinarily resulting in no or negligible disturbance of the public lands, resources, or improvements. For examples for rights of ways see 43 CFR 2801.5. For examples for locatable minerals see 43 CFR 3809.5.

Conservation Plan: The recorded decisions of a landowner or operator, cooperating with a conservation district, on how the landowner or operator plans, within practical limits, to use his/her land according to its capability and to treat it according to its needs for maintenance or improvement of the soil, water, animal, plant, and air resources.

Conserve: To cause no degradation or loss of sage-grouse habitat. Conserve can also refer to maintaining intact sagebrush steppe by fine tuning livestock use, watching for and treating new invasive species and maintaining existing range improvements that benefit sage-grouse etc.

Ecological Site: A distinctive kind of land with specific physical characteristics that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation.

Exploration: Active drilling and geophysical operations to:

a. Determine the presence of the mineral resource; or
b. Determine the extent of the reservoir.

Development: Active drilling and production of wells

Development Area: Areas primarily leased with active drilling and wells capable of production in payable quantities.

Enhance: The improvement of habitat by increasing missing or modifying unsatisfactory components and/or attributes of the plant community to meet sage-grouse objectives. Examples include modifying livestock grazing systems to improve the quantity and vigor of desirable forbs, improving water flow in riparian areas by modifying existing spring developments to return more water to the riparian area below the development, or marking fences to minimize sage-grouse hits and mortality.

General Sage-grouse Habitat: Is occupied (seasonal or year-round) habitat outside of priority habitat. These areas have been identified by state fish and wildlife agencies in coordination with respective BLM offices.

Integrated Ranch Planning: A method for ranch planning that takes a holistic look at all elements of the ranching operations, including strategic and tactical planning, rather than approaching planning as several separate enterprises.

Large Scale Anthropogenic Disturbances: Features include but are not limited to paved highways, graded gravel roads, transmission lines, substations, wind turbines, oil and gas wells, geothermal wells and associated facilities, pipelines, landfills, agricultural conversion, homes, and mines.

Late Brood Rearing Area: Habitat includes mesic sagebrush and mixed shrub communities, wet meadows, and riparian habitats as well as some agricultural lands (e.g. alfalfa fields, etc).

Lek:[vi] A traditional courtship display area attended by male sage-grouse in or adjacent to sagebrush dominated habitat. A lek is designated based on observations of two or more male sage-grouse engaged in courtship displays. Sub-dominant males may display on itinerant strutting areas during population peaks. Such areas usually fail to become established leks. Therefore, a site where less than five males are observed strutting should be confirmed active for two years before meeting the definition of a lek (Connelly et al 2000, Connelly et al. 2003, 2004).

> **Lek Complex:** A lek or group of leks within 2.5 km (1.5 mi) of each other between which male sage-grouse may interchange from one day to the next. Fidelity to leks has been well documented.

[vi] Each State may have a slightly different definition of lek, active lek, inactive lek, occupied, and unoccupied leks. Regional planning will use the appropriate definition provided by the State of interest.

Visits to multiple leks are most common among yearlings and less frequent for adult males, suggesting an age-related period of establishment (Connelly et al. 2004).

Active Lek: Any lek that has been attended by male sage-grouse during the strutting season.

Inactive Lek: Any lek where sufficient data suggests that there was no strutting activity throughout a strutting season. Absence of strutting grouse during a single visit is insufficient documentation to establish that a lek is inactive. This designation requires documentation of either: 1) an absence of sage-grouses on the lek during at least 2 ground surveys separated by at least seven days. These surveys must be conducted under ideal conditions (April 1-May 7 (or other appropriate date based on local conditions), no precipitation, light or no wind, half-hour before sunrise to one hour after sunrise) or 2) a ground check of the exact known lek site late in the strutting season (after April 15) that fails to find any sign (tracks, droppings, feathers) of strutting activity. Data collected by aerial surveys should not be used to designate inactive status as the aerial survey may actually disrupt activities.

Occupied Lek: A lek that has been active during at least one strutting season within the prior 10 years.

Unoccupied Lek: A lek that has either been “destroyed” or “abandoned.”

Destroyed Lek: A formerly active lek site and surrounding sagebrush habitat that has been destroyed and is no longer suitable for sage-grouse breeding.

Abandoned Lek: A lek in otherwise suitable habitat that has not been active during a period of 10 consecutive years. To be designated abandoned, a lek must be “inactive” (see above criteria) in at least four non-consecutive strutting seasons spanning the 10 years. The site of an “abandoned” lek should be surveyed at least once every 10 years to determine whether it has been re-occupied by sage-grouse.

Master Development Plans: A set of information common to multiple planned wells, including drilling plans, Surface Use Plans of Operations, and plans for future production.

Mitigation: Compensating for resource impacts by replacing or providing substitute resources or habitat.

Notice-level Mining Activities: To qualify for a Notice the mining activity must: 1) constitute exploration, 2) not involve bulk sampling of more than 1,000 tons of presumed ore, 3) must not exceed 5 acres of surface disturbance, and 4) must not occur in one of the special category lands listed in 43 CFR 3809.11(c). The Notice is to be filed in the BLM field office with jurisdiction over the land involved. The Notice does not need to be on a particular form but must contain the information required by 43 CFR 3809.301(b).

Offsite Mitigation: Compensating for resource impacts by replacing or providing substitute resources or habitat at a different location than the project area.

Plan of Operations: A Plan of Operations is required for all mining activity exploration greater than 5 acres or surface disturbance greater than casual use on certain special category lands. Special category lands are described under 43 CFR 3809.11(c) and include such lands as designated Areas of Critical Environmental Concern, lands within the National Wilderness Preservation System, and areas closed to off-road vehicles, among others. In addition, a plan of operations is required for activity greater than casual use on lands patented under the Stock Raising Homestead Act with Federal minerals where the operator does not have the written consent of the surface owner (43 CFR 3814). The Plan of operations needs to be filed in the BLM field office with jurisdiction over the land involved. The Plan of Operations does not need to be on a particular form but must address the information required by 43 CFR 3809.401(b).

Priority Sage-grouse Habitat: Areas that have been identified as having the highest conservation value to maintaining sustainable sage-grouse populations. These areas would include breeding, late brood-rearing, and winter concentration areas. These areas have been identified by state fish and wildlife agencies in coordination with respective BLM offices.

Range Improvement: The term range improvement means any activity, structure or program on or relating to rangelands which is designed to improve production of forage; change vegetative composition; control patterns of use; provide water; stabilize soil and water conditions; and provide habitat for livestock and wildlife. The term includes, but is not limited to, structures, treatment projects, and use of mechanical means to accomplish the desired results.

Roads, Primitive Roads and Trails: Roads, primitive roads or trails that have been specifically designated for motorized use through a public implementation-level National Environmental Policy Act process in accordance with 43 CFR, Part 8340.

Reclamation: Rehabilitation of a disturbed area to make it acceptable for designated uses. This normally involves re-contouring, replacement of topsoil, re-vegetation, and other work necessary to ensure eventual restoration of the site.

Reference State: The reference state is the state where the functional capacities represented by soil/site stability, hydrologic function, and biotic integrity are performing at an optimum level under the natural disturbance regime. This state usually includes, but is not limited to, what is often referred to as the potential natural plant community.

Restoration: Implementation of a set of actions that promotes plant community diversity and structure that allows plant communities to be more resilient to disturbance and invasive species over the long term. The long-term goal is to create functional, high quality habitat that is occupied by sage-grouse. Short-term goal may be to restore the landform, soils and hydrology and increase the percentage of preferred vegetation, seeding of desired species, or treatment of undesired species.

State: A state is comprised of an integrated soil and vegetation unit having one or more biological communities that occur on a particular ecological site and that are functionally similar with respect to the three attributes (soil/site stability, hydrologic function, and biotic integrity) under natural disturbance regimes.

Stochastic: Randomly determined event, chance event, a condition determined by predictable processes and a random element.

Surface Disruption: Resource uses and activities that are likely to alter the behavior of, displace, or cause stress to sage-grouse occurring at a specific location and/or time. Surface disruption includes those actions that alter behavior or cause the displacement of sage-grouse such that reproductive success is negatively affected, or the physiological ability to cope with environmental stress is compromised. Examples of disruptive activities may include noise, vehicle traffic, or other human presence regardless of the associated activity.

Surface Disturbance: Suitable habitat is considered disturbed when it is removed and unavailable for immediate sage-grouse use.

a. Long-term removal occurs when habitat is physically removed through activities that replace suitable habitat with long term occupancy of unsuitable habitat such as a road, powerline, well pad or active mine. Long-term removal may also result from any activities that cause soil mixing, soil removal, and exposure of the soil to erosive processes.

b. Short–term removal occurs when vegetation is removed in small areas, but restored to suitable habitat within a few years (< 5) of disturbance, such as a successfully reclaimed pipeline, or successfully reclaimed drill hole or pit.

c. Suitable habitat rendered unusable due to numerous anthropogenic disturbances

d. Anthropogenic surface disturbance are surface disturbances meeting the above definitions which result from human activities.

Transition: A shift between two states. Transitions are not reversible by simply altering the intensity or direction of factors that produced the change. Instead, they require new inputs such as revegetation or shrub removal. Practices, such as these, that accelerate succession are often expensive to apply.

Unitization: Operation of multiple leases as a single lease under a single operator

Wildcat Well: An exploratory oil well drilled in land not known to be an oil field.

Wildland Fire: Any non-structure fire that occurs in the vegetation and/or natural fuels. Includes both prescribed fire and wildfire (NWCG Memo #024-2010 April 30, 2010. www.nwcg.gov).

Winter Concentration Areas: Sage-grouse winter habitats which are occupied annually by sage-grouse and provide sufficient sagebrush cover and food to support birds throughout the entire winter (especially periods with above average snow cover). Many of these areas support several different breeding populations of sage-grouse. Sage-grouse typically show high fidelity for these areas, and loss or fragmentation can result in significant population impacts.

Literature Cited

75 FR 13910. 2010. Endangered and Threatened Wildlife and Plants: 12-month finding for petitions to list the greater sage-grouse as threatened or endangered.

Aldridge, C. L., and M. S. Boyce. 2007. Linking occurrence and fitness to persistence: Habitat-based approach for endangered greater sage-grouse. Ecological Applications 17:508–26.

Aldridge, C. L., and R. M. Brigham. 2002. Sage-grouse nesting and brood habitat use in southern Canada. Journal of Wildlife Management 66:433-444.

Aldridge, C. L., and R. M. Brigham. 2003. Distribution, abundance, and status of the greater sage-grouse, *Centrocercus urophasianus*, in Canada. Canadian Field-Naturalist 117:25-34.

Aldridge, C. L., S. E. Nielsen, H. L. Beyer, M. S. Boyce, J. W. Connelly, S. T. Knick, and M. A. Schroeder. 2008. Range-wide patterns of greater sage-grouse persistence. Diversity and Distributions 17:983-994.

Armstrong, J. C. 2007. Improving sustainable seed yield in Wyoming big sagebrush. Provo, UT: Brigham Young University. 29 p. Thesis.

Atamian, M. T., M. S. Sedinger, J. S. Heaton, and E. J. Blomberg. 2010. Landscape-level assessment of brood rearing habitat for greater sage-grouse in Nevada. Journal of Wildlife Management -74:1533-1543.

Autenrieth, R. E. 1981. Sage grouse management in Idaho. Idaho Department of Fish and Game, Wildlife Bulletin 9, Boise, Idaho, USA.

Beck, J.L., and D. L. Mitchell. 2000. Influences of livestock grazing on sage-grouse habitat. Wildlife Society Bulletin 28:993-1002.

Beck, J.L. K. P. Reese, J. W. Connelly, and M. B. Lucia. 2006. Movements and survival of juvenile greater sage-grouse in southeastern Idaho. Wildlife Society Bulletin 34:1070-1078.

Beck, J.L., J.W. Connelly, and K.P. Reese. 2009. Recovery of greater sage-grouse habitat features in Wyoming big sagebrush following prescribed fire. Restoration Ecology 17:393-403.

Beck, T.D.I., and C.E. Braun. 1980. The strutting ground count: variation, traditionalism, management needs. Proceedings of the Western Association of Fish and Wildlife Agencies 60:558-566.

Beever, E.A., and C. L. Aldridge. 2011. Influences of free-roaming equids on sagebrush ecosystems, with a focus on greater sage-grouse. Pp. 273-290 in S. T. Knick and J. W. Connelly (editors). Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press. Berkeley, CA.

Bergquist, E., P. Evangelista, T.J. Stohlgren, and N. Alley. 2007. Invasive species and coal bed methane development in the Powder River Basin, Wyoming. Environmental Monitoring and Assessment 128:381-394.

Blickely, J.L. and G.L. Patricelli. *In press*. Potential acoustical masking of greater sage-grouse display components by chronic industrial noise. Ornithological Monographs.

Blickley, J.L., D. Blackwood, and G.L. Patricelli. *In press.* Experimental evidence for the effects of chronic anthropogenic noise on abundance of greater sage-grouse at leks. Conservation Biology.

Braun, C.E. 1998. Sage grouse declines in western North America: What are the problems? Proceedings of the Western Association of State Fish and Wildlife Agencies 78:139–56.Braun, C. E., T. Britt, and R. O. Wallestad. 1977. Guidelines for maintenance of sage grouse habitats. Wildlife Society Bulletin 5:99-106.

Braun, C.E., T. Britt, and R.O. Wallestad. 1977. Guidelines for maintenance of sage grouse habitats. Wildlife Society Bulletin 5:99-106.

Briske, D.D., J.D. Derner, D.G. Milchunas, and K.W. Tate. 2011. An evidence-based assessment of prescribed grazing practices. Pp. 23-74 in D.D. Briske. Conservation benefits of rangeland resources: assessment, recommendations, and knowledge gaps. USDA National Resources Conservation Service, Washington D.C.

Brown, J.K. 1982. Fuel and fire behavior prediction in big sagebrush. US Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. Research Paper INT-290. Ogden, UT.

Burgman, M.A., D.B. Lindenmayer, and J. Elith. 2005. Managing landscapes for conservation under uncertainty. Ecology 86:2007-2017.

Cagney, J., E. Bainter, B. Budd, T. Christiansen, V. Herren, M. Holloran, B. Rashford, M. Smith and J. Williams. 2010. Grazing Influence, Objective Development, and Management in Wyoming's Greater Sage-Grouse Habitat. Cooperative Extension Service Bulletin B-1203, University of Wyoming, Laramie, WY.

Carpenter, J., C. Aldridge, and M.S. Boyce. 2010. Sage-grouse habitat selection during winter in Alberta. Journal of Wildlife Management -74:1806-1814.

Christiansen, T. 2009. Fence marking to reduce greater sage-grouse collisions and mortality near Farson, Wyoming – summary of interim results. Wyoming Game and Fish Department unpublished interim report.

Clark, L., J. Hall, R. McLean, M. Dunbar, K. Klenk, R. Bowen, and C.A. Smeraski. 2006. Susceptibility of greater sage-grouse to experimental infection with West Nile virus. Journal of Wildlife Diseases 42:14-42.

Colorado Greater Sage-grouse Steering Committee. 2008. Colorado greater sage-grouse conservation plan. Colorado Division of Wildlife, Denver, Colorado, USA.

Connelly, J.W., A. D. Apa, R. B. Smith, and K. P. Reese. 2000a. Effects of predation and hunting on adult sage grouse *Centrocercus urophasianus* in Idaho. Wildlife Biology 6:227–32.

Connelly, J.W., C.A. Hagen, and M.A. Schroeder. 2011b. Characteristics and dynamics of greater sage-grouse populations. Pp. 53-67 in S.T. Knick and J.W. Connelly. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitat. Studies in Avian Biology 38. University of California Press, Berkeley, CA.

Connelly, J.W., S.T. Knick, C.E. Braun, W.L. Baker, E.A. Beever, T. Christiansen, K.E. Doherty, E.O. Garton, S.E. Hanser, D.H. Johnson, M. Leu, R.F. Miller, D.E. Naugle, S.J. Oyler-McCance, D.A. Pyke, K.P. Reese, M.A. Schroeder, S.J. Stiver, B.L. Walker, and M.J. Wisdom. 2011. Conservation of Greater Sage-Grouse: a synthesis of current trends and future management. Pp. 549–563 in S. T. Knick and J. W. Connelly (editors). Greater Sage-Grouse: ecology and conservation of a landscape species and habitats. Studies in Avian Biology (vol. 38), University of California Press, Berkeley, CA.

Connelly, J.W., E.T. Rinkes, and C.E. Braun. 2011. Characteristics of greater sage-grouse habitats: a landscape species at micro- and macroscales. Pp. 69-83 in S.T. Knick and J.W. Connelly. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitat. Studies in Avian Biology 38. University of California Press, Berkeley, CA.

Connelly, J.W., K.P. Reese, and M.A. Schroeder. 2003. Monitoring sage-grouse habitats and populations. University of Idaho, College of Natural Resources Experiment Station Bulletin 80 Moscow, Idaho, USA.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Connelly, J.W., S.T. Knick, M.A. Schroeder and S.J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming, USA. (available at http://sagemap.wr.usgs.gov)

Crawford, J.A., R.A. Olson, N.E. West, J.C. Mosley, M.A. Schroeder, T.D. Whitson, R.F. Miller, M.A. Gregg, and C.S. Boyd. 2004. Ecology and management of sage-grouse and sage-grouse habitat._ Journal of Range Management 57: 2-19.

Davies, K.W., C.S. Boyde, J.L. Beck, J.D. Bates, T.J. Svejcar, and J.G. Gregg. 2011. Saving the sagebrush sea: an ecosystem conservation plan for big sagebrush. Biological Conservation 144:2573-2584.

Diamond, J.M., C.A. Call, and N. Devoe. 2009. Effects of targeted cattle grazing on fire behavior of cheatgrass-dominated rangeland in the northern Great Basin, USA. International Journal of Wildland Fire 18:944–950.

Doherty, M.K. 2007. Mosquito populations in the Powder River Basin, Wyoming: a comparison of natural, agricultural, and effluent coal-bed natural gas aquatic habitats. Thesis. Montana State University, Bozeman, Montanna, USA.

Doherty, K.E., D.E. Naugle, and B.L. Walker. 2010. Greater sage-grouse nesting habitat: The importance of managing at multiple scales. Journal of Wildlife Management 74:1544-1553.

Doherty, K.E., D.E. Naugle, B.L. Walker, and J.M. Graham. 2008. Greater sage-grouse winter habitat selection and energy development. Journal of Wildlife Management 72:187-195.

Doherty, K.E., D.E. Naugle, H.E. Copeland, A. Pocewicz, and J.M. Kiesecker. 2011a. Energy development and conservation tradeoffs: systematic planning for greater sage-grouse in their eastern range. Pages 505-516 in S.T. Knick and J.W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press, Berkeley, California, USA.

Doherty, K.E., D.E. Naugle, and J.S. Evans. 2010. A currency for offsetting energy development impacts: Horse-trading sage-grouse on the open market. PLoS One 5:e10339. Accessed 19 September 2011 http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone0010339

Doherty, K.E., J.L. Beck, and D.E. Naugle. 2011b. Comparing ecological site descriptions to habitat characteristics influencing greater sage-grouse nest site occurrence and success. Rangeland Ecology & Management 64: 344-351.

Eiswerth, M.E. and J.S.Shonkwiler. 2006. Examining post-wildfire reseeding on arid arid rangeland: a multivariate tobit modeling approach. Ecological Modeling 192:286-298.

Ellingson, A.R., and P.M. Lukacs. 2003. Improving methods for regional landbird monitoring: a reply to Hutto and Young. Wildlife Society Bulletin 31:896-902.

Ellis, K.L. 1984. Behavior of lekking sage grouse in response to a perched golden eagle. Western Birds 15:37–8.

Formann, R T.T., and L.E. Alexander. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics 29:207-231.

Freilich, J.E., J.M. Emlen, J.J. Duda, D.C. Freeman, and P.J. Cafaro. 2003. Ecological effects of ranching: a six-point critique. BioScience 53:759-765.

Gelbard, J.L., and J. Belnap. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. Conservation Biology 17:420-432.

Giesen, K.M. 1995. Upland bird research: evaluation of livestock grazing and residual herbaceous cover on sage grouse nest success. Job final report, project number COW-167-R/Job 18/Wk.Pl. 3. Colorado Division of Wildlife, Colorado, USA.

Graham, L., and B. Jones. 2005. Northern Eagle/Southern Routt Greater Sage-Grouse Summary Report. Colorado Division of Wildlife, Steamboat Springs, Colorado, USA.

Graham, L., and C. McConnell. 2004. Radio-collared greater sage-grouse summary report: Southern Routt and Northern Eagle Counties, Colorado. Colorado Division of Wildlife, Steamboat Springs, Colorado, USA.

Hagen, C. 2011. Greater sage-grouse conservation assessment and strategy for Oregon: a plan to maintain and enhance populations and habitats. Oregon Department of Fish and Wildlife, Portland, Oregon, USA.

Hagen, C.A., J.W. Connelly, and M.A. Schroeder. 2007. A meta-analysis for greater sage-grouse nesting and brood rearing habitats. Wildlife Biology 13 (Supplement 1):42-50.

Harju, S.M., M.R. Dzialak, R.C. Taylor, L.D. Hayden-Wing, and J.B. Winstead. 2010. Thresholds and time lags in effects of energy development on greater sage-grouse populations. Journal of Wildlife Management 74:437–48.

Holloran, M.J. 2005. Greater sage-grouse (Centrocercus urophasianus) population response to natural gas field development in western Wyoming. Dissertation, Department of Zoology and Physiology, University of Wyoming, Laramie, Wyoming.

Holloran, M.J., R.C. Kaiser, and W.A. Hubert. 2010. Yearling greater sage-grouse response to energy development in Wyoming. Journal of Wildlife Management 74:65–72.

Holloran, M.J. and S.H. Anderson. 2005. Spatial distribution of greater sage-grouse nests in relatively contiguous sagebrush habitats. Condor 107:742-752.

Jackson, S.D. 2000. Overview of Transportation Impacts on Wildlife Movement and Populations. Pp. 7-20 In Messmer, T.A. and B. West, (editors) Wildlife and Highways: Seeking Solutions to an Ecological and Socio-economic Dilemma. The Wildlife Society.

Johnson, C.J., and Martin-Hughes St-Laurent. 2011. Unifying framework for understanding impacts of human developments on wildlife. Pages 27-54 in D.E. Naugle, editor. Energy development and wildlife conservation in western North America. Island Press, Washington, D.C., USA.

Johnson, D.H., M.J. Holloran, J.W. Connelly, S.E. Hanser, C.L. Amundson, and S.T. Knick. 2011. Influences of environmental and anthropogenic features on greater sage-grouse populations. Pages 407-450 in S.T. Knick and J.W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press, Berkeley, California, USA.

Kaiser, R.C. 2006. Recruitment by greater sage-grouse in association with natural gas development in western Wyoming. Thesis, University of Wyoming, Laramie.

Knick, S.T. 2011. Historical development, principal federal legislation, and current management of sagebrush habitats: implications for conservation. Pages 13-31 in S.T. Knick and J.W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press, Berkeley, California, USA.

Knick, S.T., and J.W. Connelly. 2011a. Greater sage-grouse and sagebrush: an introduction to the landscape. Pp. 1-9 in S.T. Knick and J.W. Connelly. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitat. Studies in Avian Biology 38. University of California Press, Berkeley, CA.

Knick, S.T., and J.W. Connelly, editors. 2011b. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press, Berkeley, CA.

Knick, S.T., D.S. Dobkin, J.T. Rotenberry, M.A. Schroeder, W.M. Vander Haegen, and C. van Riper III. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. Condor 105:611-634.

Knick, S.T. and S.E. Hanser. 2011. Connecting pattern and process in greater sage-grouse populations and sagebrush landscapes. Pages 383-405 in S.T. Knick and J.W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press, Berkeley, California, USA.

Knick S.T., S.E. Hanser, R.F. Miller, D.A. Pyke, M.J. Wisdom, S.P. Finn, E.T. Rinkes and C.J. Henny. 2011. Ecological Influence and Pathways of Land Use in Sagebrush. Pp. 203-251 in S.T. Knick and J.C. Connelly (editors), Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology (vol. 38), University of California Press, Berkley, CA.

Kolada, E.J., M.L. Casazza, and J.S. Sedinger. 2009. Ecological factors influencing nest survival of greater sage-grouse in Mono County, California. Journal of Wildlife Management 73:1341-1347.

Kramer, A.T. and K. Havens. 2009. Plant conservation genetics in a changing world. Trends in Plant Science 14:599-607.

Lammers, W.M., and M.W. Collopy. 2007. Effectiveness of avian predator perch deterrents on electric transmission lines. Journal of Wildlife Management 71:2752-2758.

Launchbaugh, K., B. Brammer, M.L. Brooks, S. Bunting, P. Clark, J. Davison, M. Fleming, R. Kay, M. Pellant, D. A. Pyke, and B. Wylie. 2007. Interactions among livestock grazing, vegetation type, and fire behavior in the Murphy Wildland Fire Complex in Idaho and Nevada, July 2007. U.S. Geological Survey Open-File Report 2008–1214. (http://pubs.usgs.gov/ofr/2008/1214).

Lyon, A.G. and S.H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin 31: 486-491.

Lyons, J.E., M.C. Runge, H.P. Laskowski, and W.L. Kendall. 2008. Monitoring in the context of structured decision-making and adaptive management.

Meinke, C.W., S.T. Knick, and D.A. Pyke. 2009. A spatial model to prioritize sagebrush landscapes in the intermountain west (U.S.A.) for restoration. Restoration Ecology 17:652-659.

Miller, R.F., S.T. Knick, D.A. Pyke, C.W. Meinke, S.E. Hanser, M.J. Wisdom, and A.L. Hild. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. Pp. 145-184 in S.T. Knick and J.W. Connelly (editors). Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press, Berkeley, CA.

Moynahan, B.J. and M.S. Lindberg. 2006. Nest locations of greater sage-grouse in relationship to leks in north-central Montana. Unpublished report.

Naugle, D.E. and B.L. Walker. 2007. A collaborative vision for integrated monitoring of greater sage-grouse populations. Pp. 57-62 in K.P. Reese and R.T. Bowyer (editors). Monitoring populations of sage-grouse: proceedings of a symposium at Idaho State University. University of Idaho College of Natural Resources Station Bulletin 88. University of Idaho, Moscow, ID.

Naugle, D.E., K.E. Doherty, B.L. Walker, H.E. Copeland, M.J. Holloran, and J.D. Tack. 2011a. Sage-grouse and cumulative impacts of energy development. Pages 55-70 in D.E. Naugle, editor. Energy development and wildlife conservation in western North America. Island Press, Washington, D.C., USA.

Naugle, D.E., K.E. Doherty, B.L. Walker, M.J. Holloran, and H.E. Copeland. 2011b. Energy development and greater sage-grouse. Pages. 489-503 in S.T. Knick and J.W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press, Berkeley, California, USA.

NWCG Memo #024-2010 April 30, 2010. www.nwcg.gov

Patterson, R.L. 1952. The sage grouse in Wyoming. Sage Books, Denver, Colorado. USA.

Petersen, B.E. 1980. Breeding and nesting ecology of female sage grouse in North Park, Colorado. Thesis. Colorado State University, Fort Collins, USA.

Pyke, D.A. 2011. Restoring and rehabilitating sagebrush habitats. Pp. 531-548 in S. T. Knick and J. W. Connelly (editors). Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press. Berkeley, CA.

Reese, K.P., and R.T. Bowyer (editors). 2007. Monitoring populations of sage-grouse. College of Natural Resources Experiment Station Bulletin 88. University of Idaho, Moscow, ID.

Remington, T.E., and C.E. Braun. 1991. How surface coal mining affects sage grouse, North Park, Colorado. Proceedings, Issues and Technology in the Management of Impacted Western Wildlife. Thorne Ecological Institute 5:128-132.

Rhodes, E.C., J.D. Bates, R.N. Sharp, and K.W. Davies. 2010. Fire effects on cover and dietary resources of sage-grouse habitat. Journal of Wildlife Management 74:755-764.

Richards, R.T., J.C. Chambers, and C. Ross. 1998. Use of native plants on federal lands: policy and practice. Journal of Range Management 51:625-632.

Schroeder, M.A., C.L. Aldridge, A.D. Apa, J.R. Bohne, C.E. Braun, S.D. Bunnell, J.W. Connelly, P.A. Deibert, S.C. Gardner, M.A. Hilliard, G.D. Kobriger, S.M. McAdam, C.W. McCarthy, J.J. McCarthy, D.L. Mitchell, E.V. Rickerson, and S.J. Stiver. 2004. Distribution of sage grouse in North America. Condor 106:363–76.

Sedinger, J.S. 2007. Improving understanding and assessment of greater sage-grouse populations. Pp. 43-56 in K.P. Reese and R.T. Bowyer (editors). Monitoring populations of sage-grouse: proceedings of a symposium at Idaho State University. University of Idaho College of Natural Resources Station Bulletin 88. University of Idaho, Moscow, ID.

Steenhof, K., M.N. Kochert, and J.A. Roppe. 1993. Nesting by raptors and common ravens on electrical transmission line towers. Journal of Wildlife Management 57:271-281.

Stankey, G.H., R.N. Clark, and B.T. Bormann. 2005. Adaptive management of natural resources: theory, concepts, and management institutions. Gen. Tech. Rep. PNW-GTR-654. Portland, OR: U.S. Department of Agriculture, Forest Service, Northwest Research Station.

Stevens, B.S. 2011. Impacts of fences on greater sage-grouse in Idaho: Collision, mitigation, and spatial ecology. Thesis, University of Idaho, Moscow, Idaho, USA.

Stiver, S.J., A.D. Apa, J.R. Bohne, S.D. Bunnell, P.A. Deibert, S.C. Gardner, M.A. Hilliard, C.W. McCarthy, and M.A. Schroeder. 2006. Greater sage-grouse comprehensive strategy. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming, USA.

Stiver, S.J., E.T. Rinkes, and D.E. Naugle. 2010. Sage-Grouse Habitat Assessment Framework. , Bureau of Land Management Unpublished Report. U.S. Department of Interior, Bureau of Land Management, Idaho State Office, Boise, ID.

Swenson, J.E., C.A. Simmons, and C.D. Eustace. 1987. Decrease of sage grouse *Centrocercus urophasianus* after ploughing of sagebrush steppe. Biological Conservation 41:125–32.

Tack, J.D. 2009. Sage-grouse and the human footprint: implications for conservation of small and declining populations. Thesis. University of Montana, Missoula, Montana, USA.

Thompson, T.R. 2006. Dispersal ecology of greater sage-grouse in northwestern Colorado: evidence from genetic and demographic data: 2006 Annual Progress Report. Colorado Division of Wildlife, Grand Junction, Colorado, USA.

Thompson, T.R., K.P. Reese, and A.D. Apa. 2005. Dispersal ecology of greater sage-grouse in northwestern Colorado: Annual Report 2005. Colorado Division of Wildlife, Grand Junction, Colorado, USA.

Thurow, T.L., and C.A. Taylor, Jr. 1999. Viewpoint: the role of drought in range management. Journal of Range Management 52:413-419.

Toevs, G.R., J.J. Taylor, C.S. Spurrier, W.C. MacKinnon, and M.R. Bobo. 2011. Assessment, Inventory, and Monitoring Strategy: For integrated renewable resource management. U.S. Department of Interior, Bureau of Land Management, National Operations Center, Denver, CO.

Trombulak, S.C., and C.A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14:18-30.

Turner, M.G. 2005. Landscape ecology in North America: Past, present, and future. Ecology 86:1967-1974.

Turner, M.G., and R.H. Gardner (editors). 1991. Quantitative methods in landscape ecology. Springer-Verlag, New York, NY.

U.S. Department of Interior, Bureau of Land Management. 2004. National Sage-Grouse Habitat Conservation Strategy. WO IM2005-024. Washington D.C. Online:

Walker, B.L., and D.E. Naugle. 2011. West Nile virus ecology in sagebrush habitat and impacts on greater sage-grouse populations. Pages 127-144 in S.T. Knick and J.W. Connelly (editors). Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press, Berkeley, California, USA.

Walker, B.L., D.E. Naugle, and K.E. Doherty. 2007a. Greater sage-grouse population response to energy development and habitat loss. Journal of Wildlife Management 71:2644-2654.

Walker, B.L., D.E. Naugle, K.E. Doherty, and T.E. Cornish. 2004. Outbreak of West Nile virus in greater sage-grouse and guidelines for monitoring, handling, and submitting dead birds. Wildlife Society Bulletin 32:1000–6.

Walker, B.L. D.E. Naugle, K.E. Doherty, and T.E. Cornish. 2007b. West Nile Virus and greater sage-grouse: estimating infection rate in a wild bird population. Avian Diseases 51:691-696.

Wambolt, C.L., A.J. Harp, B.L. Welch, N. Shaw, J.W. Connelly, K.P. Reese, C.E. Braun, D.A. Klebenow, E.D.McArthur, J.G. Thompson, L.A.Torell, and J.A. Tanaka. 2002. Conservation of Greater Sage-Grouse on Public Lands in the Western U.S.: Implications of Recovery and Management Policies. PACWPL Policy Paper SG-02-02, Caldwell, ID: Policy Analysis Center for Western Public Lands. 41p.

Walsh, D.P., G.C. White, T.E. Remington, and D.C. Bowden. 2004. Evaluation of the lek count index for greater sage-grouse. Wildlife Society Bulletin 32:56-68.

Walston, L.J., B.L. Cantwell, and J.R. Krummel. 2009. Quantifying spatiotemporal changes in a sagebrush ecosystem in relation to energy development. Ecography 32:943-952.

Walters, C.J. 1986. Adaptive Management of Renewable Resources. MacMillan, New York, New York. 374pp.

Walters, C.J. 1987. Challenges in adaptive management of riparian and coastal ecosystems. Conservation Ecology 1(2):1 [online] http://www.ecologyandsociety.org/vol1/iss2/art1/

Wheatley, M., and C. Johnson. 2009. Factors limiting our understanding of ecological scale. Ecological Complexity 6:150-159.

Williams, M.I., G.B. Paige, T.L. Thurow, A.L. Hild, and K.G. Gerow. 2011. Songbird relationships to shrub-steppe ecological site characteristics. Rangeland Ecology & Management 64:109-118.

Wisdom, M.J., C.W. Meinke, S.T. Knick, and M.A Schroeder. 2011. Factors associated with extirpation of sage-grouse. Page 451-472 in S.T. Knick and J.W. Connelly (editors). Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California press, Berkeley, California, USA.

Wyoming Game and Fish Department. 2011. Wyoming Game and Fish Department Protocols for Treating Sagebrush to Benefit Sage-Grouse. Unpublished report. 5 pp.

Yoder J.M., D.A. Swanson, and E.A. Marschall. 2004. The cost of dispersal: Predation as a function of movement in ruffed grouse. Behavioral Ecology 15:469–76.

Zou, L., S.N. Miller, and E.T. Schmidtmann. 2006. Mosquito larval habitat mapping using remote sensing and GIS: Implications of coalbed methane development and West Nile virus. Journal of Medical Entomology 43:1034–41.

Code of Federal Regulations Cited

43 CFR 8340

43 CFR 3809.5

43 CFR 2801.5

43 CFR 3461.5

43 CFR 3101.1-2

43 CFR 3162.5

43 CFR 1601.0-5(b)

BLM Washington Office Instruction Memorandums

WO IM 2011–138, Sage-grouse Conservation Related to Wildland Fire and Fuels Management

WO IM 2008-204, Offsite Mitigation
Other Federal, State and Interagency Memorandums/Reports

WAFWA Sage-grouse Strategy MOU, 2008

Appendices

Appendix A. Life History Requirements of Greater Sage-grouse (excerpted from 75 FR 13910)

Greater sage-grouse depend on a variety of shrub-steppe habitats throughout their life cycle, and are considered obligate users of several species of sagebrush (e.g., *Artemisia tridentata* ssp. *wyomingensis* (Wyoming big sagebrush), *A. t.* ssp. *vaseyana* (mountain big sagebrush), and *A. t. tridentata* (basin big sagebrush)) (Patterson 1952, Braun et al. 1976, Connelly et al*.* 2000a, Connelly et al. 2004, Miller et al*.* 2011). Greater sage-grouse also use other sagebrush species such as *A. arbuscula* (low sagebrush), *A. nova* (black sagebrush), *A. frigida* (fringed sagebrush), and *A. cana* silver sagebrush (Schroeder et al*.* 1999, Connelly et al. 2004,). Thus, sage-grouse distribution is strongly correlated with the distribution of sagebrush habitats (Schroeder et al. 2004). Sage-grouse exhibit strong site fidelity (loyalty to a particular area even when the area is no longer of value) to seasonal habitats, which includes breeding, nesting, brood rearing, and wintering areas (Connelly et al. 2004, Connelly et al. 2011b). Adult sage-grouse rarely switch between these habitats once they have been selected, limiting their adaptability to changes.

During the spring breeding season, male sage-grouse gather together to perform courtship displays on areas called leks. The proximity, configuration, and abundance of nesting habitat are key factors influencing lek location (Connelly et al., 1981, and Connelly et al., 2000b, cited in Connelly et al., 2011). Leks can be formed opportunistically at any appropriate site within or adjacent to nesting habitat (Connelly et al*.* 2000a) and, therefore, lek habitat availability is not considered to be a limiting factor for sage-grouse (Schroeder et al. 1999). Nest sites are selected independent of lek locations, but the reverse is not true (Bradbury et al. 1989,Wakkinen et al*.* 1992). Thus, leks are indicative of nesting habitat.

Females have been documented to travel more than 20 km (12.5 mi) to their nest site after mating (Connelly et al. 2000a), but distances between a nest site and the lek on which breeding occurred is variable (Connelly et al. 2004, Connelly et al. 2011b). Average distance between a female's nest and the lek on which she was first observed ranged from 3.4 km (2.1 mi) to 7.8 km (4.8 mi) in five studies examining 301 nest locations (Schroeder et al. 1999).

Productive nesting areas are typically characterized by sagebrush with an understory of native grasses and forbs, with horizontal and vertical structural diversity that provides an insect prey base, herbaceous forage for pre-laying and nesting hens, and cover for the hen while she is incubating (Gregg 1991Schroeder et al*.* 1999, Connelly et al. 2000a, Connelly et al. 2004, Connelly et al. 2011b). Sage-grouse also may use other shrub or bunchgrass species for nest sites (Klebenow 1969, Connelly et al. 2000a, Connelly et al. 2004). Shrub canopy and grass cover provide concealment for sage-grouse nests and young, and are critical for reproductive success (Barnett and Crawford 1994, Gregg et al. 1994, DeLong et al.1995, Connelly et al. 2004).

Hens rear their broods in the vicinity of the nest site for the first 2-3 weeks following hatching (within 0.2-5 km (0.1-3.1 mi)), based on two studies in Wyoming (Connelly et al. 2004). Forbs and insects are essential nutritional components for chicks (Klebenow and Gray 1968, Johnson and Boyce 1991, Connelly et al. 2004). Therefore, early brood-rearing habitat must provide adequate cover (sagebrush canopy cover of 10 to 25 percent; Connelly et al. 2000a) adjacent to areas rich in forbs and insects to ensure chick survival during this period (Connelly et al. 2004, Hagen et al. 2007).

All sage-grouse gradually move from sagebrush uplands to more mesic areas (moist areas such as streambeds or wet meadows) during the late brood-rearing period (3 weeks post-hatch) in response to summer desiccation of herbaceous vegetation (Connelly et al. 2000a). Summer use areas can include sagebrush habitats as well as riparian areas, wet meadows and alfalfa fields (Schroeder et al. 1999). These areas provide an abundance of forbs and insects for both hens and chicks (Schroeder et al. 1999, Connelly et al. 2000a).

As vegetation continues to desiccate through the late summer and fall, sage-grouse shift their diet entirely to sagebrush (Schroeder et al. 1999). Sage-grouse depend entirely on sagebrush throughout the winter for both food and cover (Connelly et al. 2011a). Sagebrush stand selection is influenced by snow depth (Patterson 1952, Hupp and Braun 1989), availability of sagebrush above the snow to provide cover (Connelly et al. 2004, and references therein) and, in some areas, topography (e.g., elevation, slope and aspect, Beck 1977, Crawford et al. 2004).

Many populations of sage-grouse migrate between seasonal ranges in response to habitat distribution (Connelly et al. 2004). Migration can occur between winter and breeding and summer areas, between breeding, summer and winter areas, or not at all. Migration distances of up to 161 km (100 mi) have been recorded (Patterson 1952), however, distances vary depending on the locations of seasonal habitats (Schroeder et al. 1999). Migration distances for female sage-grouse generally are less than for males (Connelly et al. 2004), but in one study in Colorado, females travelled further than males (Beck 1977). Almost no information is available regarding the distribution and characteristics of migration corridors for sage-grouse (Connelly et al. 2004). Sage-grouse dispersal (permanent moves to other areas) is poorly understood (Connelly et al. 2004, Knick and Hanser 2011) and appears to be sporadic (Dunn and Braun 1986). Estimating an "average" home range for sage-grouse is difficult due to the large variation in sage-grouse movements both within and among populations. This variation is related to the spatial availability of habitats required for seasonal use and annual recorded home ranges have varied from 4 to 615 square kilometers (km^2) (1.5 to 237.5 square miles (mi^2)), Connelly et al. *2011b*).

Literature Cited:

Barnett, J.K., and J. A. Crawford. 1994. Pre-laying nutrition of sage grouse hens in Oregon. Journal of Range Management, 47:114-118.

Beck, T.D.I. 1977. Sage grouse flock characteristics and habitat selection in winter. Journal of Wildlife Management 41:18-26.

Bradbury, J.W., R.M. Gibson, C.E. McCarthy, and S.L. Vehrencamp. 1989. Dispersion of displaying male sage grouse. II. The role of female dispersion. Behavioral Ecology and Sociobiology 24:15-24.

Braun, C.E., M.F. Baker, R.L. Eng, J.W. Gashwiler, and M.H. Schroeder. 1976. Conservation committee report on effects of alteration of sagebrush communities on the associated avifauna. Wilson Bulletin 88:165-171.

Connelly, J.W., W.J. Arthur, and O.D. Markham. 1981. Sage grouse leks on recently disturbed sites. Journal of Range Management 52:153-154.

Connelly, J.W., C.A. Hagen, and M.A. Schroeder. 2011a. Characteristics and dynamics of greater sage-grouse populations. Pp. 53-67 in S.T. Knick and J.W. Connelly. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitat. Studies in Avian Biology 38. University of California Press, Berkeley, CA.

Connelly, J.W., E.T. Rinkes, and C.E. Braun. 2011b. Characteristics of greater sage-grouse habitats: a landscape species at micro and macro scales. Pp. 69-83 in S.T. Knick and J.W. Connelly, editors. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitat. Studies in Avian Biology 38. University of California Press, Berkeley, CA.

Connelly, J.W., S.T. Knick, C.E. Braun, W.L. Baker, E.A. Beever, T. Christiansen, K.E. Doherty, E.O. Garton, C.A. Hagen, S.E. Hanser, D.H. Johnson, M. Leu, R.F. Miller, D.E. Naugle, S.J. Oyler-McCance, D.A. Pyke, K.P. Reese, M.A. Schroeder, S.J. Stiver, B.L. Walker, and M.J. Wisdom. 2011. Conservation of greater sage-grouse: a synthesis of current trends and future management. Studies in Avian Biology. 46 pp.

Connelly, J.W., M.A. Schroeder, A.R. Sands and C.E. Braun. 2000a. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Connelly, J.W., A.D. Apa, R.B. Smith, and K.P. Reese. 2000b. Effects of predation and hunting on adult sage grouse *Centrocercus urophasianus* in Idaho. Wildlife Biology 6:227-232.

Connelly, J. W., S. T. Knick, M. A. Schroeder and S. J. Stiver. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming, USA.

Crawford, J.A., R.A. Olson, N.E. West, J.C. Mosley, M.A. Schroeder, T.D. Whitson, R.F. Miller, M.A. Gregg, and C. S. Boyd. 2004. Ecology and management of sage-grouse and sage-grouse habitat. Rangeland Ecology & Management 57:2-19.

DeLong, A.K., J.A. Crawford and D. C. DeLong, Jr. 1995. Relationships between vegetational structure and predation of artificial sage grouse nests. Journal of Wildlife Management 59:88-92.

Dunn, P.O., and C.E. Braun. 1986. Late summer-spring movements of juvenile sage grouse. Wilson Bulletin 98:83-92.

Gregg, M.A. 1991. Use and selection of nesting habitat by sage grouse in Oregon. M.S. Thesis, Oregon State University, Corvallis, Oregon, USA. 46pp.

Gregg, M.A., J.A. Crawford, M.S. Drut and A.K. DeLong. 1994. Vegetational cover and predation of sage grouse nests in Oregon. Journal of Wildlife Management 58:162-166.

Hagen, C.A., J.W. Connelly, and M.A. Schroeder. 2007. A meta-analysis for greater sage-grouse nesting and brood rearing habitats. Wildlife Biology 13 (Supplement 1):42-50.

Hupp, J.W. and C.E. Braun. 1989. Topographic distribution of sage grouse foraging in winter. Journal of Wildlife Management 53:823-829.

Johnson, G.D. and M.S. Boyce. 1991. Survival, growth, and reproduction of captive-reared sage grouse. Wildlife Society Bulletin 19: 88-93.

Klebenow, D.A. 1969. Sage grouse nesting and brood habitat in Idaho. Journal of Wildlife Management 33:649-661.

Klebenow, D.A. and G.M. Gray. 1968. Food habits of juvenile sage grouse. Journal of Range Management 21:80-83.

Knick, S.T. and S.E. Hanser. 2011. Connecting pattern and process in greater sage-grouse populations and sagebrush landscapes. Pages 383-405 *in* S.T. Knick and J.W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press, Berkeley, California, USA.

Miller, R.F., S.T. Knick, D.A. Pyke, C.W. Meinke, S.E. Hanser, M.J. Wisdom, and A.L. Hild. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. S. T. Knick and J. W. Connelly (editors). Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press, Berkeley, CA.

Patterson, R.L. 1952. The sage grouse in Wyoming. Sage Books, Denver, Colorado. USA.

Schroeder, M.A., J.R. Young, and C.E. Braun. 1999. Sage grouse (*Centrocercus urophasianus*). Pages 1-28 *in* A. Poole and F. Gill, editors. The birds of North America, No 425. The Birds of North America, Philadelphia, Pennsylvania, USA.

Schroeder, M.A., C.L. Aldridge, A.D. Apa, J.R. Bohne, C.E. Braun, S. D. Bunnell, J.W. Connelly, P.A. Deibert, S.C. Gardner, M.A. Hilliard, G.D. Kobriger, S.M. McAdam, C.W. McCarthy, J.J. McCarthy, D.L. Mitchell, E.V. Rickerson, and S.J. Stiver. 2004. Distribution of sage grouse in North America. Condor 106:363-376.

Wakkinen, W.L., K.P. Reese, and J.W. Connelly. 1992. Sage grouse nest locations in relation to leks. Journal of Wildlife Management 56:381-383.

Appendix B. Scientific Inference

When making natural resource management decisions, managers desire a high level of certainty that their management actions will have the anticipated outcome (Ratti and Garton 1994, Garton et al. 2005). Unfortunately, natural systems have inherent complexity and stochasticity that make certainty in wildlife management decisions challenging (Williams et al. 2002). In an effort to ameliorate some of this uncertainty, managers use quality, published scientific investigations which are reliant upon thoughtful research design (Ratti and Garton 1994, Garton et al. 2005) to guide population and habitat management decisions. When relevant peer reviewed literature does not exist, managers have to resort to best professional judgment and/or unpublished studies. In addition, when using published and unpublished literature, managers must also be cognizant of the research findings for certainty of the conclusions, the scientific method, and if the findings can be applied from the data and results (Murphy and Noon 1991).

Most wildlife research is located along a continuum of field studies (Ratti and Garton 1994, Garton et al. 2005; Fig. 1) and provides varying degrees of reliable knowledge (Romesburg 1981, Hurlbert, 1984, Eberhardt and Thomas 1991). The more rigorous the research design, results, and conclusions, the more confident managers can be in the anticipated outcome (Ratti and Garton 1994, Garton et al. 2005). Research that bases its results and interpretation on an integrated research process includes field level experiments, field study, and modeling (Fig. 1). If designed appropriately, these research efforts can provide for a more broad-based application of research results as opposed to descriptive natural history studies (Ratti and Garton 1994, Garton et al. 2005) (Fig. 1).

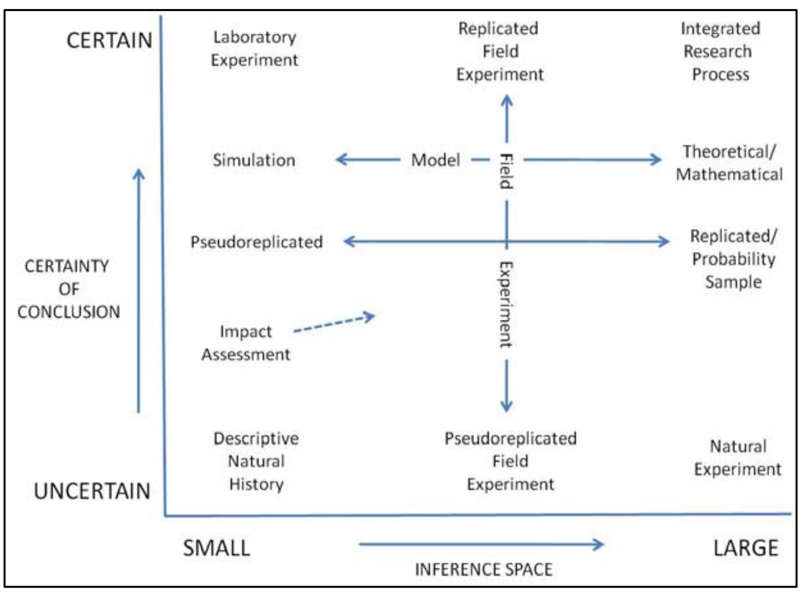

Figure 1. The spectrum of types of wildlife studies that can produce results and conclusions with a large amount of certainty over a very large area of applicability (adapted from Ratti and Garton 1994 and Garton et al. 2005).

Because sage-grouse research has been on-going for over 60 years, managers have access to published literature from several studies (metareplication (Johnson 2002)) that includes different years, study areas, methods, and investigators (Johnson 2002) which leads to more certainty in conclusions (for example see Hagen et al. 2007). In contrast, for some management actions, access to published and unpublished literature may be limited to a single descriptive study. A single descriptive study and/or professional judgment has the lowest level of certainty and lowest inference space. Unfortunately, it may be the only information available on the subject. Ultimately, the result is succinctly summarized by Anderson et al. (2001:312) who stated, "In the long run, science is safeguarded by repeated studies to ascertain what is real and what is merely a spurious result from a single study."

Management in sagebrush ecosystems is further complicated by new forms of development or the unprecedented pace at which traditional uses are increasing. Wind and other renewable energy sources are being proposed and developed in areas that previously had undergone little development. The applicability of results from previous research in other regions on oil and gas development to these new forms of land use is unknown, but is the best information currently available. We also do not know how sagebrush and sage-grouse respond to the increasing intensity of all uses ranging from traditional commodity development to nonconsumptive activities, such as recreation and OHV travel that is occurring across their range. Although previous research can guide management decisions, the changes due to the cumulative effect of this new level of increased development may take years to be fully expressed in habitat and population response.

No single research study, or even a series of studies, regardless of design, and/or inference extent can provide complete certainty in their conclusion(s). As a result, managers must be vigilant in their judgment of research study design, its inference space, and applicability to their management issue when making management decisions. This report cites a large number of published and unpublished studies that can be placed along the continuum of certainty of conclusion and inference space (Fig. 1). Many of the studies cited are from different researchers, study sites, methodologies, and/or years which assists and improves the certainty of the conclusion and inference space (Fig. 1), but ultimately, it is incumbent upon managers to assess their level of risk (consequences of being wrong) with management decisions based upon the cited findings.

The large spatial scales occupied by sage-grouse seasonally (as much as 1,700 mi^2; Leonard et al. 2000) have made research on how they respond to habitat perturbations difficult to conduct. Although strength of inference is strongest for replicated experiments, studies of this nature have not been conducted on large scale perturbations such as oil and gas developments, wind farms, coal mines, powerlines, etc. We therefore relied on retrospective and correlational studies that looked at changes in sage-grouse distribution, abundance or demographic rates over time following these developments. We gave greater credence to conclusions obtained from multiple studies conducted at different locations at different times that showed similar results.

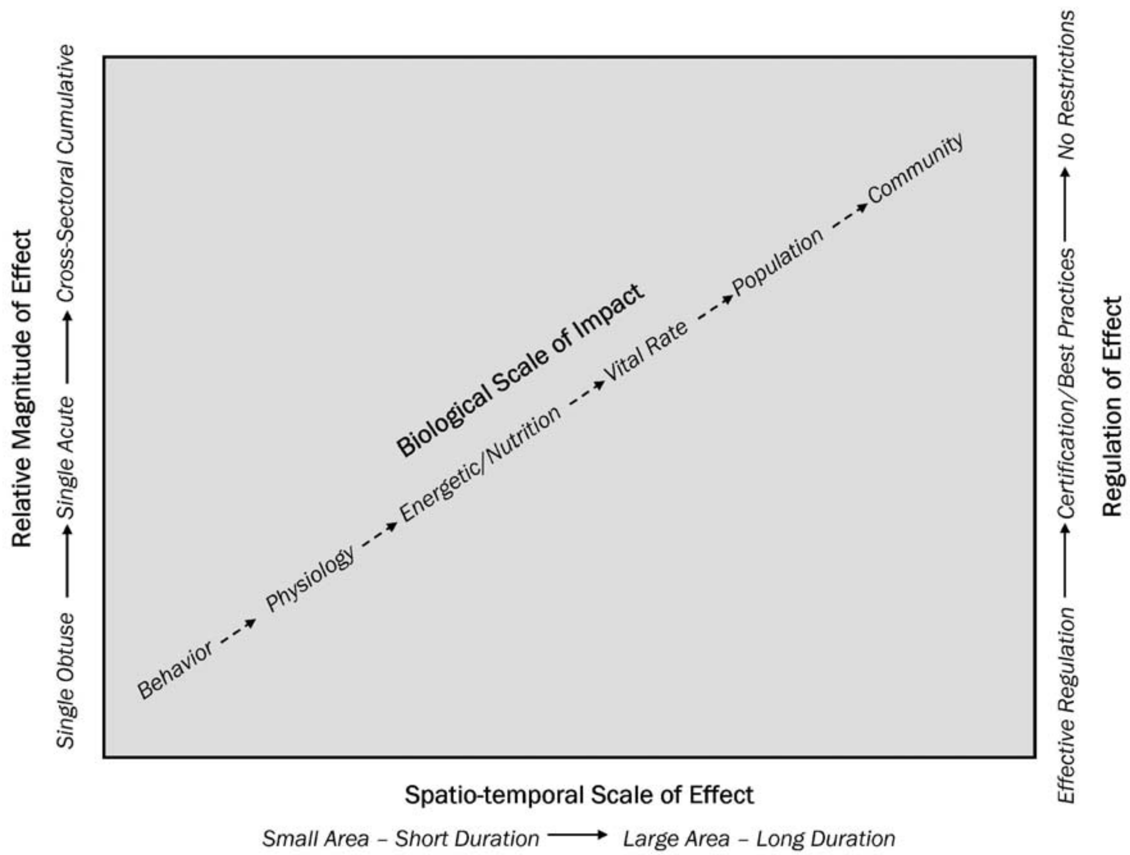

Figure 2. Schematic representation of a typology for classifying and predicting the impacts of human-wildlife interactions (as modified from Johnson and St-Laurent 2011).

Conservation measures described in this report are derived from interpretation of the best available scientific studies using our best professional judgment. Because there is a degree of uncertainty about the effectiveness of these conservation measures, we recommend a rigorous adaptive management process be employed, with population and habitat monitoring as well as feedback loops so that conservation measures or policies that are ineffective can be changed (Lyons et al. 2008).

Literature Cited:

75 FR 13910. 2010. Endangered and Threatened Wildlife and Plants: 12-month finding for petitions to list the greater sage-grouse as threatened or endangered.

Anderson, D.R., K.P. Burnham, W.R. Gould, and S. Cherry. 2001. Concerns about finding effects that are actually spurious. Wildlife Society Bulletin 29:311-316.

Eberhardt, L.L., and J.M. Thomas. 1991. Designing environmental field studies. Ecological Monographs 61:53-73.

Garton, E.O., J.T. Ratti, and J.H. Giudice. 2005. Research and Experimentation Design. Pages 43 – 71 in C. E. Braun, editor. Techniques for wildlife investigations and management. Sixth edition. The Wildlife Society, Bethesda, Maryland, USA.

Hagen, C.A., J.W. Connelly, and M.A. Schroeder. 2007. A meta-analysis for greater sage-grouse nesting and brood rearing habitats. Wildlife Biology 13 (Supplement 1):42-50.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54:187-211.

Johnson, D.H. 2002. The importance of replication in wildlife research. Journal of Wildlife Management 66:919-932.

Knick, S.T., and J.W. Connelly (editors). 2011. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology Series (vol. 38), University of California Press, Berkeley, CA.

Leonard, KM., K.P. Reese, and J.W. Connelly2000. Distribution, movements and habitats of sage grouse *Centrocercus urophasianus* on the Upper Snake River Plain of Idaho: changes from the 1950s to the 1990s. Wildl. Biol. 6:265-270.

Lyons, J.E., M.C. Runge, H.P. Laskowski, and W.L. Kendall. 2008. Monitoring in the context of structured decision-making and adaptive management.

Murphy, D.D. and B.D. Noon. 1991. Coping with uncertainty in wildlife biology. Journal of Wildlife Management 55:773-782.

Patterson, R.L. 1952. The sage grouse in Wyoming. Sage Books, Denver, Colorado. USA.

Ratti, J.T. and E.O. Garton. 1994. Research and Experimental Design. Page 1-23 in T.A. Bockhout (editor). Research and management techniques for wildlife and habitats. Fifth edition. The Wildlife Society, Bethesda, Maryland, USA.

Romesburg, H.C. 1981. Wildlife science: gaining reliable knowledge. Journal of Wildlife Management 45:293-313.

Schroeder, M.A., J.R. Young, and C.E. Braun. 1999. Sage grouse (*Centrocercus urophasianus*). Pages 1-28 *in* A. Poole and F. Gill, editors. The birds of North America, No 425. The Birds of North America, Philadelphia, Pennsylvania, USA.

Williams, B.K., J.D. Nichols, and M.J. Conroy. 2002. Analysis and Management of Animal Populations. Academic Press, New York, New York, USA.

Appendix C. BMPs for how to make a pond that won't produce mosquitoes that transmit West Nile virus (from Doherty (2007)).

The following are seven distinct site modifications that if adhered to, would minimize exploitation of CBNG ponds by *Culex tarsalis*:

1. Increase the size of ponds to accommodate a greater volume of water than is discharged. This will result in un-vegetated and muddy shorelines that breeding *Cx. tarsalis* avoid (De Szalay and Resh 2000). This modification may reduce *Cx. tarsalis* habitat but could create larval habitat for *Culicoides sonorensis*, a vector of blue tongue disease, and should be used sparingly (Schmidtmann et al. 2000). Steep shorelines should be used in combination with this technique whenever possible (Knight et al. 2003).

2. Build steep shorelines to reduce shallow water (>60 cm) and aquatic vegetation around the perimeter of impoundments (Knight et al. 2003). Construction of steep shorelines also will create more permanent ponds that are a deterrent to colonizing mosquito species like *Cx. tarsalis* which prefer newly flooded sites with high primary productivity (Knight et al. 2003).

3. Maintain the water level below that of rooted vegetation for a muddy shoreline that is unfavorable habitat for mosquito larvae. Rooted vegetation includes both aquatic and upland vegetative types. Avoid flooding terrestrial vegetation in flat terrain or low lying areas. Aquatic habitats with a vegetated inflow and outflow separated by open water produce 5-10 fold fewer Culex mosquitoes than completely vegetated wetlands (Walton and Workman 1998). Wetlands with open water also had significantly fewer stage III and IV instars which may be attributed to increased predator abundances in open water habitats (Walton and Workman 1998).

4. Construct dams or impoundments that restrict down slope seepage or overflow by digging ponds in flat areas rather than damming natural draws for effluent water storage, or lining constructed ponds in areas where seepage is anticipated (Knight et al. 2003).

5. Line the channel where discharge water flows into the pond with crushed rock, or use a horizontal pipe to discharge inflow directly into existing open water, thus precluding shallow surface inflow and accumulation of sediment that promotes aquatic vegetation.

6. Line the overflow spillway with crushed rock, and construct the spillway with steep sides to preclude the accumulation of shallow water and vegetation.

7. Fence pond site to restrict access by livestock and other wild ungulates that trample and disturb shorelines, enrich sediments with manure and create hoof print pockets of water that are attractive to breeding mosquitoes.

Literature Cited:

De Szalay, F.A. and V.H. Resh. 2000. Factors influencing macroinvertebrate colonization of seasonal wetlands: responses to emergent plant cover. Freshwater Biology. 45: 295-308.

Doherty, M.K. 2007. Mosquito populations in the Powder River Basin, Wyoming: a comparison of natural, agricultural and effluent coal bed natural gas aquatic habitats. M.S. Thesis. Montana State University, Bozeman, U.S.A.

Knight, R.L., W.E. Walton, G.F. Meara, W.K. Riesen and R. Wass. 2003. Strategies for effective mosquito control in constructed treatment wetlands. Ecological Engineering. 21: 211-232.

Schmidtmann, E.T., R.J. Bobian, R.P. Beldin. 2000. Soil chemistries define aquatic habitats with immature populations of the *Culicoides variipennis* complex (Diptera: *Ceratopogonidae*). Journal of Medical Entomology. 37: 38-64.

Walton, W.E., and P.D. Workman. 1998. Effect of marsh design on the abundance of mosquitoes in experimental constructed wetlands in Southern California. Journal of the American mosquito control Association 14:95-107.

Appendix D. Best Management Practices for Fluid Mineral Development

Priority Habitats - BMPs are continuously improving as new science and technology become available and therefore are subject to change. Include from the following BMPs those that are appropriate to mitigate effects from the approved action.

Roads

- Design roads to an appropriate standard no higher than necessary to accommodate their intended purpose.
- Locate roads to avoid important areas and habitats.
- Coordinate road construction and use among ROW holders.
- Construct road crossing at right angles to ephemeral drainages and stream crossings.
- Establish speed limits on BLM system roads to reduce vehicle/wildlife collisions or design roads to be driven at slower speeds.
- Establish trip restrictions (Lyon and Anderson 2003) or minimization through use of telemetry and remote well control (e.g., Supervisory Control and Data Acquisition).
- Do not issue ROWs to counties on newly constructed energy development roads, unless for a temporary use consistent with all other terms and conditions included in this document.
- Restrict vehicle traffic to only authorized users on newly constructed routes (use signing, gates, etc.)
- Use dust abatement practices on roads and pads.
- Close and rehabilitate duplicate roads.

Operations

- Cluster disturbances, operations (fracture stimulation, liquids gathering, etc.), and facilities.
- Use directional and horizontal drilling to reduce surface disturbance.
- Place infrastructure in already disturbed locations where the habitat has not been restored.
- Consider using oak (or other material) mats for drilling activities to reduce vegetation disturbance and for roads between closely spaced wells to reduce soil compaction and maintain soil structure to increase likelihood of vegetation reestablishment following drilling.
- Apply a phased development approach with concurrent reclamation.
- Place liquid gathering facilities outside of priority areas. Have no tanks at well locations within priority areas (minimizes perching and nesting opportunities for ravens and raptors and truck traffic). Pipelines must be under or immediately adjacent to the road (Bui et al. 2010).

- Restrict the construction of tall facilities and fences to the minimum number and amount needed.
- Site and/or minimize linear ROWs to reduce disturbance to sagebrush habitats.
- Place new utility developments (power lines, pipelines, etc.) and transportation routes in existing utility or transportation corridors.
- Bury distribution power lines.
- Corridor power, flow, and small pipelines under or immediately adjacent to roads.
- Design or site permanent structures which create movement (e.g. a pump jack) to minimize impacts to sage-grouse.
- Cover (e.g., fine mesh netting or use other effective techniques) all drilling and production pits and tanks regardless of size to reduce sage-grouse mortality.
- Equip tanks and other above ground facilities with structures or devices that discourage nesting of raptors and corvids.
- Control the spread and effects of non-native plant species (Evangelista et al. 2011). (E.g. by washing vehicles and equipment.)
- Use only closed-loop systems for drilling operations and no reserve pits.
- Restrict pit and impoundment construction to reduce or eliminate threats from West Nile virus (Doherty 2007).
- Remove or re-inject produced water to reduce habitat for mosquitoes that vector West Nile virus. If surface disposal of produced water continues, use the following steps for reservoir design to limit favorable mosquito habitat:
    - Overbuild size of ponds for muddy and non-vegetated shorelines.
    - Build steep shorelines to decrease vegetation and increase wave actions.
    - Avoid flooding terrestrial vegetation in flat terrain or low lying areas.
    - Construct dams or impoundments that restrict down slope seepage or overflow.
    - Line the channel where discharge water flows into the pond with crushed rock.
    - Construct spillway with steep sides and line it with crushed rock.
    - Treat waters with larvicides to reduce mosquito production where water occurs on the surface.
- Limit noise to less than 10 decibels above ambient measures (20-24 dBA) at sunrise at the perimeter of a lek during active lek season (Patricelli et al. 2010, Blickley et al. *In preparation*).
- Require noise shields when drilling during the lek, nesting, broodrearing, or wintering season.
- Fit transmission towers with anti-perch devices (Lammers and Collopy 2007).

- Require sage-grouse-safe fences.
- Locate new compressor stations outside priority habitats and design them to reduce noise that may be directed towards priority habitat.
- Clean up refuse (Bui et al. 2011).
- Locate man camps outside of priority habitats.

Reclamation

- Include objectives for ensuring habitat restoration to meet sage-grouse habitat needs in reclamation practices/sites (Pyke 2011). . Address post reclamation management in reclamation plan such that goals and objectives are to protect and improve sage-grouse habitat needs.
- Maximize the area of interim reclamation on long-term access roads and well pads including reshaping, topsoiling and revegetating cut and fill slopes.
- Restore disturbed areas at final reclamation to the pre-disturbance landforms and desired plant community.
- Irrigate interim reclamation if necessary for establishing seedlings more quickly.
- Utilize mulching techniques to expedite reclamation and to protect soils.

<u>General sage-grouse habitat</u>

Best Management Practices

Make applicable BMPs mandatory as Conditions of Approval within general sage-grouse habitat. BMPs are continuously improving as new science and technology become available and therefore are subject to change. At a minimum include the following BMPs:

Roads

- Design roads to an appropriate standard no higher than necessary to accommodate their intended purpose.
- Do not issue ROWs to counties on energy development roads, unless for a temporary use consistent with all other terms and conditions included in this document.
- Establish speed limits to reduce vehicle/wildlife collisions or design roads to be driven at slower speeds.
- Coordinate road construction and use among ROW holders.
- Construct road crossing at right angles to ephemeral drainages and stream crossings.
- Use dust abatement practices on roads and pads.

- Close and reclaim duplicate roads, by restoring original landform and establishing desired vegetation.

Operations

- Cluster disturbances, operations (fracture stimulation, liquids gathering, etc.), and facilities.
- Use directional and horizontal drilling to reduce surface disturbance.
- Clean up refuse (Bui et al. 2010).
- Restrict the construction of tall facilities and fences to the minimum number and amount needed.
- Cover (e.g., fine mesh netting or use other effective techniques) all drilling and production pits and tanks regardless of size to reduce sage-grouse mortality.
- Equip tanks and other above ground facilities with structures or devices that discourage nesting of raptors and corvids.
- Use remote monitoring techniques for production facilities and develop a plan to reduce the frequency of vehicle use.
- Control the spread and effects from non-native plant species. (e.g. by washing vehicles and equipment.)
- Restrict pit and impoundment construction to reduce or eliminate augmenting threats from West Nile virus (Dougherty 2007).

Reclamation

- Include restoration objectives to meet sage-grouse habitat needs in reclamation practices/sites (Pyke 2011). Address post reclamation management in reclamation plan such that goals and objectives are to enhance or restore sage-grouse habitat.

Literature Cited:

Blickley, J.L., D. Blackwood, and G.L. Patricelli. In preparation. Experimental evidence for avoidance of chronic anthropogenic noise by greater sage-grouse. University of California-Davis, California, USA.

Bui, T.D., J.M. Marzluff, and B. Bedrosian. 2010. Common raven activity in relation to land use in western Wyoming: implications for greater sage-grouse reproductive success. Condor 112:65-78.

Doherty, M.K. 2007. Mosquito populations in the Powder River Basin, Wyoming: a comparison of natural, agricultural and effluent coal-bed natural gas aquatic habitats. M.S. thesis, Montana State University, Bozeman, Montana, USA.

Evangelista, P.H., A.W. Crall, and E. Bergquist. 2011. Invasive plants and their response to energy development. Pages 115-129 in D.E. Naugle, editor. Energy development and wildlife conservation in western North America. Island Press, Washington, D.C., USA.

Lammers, W.M., and M.W. Collopy. 2007. Effectiveness of avian predator perch deterrents on electric transmission lines. Journal of Wildlife Management 71:2752-2758.

Lyon, A.G. and S.H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin 31: 486-491.

Patricelli, G.L., J.L. Blickley, and S. Hooper. 2010. Incorporating the impacts of noise pollution into greater sage-grouse conservation planning. 27th Meeting of the Western Agencies Sage and Columbian Sharp-tailed Grouse Technical Committee Workshop. Twin Falls, Idaho, USA.

Pyke, D.A. 2011. Restoring and rehabilitating sagebrush habitats. Pp. 531-548 in S.T. Knick and J.W. Connelly (editors). Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. University of California Press. Berkeley, CA.

Appendix E. Best Management Practices for Locatable Mineral Development

BMPs are continuously improving as new science and technology become available and therefore are subject to change. Include from the following BMPs those that are appropriate to mitigate effects from the approved action.

Roads

- Design roads to an appropriate standard no higher than necessary to accommodate their intended purpose.
- Locate roads to avoid important areas and habitats.
- Coordinate road construction and use among ROW holders.
- Construct road crossing at right angles to ephemeral drainages and stream crossings.
- Establish speed limits on BLM system roads to reduce vehicle/wildlife collisions or design roads to be driven at slower speeds.
- Do not issue ROWs to counties on mining development roads, unless for a temporary use consistent with all other terms and conditions included in this document.
- Restrict vehicle traffic to only authorized users on newly constructed routes (e. g., use signing, gates, etc.)
- Use dust abatement practices on roads and pads.
- Close and reclaim duplicate roads, by restoring original landform and establishing desired vegetation.

Operations

- Cluster disturbances associated with operations and facilities as close as possible.
- Place infrastructure in already disturbed locations where the habitat has not been restored.
- Restrict the construction of tall facilities and fences to the minimum number and amount needed.
- Site and/or minimize linear ROWs to reduce disturbance to sagebrush habitats.
- Place new utility developments (power lines, pipelines, etc.) and transportation routes in existing utility or transportation corridors.
- Bury power lines.
- Cover (e.g., fine mesh netting or use other effective techniques) all pits and tanks regardless of size to reduce sage-grouse mortality.
- Equip tanks and other above ground facilities with structures or devices that discourage nesting of raptors and corvids.

- Control the spread and effects of non-native plant species (Gelbard and Belnap 2003, Bergquist et al. 2007).
- Restrict pit and impoundment construction to reduce or eliminate threats from West Nile virus (Doherty 2007).
- Remove or re-inject produced water to reduce habitat for mosquitoes that vector West Nile virus. If surface disposal of produced water continues, use the following steps for reservoir design to limit favorable mosquito habitat:
    - Overbuild size of ponds for muddy and non-vegetated shorelines.
    - Build steep shorelines to decrease vegetation and increase wave actions.
    - Avoid flooding terrestrial vegetation in flat terrain or low lying areas.
    - Construct dams or impoundments that restrict down slope seepage or overflow.
    - Line the channel where discharge water flows into the pond with crushed rock.
    - Construct spillway with steep sides and line it with crushed rock.
    - Treat waters with larvicides to reduce mosquito production where water occurs on the surface.
- Require sage-grouse-safe fences around sumps.
- Clean up refuse (Bui et al. 2010).
- Locate man camps outside of priority sage-grouse habitats.

Reclamation

- Include restoration objectives to meet sage-grouse habitat needs in reclamation practices/sites. Address post reclamation management in reclamation plan such that goals and objectives are to protect and improve sage-grouse habitat needs.
- Maximize the area of interim reclamation on long-term access roads and well pads including reshaping, topsoiling and revegetating cut and fill slopes.
- Restore disturbed areas at final reclamation to pre-disturbance landform and desired plant community.
- Irrigate interim reclamation as necessary during dry periods.

Utilize mulching techniques to expedite reclamation.

Literature Cited:

Bergquist, E., P. Evangelista, T. J. Stohlgren, and N. Alley. 2007. Invasive species and coal bed methane development in the Powder River Basin, Wyoming. Environmental Monitoring and Assessment 128:381-394.

Bui, T.D., J.M. Marzluff, and B. Bedrosian. 2010. Common raven activity in relation to land use in western Wyoming: implications for greater sage-grouse reproductive success. Condor 112:65-78.

Doherty, M.K. 2007. Mosquito populations in the Powder River Basin, Wyoming: a comparison of natural, agricultural and effluent coal bed natural gas aquatic habitats. Thesis. Montana State University, Bozeman, U.S.A.

Gelbard, J.L., and J. Belnap. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. Conservation Biology 17:420-432.

Appendix F. Best Management Practices for Fire & Fuels (wo IM 2011-138)

Fuels Management BMPs:

1. Where applicable, design fuels treatment objective to protect existing sagebrush ecosystems, modify fire behavior, restore native plants, and create landscape patters which most benefit sage-grouse habitat.

2. Provide training to fuels treatment personnel on sage-grouse biology, habitat requirements, and identification of areas utilized locally.

3. Use fire prescriptions that minimize undesirable effects on vegetation or soils (e.g., minimize mortality of desirable perennial plant species and reduce risk of hydrophobicity).

4. Ensure proposed sagebrush treatments are planned with interdisciplinary input from BLM and /or state wildlife agency biologist and that treatment acreage is conservative in the context of surrounding sage-grouse seasonal habitats and landscape.

5. Where appropriate, ensure that treatments are configured in a manner (e.g., strips) that promotes use by sage-grouse (See Connelly et al., 2000*)

6. Where applicable, incorporate roads and natural fuel breaks into fuel break design.

7. Power-wash all vehicles and equipment involved in fuels management activities prior to entering the area to minimize the introduction of undesirable and/or invasive plant species.

8. Design vegetation treatment in areas of high frequency to facilitate firefighting safety, reduce the risk of extreme fire behavior; and to reduce the risk and rate of fire spread to key and restoration habitats.

9. Give priority for implementing specific sage-grouse habitat restoration projects in annual grasslands first to sites which are adjacent to or surrounded by sage-grouse key habitats. Annual grasslands are second priority for restoration when the sites not adjacent to key habitat, but within 2 miles of key habitat. The third priority for annual grasslands habitat restoration projects are sites beyond 2 miles of key habitat. The intent is to focus restoration outward from existing, intact habitat.

10. As funding and logistics permit, restore annual grasslands to a species composition characterized by perennial grasses, forbs, and shrubs.

11. Emphasize the use of native plant species, recognizing that non-native species may be necessary depending on the availability of native seed and prevailing site conditions.

12. Remove standing and encroaching trees within at least 100 meters of occupied sage-grouse leks and other habitats (e.g., nesting, wintering, and brood rearing) to reduce the availability of perch sites for avian predators, as appropriate, and resources permit.

13. Protect wildland areas from wildfire originating on private lands, infrastructure corridors, and recreational areas.

14. Reduce the risk of vehicle or human-caused wildfires and the spread of invasive species by planting perennial vegetation (e.g., green-strips) paralleling road rights-of-way.

15. Strategically place and maintain pre-treated strips/areas (e.g., mowing, herbicide application, and strictly managed grazed strips) to ail in controlling wildfire should wildfire occur near key habitats or important restoration areas (such as where investments in restoration have already been made).

Fire Management BMPs:

1. Develop state-specific sage-grouse toolboxes containing maps, a list of resource advisors, contact information, local guidance, and other relevant information.

2. Provide localized maps to dispatch offices and extended attack incident commanders for use in prioritizing wildfire suppression resources and designing suppression tactics.

3. Assign a sage-grouse resource advisor to all extended attack fires in or near key sage-grouse habitat areas. Prior to the fire season, provide training to sage-grouse resource advisors on wildfire suppression organization, objectives, tactics, and procedures to develop a cadre of qualified individuals.

4. On critical fire weather days, pre-position additional fire suppression resources to optimize a quick and efficient response in sage-grouse habitat areas.

5. During periods of multiple fires, ensure line officers are involved in setting priorities.

6. To the extent possible, locate wildfire suppression facilities (i.e., base camps, spike camps, drop points, staging areas, heli-bases) in areas where physical disturbance to sage-grouse habitat can be minimized. These include disturbed areas, grasslands, near roads/trails or in other areas where there is existing disturbance or minimal sagebrush cover.

7. Power-wash all firefighting vehicles, to the extent possible, including engines, water tenders, personnel vehicles, and ATVs prior to deploying in or near sage-grouse habitat areas to minimize noxious weed spread.

8. Minimize unnecessary cross-country vehicle travel during fire operations in sage-grouse habitat.

9. Minimize burnout operations in key sage-grouse habitat areas by constructing direct fireline whenever safe and practical to do so.

10. Utilize retardant and mechanized equipment to minimize burned acreage during initial attack.

11. As safety allows, conduct mop-up where the black adjoins unburned islands, dog legs, or other habitat features to minimize sagebrush loss.

Literature Cited:

Connelly, J.W., M.A Schroeder, A.R. Sands, and C.E. Braun 2000. Guidelines to Manage Sage-grouse Populations and Their Habitats. Wildlife Society Bulletin 28:967-985.

Appendix G. National Technical Team Members

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Recommended management strategies to limit anthropogenic noise impacts on greater sage-grouse in Wyoming

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Abstract: Recent research has demonstrated that noise from natural gas development negatively impacts sage-grouse (*Centrocercus urophasianus*) abundance, stress levels, and behaviors. Other types of anthropogenic noise sources are similar to gas-development noise and, thus, the response by sage-grouse is likely to be similar. The results of research suggest that effective management of the natural soundscape is critical to the conservation and protection of sage-grouse. The goals of this review are to discuss current approaches in the management of new and existing noise sources in Wyoming and recommend research priorities for establishing effective noise management strategies. We make 4 interim recommendations: (1) that noise-management objectives should be set relative to typical ambient noise levels in sage-grouse habitat before development; the best currently available measuremenet of residual noise levels levels (L_{90}) in undisturbed areas suggest an ambient level of 16 to 20 dBA; (2) that an increase in median noise levels (L_{50}) of 10 dBA above ambient be allowed; (3) that management strategies be expanded to protect the soundscape in areas critical for mating, foraging, nesting, and brood-rearing activities of sage-grouse, rather than protecting the lek area alone; and (4) management strategies be focused on the siting of roads or limiting of traffic volumes during crucial times of the day (0600 to 0900 hours) and season (i.e., breeding season), rather than setting targets for vehicle noise exposure. Roads should be sited or traffic should be seasonally limited within 1.3 to 1.7 km from the edge of critical areas for nesting, foraging and breeding. We emphasize that protections based on these interim recommendations may need to be revised upon completion of ongoing and future research.

Key words: anthropogenic noise, *Centrocercus urophasianus,* chronic noise, energy development, human–wildlife conflicts, natural gas development, natural soundscape, noise management strategies, sage-grouse, roads

Greater sage-grouse (*Centrocercus urophasianus*) populations have declined throughout their range, leading to their designation as a candidate for listing under the Endangered Species Act. Among the factors identified as a threat to sage-grouse is the expansion of energy development across much of the remaining sage-grouse habitat (e.g., Aldridge and Boyce 2007, Doherty et al. 2010, Holloran et al. 2010, Naugle et al. 2011). One potential means by which energy development and other human activities might impact sage-grouse populations is through the production of noise (e.g., Rogers 1964, Braun 1998, Holloran 2005, Connelly et al. 2011).

Acoustic communication is very important in the reproductive behaviors of sage-grouse, and energy exploration and development activities generate substantial noise (Blickley and Patricelli 2012). Therefore, it is important to determine whether noise produced by energy development affects sage-grouse breeding biology. Female sage-grouse use male vocalizations to find males on the lek (Gibson 1989), and, during courtship, females assess male vocalizations and other aspects of male display when choosing a mate (Wiley 1973, Gibson and Bradbury 1985, Gibson 1996, Patricelli and Krakauer 2010). Noise from natural gas development primarily is produced by drilling rigs, compressors, generators, and traffic on access roads. All of these noise sources are loudest in frequencies (i.e., pitch) <2.0 kHz (Blickley and Patricelli 2012). Male sage-grouse produce acoustic signals in a similar frequency range, between 0.2 and 2.0 kHz, so the potential exists for industrial noise to mask sage-grouse communication and, thus, interfere with the ability of females to find and choose mates (Blickley and Patricelli 2012). For a prey species, such as sage-grouse, noise also may increase predation risk by masking the sounds of approaching predators and increase stress levels by increasing the perception of predation risk (Quinn et al. 2006, Rabin et al. 2006). In other vertebrate species, noise has been found to impact individuals directly, for example, by causing startling behaviors, increased heart rate, or increased annoyance. All of these factors may interfere with normal foraging, resting, and breeding behaviors and contribute to higher stress levels and reduced fitness (reviewed in Barber et al. 2009, Kight and Swaddle 2011).

Figure 1. Male sage-grouse displaying on a lek in the Pinedale Anticline Project Area with natural gas drilling rigs in the background (*Photo © courtesy Gerrit Vyn*)

Holloran (2005) found observational evidence that noise may be at least partly responsible for impacts from natural gas development on sage-grouse populations in the Pinedale Anticline Project Area (PAPA), Wyoming, one of the largest natural gas fields in the United States (Figure 1). Juvenile males avoided leks located near natural-gas drilling sites, even if the leks previously had high attendance by males (Holloran et al. 2010). These effects were more pronounced downwind of the drilling sites where noise levels were higher, suggesting that noise contributed substantially to these declines (Holloran 2005).

To investigate potential impacts from noise on greater sage-grouse lekking activity, we experimentally introduced noise from natural gas drilling rigs and traffic on access roads at 8 leks and compared lek attendance to 8 paired control leks near Hudson, Wyoming, between 2006 and 2008 (Blickley et al. 2012*a*). Speakers were placed in a line along an edge of the lek, creating a noise gradient across the lek. The mean noise level (measured as an equivalent noise level, L_{eq}) at 10 m from the speakers was 56.1 dBA on drilling-noise leks and 43.2 dBA on traffic noise leks, while the maximum noise level, L_{max}, was 59.1 dBA and 59.4 dBA for drilling and traffic leks, respectively (see Appendix for glossary of noise terms). We found immediate and sustained declines in male attendance on noise leks (29% decline on drilling noise leks and 73% decline on traffic noise leks relative to paired control leks) and evidence of similar declines in female attendance. These results suggest a strong noise avoidance in male and, possibly, female sage-grouse (Blickley et al. 2012*a*). In addition, we found evidence of elevated levels of corticosterone metabolites in fecal samples collected from noise leks compared to samples collected from control leks. Because elevated corticosterone levels are associated with increased physiological stress (Wasser et al. 2000, Wingfield 2005, Bonier et al. 2009), these results suggest that even males that do not abandon noisy leks are physiologically impacted (Blickley et al. 2012*b*). Further, our analyses of behaviors on playback leks suggest that males alter the timing of their vocalizations in response to noise, increasing display rates during close courtship on leks with drilling noise, and waiting for gaps of quiet on leks with vehicle noise (Blickley 2012). These results are consistent with males avoiding the impacts of masking noise on courtship communication; other types of disturbance, such as startling or learned aversion to vehicular noise, also may contribute to this response. Other types of anthropogenic noise sources (e.g., infrastructure from oil, geothermal, and mining, as well as wind development, off-road vehicles, highway traffic, and urbanization) are similar in acoustic frequency, amplitude, and timing to the noise played in this experiment, and response by sage-grouse to these other noise sources may be similar. These results suggest that effective management of the natural soundscape is critical to the conservation and protection of sage-grouse.

Table 1. Spring 2009 noise levels on leks in the Pinedale Anticline Project area, Wyoming. Data were collected by KC Harvey Environmental L.L.C. for the Pinedale Anticline Project office (KC Harvey Environmental L.L.C. 2009); raw data were re-analyzed and summarized here. All measures are presented in dBA. All leks are close enough to development sites, access roads or highways to experience anthropogenic noise; noise levels may also include sounds from male sage-grouse displaying on the leks (displaying males on these relatively small leks are unlikely to significantly impact L_{50} or L_{90} measures, but may affect other metrics). Measurements are from the full 24 hours/day, so they are not focused on the night and morning periods likely critical to greater sage-grouse (0600 to 0900 hours). Further, weather data are not available and windy periods were not excluded, so these values likely include substantial energy from wind. Finally, these data were collected with a Type-2 SLM and, therefore, are likely higher than true ambient levels (see Appendix).

Lek name	Dates	Duration (hrs)	L_{90}	L_{50}	L_{10}	L_{avg} (L_{eq})	L_{max}	L_{min}
Alkali Draw	April 2, 6	121.0	23.6	28.8	41.2	44.1	92.6	19.6
Big Fred	April 12, 16, May 12	123.0	27.6	33.9	44.0	42.4	80.2	22.0
Bloom Reservoir	April 22, 27	120.0	22.2	29.2	44.7	41.9	83.9	19.4
Cat	May 2, 7	120.3	22.8	28.1	44.1	44.3	86.9	19.6
Little Fred	April 12, 16, May 7	85.5	32.7	36.7	45.5	44.2	80.8	31.8
Lovatt West	April 22, 23, May 12	127.0	30.4	33.7	48.3	47.4	84.5	28.2
Lower Sand Springs Draw	May 7	111.3	25.9	29.8	41.5	39.7	73.4	23.6
Mesa Road 3	May 12	141.3	31.9	32.1	33.1	32.5	53.4	31.7
Oil Fork Road	April 17, 22, 27	120.4	24.5	33.0	46.7	42.8	78.0	22.8
The Rocks	April 6	147.5	32.1	33.1	46.8	44.4	95.3	31.7
Shelter Cabin Reservoir	April 6, 12, May 27	99.1	27.1	32.4	41.9	40.5	78.0	23.3
South Rocks	May 2	121.0	27.4	33.3	46.2	42.7	73.7	23.8
MEAN		119.8	27.4	32.0	43.7	42.2	80.1	24.8
MEDIAN		120.7	27.2	32.7	44.4	42.8	80.5	23.4
SD		16.4	3.7	2.5	4.0	3.7	10.8	4.8
SE		3.3	0.7	0.5	0.8	0.7	2.2	1.0
Maximum		147.5	32.7	36.7	48.3	47.4	95.3	31.8
Minimum		85.5	22.2	28.1	33.1	32.5	53.4	19.4

In 2008, Governor Dave Freudenthal issued an executive order, titled "Greater Sage-Grouse Core Population Area Strategy" (State of Wyoming 2008), stating that "new development or land uses within Core Population Areas should be authorized or conducted only when it can be demonstrated by the state agency that the activity will not cause declines in Greater Sage-Grouse populations." The core area strategy was reaffirmed and refined by Governor Matt Mead (State of Wyoming 2010, 2011). To better achieve the goals of the core area strategy, here we discuss management approaches for limiting noise impacts on greater sage-grouse. Specifically, our goals are 3-fold: (1) to discuss current approaches in the management of new and existing noise sources in Wyoming; (2) to recommend research priorities for establishing effective noise management strategies; and (3) to provide managers and policy makers with recommendations for the interim protection of sage-grouse from known or expected impacts of increased noise levels using the best available science.

Current noise management strategies in Wyoming

Noise management strategies in greater sage-grouse habitat inside and outside of the core area typically share 3 common components: (1) the management objective for noise is established relative to ambient levels; (2) noise is limited to

10 dB above ambient levels; and (3) compliance with this objective is measured at the edge of the lek. In light of the research reviewed above, we discuss potential problems with these 3 components of noise management strategies, both in terms of whether they are practical to implement and their likely efficacy in reducing disturbance to sage-grouse populations. In addition, we discuss special issues related to management of noise from traffic.

Ambient noise levels

Management strategies on Wyoming public lands outside sage-grouse core areas (and before the core area strategy was implemented) typically allow for noise exposure on leks to 10 dB above the ambient level, which typically is defined as 39 dBA, which sets the limit of exposure at 49 dBA (e.g., Bureau of Land Management [BLM] 1999, 2003, 2008). However, there is evidence that 39 dBA is not an appropriate estimate of ambient levels in sagebrush habitat. This value originated in a 1971 U.S. Environmental Protection Agency (EPA) report from a single, afternoon measurement from a farm in Camarillo, California. The farm is described in the report as follows:

> Rural agricultural near tomato field; 50 yards to the trees around the yard and dwelling area; 160 yds to Walnut Ave., a lightly travelled surface road; 0.6 mi to State Hwy 118, a 2-lane moderately travelled highway; 0.6 mi to LeLeror Ave. and 0.75 mi to La Vista Ave, both lightly travelled surface roads; 3.5 mi to Santa Paula Freeway; 3.6 mi to the Ventura Freeway; 4.5 mi to Camarillo. The major intruding events were created by jet propeller aircraft flyovers and dogs barking. Other intruding events were background traffic noise…. During the day an orchard pruner in the distance controlled the minimum noise level.

It is clear from this description that the farm was very different from undisturbed sage-grouse habitat. The EPA report presented this value (i.e., 39 dBA) as an example of an afternoon noise level in an active rural area; the value was not recommended as a default level for undisturbed landscapes. Further, this value is an L_{50}, a median noise level (see Appendix), which, in a busy area, such as this, will include noise from anthropogenic sources, as well as from birds, insects, wind gusts, etc. A more appropriate metric for measuring ambient noise levels is L_{90}, the level that is exceeded 90% of the time (see Appendix). The L_{90} is accepted by the American National Standards Institute (ANSI) as a measure of background or "residual noise level" (2003). Indeed, the same EPA report (1971) found residual noise levels of 30 to 34 dBA on rural farms and 16 to 22 dBA in wilderness areas, whereas 39 dBA residual values were more typical of residential areas in Los Angeles, Detroit, and Boston. Further, this 39 dBA measurement was collected during an afternoon, when noise levels are typically higher; this same Camarillo farm had L_{50} measurements of 32 to 34 dBA at night and in the early morning (the L_{90} levels at this time were <30 dBA). Because calm nights and mornings (0600 to 0900 hours) are the window of time when sound is most critical for communication in sage-grouse, as well as for the auditory detection of approaching predators, this is the most important period for noise measurement. Afternoons in much of the habitat of the sage-grouse are windy, making noise measurements difficult and impeding communication and predator detection by sage-grouse and other wildlife. Daytime noise levels are not irrelevant, but because anthropogenic noise will often be masked by wind, such noise is less likely to have an impact on breeding. Further, because measurements in the afternoon are more difficult and results are more variable, it is less practical to use afternoon measures for ambient or exceedance values. Ideally, anthropogenic contributions to noise levels throughout the day would be kept as close to nighttime and morning target levels as possible.

Noise levels measured in disturbed and undisturbed areas in Wyoming further suggest that 39 dB is inappropriate as an ambient value for most sage-grouse habitat. In a report for the Pinedale Anticline Project Office (PAPO, an interagency office that oversees energy development activities on the PAPA), KC Harvey Environmental L.L.C. (2009) measured noise exposure near leks on the PAPA. Data were collected by multi-day deployment of 4

Type-2 sound-level meters (Quest-SoundPRO-DL-2-1/3-10). We analyzed the raw data collected by KC Harvey with permission of the PAPO, and found that that most leks, even those with multiple, active drilling rigs nearby, had residual (L_{90}) and median (L_{50}) levels much lower than 39 dBA (Table 2). These measurements from disturbed areas are almost all <39 dBA, demonstrating that this value is inappropriately high as an estimate for ambient noise in undisturbed areas.

Based on our review of reports and empirical measurements collected in Wyoming, we estimate that pre-development ambient values from nights and calm mornings in sagebrush habitat are closer to 16 to 20 dBA (see recommendations section for details). Assuming that 16 dBA is a more representative ambient value, a noise source at currently allowable levels (i.e., 49 dBA) would exceed ambient by 33 dB. This represents a 44-fold increase in the noise level, which would be perceived by humans as at least 10 times louder than ambient (see Appendix). Such a level of sound would dominate the soundscape and cause significant disruption. Results from our experiments further indicate that 49 dBA is too loud as an allowable exposure level within sage-grouse leks. Our noise-playback leks (described above, Blickley et al. 2012*a*) experienced levels that were mostly in compliance with the 49 dB noise limit (<49 dBA across most of the lek area, except for the area within ~20 m of the speakers). Yet, we found large declines in attendance by sage-grouse, increases in stress levels and altered display behaviors across the lek (Blickley 2012, Blickley et al. 2012*a*, *b*). Therefore, the available scientific evidence shows that 39 dBA is inappropriate for use as a default ambient value for sage-grouse habitat and suggests that allowing 49 dBA of noise exposure on leks and other sensitive areas will cause significant disturbance to sage-grouse populations.

In 2010, stipulations for sage-grouse core areas in Wyoming were created by executive order (State of Wyoming 2010). These stipulations used measured ambient values, rather than a 39 dBA default ambient value. A more recent executive order (State of Wyoming 2011) affirms this approach, stating:

"New noise levels, at the perimeter of a lek, should not exceed 10 dBA above ambient noise (existing activity included) from 6:00 p.m. to 8:00 am during the initiation of breeding (March 1 to May 15). Ambient noise levels should be determined by measurements taken at the perimeter of a lek at sunrise."

Because measured ambient noise levels are likely to be <39 dBA in most places, the core area stipulations will typically limit noise to levels <49 dBA and, thus, offer greater protection for sage-grouse. But because existing activity is explicitly included in measurements of ambient noise, there may be some areas where existing sources lead to ambient measures >39 dBA, thereby allowing for >49 dBA of noise exposure. Further, each new development may add 10 dB to existing noise levels, potentially causing an incremental increase in noise over time. Such increasing noise would likely cause increasing impacts, because sage-grouse do not appear to habituate to anthropogenic noise over time. The declines in male attendance that we observed on our noise-playback leks were immediate and sustained throughout the 3-year experiment (Blickley et al. 2012*a*), and elevated stress hormones were observed in both the second and third years of noise playback (Blickley et al. 2012*b*), indicating that sage-grouse do not adapt to increased noise levels over time. Therefore, the combined impact of all anthropogenic noise sources should be considered when assessing disturbance to sage-grouse habitat. To do so, management objectives should be set relative to the undisturbed soundscape, capping the total noise exposure at or near 10 dB above a "pre-development" ambient value. Such a cap would not preclude further development at sites that already have sources exceeding ambient by nearly 10 dB due to the complex way that multiple sound sources combine to determine overall noise levels. For example, a new source with an L_{50} 9 dB quieter than the L_{50} of an existing source at the measurement site would add only 0.5 dB to the total noise exposure.

Collecting measurements of ambient noise levels in quiet areas is extremely challenging and requires expensive, specialized equipment, which makes the requirement to collect ambient values at each lek difficult to implement. Unfortunately, ambient measures will be inflated by non-ideal weather—especially wind, even at low levels. Measures will also be inflated by almost all errors made by the person deploying the noise meter, such as poor placement of the meter for long-term deployment, rustling from clothing, crunching leaves underfoot, and even breathing close to the meter when it is handheld. Even professional measurements on a Type-1 sound level meter (SLM; see Appendix) will typically overestimate ambient levels in quiet areas (<27 dBA). This is because A-weighting approximates human hearing by boosting the amplitudes of the mid-frequencies, which in very quiet areas will include noise from the pre-amplifier on the sound-level meter. All of these sources of measurement inaccuracy will inflate ambient values and, therefore, allow more noise exposure at leks.

In summary, further research is needed to establish pre-development ambient noise values; in the interim, neither an unrealistic default value (39 dBA) nor ambient values measured at the edge of the lek will offer sufficient protection to sage-grouse.

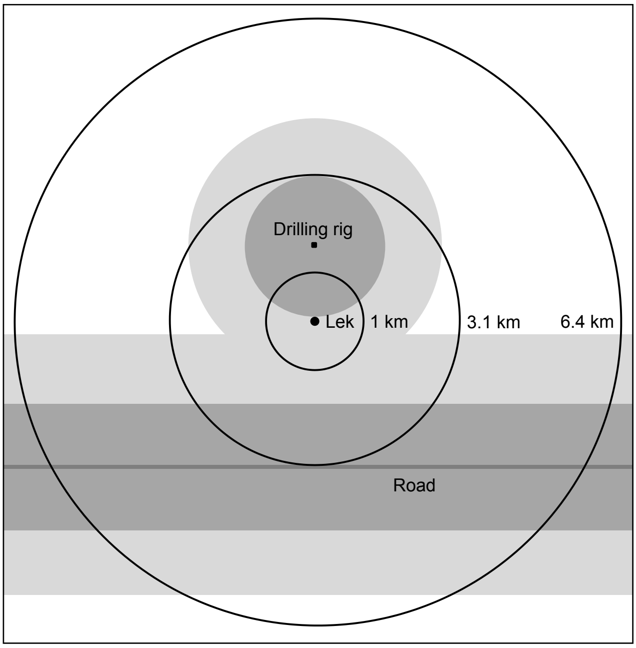

Figure 2. Traffic and drilling noise surrounding a lek. This illustration shows a lek in the center, surrounded by a 1.0-km buffer, a 3.1-km buffer and a 6.4-km buffer. Noise from an example natural gas drilling rig exceeds 10 dBA above ambient (here ambient is assumed to be 20 dBA) for a radius of approximately 1.5 km (dark gray), and is audible above ambient for at least 2.7 km (light gray). This is an example drilling rig measured in the Pinedale Anticline Project Area (PAPA)and is not meant to be representative of all drilling rigs. An average road at the lower edge of the 3.1-km buffer will have noise levels (L_{max}) exceeding ambient by 10 dBA for a distance of 1.3 km and will be audible above ambient for at least 2.7 km with each passing vehicle. Distances are approximately to scale and calculations assume no temperature inversions, which nearly double sound propagation distances, and no topographical effects on sound propagation (excess attenuation of sound is calculated as described in BLM 1999). The lek area is in compliance with the upper limit of recommended noise levels, but much of the surrounding area critical for foraging, nesting and brood-rearing is exposed to higher levels of noise.

The 10-dB threshold

Once an ambient noise value is established, most current noise management strategies limit new noise levels to 10 dB above this ambient value. The 10-dB threshold is used commonly inside and outside of Wyoming core areas and in other states; however, we do not yet know whether this threshold is sufficient to protect greater sage-grouse. This threshold is based on a small number of songbird studies (Nicholoff 2003, Dooling and Popper 2007), and there is no scientific basis for assuming that sage-grouse will respond to noise in a manner similar to songbirds. Indeed, the low-frequency vocalizations of sage-grouse might make them more vulnerable to masking by anthropogenic noise than many songbirds (Blickley and Patricelli 2012). Recent studies of songbirds have found that species with larger body size and lower-frequency vocalizations (i.e., more similar to sage-grouse) are more prone to population declines in response to noise (Francis et al. 2009, Hu and Cardoso 2009).

Further, 10 dB is a significant increase in the amount of noise. For an animal vocalizing to communicate with potential mates or offspring, a 10-dB increase in noise levels corresponds to a 10-fold decrease in the active space of the vocalization (i.e., listening area; Brenowitz 1982, Barber et al. 2009; see Appendix). This same increase in noise will lead to up to a 3-fold decrease in the detection distance between 2 receivers (Barber et al. 2009). This means that, in a noisy environment, the receiver must be 3 times closer to hear a vocalization than in quiet conditions, and perhaps more critically, a predator would be able to approach 3 times closer in noisy conditions before it was detected by a sage-grouse. Indeed, the night-time capture success of sage-grouse by spotlighting is greatly improved with a noise source to mask the sound of footsteps from approaching biologists (Connelly et al. 2003); predators likely gain a similar advantage in noise.

Masking of vocalizations and the sounds of predator approach is only a single source of impacts from noise. Animals may also experience behavioral disruptions, elevated heart rate, interrupted rest, and increased stress levels, all of which may affect health and reproduction or cause avoidance of noisy areas (reviewed in Barber et al. 2009, Kight and Swaddle 2011). Many of these behavioral and physiological impacts could occur at or below the 10-dB threshold. Further studies are needed on sage-grouse to determine whether the 10-dB threshold is insufficient, sufficient, or even too conservative.

Importance of measurement location

Current management strategies that limit noise to 10 dB above ambient levels inside and outside of greater sage-grouse core areas, typically specify that measurements should be collected at the edge of the lek to assess compliance (e.g., State of Wyoming 2011; BLM 1999, 2003, 2008). This strategy introduces 2 potential problems. First, one could find ambient noise measures of 50 to 60 dBA L_{eq} on the edge of a lek due to the vocalizing sage-grouse (Blickley and Patricelli 2012), allowing anthropogenic noise under the 10-dB-over-ambient rule to reach 60 to 70 dBA. After an ambient value is established, determining whether a development project is compliant would require again measuring noise exposure at the lek edge. This could lead to a scenario where increasing development noise could cause declines in lek attendance, which could reduce noise readings over time, as fewer birds contribute to the sound of the lek. Such data would be misleading and provide inaccurate noise measurements of anthropogenic sources. There are methods available to reduce this problem, such as using appropriate noise metrics (such as L_{50} and L_{90}; see Appendix) and collecting measurements before birds arrive on the lek or after birds are flushed. But this issue makes the current stipulations more difficult, disruptive, and ambiguous to implement.

A second potential problem with measuring compliance at edge of the lek is that much of the area surrounding a lek may be exposed to higher noise levels, even if the lek area per se is in compliance (Figure 2). This management strategy, therefore, protects only a fraction of sage-grouse activities during the breeding season (e.g., mate assessment and copulation on the lek) leaving unprotected other critical activities that occur in areas around the lek, such as foraging, roosting, nesting, and brood rearing. Our experimental design allowed us to examine noise impacts only on the lek (Blickley et al. 2012*a*), and, therefore, we cannot provide direct evidence that off-lek noise will impact sage-grouse populations. However, there is indirect evidence that male display and copulation activities on the lek may be affected by noise occurring around the lek area. To sustain their costly display behaviors, males must forage off lek, potentially exposing themselves to higher noise disturbance levels (Figure 2). Vehrencamp et al. (1989) found that males on the lek that are in good condition and are successful in mating forage farther from the lek during the day, compared to unsuccessful, poor-condition males (range 0.2 to 0.8 km off lek). Other studies have found males travelling an average of 1.0 km and a maximum of 2.4 km to forage off lek (e.g., Wallestad and Schladweiler 1974, Schoenberg 1982). If foraging in noisy areas increases male stress levels or predation risk, or decreases foraging efficiency, as has been found in other vertebrate species (Quinn et al. 2006, Rabin et al. 2006), then these noise impacts may affect subsequent male display behaviors on the lek. More importantly, there is evidence that females and juvenile males use male vocalizations to find males on the lek (Gibson 1989). Blickley and Patricelli (2012) found that industrial noise masks these sounds, which may make it more difficult for females and juvenile males in noisy areas surrounding a lek to find the lek itself. Reduced female visitation would decrease copulation activities on the lek, and reduced juvenile male recruitment would lead to male attendance declines over time. For these reasons, the protection of lekking activities may require protection of more than just the lek surface alone.

Additionally, many critical breeding activities occur off-lek, potentially in areas with higher noise levels. Because ~45% of females nest within a 3.1-km radius of the lek and 74 to 80% of females nest within a 6.4-km radius of the lek (Moynahan 2004, Holloran and Anderson 2005), many of these nesting females will experience noise levels exceeding management objectives for the lek (Figure 2). Most vocalizations used between hens and chicks are much quieter than sounds produced by males on leks (Schroeder et al. 1999), and, therefore, are much more prone to masking (Blickley and Patricelli 2012). Additionally, predation rates can be high for chicks and females on nests in disturbed habitats (Hagen 2011), and females likely rely mainly on acoustic rather than visual cues to detect the approach of predators at night. Thus, when noise masks the sounds of predator approach, females and chicks may be more at risk in noisy areas than males on the lek. Further, breeding females may suffer detrimental health impacts from elevated stress at a time when stress levels are already elevated (Jankowski 2007). While we do not have direct evidence for an impact of noise on these off-lek activities, there is evidence that proximity to roads and infrastructure (which raises noise levels) affects nest placement, nest initiation rates, chick survival, and brood-rearing activities (Lyon and Anderson 2003, Holloran and Anderson 2005, Aldridge and Boyce 2007, Holloran et al. 2010).

Other types of disruptive activities in sage-grouse habitat are managed throughout areas critical for lekking, nesting, and early brood rearing (e.g., State of Wyoming 2011, BLM 2012); there is no scientific basis for focusing the monitoring and management of noise on the lek area alone without including these other critical areas.

Traffic noise

There is evidence that noise from traffic has a significant impact on sage-grouse. Blickley et al. (2012*a*) found a 73% decline in male attendance on leks exposed to traffic noise compared to their paired controls, more than twice the decline observed on leks exposed to drilling noise (29%). Traffic noise also was also found to cause an increase in stress hormone levels (Blickley et al. 2012*b*) and a disruption of strutting patterns on the lek (Blickley 2012). Further evidence comes from other studies not focused on noise alone. Lyon and Anderson (2003) found that even light vehicular traffic (1 to 12 vehicles/day) substantially reduced nest initiation rates and increased the distance of nests from lek sites. Holloran (2005) found that traffic on roads within 1.3 km of the lek during the early morning, while males are strutting, is related to declines in male attendance. These results suggest that effective management strategies should include efforts to minimize traffic near areas critical for sage-grouse reproduction.

However, management strategies that allow up to 10 dB of noise above ambient are not sufficient to protect sage-grouse from the impacts of traffic noise. Because traffic noise in sage-grouse habitat is typically intermittent and interspersed with periods of quiet, a high volume of traffic would be needed to raise overall noise levels by 10 dBA. In general, a 10-dB increase in average noise levels is associated with a 10-fold increase in traffic, which would represent an increase from 2 to 20 vehicles or from 20 to 200 vehicles over a given time interval. A 10-fold increase in traffic would likely have a major impact on sage-grouse, yet may not exceed current noise management objectives inside and outside of core areas. This suggests that approaches for the management of more continuous noise sources, such as noise from compressors stations, drilling rigs, and other permanent or temporary infrastructure, may not be suitable for the management of traffic noise.

Recommendations for research priorities

Our understanding of impacts of noise on sage-grouse has improved over the last few years, but there is still much to learn. Below, we outline recommendations for research that would help to develop more effective management strategies for anthropogenic noise.

Establishing ambient values

As discussed above, management objectives for noise are typically established relative to ambient noise levels. The choice of ambient value, thus, has important consequences, setting the upper limit of allowable noise. It is, therefore, critical to establish accurate ambient noise values for such management strategies to succeed in protecting vulnerable species.

Due to the previously discussed difficulty of measuring ambient values at quiet locations, we suggest that it is not feasible or practical to establish baseline noise levels by having personnel with little specialized training measure ambient noise at each lek prior to development. Further, experimental evidence discussed above indicates that ambient values should represent the pre-development ambient levels, such that new developments do not further impact already impacted soundscapes. One approach to establish pre-development ambient noise levels is to commission the measurement of ambient levels by professionals with experience in environmental acoustics. Such professionals would need to measure ambient values for each site prior to development (or if there are already noise sources in an area, they could choose a similar but undisturbed area to estimate natural ambient levels; e.g., Ambrose and Florian 2013). Alternatively these professionals could sample noise levels at representative undisturbed areas across the state, using such measurements to establish ambient values by region or habitat type.

We recommend that ambient measurements should be collected using a Type-1 precision SLM enclosed in environmental housing for long-term deployment at each site. The meter should log unweighted one-third-octave spectra of noise at 1-second intervals. The L_{90} and other metrics listed in the Appendix should each be collected as A-weighted values, and, if possible, as dBF (i.e., dB-flat or unweighted) and C-weighted. With a logging SLM, one can save the time history, showing how noise levels change over time in the sampling period. This can be very useful in isolating the causes of change in noise levels. One can also calculate each metric hourly or over the entire sampling period. Hourly metrics are useful when focusing on a critical time window (e.g., 0600 to 0900 hours). The meter (or a nearby station) should also log wind speed, so that measurements can be excluded when wind likely contributes to noise levels. In addition to using SLMs, alternative methods to collect noise measurements, such as carefully calibrated audio recording units that can be used to calculate appropriate metrics, would also be appropriate (Patricelli et al. 2007, Lynch et al. 2011).

Such empirical sampling of noise levels also could be combined with noise modeling to create a map of natural ambient noise across focal areas. There are a number of suitable freeware programs for predicting sound propagation, such as NMSim (Wyle Laboratories Consulting, Arlington, Va., and Blue Ridge Research and Consulting, LLC, Asheville, N.C.), and SPreAD-GIS (Reed et al. 2012), as well as commercial software, such as SoundPLAN (Braunstein + Berndt GmbH, Germany) and Predictor-LimA (Brüel and Kjær Sound and Vibration Measurement A/S, Nærum, Denmark). This map would be useful for multiple public and private agencies interested in tracking noise exposure, because the data are not sage-grouse specific.

Determining an appropriate threshold

Once an ambient value is determined, we must then resolve whether the current threshold of 10 dB above ambient is sufficient to protect sage-grouse. The most feasible way to determine the threshold level at which sage-grouse are impacted by noise is by analyzing nesting success, lek attendance, and other population variables relative to existing variation in noise levels in a spatially-explicit manner using habitat-selection modeling. This method would examine the impact of variation in noise exposure across a disturbed landscape, while statistically controlling for other possible contributors. The resulting slope of the relationship between noise and measures of population change can then be used to predict the threshold level at which a minimal (or acceptable) level of impact on sage-grouse occurs. Such an approach would also be useful for examining noise impacts outside of the breeding season, especially in winter, where changes in habitat quality and availability can lead to significant impacts on population health (Beck 1977, Swenson et al. 1987, Doherty et al. 2008).

Measuring traffic noise

The evidence reviewed above demonstrates that traffic noise negatively impacts sage-grouse; however, we do not know the best metrics to use for management objectives in limiting traffic noise. This is because intermittent traffic, such as the traffic in most sage-grouse habitats, produces short periods of loud noise interspersed with longer periods of quiet. We do not know whether it is the total noise exposure throughout the day (or in a critical time period, such as nights and mornings) or the maximum noise level as a vehicle passes that best predicts impacts on sage-grouse. Lyon and Anderson (2003) found that nesting activities can be disturbed by only 1 to 12 vehicles/day, suggesting that the chosen noise metric should be sensitive to infrequent sounds. Infrequent to low traffic levels would barely register using measures of average or median amplitude (e.g., Leq or L_{50}). Even measures of maximum noise levels (e.g., L_{max} of L_{peak}; see Appendix) can be problematic, because other sound sources besides vehicles can affect these measures. For example a single bird singing near the meter could lead to extremely high maximum noise measurements. Such events can be excluded using synchronized audio or video recordings, direct observations or by analyzing the frequency profile of the noise (Lynch et al. 2011). Even with such an exclusion protocol in place, maximum values may be more informative when combined with a measure of overall exposure, such as L_{eq} or axle counts.

To determine which noise metrics best predict traffic impacts on sage-grouse, traffic noise can be included in habitat-selection models. This approach will allow estimation of the relationships between population variables and traffic variables (distance, traffic level and noise level). This would help to establish whether the impacts from traffic noise are better mitigated through setting noise objectives or by managing the siting and traffic levels of roads directly. Many of the noise-prediction models discussed in the previous section allow modeling of moving sources, such as different kinds of vehicles.

Recommendations for interim protections

The research described above will take time. Below, we provide managers and policy makers with recommendations for the interim protection of sage-grouse from known or expected impacts of increased noise levels using the best available science. We emphasize that protections based on these interim recommendations may need to be revised upon completion of ongoing and future research.

Setting an ambient value

Based on our review of reports and empirical measurements collected in Wyoming, we have concluded that true ambient values pre-development in nights and calm morning in sagebrush habitat are likely to be 16 to 20dBA. The first source for this conclusion is the 1971 EPA report from which the original 39 dBA ambient value was drawn (EPA 1971). This report finds residual noise levels (L_{90}) in wilderness areas of 16-22 dBA, measured during day and nighttime at a campsite on the north rim of the Grand Canyon National Park (excluding evenings from 1900 to 2200 hours, which were dominated by insect noise that is minimal during the sage-grouse breeding season due to low temperatures). The EPA report concludes that "these increases in [residual] noise level, from wilderness to farm and to city, are the result of man's activities and his use of machines." Lynch et al. (2011) more recently measured noise exposure at 189 sites in 43 U.S. National Parks, finding an average 24-hour residual noise level of 21.6 dBA. Note that these measures include only the one-third octave bands from 12.5 Hz to 800 Hz, so they are not directly comparable to the full-spectrum measures; however, these frequencies span most anthropogenic noise and residual noise in undisturbed areas, so this measure provides an appropriate estimate of ambient noise levels at these sites (Lynch et al. 2011).

In addition, in our analysis of the data from long-term deployment of SLMs by KC Harvey Environmental L.L.C. (2009) on the PAPA, the median L_{90} among 12 monitored leks was 27.2 dBA and the minimum lek was 22.2 dBA (Table 1). These are likely overestimates of pre-development ambient, given that (1) all of these leks experienced some noise from natural gas infrastructure and highways and (2) that measurements included afternoons and windy periods, and (3) that this Type-2 SLM had a noise floor of 20 to 22 dBA and, thus, could not measure quieter values (and likely overestimated levels near this lower limit; see Appendix for more information).

A more recent study, which measured noise using highly-sensitive Type-1 SLM with a noise floor of 14 dB, found that the mean day-long residual noise level (L_{90}) of 3 undisturbed leks near the PAPA was 15.5 dBA (range 14.2 dBA to 17.1 dBA). Even on the heavily-developed PAPA, the 19 monitored leks ranged from 16.0 dBA to 34.8 dBA, with 4 of the leks having L90 values < 20 dBA (Ambrose and Florian 2013). Therefore, we recommend that an ambient value of 16 to 20 dBA should be used for interim protections in sage-grouse habitat. In revised management strategies, this new default ambient would replace the previous default of 39 dBA or replace empirical measurements of ambient at lek edge.

Setting a threshold above ambient

As discussed above, we do not yet know whether limiting noise to 10 dB above ambient is appropriate for protecting sage-grouse. However, we recommend continuing to use the 10 dB threshold as an interim measure, combined with appropriate measures of ambient (i.e., 16 to 20 dBA). This threshold value is based on the best available science, but should be revised as needed when better information becomes available. Using 16 dBA as the ambient value would allow up to 26 dBA of noise exposure; using 20 dBA as ambient would allow up to 30 dBA of noise exposure.

How should compliance with this management objective be measured? Noise can be variable over time, space, and frequency spectrum, and no single metric can capture this complexity. However, using multiple metrics to assess compliance may be complicated to implement, at least in the interim. Therefore, we recommend using the A-weighted L_{50} as a measure of median noise exposure. This metric is useful because it is less influenced by the brief, intruding sounds (e.g., birds, insects and airplanes) that can dominate other metrics. This metric also may exclude some types of noise produced by the development activities being monitored, including vehicles (unless traffic is very heavy). For that reason, it will typically not be effective at reflecting any impacts caused by traffic noise. Despite this concern, the L_{50} is recommended because, otherwise, birds, insects, and other indicators of a healthy habitat may be counted against compliance (unless audio recordings are produced, allowing monitors to exclude time periods with such activity; this may be a preferable solution in the long run, but it will require time to develop such a protocol).

We recommend that measurements be made during times when noise exposure is most likely to affect greater sage-grouse; that is, nights and mornings (i.e., 0600 to 0900 hours). Further, we recommend using the average of L_{50} values at multiple (3 to 4) locations between each noise source and the edge of the protected area. This will reduce the impact of aberrant measurements (high or low) at particular locations, because noise values can change with topography and local ground cover. Measurements should be taken with a Type-1 sound level meter (or a method with similar accuracy and a noise floor <20 dBA). We recommend making measurements of ≥1 hour at each site, ideally over multiple days and climatic conditions, because temperature (especially temperature inversions), humidity, and wind can affect noise levels. Whenever possible, we recommend collecting additional metrics for research and long-term monitoring (see recommended metrics in the "Establishing ambient values" section above).

It should be noted that, based on the measurements presented in Table 1, four of the 12 monitored leks on the Pinedale Anticline are in compliance with the noise management objectives recommended here based on a 20-dBA ambient value (i.e., they do not exceed an L_{50} of 30 dBA). These leks are in a heavily developed area that has experienced declines in sage-grouse populations (Holloran 2005, Holloran et al. 2010). This suggests that (1) these recommended protections are not as onerous as they may initially seem, and (2) even these stricter recommendations may not suffice to avoid population declines if noise levels are measured at lek edge (as in Table 1), rather than across nesting and brood-rearing habitats, as discussed below.

Redefining the protected area

Current noise management strategies typically recommend noise measurements at the edge of the lek to assess compliance (e.g., State of Wyoming 2011; BLM 1999, 2003, 2008). This approach manages noise levels on the lek itself, but not in the surrounding habitat that is critical to successful reproduction of sage-grouse. As discussed above, there is evidence that this off-lek noise will affect on-lek activities and successful reproduction. Therefore, we recommend that interim and longer-term management strategies aim to protect the soundscape in areas critical for mating, foraging, nesting, and brood-rearing activities. Thus, we recommend that noise >10 dB above ambient be managed as a disruptive activity throughout sage-grouse lekking, nesting, and brood-rearing habitat (e.g., BLM 2012). To accomplish this, we recommend measuring compliance with noise objectives at the edge of the critical area encompassing lekking, nesting, and brood-rearing activities, rather than at the edge of the lek. These critical areas are typically defined as buffers surrounding the edge of the lek, with a 3.1-km buffer encompassing ~45% of nests and a 6.4-km buffer encompassing 74 to 80% of nests (Moynahan 2004, Holloran and Anderson 2005). Where possible, mapping of utilized areas would be preferable. The size and shape of the protected area should be determined based on management objectives.

Limiting traffic noise

Given the difficulty of measuring intermittent traffic noise and the uncertainty about which metrics are informative, we recommend that interim protections focus not on setting objectives for traffic noise levels, but, rather, on the siting of roads or the limitation of traffic during critical times of the day (0600 to 0900 hours) and year (breeding season).

To develop interim recommendations for the siting of roads, we estimated the distance from a road at which noise levels (L_{max} as a single vehicle passes) will drop down to 10 dB above ambient. To calculate this estimate of impact distances from roads, we used our measurements of noise levels from 17 vehicles (flatbed trucks and big rigs) on the Luman Road and 8 vehicles on North Jonah Road on the Jonah Natural Gas Field in Sublette County, Wyoming (collected in 2006). All measurements were made at 0.4 km from the road. A-weighted L_{max} values were averaged for each road and the average of the 2 roads was 45.5 dBA (S.E. = 1.3 dBA; range 37 to 58.7 dBA). We similarly calculated average A-weighted levels for each octave from 16 to 16,000 Hz. In each octave band, we calculated propagation using the assumption of spherical spreading and octave-specific excess attenuation values from the Pinedale Anticline noise analysis report prepared by the BLM with assistance from the U.S. Army Corps of Engineers and U.S. Forest Service (BLM 1999). Using these methods, we extrapolated noise propagation beyond our quarter-mile measurements until levels reached 30, 26, 20 and 16 dBA (Figure 2). The same calculations were used to estimate propagation distances around an example drilling rig measured on the PAPA in 2006 (an L_{eq} of 66.7 dBA at 0.1 km; Figure 2).

Using an ambient of 20 dBA, we calculated that vehicle noise will diminish to 30 dB at ~1.3 km from the road. Using an ambient of 16 dB, we calculated that vehicle noise will diminish to 26 dBA at ~1.7 km from the road. Therefore, to avoid disruptive activity in areas crucial to mating, nesting, and brood-rearing activities, we recommend that managers consider siting roads (or seasonally limiting traffic) within 1.3 to 1.7 km from the edge of these areas. We emphasize that we are recommending restrictions within this distance of the edge of sage-grouse nesting and brood-rearing habitat, not the lek edge. Further, note that noise from traffic will be audible at least until levels drop down to ambient values, which will occur 2.7 to 3.6 km from the road. Therefore, adopting these recommendations will not eliminate traffic noise in critical areas, but should reduce its impact.

Under certain conditions, noise may propagate much farther than predicted above. The above estimates are based on the maximum noise levels as a single vehicle passes; however, on roads with sufficient traffic to create a steady stream of vehicles, noise drops off more slowly (levels would follow predictions of cylindrical spreading, dropping only 3 dB with every doubling of distance, rather than 6 dB, as assumed here). Similarly, noise levels drop off according to predictions of cylindrical spreading during early morning temperature inversions, which are common in sage-grouse habitat (Schnell et al. 2009). For an ambient of 20 dB and 16 dB, respectively, traffic noise under conditions of cylindrical spreading would reach 10 dB above ambient at 2.3 to 3.3 km from the road, and this noise would reach ambient at 5.3 to 6.4 km from the road. For these reasons, the recommendations presented here will not protect sage-grouse breeding activities under all conditions, but will be a significant improvement over current policy in most cases.

Given that traffic noise was found to have more than twice the impact of continuous noise on lek attendance (Blickley et al. 2012*a*), minimizing traffic noise as a disruptive activity in all areas critical for successful reproduction should be a priority in any revised noise management strategy. In areas where implementing recommended limits on siting or traffic is not possible, other measures may reduce traffic noise impacts. One possibility would be to adjust the times at which personnel begin and end work shifts in development areas to avoid causing an increase in traffic during critical times. Avoiding shift changes between 1800 and 0900 hours would be ideal, but if this is not possible, then avoiding 2400 to 0900 hours would likely be a significant improvement.

Conclusions

Over the last decade, interest in understanding noise impacts on wildlife has been increasing rapidly (Barber et al. 2009, Blickley and Patricelli 2010, Kight and Swaddle 2011). Recent research has demonstrated that noise can cause avoidance (Habib et al. 2007, Bayne et al. 2008, Blickley et al. 2012*a*), flight (Brown 1990, Delaney et al. 1999), altered communication (Slabbekoorn and Peet 2003, Leonard and Horn 2005), reduced pair-bonding (Swaddle and Page 2007), reduced breeding success (Francis et al. 2009), increased stress (Weisenberger et al. 1996, Blickley et al. 2012*b*), increased mortality in some species, and no effect or even the opposite effects in other species (Francis et al. 2009, Crino et al. 2013). As a result of the increased interest in noise impacts, the methods available to measure noise and noise impacts have been improving rapidly, as have industry standards (Pater et al. 2009, Lynch et al. 2011). The recommendations presented here for further research, for noise measurement protocols, and for interim protection are based on the best available science, reflecting our current understanding of noise impacts on greater sage-grouse. However, we emphasize the importance of building flexibility into sage-grouse protections in Wyoming and other states so that the results of ongoing and future research can be used to improve upon the recommendations presented here.

Finally, it is critical to note that noise is only one of multiple types of disturbance impacting greater sage-grouse habitat (Connelly et al. 2004, Connelly et al. 2011). Noise mitigation alone is unlikely to suffice in offering protection for this species. Indeed, in some cases, restrictions on the density of developments (e.g., well density in areas of natural gas development) may offer more or equivalent protection from noise and other types of disturbance than the recommendations we make here, if those restrictions lead to larger distances between developments and critical habitat for sage-grouse. Therefore, we are not recommending that the protections described here supplant all existing protection. Rather, we hope that these recommendations for protecting the soundscape be considered as part of a comprehensive conservation strategy for sage-grouse that addresses many types of disturbance.

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Literature cited

Aldridge, C. L., and M. S. Boyce. 2007. Linking occurrence and fitness to persistence: habitat-based approach for endangered greater sage-grouse. Ecological Applications 17:508–526.

Ambrose, S., and C. Florian. 2013. Draft: sound levels of gas field activities at greater sage-grouse leks, Pinedale Anticline Project Area, Wyoming, April 2013. Sandhill Company, Pinedale Anticline Project Office, Pinedale, Wyoming, USA.

American National Standards Institute. 1983. American national standard specification for sound level meters, S1.4. American National Standards Institute, Washington, D.C., USA.

American National Standards Institute. 2001. American national standard specification for sound level meters, S1.4. American National Standards Institute, Washington, D.C., USA.

American National Standards Institute. 2003. American national standard quantities and procedures for description and measurement of environmental sound, S12.9P1. American National Standards Institute, Washington, D.C., USA.

American National Standards Institute. 2005. American national standard quantities and procedures for description and measurement of environmental sound: noise assessment and prediction of long-term community response, S12.9P4. American National Standards Institute, Washington, D.C., USA.

Barber, J. R., K. R. Crooks, and K. M. Fristrup. 2009. The costs of chronic noise exposure for terrestrial organisms. Trends in Ecology and Evolution 25:180–189.

Bayne, E., L. Habib, and S. Boutin. 2008. Impacts of chronic anthropogenic noise from energy-sector activity on abundance of songbirds in the boreal forest. Conservation Biology 22:1186–1193.

Beck, T. D. I. 1977. Sage-grouse flock characteristics and habitat selection in winter. Journal of Wildlife Management 41:18–26.

Blickley, J. L. 2012. The effects of anthropogenic noise on lek attendance, communication, and behavior in greater sage-grouse (*Centrocercus urophasianus*). Dissertation, University of California, Davis, California, USA.

Blickley, J. L., D. Blackwood, and G. L. Patricelli. 2012*a*. Experimental evidence for the effects of chronic anthropogenic noise on abundance of greater sage-grouse at leks. Conservation Biology 26:461–471.

Blickley, J. L., and G. L. Patricelli. 2010. Impacts of anthropogenic noise on wildlife: research priorities for the development of standards and mitigation. Journal of International Wildlife Law and Policy 13:274–292.

Blickley, J. L., and G. L. Patricelli. 2012. Potential acoustic masking of greater sage-grouse display components by chronic industrial noise. Ornithological Monographs 74:23–35.

Blickley, J. L., K. R. Word, A. H. Krakauer, J. L. Phillips, S. N. Sells, J. C. Wingfield, and G. L. Patricelli. 2012*b*. Experimental chronic noise exposure is related to elevated fecal corticosteroid metabolites in lekking male greater sage-grouse (*Centrocercus urophasianus*). PLoS ONE 7:e50462.

Bonier, F., P. R. Martin, I. T. Moore, and J. C. Wingfield. 2009. Do baseline glucocorticoids predict fitness? Trends in Ecology and Evolution 24:634–642.

Braun, C. E. 1998. Sage-grouse declines in western North America: what are the problems? Proceedings of the Western Association of State Fish and Wildlife Agencies 78:139–156.

Brenowitz, E. A. 1982. The active space of red-winged blackbird song. Journal of Comparative Physiology 147:511–522.

Brown, A. L. 1990. Measuring the effect of aircraft noise on sea birds. Environment International 16:587–592.

Bureau of Land Mangement. 1999. Noise analysis for the Pinedale Anticline Oil and Gas Exploration and Development Project, Sublette County, Wyoming. U.S. Bureau of Land Management, Pinedale Field Office, Pinedale, Wyoming, USA.

Bureau of Land Mangement. 2003. Final environmental impact statement and proposed plan amendment for the Powder River Basin oil and gas project. U. S. Bureau of Land Mangement, Buffalo Field Office, Buffalo, Wyoming, USA.

Bureau of Land Mangement. 2008 Record of decision for the supplemental environmental impact statement, Pinedale Anticline Oil and Gas Exploration and Development Project. U.S. Bureau of Land Mangement, Pinedale Field Office, Pinedale, Wyoming, USA.

Bureau of Land Management. 2012. Instruction memorandum 2012-19: greater sage-grouse habitat management policy on Wyoming Bureau of Land Management administered public lands, including the federal mineral estate.

U. S. Bureau of Land Management, Wyoming State Office, Cheyenne, Wyoming, USA.

Connelly, J. W., K. P. Reese, and M. A. Schroeder. 2003. Monitoring of greater sage-grouse habitats and populations. College of Natural Resources Experiment Station Bulletin. University of Idaho, Moscow, Idaho, USA.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004 Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming, USA.

Connelly, J. W., S. T. Knick, C. E. Braun, W. T. Baker, E. A. Beever, T. Christiansen, K. E. Doherty, E. O. Garton, S. E. Hanser, D. H. Johnson, M. Leu, R. F. Miller, D. E. Naugle, S. J. Oyler-McCance, D. A. Pyke, K. P. Reese, M. A. Schroeder, S. J. Stiver, B. L. Walker, and M. J. Wisdom. 2011. Conservation of greater sage-grouse: a synthesis of current trends and future management. Pages 549–563 *in* S. T. Knick and J. W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Volume 38. University of California Press, Berkeley, California, USA.

Crino, O. L., E. E. Johnson, J. L. Blickley, G. L. Patricelli, and C. W. Breuner. 2013. Effects of experimentally elevated traffic noise on nestling white-crowned sparrow stress physiology, immune function and life history. Journal of Experimental Biology 216:2055–2062.

Delaney, D. K., T. G. Grubb, P. Beier, L. L. Pater, and M. H. Reiser. 1999. Effects of helicopter noise on Mexican spotted owls. Journal of Wildlife Mangement 63:60–76.

Doherty, K. E., D. E. Naugle, B. L. Walker, and J. M. Graham. 2008. Greater sage-grouse winter habitat selection and energy development. Journal of Wildlife Management 72:187–195.

Doherty, K. E., D. E. Naugle, and B. L. Walker. 2010. Greater sage-grouse nesting habitat: the importance of managing at multiple scales. Journal Of Wildlife Management 74:1544–1553.

Dooling, R. J., and A. N. Popper. 2007. The Effects of highway noise on birds. California Department of Transportation Division of Environmental Analysis, Sacramento, California, USA.

Environmental Protection Agency. 1971. Community noise. U.S. Environmental Protection Agency, Washington, D.C., USA.

Francis, C. D., C. P. Ortega, and A. Cruz. 2009. Noise pollution changes avian communities and species interactions. Current Biology 19:1415–1419.

Gibson, R. M. 1989. Field playback of male display attracts females in lek breeding sage grouse. Behavioral Ecology and Sociobiology 24:439–443.

Gibson, R. M. 1996. Female choice in sage grouse: the roles of attraction and active comparison. Behavioral Ecology and Sociobiology 39:55–59.

Gibson, R. M., and J. W. Bradbury. 1985. Sexual selection in lekking grouse: phenotypic correlates of male strutting success. Behavioral Ecology and Sociobiology 18:117–123.

Habib, L., E. M. Bayne, and S. Boutin. 2007. Chronic industrial noise affects pairing success and age structure of ovenbirds *Seiurus aurocapilla*. Journal of Applied Ecology 44:176–184.

Hagen, C. A. 2011. Predation on greater sage-grouse: facts, process, and effects. Pages 95–100 *in* S. T. Knick and J. W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Volume 38. University of California Press, Berkeley, California, USA.

Holloran, M. J. 2005. Greater sage-grouse (*Centrocercuc urophasianus*) population response to natural gas field development in western Wyoming. Dissertation, University of Wyoming, Laramie, Wyoming, USA.

Holloran, M. J., and S. H. Anderson. 2005. Spatial distribution of greater sage-grouse nests in relatively contiguous sagebrush habitats. Condor 107:742–752.

Holloran, M. J., R. C. Kaiser, and W. A. Hubert. 2010. Yearling greater sage-grouse response to energy development in Wyoming. Journal Of Wildlife Management 74:65–72.

Hu, Y., and G. C. Cardoso. 2009. Are bird species that vocalize at higher frequencies preadapted to inhabit noisy urban areas? Behavioral Ecology 20:1268–1273.

Jankowski, M. D. 2007. The influence of habitat disturbance and synergized resmethrin on avian immunocompetence. Dissertation, University of Wisconsin, Madison, Wisconsin, USA.

KC Harvey Environmental L.L.C. 2009. Pinedale Anticline Project area sage-grouse monitoring: noise monitoring report. Pinedale Anticline Project Office, Pinedale, Wyoming, USA.

Kight, C. R., and J. P. Swaddle. 2011. How and why environmental noise impacts animals: an integrative, mechanistic review. Ecology Letters 14:1052–1061.

Leonard, M. L., and A. G. Horn. 2005 Ambient noise and the design of begging signals. Proceedings of the Royal Society of London, Series B. 272:651–656.

Lynch, E., D. Joyce, and K. Fristrup. 2011. An assessment of noise audibility and sound levels in U.S. National Parks. Landscape Ecology 26:1297–1309.

Lyon, A. G., and S. H. Anderson. 2003. Potential gas development impacts on sage-grouse nest initiation and movement. Wildlife Society Bulletin 31:486–491.

Moynahan, B. J. 2004. Landscape-scale factors affecting population dynamics of greater sage-grouse (*Centrocercus urophasianus*) in north-central Montana. Dissertation, University of Montana, Missoula, Montana, USA.

Naugle, D. E., K. E. Doherty, B. E. Walker, M. J. Holloran and H. J. Copeland. 2011. Energy development and greater sage-grouse. Pages 489–504 *in* S. T. Knick and J. W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Volume 38. University of California Press, Berkeley, California, USA.

Nicholoff, S. H., compiler. 2003 Wyoming Bird Conservation Plan, Version 2.0. Wyoming Partners In Flight, Wyoming Game and Fish Department, Lander, Wyoming, USA.

Pater, L. L., T. G. Grubb, and D. K. Delaney. 2009. Recommendations for improved assessment of noise impacts on wildlife. Journal Of Wildlife Management 73:788–795.

Patricelli, G. L., M. S. Dantzker, and J. W. Bradbury. 2007. Differences in acoustic directionality among vocalizations of the male red-winged blackbird (*Agelaius phoeniceus*) are related to function in communication. Behavioral Ecology and Sociobiology 61:1099–1110.

Patricelli, G. L., and A. H. Krakauer. 2010. Tactical allocation of effort among multiple signals in sage grouse: an experiment with a robotic female. Behavioral Ecology 21:97–106.

Quinn, L., J. Whittingham, J. Butler, and W. Cresswell. 2006. Noise, predation risk compensation and vigilance in the chaffinch *Fringilla coelebs*. Journal of Avian Biology 37:601–608.

Rabin, L. A., R. G. Coss, and D. H. Owings. 2006. The effects of wind turbines on antipredator behavior in California ground squirrels (*Spermophilus beecheyi*). Biological Conservation 131:410–420.

Reed, S. E., J. L. Boggs, and J. P. Mann. 2012. A GIS tool for modeling anthropogenic noise propagation in natural ecosystems. Environmental Modelling and Software 37:1–5.

Rogers, G. E. 1964. Sage-grouse investigations in Colorado, Technical Publication No. 16, Colorado Game, Fish and Parks Department, Denver, Colorado, USA.

Schnell, R. C., S. J. Oltmans, R. R. Neely, M. S. Endres, J. V. Molenar, and A. B. White. 2009. Rapid photochemical production of ozone at high concentrations in a rural site during winter. Nature Geoscience 2:120–122.

Schoenberg, T. J. 1982. Sage- grouse movements and habitat selection in North Park, Colorado. Thesis, Colorado State University, Fort Collins, Colorado, USA.

Schroeder, M. A., J. R. Young, and C. E. Braun. 1999 Sage-grouse: *Centrocerus urophasianus*. Birds of North America 425:1–28.

Slabbekoorn, H., and M. Peet. 2003. Ecology: Birds sing at a higher pitch in urban noise. Nature 424:267.

State of Wyoming. 2008. Greater sage-grouse core area protection. Office of the Governor, State of Wyoming Executive Order 2008-2, Cheyenne, Wyoming, USA, <http://www-wsl.state.wy.us/sis/wydocs/execorders/WS-GOV-EO-2008-02.pdf>. Accessed August 20, 2013.

State of Wyoming. 2010. Greater sage-grouse core area protection. Office of the Governor, State of Wyoming Executive Order 2010-4, Cheyenne, Wyoming, USA, <http://www-wsl.state.wy.us/sis/wydocs/execorders/WS-GOV-EO-2010-04.pdf>. Accessed August 20, 2013.

State of Wyoming. 2011. Greater sage-grouse core area protection. Office of the Governor, State of Wyoming Executive Order 2011-5, Cheyenne, Wyoming, USA, <http://www-wsl.state.wy.us/sis/wydocs/execorders/EO2011-05.pdf>. Accessed August 20, 2013.

Swaddle, J. P., and L. C. Page. 2007. High levels of environmental noise erode pair preferences in zebra finches: implications for noise pollution. Animal Behaviour 74:363–368.

Swenson, J. E., C. A. Simmons, and C. D. Eustace. 1987. Decrease of sage-grouse *Centrocerus urophasianus* after ploughing of sagebrush steppe. Biological Conservation 41:125–132.

Vehrencamp, S. L., J. W. Bradbury, and R. M. Gibson. 1989. The energetic cost of display in male sage grouse. Animal Behaviour 38:885–896.

Wallestad, R. O., and P. Schladweiler. 1974. Breeding season movements and habitat selection of male sage grouse. Journal of Wildlife Management 38:634–637.

Wasser, S. K., K. E. Hunt, J. L. Brown, K. Cooper, C. M. Crockett, U. Bechert, J. J. Millspaugh, S. Larson, and S. L. Monfort. 2000. A generalized fecal glucocorticoid assay for use in a diverse array of nondomestic mammalian and avian species. General and Comparative Endocrinology 120:260–275.

Weisenberger, M. E., P. R. Krausman, M. C. Wallace, D. W. D. Young, and O. E. Maughan. 1996. Effects of simulated jet aircraft noise on heart rate and behavior of desert ungulates. Journal of Wildlife Management 60:52–61.

Wiley, R. H. 1973. Territoriality and non-random mating in sage-grouse (*Centrocercus urophasianus*). Animal Behaviour Monographs 6:85–169.

Wingfield, J. C. 2005. The concept of allostasis: coping with a capricious environment. Journal of Mammalogy 86:248–254.

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Appendix:

Glossary of noise terms

A-weighting: A-weighting (usually denoted as dBA or dB-A) is used to account for changes in level sensitivity as a function of frequency (ANSI 2001). In an effort to simulate the relative response of the human ear, A-weighting de-emphasizes the high (>6.3 kHz) and low (<1 kHz) frequencies, and emphasizes the frequencies in between. Unfortunately, there is no weighting specific to sage-grouse or other wildlife. Most birds, besides owls, have hearing capabilities similar to or slightly worse than humans; therefore, some experts recommend that A-weighting may be a suitable if not ideal metric for studies of birds (Dooling and Popper 2007).

Ambient noise: Ambient noise, often called background noise, is typically defined as any sound other than the sound being monitored. Ambient noise can be measured to include all of the non-focal sounds in the environment, such as wind, birds, insects, and other sources, including anthropogenic noise. Here we recommend that, to improve repeatability and generality of measurements and better limit anthropogenic noise impacts, we should minimize the contribution of these sources of noise in our ambient measures when setting a baseline for noise management strategies.

Decibel: The amplitude of a sound, perceived as loudness, is typically measured in decibels (dB). The decibel scale is logarithmic, and, therefore, small changes in decibel level represent large changes in loudness. Every 6 dB increase in noise levels is a doubling in amplitude, measured as changes in air pressure. One often hears the rule of thumb that a 10 dB increase in noise is subjectively perceived by humans as a doubling in loudness. However, this perception depends on the frequencies (i.e., pitch) of the sounds and can vary with amplitude. In humans, a 6 dBA increase in noise level leads to an approximate doubling in the number of noise complaints (ANSI 2005), suggesting that humans are more sensitive than this 10 dB rule of thumb implies. Because we do not know if sage-grouse or other nonhuman animals perceive sounds similarly to humans, the non-subjective "6 dB doubling" rule of thumb is preferable. Noise measurements are typically made relative to the threshold of human hearing (20μPa) and denoted as sound pressure level (SPL), or dB SPL (though the SPL is often assumed). A value of 0 dB SPL is equal to the threshold of human hearing; 60 to 70 dB SPL is typical conversational level and 130 dB SPL is the threshold of pain.

Detection distance and listening area: Detection distance is the maximum distance between the sender and receiver where the signal is still audible. The listening area is the total area around the sender over which a sound can be detected (also called the active space; Brenowitz 1982). Barber et al. (2009) offered simple formulas for estimating the reduction in detection distance and listening area resulting from an increase in background noise. The formula for calculating how the detection distance changes with an increase in noise is: detection distance $=10^{(-\text{ (dB change in noise)}/20)}$. This shows a halving of detection distance for each 6 dB increase in noise; therefore, a >3-fold decrease (69% decrease) in detection distance with a 10 dB increase in noise and a 10-fold reduction in detection distance (90% decrease) with a 20 dB increase in noise. The formula for calculating how the listening area changes with an increase in noise is: listening area = $10^{(-\text{(dB change in noise)}/10)}$. The area of a circle (i.e., listening area around the vocalizing animal) decreases with the square of the radius (i.e., detection distance between the vocalizing animal and the receiver), which leads to a halving of listening area with every 3 dB increase in noise and 10-fold reduction with every 10 dB. These decreases in active space and detection distance are less extreme when environmental attenuation of noise is considered, but are nonetheless very large (Blickley and Patricelli 2012).

One-third octave bands: An octave is a band of frequencies whose lower limit is half of the upper limit, and is named for its central frequency. The range of human hearing is divided into 10 standardized octave bands; each octave-band can be broken down into 3 parts, or one-third-octave bands typically ranging from 12.5 Hz to 20 kHz. One-third-octave band levels can be used to construct power spectra that show the relative power of different frequencies. One-third octave band measures can be used to calculate a number of other metrics, especially if they are collected continuously at short intervals. Measurements of the relative amplitude of the noise at different frequencies is important for calculating the potential of a noise source to mask sound relevant to the species of interest and can sometimes be used to identify the source of the sound.

L_{eq} (also called L_{avg}): The equivalent noise level. This can be thought of as the average noise level across the sample period; more precisely, it is the level of a constant sound over a specific time period that has the same sound energy as the actual (variable) sound.

L_{max} and L_{min}: The RMS (root-mean squared) maximum and minimum noise levels integrated over a specified time interval and measured during a single noise event or specified time period. The L_{max} characterizes the maximum noise level, defined by the loudest single noise event. Similarly, L_{min} is the minimum noise level or quietest period.

L_{50}: The median noise level is the level that is exceeded 50% of the time. This measure is collected over some time period (e.g., 1 hour, or from 0600 to 0900 hours) with this period being broken down into much smaller intervals (typically 1 second); an L_{50} of 30 dBA would mean that half of the intervals measured were <30 dBA, and half of them were >30 dBA. This metric is recommended rather than a measure of average noise over a longer interval, like L_{eq} or L_{avg}, because these average metrics are more influenced by occasional loud events, such as those caused by a songbirds, insects, aircraft, wind gusts, etc. These intruding sounds will have no impact on the L_{50}, unless they are present more than 50% of the time.

L_{90}: This is accepted by the American National Standards Institute (ANSI 2003) as a measure of background or "residual noise level". As with the L_{50}, the L_{90} is collected over some time period (e.g., 1 hour, or from 0600 to 0900 hours) with this period being broken down into much smaller intervals (typically 1 second); an L_{90} of 20 dBA would mean that 10% of the intervals measured were <20 dBA and 90% of them were >20 dBA. Residual noise levels reflect background noise level at a site, since they exclude most intruding noise from birds, insects, wind gusts and sporadic anthropogenic noises (passing vehicles or aircraft) that raise the average (e.g., L_{eq} or L_{avg}) and peak values (e.g., L_{peak}, L_{max}, and L_{10}) over a measurement period. This metric is the most suited for estimating ambient values to set the baseline for management objectives. Note that in an area with anthropogenic noise sources producing continuous noise (like most energy development infrastructure), the L_{90} measurement will not represent pre-development ambient values since the continuous noise source will contribute to the residual levels. To estimate predevelopment ambient for a disturbed site, measurements must be collected in a similar but undisturbed area, or estimated through modeling.

L_{10}: The L_{10} is the noise level that is exceeded 10% of the time and is a metric that characterizes the maximum of noise level in an area. The L_{10} is collected over some time period with this period being broken down into much smaller intervals (typically, 1 second); an L_{10} of 60 dBA would mean that 90% of the intervals measured were <60 dBA, and 10% of them were >60 dBA.

Noise: Any unwanted sound is considered noise. Thus, signals produced by 1 animal, such as crickets, may be noise to another animal. When managing noise impacts on wildlife, we typically consider only sounds produced by humans and human-produced infrastructure to be noise.

Sound level meter (SLM): A sound level meter is a tool used to measure the amplitude of a noise source in decibels. Most Type-1 (ANSI 1983) precision SLMs have a "noise floor" of ~17 dB, meaning that they cannot measure quieter sounds, because these sounds will be masked by the noise from the SLM itself. Recently, highly-sensitive Type-1 SLMs with noise floors of 12 to 14 dBA have become available. Some SLM noise is typically detected ≤10 dB above the noise floor (i.e., 27 dB), especially when using A-weighting, as discussed above. This is not a problem when measuring louder sounds (i.e., many noise sources associated with development) that overwhelm any contribution of the noise from the SLM (as well as noise from a slight breeze or other incidental sounds). Measurements of quiet sounds are, thus, particularly challenging. Type-2 SLMs are more affordable but can have noise floors of ~35 dB and should, therefore, never be used to measure ambient noise or quiet sound sources (expected to be <35 to 40 dBA); some more expensive Type-2 meters have noise floors approaching 22 dBA and would, therefore, be more useful for measuring quiet sounds, but not ambient levels. The importance of the noise floor of the meter can be seen clearly when comparing the data from Ambrose and Florian (2013), who found an L_{90} of 16.0 dB on the quietest lek on the PAPA with a Type-1 SLM (14 dB noise floor), and the data of KC Harvey (2009; Table 1), who found an L_{90} of 22.2 dBA on the quietest lek on the PAPA with a Type-2 SLM (20 to 22 dB noise floor). These data suggest that the L90 values from the KC Harvey study were likely determined by the noise floor of the SLM rather than by the ambient noise levels in this area Within a few decibels above the noise floor, the accuracy of Type-2 meters is typically only slightly lower than Type-1 meters. Type-3 SLMs have higher noise floors and lower accuracy and should not be used for measuring ambient or assessing compliance.

Soundscape: All of the sounds at a particular location.

FACTORS AFFECTING GUNNISON SAGE-GROUSE (*CENTROCERCUS MINIMUS*)

CONSERVATION IN SAN JUAN COUNTY, UTAH

by

Phoebe R. Prather

A dissertation submitted in partial fulfillment
of the requirements for the degree

of

DOCTOR OF PHILOSOPHY

in

Ecology

Approved:

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UTAH STATE UNIVERSITY
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ABSTRACT

Factors Affecting Gunnison Sage-Grouse (*Centrocercus minimus*)

Conservation in San Juan County, Utah

by

Phoebe R. Prather, Doctor of Philosophy

Utah State University, 2010

Major Professor: Terry A. Messmer
Department: Wildland Resources

Due to loss of habitat, Gunnison sage-grouse (*Centrocercus minimus*) currently occupy 8.5% of their presumed historical range. One population survives in Utah, occurring in San Juan County. The Gunnison Sage-grouse Rangewide Conservation Plan and the San Juan County Gunnison Sage-grouse Conservation Plan recommended management strategies to address identified conservation threats to the Utah population. I addressed three conservation strategies identified in the plans: 1) creation and enhancement of brood-rearing areas; 2) assessment of habitat conditions within the Gunnison Sage-grouse Conservation Area; and 3) prevention or reduction of perching events by avian predators on distribution line power poles.

From 2007-2009, I addressed the conservation strategy of creating mesic brood-rearing areas in Conservation Reserve Program fields and native sagebrush areas by evaluating the role of irrigation and dormant season cattle grazing on habitat. Vegetation and arthropod diversity in irrigated versus non-irrigated plots did not differ (p>0.01).

Conservation Reserve Program plots exhibited greater arthropod abundance and cover of perennial grass than the native sagebrush plots, but lower diversity of perennial grasses and abundance and diversity of forbs ($p<0.01$).

The second conservation strategy I addressed was the completion of an assessment of habitat conditions within the Gunnison Sage-grouse Conservation Area. I measured vegetation conditions within habitat occupied and unoccupied by Gunnison sage-grouse. Cover and height of grasses exceeded guidelines for occupied and unoccupied habitats. Forb cover was below recommended guidelines in occupied habitat. Sagebrush cover was below guidelines for winter habitat. Habitat restoration efforts should focus on retaining existing sagebrush cover and establishment of sagebrush, forb, and grass cover within Conservation Reserve Program fields.

The third conservation strategy I evaluated was the retrofitting of distribution line power poles with perch deterrents to discourage avian predators from perching. I evaluated the efficacy of five perch deterrents. The perch deterrents did not mitigate potential avian predators from perching. A deterrent designed for insulators, in combination with physical deterrents we tested, has potential to prevent perching.

These studies provided a sound first step that can be built upon by the Monticello/Dove Creek Local Working Group to improve habitat conditions, reduce the threat of avian predation, and plan future conservation activities within the Conservation Area.

(138 pages)

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Phoebe R. Prather

CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

INTRODUCTION

BACKGROUND

In the mid 1970s the Colorado Division of Wildlife began studying sage-grouse (*Centrocercus* spp.) populations located within the state. These studies included the collection of wings from hunted sage-grouse (Young et al. 2000). Biologists noted that primary wings collected from sage-grouse in the Gunnison Basin of Colorado were smaller than those of other populations. These observations prompted further studies on the Gunnison Basin populations. The subsequent studies discovered differences in morphometrics, breeding behavior, plumage and genetics, leading to the reclassification of the grouse species that inhabits the Gunnison Basin in Colorado and southeastern Utah as the Gunnison sage-grouse (*C. minimus*) in 2000 by the American Ornithologists' Union (Young et al. 2000, AOU Checklist Committee 2002, Gunnison Sage-grouse Rangewide Steering Committee 2005).

Species Description

The Gunnison sage-grouse is substantially smaller than the greater sage-grouse (*C.urophasianus*), with shorter tarsus, culmen, and carpal measurements (Schroeder et al. 1999). The average mass of male Gunnison sage-grouse ranges from 1.5-1.82 kg., while the average mass of a male Greater sage-grouse ranges from 2-3 kg. The male Gunnison sage-grouse has considerably larger and thicker filoplumes and shorter rectrices that have more distinct barring. The males of the two species also differ in their strutting displays.

Species Distribution

Gunnison sage-grouse currently occupy 8.5% of their presumed historical range (Schroeder et al. 2004). The Gunnison sage-grouse was thought to have historically occurred in Colorado, Kansas, Oklahoma, New Mexico, Arizona, and Utah before rapid settlement of the west in the 1800s (Young et al. 2000). After a more thorough investigation the species is now believed to have occurred in southwestern Colorado, northwestern New Mexico, northeastern Arizona, and southeastern Utah (Schroeder et al. 2004). The distribution of presumed historic habitat encompassed 46,521 km^2 (21,376 mi^2), but the species is now estimated to have a range of 4,787 km^2 (1,822 mi^2, Schroeder et al. 2004, Fig. 1.1). This decline in the range of the species has been attributed to the loss or conversion of sagebrush (*Artemisia* spp.) to other land uses. The quality of the remaining habitat has been impacted by urbanization, grazing, agriculture and fragmentation (Schroeder et al. 2004). The historic distribution of the species was probably always somewhat patchy, but the patchiness has been greatly exacerbated by habitat loss (Gunnison Sage-grouse Rangewide Steering Committee 2005).

Habitat fragmentation has reduced the Gunnison sage-grouse to seven known populations in Colorado and one population in southeastern Utah (Fig. 1.2). In 2004, the Gunnison sage-grouse population was estimated to be fewer than 3,200 birds; with 2,400 occurring in the Gunnison Basin, Colorado, population (Gunnison Sage-grouse Rangewide Steering Committee 2005). The only known Gunnison sage-grouse population in Utah occurs in San Juan County, Utah, near the town of Monticello. The Monticello, Utah, and the Dove Creek, Colorado, populations are now treated as one population due to genetic similarities and close geographical proximity (Gunnison Sage-grouse Rangewide Steering Committee 2005).

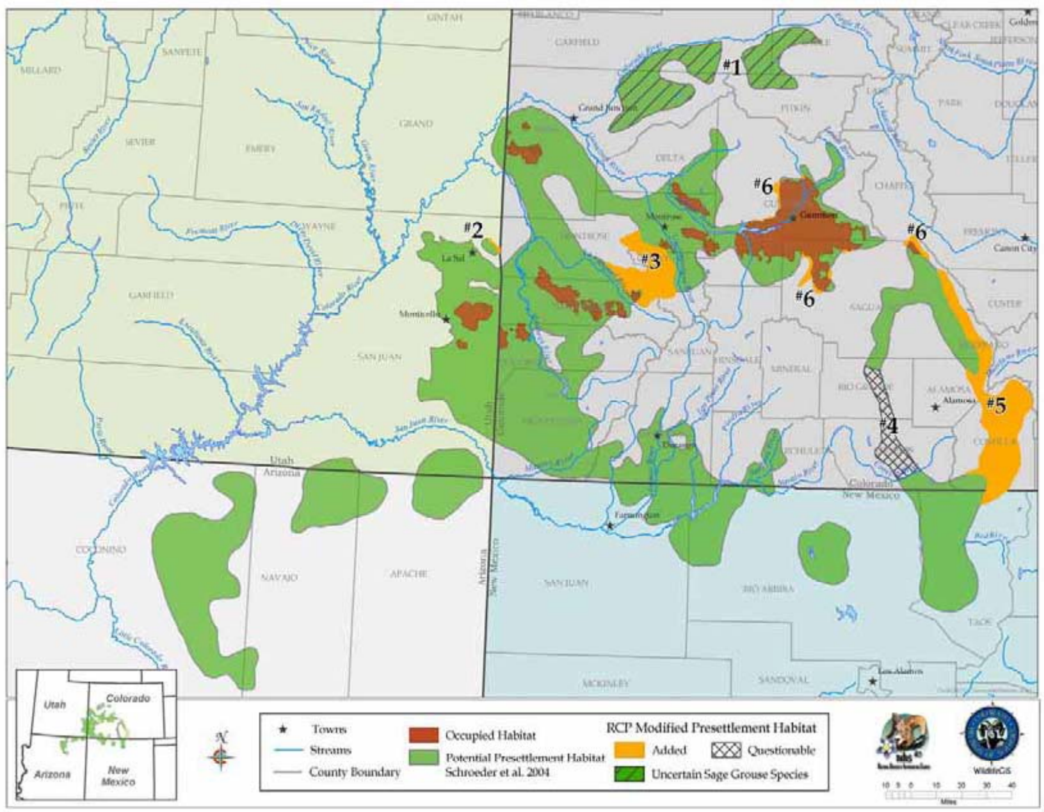

Figure 1.1. Current and historical Gunnison sage-grouse (*Centrocercus minimus*) range (Gunnison Sage-grouse Rangewide Steering Committee 2005).

Species Status and Conservation

Gunnison sage-grouse are considered a species of special concern for management purposes because the rapid decline in the species distribution and abundance

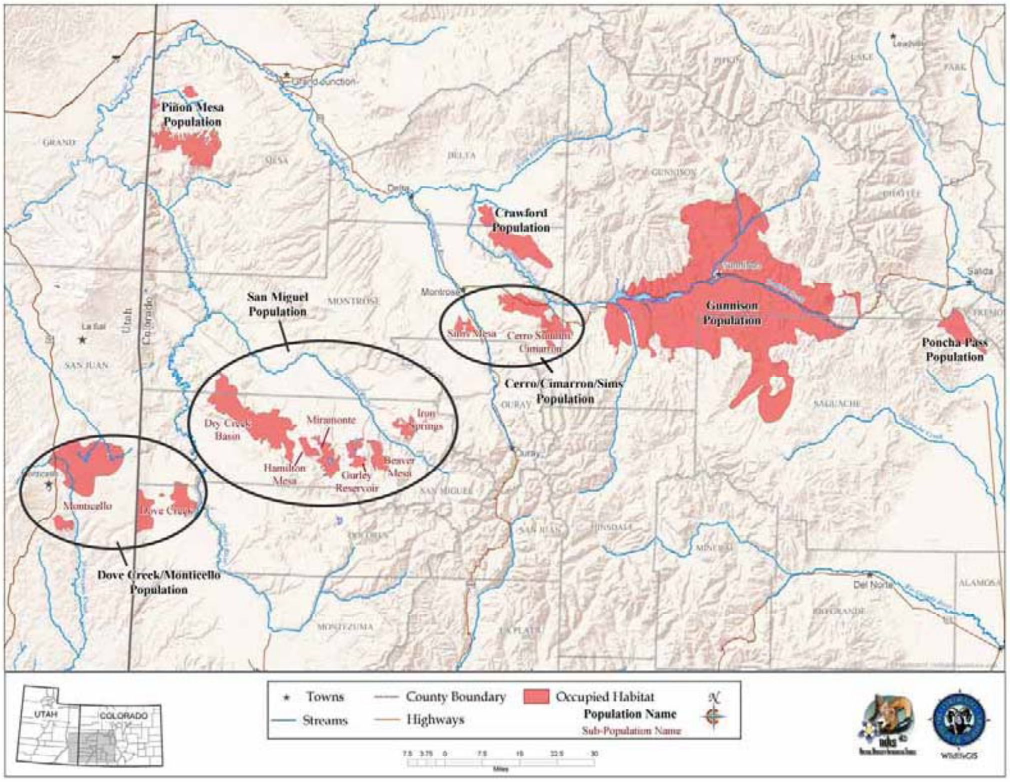

Figure 1.2. Locations of current Gunnison sage-grouse (*Centrocercus minimus*) populations. The discontinuity in occupied habitat at the state line in the Dove Creek/Monticello area is where there is an abrupt change from occupied habitat on the Colorado side to cropland on the Utah side of the border (Gunnison Sage-grouse Rangewide Steering Committee 2005).

has caused the remaining populations to be unusually small and isolated (Oyler-McCance et al. 2005). Identified potential threats to the Gunnison sage-grouse include low genetic diversity, genetic drift from small population sizes, habitat loss, degradation and fragmentation, impacts of drought, predator communities, and the interactions of all these threats (Gunnison Sage-grouse Rangewide Steering Committee 2005). The greatest threat is the loss, fragmentation, and degradation of sagebrush habitats because of urban development and conversion.

Concern about the small population sizes began in the 1990's. In 1995, before the separation of the sage-grouse into two separate species, the first local working group had formed in the Gunnison Basin of Colorado, with a conservation plan created in 1997 (Gunnison Sage-grouse Rangewide Steering Committee 2005). The formation of local working groups and conservation plans for the other populations soon followed. The San Juan County Gunnison Sage-grouse Local Working Group (SWOG) was formed in 1996 with the purpose of implementing management strategies that would conserve the local population (SWOG 2000). SWOG completed the San Juan County Gunnison Sage-grouse Conservation Plan (SJCCP) in 2000. The local working group in Dove Creek, Colorado published a local conservation plan in 1997 with the same purpose.

Continued concerns lead environmental groups to petition the United States Fish and Wildlife Service (USFWS) in January, 2000 to list the Gunnison sage-grouse as endangered (Gunnison Sage-grouse Rangewide Steering Committee 2005). In March, 2000 the USFWS designated the Gunnison sage-grouse as a candidate species for threatened or endangered species status. Under this designation the status of the species was reviewed annually to determine if a listing was still warranted and to determine its listing priority. In 2006 the USFWS ruled to remove the Gunnison sage-grouse from the Candidate Species list.

In 2005, the Gunnison Sage-grouse Rangewide Steering Committee produced the Gunnison Sage-grouse Rangewide Conservation Plan (RCP) to help guide local working groups (Gunnison Sage-grouse Rangewide Steering Committee 2005). In 2006, SWOG merged with the Dove Creek, Colorado, local working group to form the Monticello/Dove Creek Local Working Group (LWG). The merger took place in response to treatment of sage-grouse in Dove Creek and Monticello as one distinct subpopulation in the RCP (Gunnison Sage-grouse Rangewide Steering Committee 2005).

Gunnison Sage-grouse Rangewide Conservation Plan (RCP)

The Gunnison sage-grouse Rangewide Conservation Plan (RCP) was published in 2005 by the Gunnison sage-grouse Rangewide Steering Committee to serve as a guide to aid in the Gunnison sage-grouse conservation efforts (Gunnison sage-grouse Rangewide Steering Committee 2005). The RCP is the first up-to-date and rigorous assessment of the rangewide population and habitat data for the Gunnison sage-grouse. The RCP is intended to supplement local plans and offer a rangewide perspective to help ensure that the cumulative result of conserving local populations is in fact conserving the species. One of the guiding principles of the RCP is to create a plan that will be flexible enough to incorporate Gunnison sage-grouse research findings and successful management practices into conservation actions.

San Juan County Gunnison Sage-grouse Conservation Plan (SJCCP)

The San Juan County Gunnison Sage-grouse Working Group (SWOG) was formed in 1996 to identify and implement community-based conservation strategies to reverse the decline in the Gunnison sage-grouse population in San Juan County, Utah (SWOG 2000). The purpose of SWOG was to develop a conservation plan that could be implemented by state and federal wildlife resource agencies, private landowners, and local governments. Implementation of the San Juan County Gunnison Sage-grouse Conservation Plan (SJCCP) helped ensure local ownership of future management and land use decisions, and respect for private property rights.

The SJCCP was initiated to conserve the species by reducing threats, stabilizing populations, and maintaining ecosystems. It was committed to conserving and enhancing Gunnison sage-grouse populations that occurred on privately owned land in the county and to contribute to the economic viability of farms, ranches and the local community. The SJCCP identified conservation strategies that have been and will continue to be implemented by private and public partners to restore Gunnison sage-grouse habitats and populations. The plan's primary purpose was to conserve the species by implementing voluntary conservation actions.

The SJCCP contained two main parts: Habitat Conservation Assessment and Conservation Strategies. The Habitat Conservation Assessment described SWOG's current understanding about the status of the Gunnison sage-grouse distributions, habitat conditions, and factors that may be affecting the county's population. The Conservation Strategies identified goals and objectives, conservation actions, implementation schedules and responsibilities, evaluation guidelines, and monitoring requirements. The SJCCP was designed to be an adaptive document, capable of being updated with new information, identified issues, and ongoing management and research activities conducted in the county to guide future implementation.

Utah Population Status

The historic range and population size of the Utah population of the Gunnison Sage-grouse is not well documented (Gunnison Sage-grouse Rangewide Steering Committee 2005). Prior to 1968 there is no known written documentation of Gunnison sage-grouse in the Monticello area, but personal accounts of sage-grouse observations from long-time residents indicate that the sage-grouse range extended considerably farther in all directions than the currently occupied area (Fig. 1.3). The Gunnison sage-grouse occur primarily on private land and population declines in the county coincided with land use changes. Lek counts and population monitoring began in 1968. Since 1968, three active leks have been converted from sagebrush to crops or grazed pastures (Gunnison Sage-grouse Rangewide Steering Committee 2005). The number of birds on these leks declined rapidly and the leks were eventually abandoned (Fig. 1.4). In 2003, the population was estimated to be between 100-120 individuals (Gunnison Sage-grouse Rangewide Steering Committee 2005).

Land use in this area changed between 1984 and 1998. These land use changes included declines in non-irrigated agricultural land, black sagebrush (*Artemisia nova*), water areas, pinyon-juniper (*Pinus* spp., *Juniperus* spp.) and big sagebrush (*A. tridentata*), and conversion of land to Conservation Reserve Program (CRP) fields.

In 1997, SWOG designated a Gunnison sage-grouse priority conservation area northeast of the town of Monticello (Fig. 1.5, SWOG 2000). The Conservation Area (CA) contains 1,392,812 ha, 38% (127,170 ha) of which are privately owned. The CA was identified by encompassing historic and current lek sites, potentially suitable sage-grouse habitat, and sage-grouse observations. Within the CA, SWOG identified a Core Conservation Area (CCA) that consists of 136,249 ha, of which 89% (88,420 ha) are privately owned. Within the CCA, a Conservation Study Area (CSA) has been identified. The CSA consists of 24,177 ha, over 93% (22,556 ha) of which is privately owned. The CSA contains the year-round range of the population.

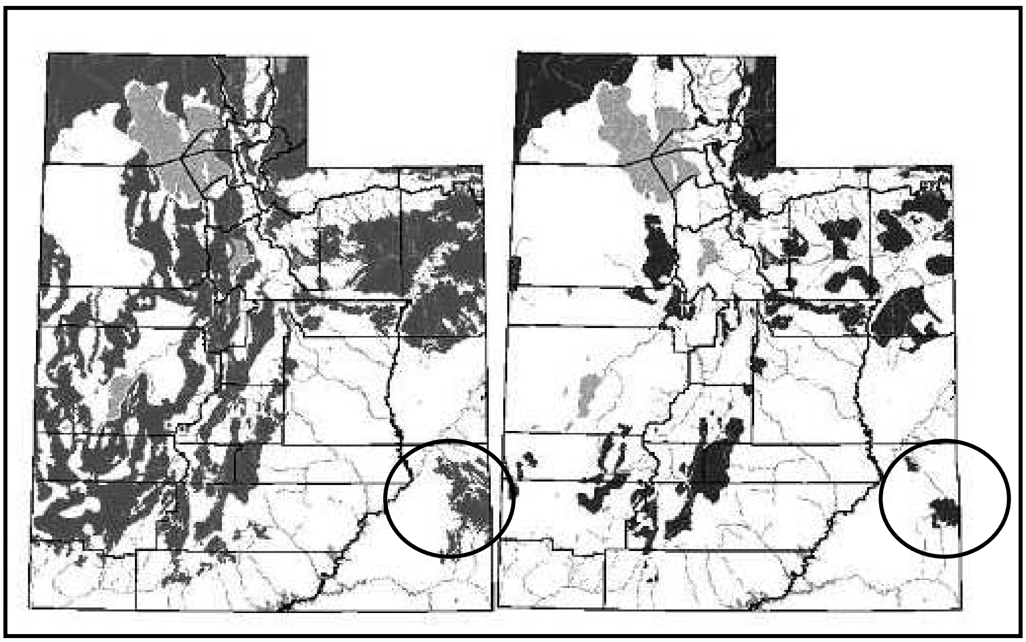

Figure 1.3. Historic (left) and current (right) distribution of Greater sage-grouse (*Centrocercus urophasianus*) and Gunnison sage-grouse (*C. minimus*) in Utah (Beck et al. 2003). Gunnison sage-grouse distributions in San Juan County, Utah are circled.

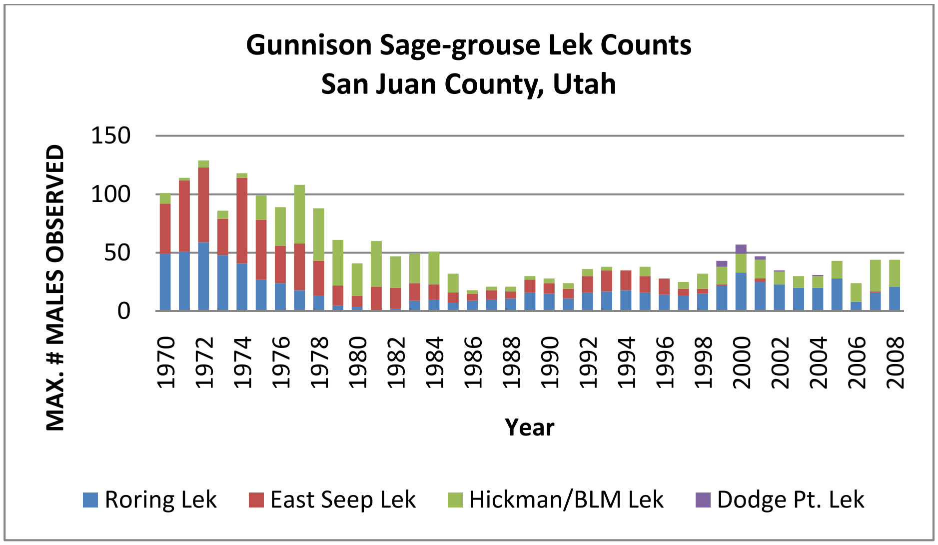

Figure 1.4. Gunnison Sage-grouse (*Centrocercus minimus*) lek counts from San Juan County, Utah. Maximum number of males observed is recorded. Data from Hickman and BLM leks have been combined because of daily movements of males between these two leks (SWOG 2008).

Ecology of the Utah Population

Intensive monitoring of radio-collared Gunnison sage-grouse and their habitats began in 2001 to initiate the process of implementing the SJCCP (SWOG 2003). These studies provided SWOG with information on the basic population ecology and dynamics, habitat use, and the response of the population to management actions. These were the first studies conducted on the Monticello, Utah, population.

Lupis (2005) investigated the movement and habitat use patterns, nesting, brood-rearing and summer habitat use, and factors that might be limiting the San Juan County population in a study conducted from March to September of 2001 and 2002. The objectives of the study were to: 1) Identify and evaluate nesting and brood-rearing habitat

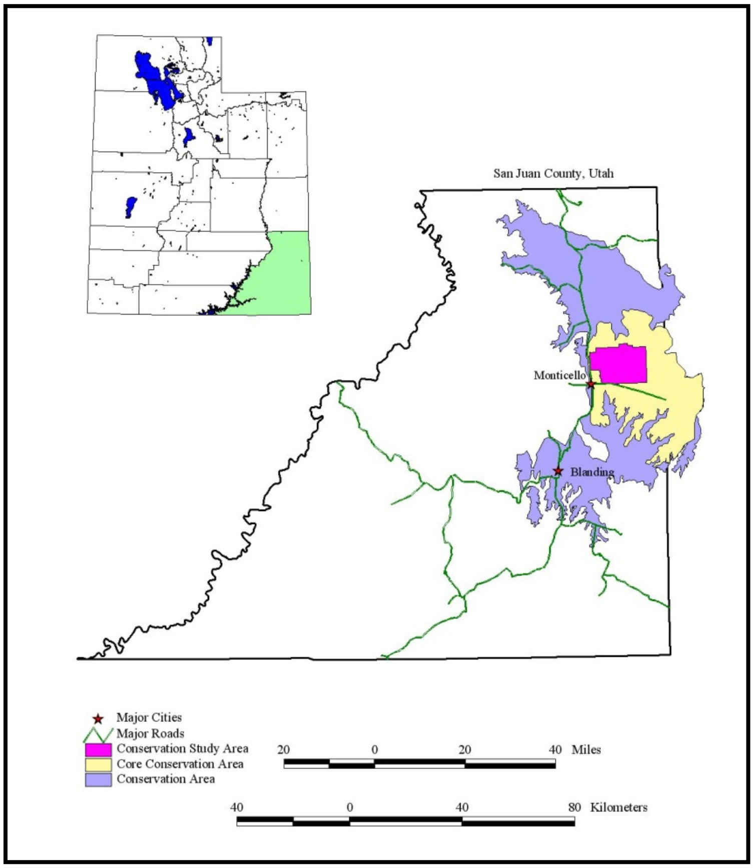

Figure 1.5. Gunnison sage-grouse (*Centrocercus minimus*) Conservation Area, San Juan County, Utah (Lupis 2005).

used by radio-collared hens; 2) Identify and evaluate summer habitat used by radio-collared males and broodless hens; 3) Assess movement patterns, reproductive success, survival, and mortality for radio-collared grouse; and 4) Determine use of CRP lands by Gunnison sage-grouse and their response to management practices. The information gained was compared to that of other Gunnison sage-grouse populations.

Ward (2007) conducted a study from 2002-2004 to determine: 1) reproductive success, survival, and mortalities of Gunnison sage-grouse in San Juan County; 2) nesting and brood-rearing success for Gunnison sage-grouse hens; 3) winter habitat use of Gunnison sage-grouse; and 4) arthropod abundance and diversity related to vegetative composition at nest (sagebrush) and potential brood-rearing sites (CRP lands) for Gunnison sage-grouse hens.

Nesting. - Three nests, located 0.48 km to 3.3 km from the nearest active lek site, were monitored (Lupis 2005). All nests successfully hatched some eggs between 21-23 May, with clutch sizes ranged from 6-10 eggs. Using background research from other populations in combination with the hatch dates from this study, nest initiation was estimated to occur between 25-27 April, with peak mating occurring between 14-16 April. All nests were laid under sagebrush, with one hen nesting in black sage (*A. nova*) and two nesting in CRP/grassland. The dominant shrub at nest sites was big sagebrush (*A. tridentata*), the dominant grass was crested wheatgrass (*Agropyron cristatum*), and the dominant forb was alfalfa (*Medicago* spp.). The height of the nest bush ranged from 21-22 cm.

The SJCCP identifies the breeding complex as all land within two miles of a known lek site (SWOG 2000). The desired vegetation conditions include a canopy cover of 20-40% big sagebrush with an average height of 40 cm, a 30% minimum grass canopy cover, and a 10% minimum forb canopy cover. From 2000-2001 the mean percentages of vegetation cover types at monitored nest sites included 27.5% shrubs, 6% grass and 0.5% forbs (Lupis 2005, Ward 2007). Reference sites randomly selected in black sagebrush and CRP/grassland cover types were composed of 10.4% shrubs, 34.7% grass and 8.8% forbs. From 2003-2004 the mean percentage of vegetation cover types at monitored nest sites included 42.9% shrubs, 2.7% grass and 1.4% forbs, compared to reference sites with 36.8% shrubs, 0.7% grass and 0.4% forbs (Ward 2007).

Brood-rearing. - Two radio-collared hens with broods and one uncollared hen with a brood were monitored for approximately 90 days post-hatch (Lupis 2005). One hen fledged two or three chicks (a final count was unattainable), one hen fledged two chicks, and one hen failed to fledge any chicks. The two broods moved a distance of 2.7 km and 3.0 km from the nest site, with home range size ranging from 3.03 km^2 to 3.54 km^2. The hen with no brood had a home range of 12.6 km^2. Broods preferred CRP/grassland and big sagebrush habitat to any other cover type, such as black sage, bare ground, and grazed lands with little vegetation. Brood locations supported more forb cover, and less grass and shrub cover than reference sites.

The SJCCP identifies the need to establish and maintain a canopy cover of 20-40% big sagebrush, 30% minimum grass canopy cover, and a 10% minimum forb canopy cover in brood-rearing areas (SWOG 2000). In 2001, brood location sites consisted of 6.1% shrubs, 14.8% grass and 9.5% forbs with an average height of 18.8 cm (Lupis 2005, Ward 2007). In 2002 the vegetation characteristics of brood locations consisted of 2.8% shrubs, 5.7% grass and 1.7% forbs with an average height of 12.2 cm. The percent cover types for reference sites consisted of 10.4% shrubs, 34.7% grasses, and 8.8% forbs.

From May to August of 2003 and 2004 female Gunnison sage-grouse were monitored to determine nest site selection and nest success (Ward 2007). Vegetation characteristics and arthropod abundance and diversity were collected in sagebrush cover types and compared with random CRP sites. The CRP fields yielded a greater forb and grass cover than other habitats. Seventy-five percent of the bird habitat use locations and 60% of the total number of arthropods collected were in CRP fields. A larger number of arthropod families were found within CRP fields than other habitats. A higher number of arthropods were collected in 2004 than 2003 possibly because the higher amount of precipitation that year contributed to more vegetation growth. In San Juan County CRP appears to serve as a substitute habitat for arthropod populations in lieu of irrigated pastures, and wheat and bean fields. It now appears to provide critical seasonal use for grouse. Because of this, it has become a conservation priority for continued enrollment and management of current CRP fields and the enrollment of other fields in the program.

Males and Broodless Hens. - Radio-collared males and hens without broods used similar habitats to those utilized by hens with broods described above (Lupis 2005). Males remained within 3.6 km of the lek of capture and selected CRP/grassland and big sagebrush habitats in preference to the other cover types available. Broodless hens selected woodlands, CRP/grasslands and rangelands, and remained within an average of 4.4 km of the lek of capture, but one hen moved a distance of 7.4 km. Birds captured on the Hickman Flat lek were found in mixed-sex flocks of two to eighteen individuals. Birds captured on the Roring lek remained in single-sex flocks of one to sixteen individuals.

Winter. - In the winter of 2002-2003 the Gunnison sage-grouse used black sagebrush and big sagebrush with a canopy cover of 15-25% more than expected based on availability (Ward 2007). In the winter of 2003-2004 black sagebrush and big sagebrush within CRP were selected in greater proportion based on availability. Shrub height at bird locations ranged from 17.8-91.4 cm. The ideal combination appeared to consist of black sage intermixed with patches of Wyoming big sagebrush. Black sage only occupies 7% of the eastern portion of the area occupied by the population and it was discovered that the majority of the radio-collared birds moved to the eastern side of the study area to winter in the black sage area. The distance traveled between summer and winter habitats for adult grouse ranged from 0.3 to 5.6 km for males, and 2.5 to 8.2 km for females. Average winter home range for males was 2.5 km^2 and 3.0 km^2 for females. Flock sizes were found to be between two and thirty plus individuals.

Suitable winter habitat appears to be limited in the area occupied by the Gunnison sage-grouse population. Because of this, conservation efforts should be directed at preserving and enhancing the remaining black sage patches and establishing additional areas of Wyoming big sagebrush and black sage within CRP fields throughout the study area. The SJCCP calls for the establishment of vegetation conditions on 50% of the areas within the CSA and 25% of the buffer area around the CSA (SWOG 2000). The desired conditions stated within the plan consist of a minimum 15% canopy cover of big sagebrush averaging a height of 30 cm on south and west facing slopes interspersed with small areas of dense big sagebrush with a canopy cover of 40% and an average height of 40 cm. Drainages should support a minimum canopy cover of 30% big sagebrush with an average height of 50 cm.

STUDY PURPOSE

This study addressed three conservation strategies identified in both the RCP and SJCCP. The first conservation strategy addressed was the creation or enhancement of brood-rearing habitats. I attempted to create or enhance brood-rearing habitats using irrigation and dormant season cattle grazing. My objective was to evaluate the effect of irrigation and dormant season cattle grazing of CRP fields and native sagebrush fields on sage-grouse productivity potentials as measured by changes in vegetation composition and structure, arthropod diversity and abundance, and bird use.

The second conservation strategy I addressed was the assessment of vegetation conditions and habitat quality of current and historical Gunnison sage-grouse habitats in Utah. My objective was to collect vegetation data in occupied and potential habitats as identified in the RCP and SJCCP to assess the status of existing and potential Gunnison sage-grouse habitat in the CA. Managers will be able to use this information to quantify the relative contribution of occupied and potential habitats to the overall RCP goals. This information can also be used to update the current SJCCP and the information in the RCP and prioritize conservation efforts.

The RCP and SJCCP also identified the presence of man-made vertical structures such as power poles and fence lines as a threat to Gunnison sage-grouse conservation. Connelly et al (2000) reported that vertical structures in areas occupied by sage-grouse provide raptors and corvids with new perches that could result in increased predation on adults, chicks, and nests. The RCP and SJCCP recommended as a conservation strategy that power poles within areas occupied by Gunnison sage-grouse be fitted with deterrents to discourage perching by potential sage-grouse avian predators. However, little information was currently available regarding the efficacy of commercially available perch deterrents. To address this management need, I evaluated the effectiveness of five types of perch deterrents in the reduction or prevention of corvid and raptor perching events on poles of a power distribution line with the objective of determining if raptor or corvid use of the distribution line differed by perch deterrent type and/or control.

LITERATURE CITED

AOU Checklist Committee. 2002. Forty-second supplement to the American Ornithologists' Union checklist of North American Birds. The Auk 117: 847-858.

Beck, J.L., D.L. Mitchell, and B.D. Maxfield. 2003. Changes in the distribution and status of sage-grouse in Utah. Western North American Naturalist. 63:203–214.

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulleting 28(4): 967-985.

Gunnison Sage-grouse Rangewide Steering Committee. 2005. Gunnison Sage-grouse rangewide conservation plan. Colorado Division of Wildlife, Denver, Colorado, USA.

Lupis, S.G. 2005. Summer ecology of Gunnison Sage-grouse (*Centrocercus minimus*) in San Juan County, Utah. Thesis, Utah State University, Logan, USA.

Oyler-McCance, S.J., J. St. John, S.E. Taylor, A.D. Apa, and T.W. Quinn. 2005.

Population genetics of Gunnison Sage-grouse: implications for management. Journal of Wildlife Management 69: 630-637.

San Juan County Gunnison Sage-grouse Working Group (SWOG). 2000. Gunnison Sage-grouse Conservation Plan, San Juan County, Utah, USA.

San Juan County Gunnison Sage-grouse Working Group (SWOG). 2003. Gunnison Sage-grouse Conservation Plan Update, San Juan County, Utah, USA.

San Juan County Gunnison Sage-grouse Working Group (SWOG). 2008. Gunnison Sage-grouse Annual Report, San Juan County, Utah, USA.

Schroeder, M.A., J.R. Young, and C.E. Braun. 1999. Sage grouse (*Centrocercus urophasianus*). Pages 1-28 *in* Poole and F. Gill, editors. The birds of North America, No. 425. The Birds of North America, Philadelphia, Pennsylvania, USA.

Schroeder, M.A., C.L. Aldridge, A.D. Apa, J.R. Bohne, C.E. Braun, S.D. Bunnell, J.W. Connelly, P.A. Deibert, S.C. Gardner, MA. Hilliard, G.D. Kobriger, S.M. McAdam, C.W. McCarthy, J.J. McCarthy, D.L. Mitchell, E.V. Rickerson, and S.J. Stiver. 2004. Distribution of sage-grouse in North America. The Condor 106:363-376.

Ward, S. 2007. Gunnison sage-grouse winter ecology and arthropod abundance diversity in relation to habitat use in San Juan County, Utah. Thesis, Utah State University, Logan, USA.

Young, J.A., C.E Braun, S.J. Oyler-McCance, J.W. Hupp, and T.W. Quinn. 2000. A new species of sage-grouse (Phasianidae: *Centrocercus*) from southwestern Colorado. Wilson's Bulletin 112: 445-453.

CHAPTER 2

EFFECT OF IRRIGATION AND DORMANT SEASON CATTLE GRAZING ON VEGETATION DIVERSITY AND ARTHROPOD ABUNDANCE IN CONSERVATION RESERVE PROGRAM AND NATIVE SAGEBRUSH IN SAN JUAN COUNTY, UTAH

ABSTRACT Gunnison sage-grouse (*Centrocercus minimus*) populations currently occupy 4,787 km^2 (8.5% of the original range) in Colorado and Utah. Declining populations are characterized by reduced recruitment attributed to breeding habitat (lekking, nesting, and brood-rearing) loss and fragmentation. Increased availability of forbs and arthropods in brood-rearing habitats has been positively associated with survival and recruitment of sage-grouse chicks. Concomitantly, the Gunnison Sage-grouse Rangewide Conservation Plan (RCP) and the San Juan County Gunnison Sage-grouse Conservation Plan (SJCCP) identified protection and enhancement of mesic brood-rearing habitats as a priority conservation strategy.

From 2007-2009, I evaluated Gunnison sage-grouse use, vegetation and arthropod responses to irrigation and dormant season cattle grazing on 32 randomly selected 0.1 ha plots, with 12 plots located in agricultural lands enrolled in the Conservation Reserve Program (CRP) and 12 plots in a native sagebrush area. Specifically, I evaluated the role of irrigation and dormant season cattle grazing in creating mesic wet meadow environments and their effect on habitat quality as measured by changes in vegetation structure and composition, arthropod abundance and diversity, and sage-grouse use. Vegetation in the irrigated plots remained greener longer through the growing season

than in the non-irrigated plots, but vegetation diversity did not differ (p>0.01). Overall, the CRP plots exhibited greater arthropod abundance and percent cover of perennial grass than the native sagebrush plots, but lower diversity of perennial grasses and abundance and diversity of forbs (p<0.01). Crested-wheatgrass (*Agropyron cristatum*) was the dominate vegetation in CRP and may have out-competed native forbs. Dormant season grazing of CRP did not have a positive or negative effect on crested wheatgrass cover. Lastly, I did not detect any increased sage-grouse use of the treatment plots.

The sprinkler irrigation system used in this study allowed quantification of water application rates leaving the nozzle but not actual application rates because of frequent winds that resulted in non-uniform plot coverage and increased evaporation. Thus, creation of mesic areas in brood-rearing habitats may best be accomplished by a system of terraces, ditch plugs or small check dams that retain moisture longer, and by providing flood irrigation. To increase forb and grass diversity in CRP, managers should evaluate the use of mechanical treatments, coupled with spring grazing, and reseeding to mitigate the potential competitive effects of crested wheatgrass.

INTRODUCTION

Connelly et al. (2000) identified several factors contributing to the continued decline of sage-grouse (*Centrocercus* spp.) populations range-wide. Of these, the loss, degradation, and fragmentation of the sagebrush (*Artemisia* spp.) ecosystem remain paramount. As sagebrush obligates, sage-grouse require this habitat type to complete their life cycle. The structure and composition of plant communities within sagebrush ecosystems influence sage-grouse nesting, breeding, brood-rearing, fall, and winter habitat selection.

Gunnison sage-grouse (*C. minimus*) currently occupy 4,787 km^2 (8.5% of their original range) in Colorado and Utah. The Gunnison Sage-grouse Rangewide Conservation Plan (RCP) and the San Juan County Gunnison Sage-grouse Conservation Plan (SJCCP) recommend management strategies to conserve the species by restoring impacted habitats (SWOG 2000, GSRSC 2005). Both plans identified the lack of brood-rearing habitat as limiting sage-grouse productivity and recommended the creation of mesic areas for broods as a priority conservation strategy.

Sage-grouse Brood-rearing Habitats

Good brood-rearing habitat includes areas with an abundant diversity of forbs and insects high in calcium, phosphorus and protein, and the availability of herbaceous plant species during the late-growing season (Peterson 1970, Wallestad 1971, Klott and Lindzey 1990, Johnson and Boyce 1990, Sveum et al. 1998, Connelly et al. 2000, Crawford et al. 2004). The quality of brood-rearing habitats changes as summer progresses and food availability shifts. The habitats tend to become more xeric resulting in desiccation of forbs. Increased sage-grouse brood use of wet meadow areas has been related to the amount of desiccation occurring.

Wallestad (1971) documented the summer movements and habitats used by broods in central Montana. He observed that hens with broods occupied areas characterized by mixed sagebrush and open areas exhibiting succulent forbs and clumps of tall sagebrush for hiding and roosting cover. As the season progressed into late August and early September the broods shifted to areas where sagebrush was more common and dense. He concluded that large tracts of dense sagebrush appeared to have little value as sage-grouse brood habitat, even though it is essential as winter habitat. Peterson (1970) and Klott and Lindzey (1990) reported that an important component of juvenile sage-grouse habitat appears to be an abundance and diversity of forbs with sagebrush cover <20%. Broods used areas with less shrub cover than what was average for that habitat.

Crawford et al. (2004) suggested that the availability of forbs and invertebrates is positively associated with survival and recruitment of chicks. Johnson and Boyce (1990) conducted a study on captive-reared sage-grouse chicks and the influence of insect reduction in their diet on survival. They reported a correlation between the quantity of insects in the diet and chick survival and growth. Chicks less than 21 days old needed insects to develop and survive. All chicks hatched in captivity that were not given insects died between the ages of 4 and 10 days. Insects decreased in the diets of chicks >21 days of age but were still required for optimum development.

The diets of broods in Oregon included 34 genera of forbs and 41 families of invertebrates (Drut et al. 1994). Klebenow and Gray (1968) recorded weekly diet selection data for age classes of sage-grouse chicks from hatch until brood break up at eight to ten weeks of age. During the first week insects were predominant, composing 52% of the total diet. After the first week, insects decreased in importance but were still part of the diet. As insects decreased, forbs became the most important food source for chicks. At four weeks, as plants began to dry, sagebrush appeared in the diet in small amounts, progressively increasing as the season progressed and the availability of forbs decreased. Similar findings were also reported by Peterson (1970).

Drut et al. (1994) quantified the importance of forbs and invertebrates in sage-grouse productivity in Oregon. They reported higher productivity in a population where 80% of the dietary mass in chicks diets consisted of forbs and arthropods compared to another study area where chick diets consisted of 65% sagebrush. Sveum et al. (1998) suggested a brood that needs a larger home range due to limited availability of forbs may also have a lower survival rate than a brood using a smaller area exhibiting a greater abundance in forbs.

Gunnison Sage-grouse Brood-Rearing Habitat in Utah

In 1997, the San Juan County Gunnison Sage-grouse Local Working Group (SWOG) designated an area northeast of the town of Monticello, Utah, as a sage-grouse priority conservation area (SWOG 2000). The Conservation Area (CA) consisted of 1,392,812 ha, 38% (127,170 ha) of which is privately owned. The CA was identified by encompassing historic and current lek sites, potentially suitable sage-grouse habitat, and sage-grouse observations. Within the CA, SWOG identified a Core Conservation Area (CCA) that consisted of 136,249 ha, of which 89% (88,420 ha) is privately owned. Within the CCA, a Conservation Study Area (CSA) was identified. The CSA consisted of 24,177 ha, over 93% (22,556 ha) of which is privately owned. The CSA contains the year round range of the Utah population.

The SJCCP stated that the desired brood-rearing habitat conditions should include a canopy cover of 20-40% sagebrush with an average height of 40 cm, a minimum of 30% grass canopy cover, and a minimum of 10% forb canopy cover. The SJCCP further recommended that the height of the vegetation in wet meadow areas is to be greater than 10 cm between 15 June and 31 July on over 75% of the area considered to be brood-rearing habitat.

The Farm Program and Sage-grouse Conservation

Because over 90% percent of the habitat occupied by Gunnison sage-grouse in San Juan County is privately owned, the implementation of the Conservation Reserve Program (CRP) under the Food Security Act of 1985 was recognized by SWOG as a major species conservation action. The CRP is a voluntary program that provided financial incentives to encourage private landowners to retire cropland from agricultural production by establishing an approved permanent vegetation cover. During the period of the contract, the land could not be cultivated to produce an agricultural commodity. Haying and grazing were allowed on a case-by-case basis to mitigate the effects of drought on local livestock producers. The only techniques allowed to manage CRP fields are burning, spraying for noxious weeds, and mowing.

The Farm Security and Rural Investment Act of 2002 (Farm Bill) reauthorized the Environmental Quality Incentives Program (EQIP) to provide a voluntary conservation program for farmers and ranchers that promotes agricultural production and environmental quality. This program offered financial and technical help to assist eligible participants to install or implement structural and management practices on eligible agricultural land. This study used management practices within EQIP that could be employed by landowners in the CCA to enhance sage-grouse brood-rearing habitat through the creation of mesic environments. These environments could potentially increase forb cover and arthropod diversity in existing CRP fields and provide important seasonal habitats for sage-grouse broods.

Land Use Changes in San Juan County and Gunnison Sage-grouse

Gunnison sage-grouse population declines in San Juan County have coincided with land use changes. The population was at its highest in the 1970s and 1980s (SWOG 2003, Lupis 2005). During this period, the primary agricultural crops in the county were winter wheat (*Triticum* spp.) and dryland alfalfa (*Medicago* spp.). Many growers did not use herbicides or insecticides because of the slim profit margin in growing these crops (J. Keyes, Utah State University Extension, personal communication). These practices may have resulted in a greater arthropod abundance as a result of increased green vegetation and forb availability. During this period landowners also frequently reported observing flocks of grouse in their fields during harvest and post-harvest periods.

In the past, many landowners in San Juan County did not have automatic control valves on wells used to fill livestock water tanks (SWOG 2000). This would cause tanks to overflow, inadvertently creating mosaics of ephemeral wet meadow or mesic habitats below the tanks. These overflow areas were not grazed by livestock until late fall when the herds were moved to winter pasture. Landowners reported these holding corrals continually produced more forage, greened-up earlier, stayed greener longer than adjacent areas, and often supported sage-grouse broods. The SWOG believed this activity enhanced Gunnison sage-grouse productivity (SWOG 2000). But with more efficient watering devices the seasonal wet meadows disappeared. The SWOG believed that the loss of these wet meadow or mesic sites in brood-rearing areas could be a potential reason for low sage-grouse numbers and low recruitment because the quality and quantity of herbaceous cover has been reduced.

CRP and Sage-grouse

One of the most comprehensive land use changes to occur in the county was the conversion of thousands of hectares of cropland to CRP. Because of drought conditions many of these CRP fields had to be reseeded, and thus were devoid of vegetation for almost two years (G. Wallace, Utah Division of Wildlife Resources, personal communication). In the two years post-CRP signup the number of males counted on lek sites decreased by 50%.

In 1997 the habitat for the San Juan County population was designated as a priority conservation area for the species (Lupis 2005). This designation increased the amount of land that could qualify as CRP, adding an additional 150 km^2 of land enrolled in the program (Fig. 2.1). However, based on lek counts, the San Juan County population is at a historic low with a 2004 population estimate of 155 to 174 birds (SWOG 2005). Research suggested that CRP habitats appear to provide the greatest arthropod abundance

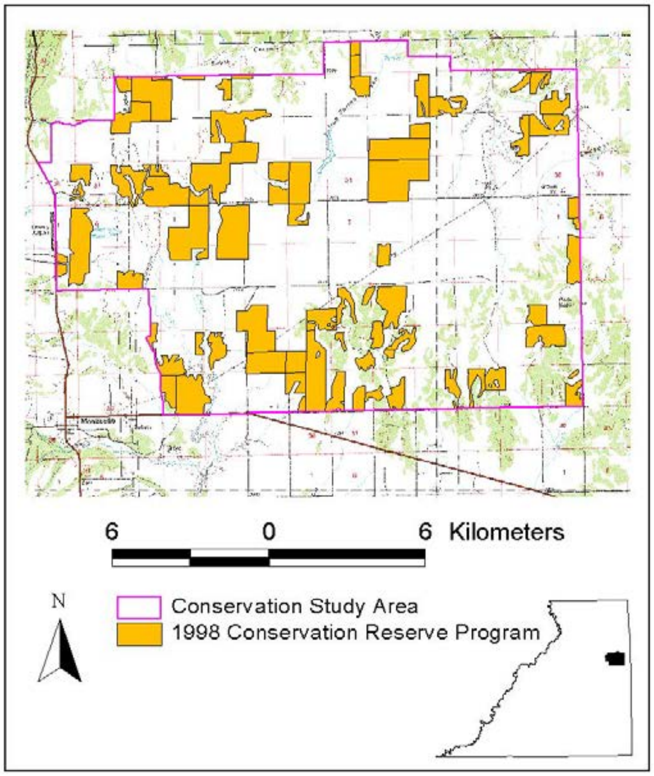

Figure 2.1. Agricultural lands enrolled in the Conservation Reserve Program under the conservation priority initiative in the Conservation Study Area, San Juan County, Utah (Lupis 2005).

(Lupis 2005, Ward 2006). These CRP fields are also preferred over other cover types during the brood-rearing period (Lupis et al. 2006).

Beginning in late 2001, San Juan County experienced a major drought. In response to drought conditions, the FSA opened CRP for late season grazing. Grazing was allowed on several CRP fields in the CSA. Lupis (2005) investigated the effects of domestic livestock grazing of the CRP fields on the movement patterns of Gunnison sage-grouse.

Three males, 2 broodless hens, and 1 hen with a brood were monitored before, during, and after grazing. Males avoided the grazed CRP fields during grazing and did not return after the livestock were removed. Two of the males were located within a CRP field during grazing 15-20% of the time, and 1 male was recorded in a grazed field 40% of the time. Broodless hens also avoided CRP fields during grazing to varying degrees. One hen was in a CRP field during grazing 78% of the time, 1 female 12.8% of the time and returned twice after livestock was removed, and 1 hen was never located within a CRP field during or after grazing. The monitored brood remained within the CRP field during grazing and successfully recruited 2 chicks into the fall population.

STUDY PURPOSE

This study addressed the RCP and SJCCP conservation strategy of evaluating methods to create or enhance brood-rearing habitats. The specific objectives of my research were to evaluate; 1) the role of irrigation in CRP and native sagebrush on sage-grouse habitat potentials as measured by changes in vegetation composition and structure, arthropod diversity and abundance, and bird use; and 2) the role of dormant season cattle grazing on these same potentials.

STUDY AREA

The study was conducted in San Juan County, located in the extreme southeastern corner of Utah (Fig. 2.2). The county is bordered by the Colorado River to the north and west, Arizona to the south, and Colorado to the east. The CSA is part of the Colorado Plateau Province and sits on the extensive Sage Plains tableland on the northeast side of the Abajo Mountains with an elevation between 2,042 m and 2,133 m (Olsen et al. 1962). The surface of the plateau consists of undulating to rolling, low hills of eolian deposits of variable thickness derived from sandstone over colluvium and/or residuum weathered from sandstone. The area is characterized by large grass pastures and agricultural fields interspersed with fragmented patches of Wyoming big sagebrush (*A. tridentate* spp. *wyomingensis*) and black sagebrush (*A. nova*). There are no perennial water sources on the plateau. The CSA consists of 95% privately owned land, most of which is currently enrolled in CRP. The remaining privately owned lands are used as rangeland pastures for cattle grazing or dryland farming.

Long term (1902-2009) precipitation and temperature for the CSA was summarized from local weather station data archived by the Utah Climate Center, Logan, Utah. Precipitation and temperature measurements for the study period (2007-2009) are summarized from data recorded on a portable weather station. The long-term average annual precipitation (1902-2009) in the study area was 39.55 cm, with most arriving from July to October in the form of rain. The mean annual high and low temperatures on the study area were 35.9° C and -21.2° C, respectively. From 2007-2009 the average annual precipitation on the study area was 30.23 cm, with average annual high and low temperatures of 37.5 ° C and -18.3 ° C, respectively.

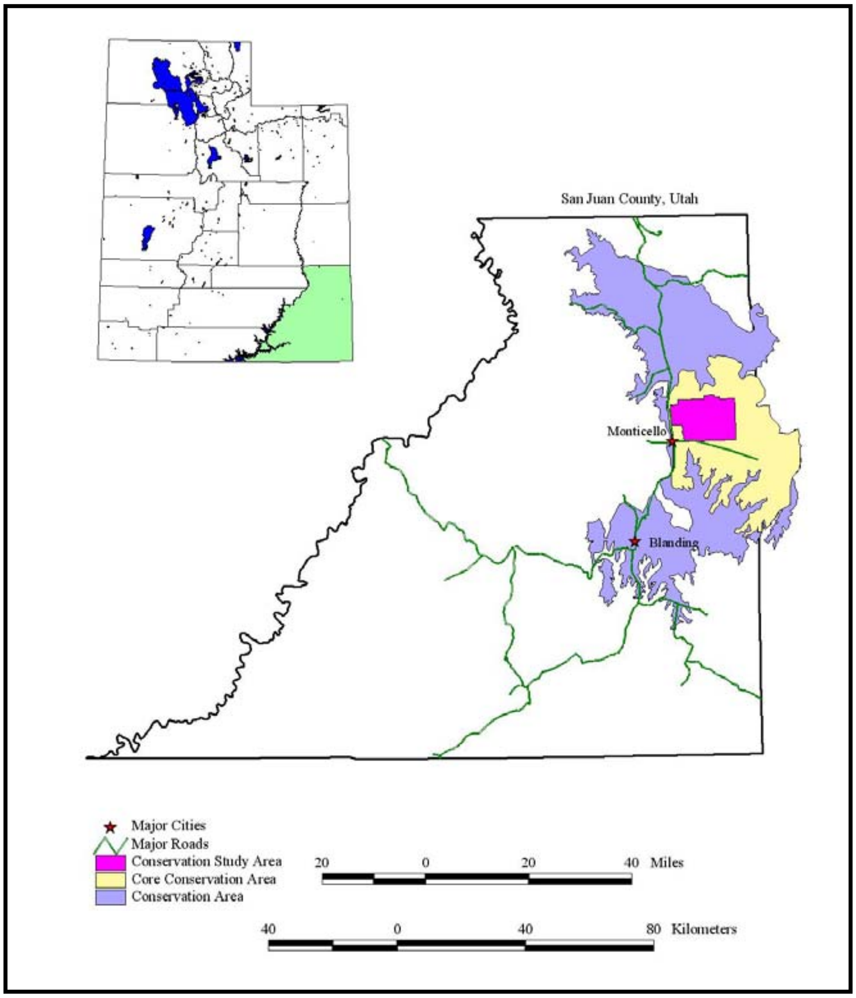

Figure 2.2. Gunnison Sage-grouse Conservation Area, San Juan County, Utah (Lupis 2005).

The CSA is relatively flat with elevations ranging from 2,065-2,149 m. The CSA is a mosaic of habitat types dominated by CRP/grassland and sagebrush cover types (SWOG 2000). The original seed mixture for the CRP fields and the plant species recorded within the CRP and sagebrush plots during this study can be found in

Appendices A and B, respectively. The dominant forb species recorded in the sagebrush plots were scaly globemallow (*Sphaeralcea leptophylla*), sulphur buckwheat (*Eriogonum umbellatum*), hairy golden aster (*Heterotheca villosa*), and cryptantha (*Cryptantha* spp.). Few plants from the original CRP seed mixture were found in the CRP plots. The dominant species in the CRP plots was crested wheatgrass (*Agropyron cristatum*) with occasional patches of Wyoming big sagebrush and rubber rabbitbrush (*Chrysothamnus nauseosus*). Dominant forb species within the CRP plots were Russian knapweed (*Centaurea repens*), African mustard (*Malcomia africana*), and Russian thistle (*Salsola pestifer*). Forbs within Wyoming big sagebrush patches in CRP plots were the same as those found in the sagebrush plots. Cheatgrass (*Bromus tectorum*) was present in both CRP and native sagebrush plots.

METHODS

Experimental Design

I identified one study site in a native sagebrush area and one study site in a CRP field, both sites within the CSA. I identified 16 0.1 ha plots in each study site. I arranged the plots in an experimental randomized block design that controlled for differences in vegetation and landscape topography that could affect the vegetation present at each plot. Each plot was considered a separate experimental unit. At each site, the plots were arranged in 4 blocks, with each block consisting of 4 plots (Fig. 2.3). Within a block, each plot was randomly assigned to one of the 3 irrigation treatments or control. Half of each plot was grazed by cattle. Vegetation transects and arthropod trapping grids were established in both halves of each plot to measure the effects of irrigation and irrigation combined with grazing. This layout resulted in four replications of each irrigation and grazing treatment and control in each habitat.

Irrigation

I evaluated 3 irrigation treatments base on application rates: once a week, every 2 weeks and every 3 weeks. Plots receiving no water served as reference or control sites. Plots were randomly assigned to each irrigation treatment or control within each block. Three groundwater wells in close proximity to the identified treatment plots were used to distribute water to each plot for irrigation. Treatment plots were irrigated with a Rain Bird sprinkler model 65PJ™ with a 30 meter spraying radius (Rain Bird Corporation, Azusa, CA). The treatment and control plots were established in the summer of 2007. Given that there were site-specific differences, we conducted tests before the study began to standardize the capacity of the pumps at each treatment plot. During this period I measured the amount of water distributed on each plot by time. These experiments allowed us to establish a standard rate of flow. Irrigation of the plots began in May 2007 and continued to the end of July. This time period coincided with peak nesting and brood-rearing periods (Lupis 2005, Ward 2006).

All plots were irrigated for an 8-hour period. Due to strong afternoon winds, the irrigation periods occurred in the early morning and evening. Each plot assigned to an irrigation treatment received 1.4 cm of water each irrigation period, the equivalent of the

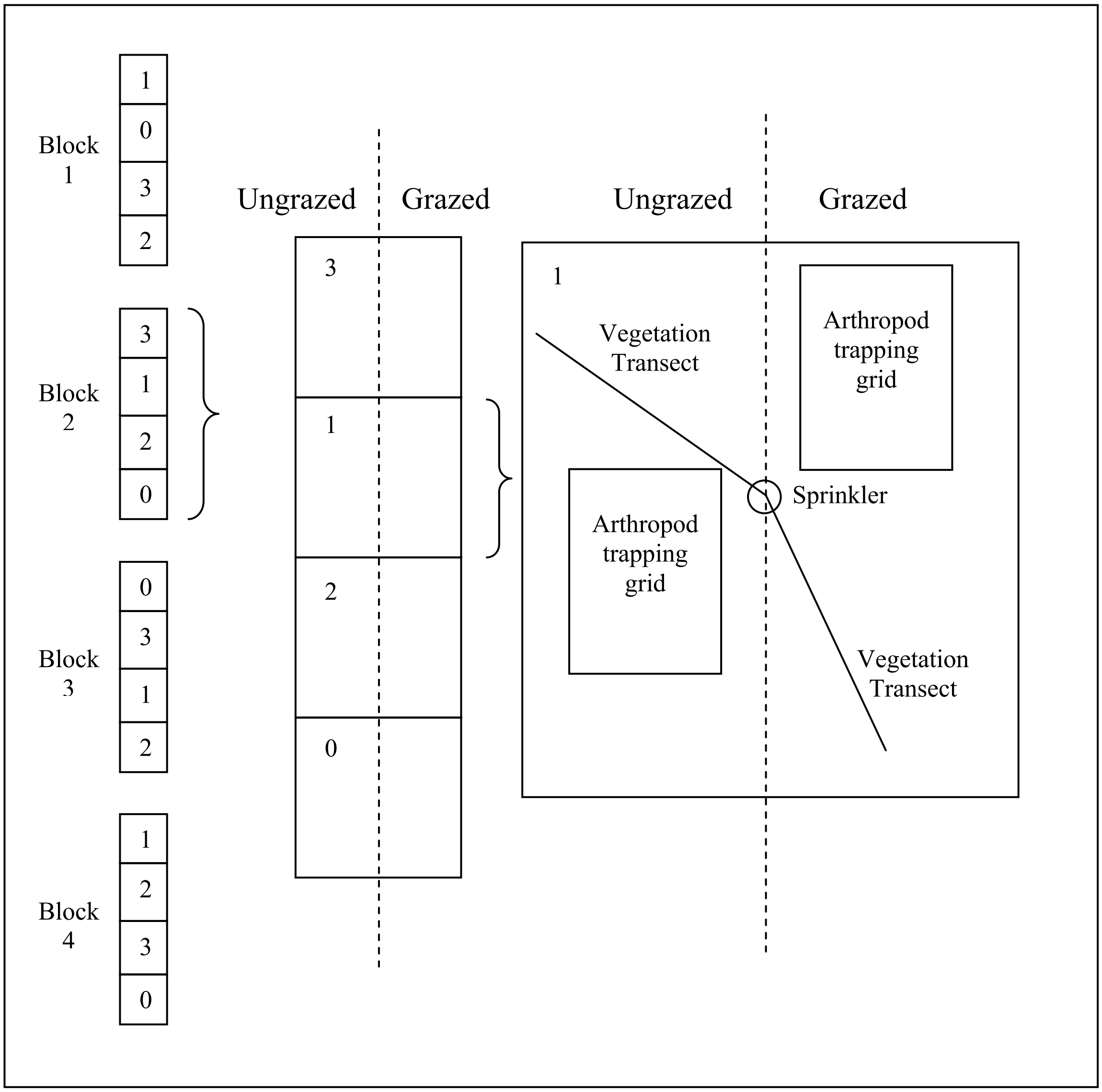

Figure 2.3. Experimental design schematic showing the layout of 4 blocks each containing 3 plots randomly assigned to watering treatments (once a week, every 2 weeks, every 3 weeks) and 1 control (no water) plot, ungrazed and grazed treatments, and location of arthropod trapping grid, vegetation transects, and rain-bird sprinkler. Each 4 block layout occurs within the Conservation Reserve Program and native sagebrush study sites. San Juan County, Utah, 2007-2009.

long term average precipitation this area receives in the months of May, June, and July. Within both CRP and sagebrush sites, the plots that were irrigated weekly over the 7-week period received the equivalent of an additional 10.2 cm of water as measured on test gauges. The plots that were irrigated every two weeks or 4 times over the 7-week period received the equivalent of 5.1 cm of additional water. The plots irrigated every three weeks or 3 times over the 7-week period received the equivalent of 3.8 cm of additional water. The irrigation occurred in May-July of 2007, 2008, and 2009.

Vegetation Monitoring

I used the GSRSC Structural Vegetation Collection Guidelines (SVCG) to measure vegetation parameters (GSRSC 2007). At each site, each treatment and control plot contained one 30-meter vegetation transect. Transects were permanently marked with t-posts with the same transects used in consecutive years. Percent canopy cover of grasses, forbs, and shrubs was visually estimated by placing a Daubenmire frame every 3 m along each 30-m transect (Daubenmire 1959). The SVCG identified six cover classes based on the standardized Daubenmire method. The Daubenmire method lumped too much vegetation into the 5-25% class for the Gunnison sage-grouse vegetation variables, so it was into 2 cover classes. The canopy cover classes used in this study were: 0-5%, 5-15%, 15-25%, 25-50%, 50-75%, and 75-100% (GSRSC 2007).

One height measurement of sagebrush, forb, annual grass, and perennial grass was taken at each Daubenmire frame by selecting the plant closest to the lower left hand corner of the frame. If sagebrush was not found within the frame then the closest sagebrush within 10 m of the frame was used. If no sage was within 10 m of the frame it was marked as not present. Only forbs and grasses within the frame were used to measure height. If no forb or grass was within the frame the plant group was marked as not being present. Height and percent cover of grasses, forbs and shrubs was measured in early June and again the last week of July.

Vegetation was clipped and weighed to measure the forage production of each plot using a 0.5m x 1m frame. All vegetation within the frame was clipped, stored in paper bags, dried, and weighed. The vegetation was separated into the categories perennial grasses, annual grasses, and forbs. Vegetation was clipped along a 30 m transect radiating from the center sprinkler. Frames were placed every 3 m, resulting in 10 frames. A different transect was used each year to prevent clipping the same location more than once. The clipping transect did not overlap the permanent vegetation monitoring transect. Forage production was measured the last week of July.

Any uncertainties in identification of a plant species were documented with photos and pressings. The same survey method and transect lines were used during the collection of data in 2007, 2008, and 2009 to evaluate the effectiveness of treatment plots in increasing grass and forb abundance and diversity.

Arthropod Surveys

Terrestrial arthropods were sampled by using pitfall traps arranged in a pattern that allowed capture data to be used with DISTANCE software to estimate density of total arthropods and of individual taxa (Buckland et al. 2001, Lukacs et al. 2004, Graham et al. 2008). Pitfall traps in each plot were arranged to meet the assumptions of DISTANCE sampling, which are that all invertebrates on the center line are detected and that distances from the center line are accurately measured. Sixty pitfall traps were used at each plot in the arrangement shown in Fig. 2.4. This pattern was generated by using WebSim to simulate a hazard-rate model of invertebrate captures that resulted in estimates with small confidence intervals, and matched trapping results in a pilot study of invertebrate pitfall trapping in Colorado (Lukacs 2001, 2002; Graham et al. 2008).

Pitfall traps were placed by carefully measuring and marking correct locations with flags, then digging in the traps. Pitfall traps were constructed as described by New (1998). For each trap, a 1.5-liter plastic jar was buried below ground level and a 500-mL cup containing 125 mL of soapy water was placed in the cup (Graham et al. 2008). A 15-cm diameter funnel was placed over the jar, centered over the cup, with the top of the funnel at ground level. Each water treatment and grazing treatment plot in each of the sagebrush and CRP habitats contained a pitfall trapping arrangement with 60 traps. I sampled in early June, during the estimated first week after hatch for Gunnison sage-grouse nests in San Juan County, Utah (Lupis 2005, Ward 2006). The traps were opened in sequence and remained open for three days during a 7-day period. When closed, each trap was poured into a 150 mL sample container with the remaining space filled with 91% isopropyl alcohol to assure the sample was stored in a 70% isopropyl alcohol and water solution. Labels affixed to the outside of the sample containers recorded habitat, plot number, treatment, date, and trap number. Samples were stored at room temperature once they were returned to the lab.

In the lab, each sample was washed through a 0.5-mm mesh net (Graham et al. 2008). Everything remaining in the net was placed in a Petri dish. Arthropods were sorted to order. Taxa were identified following Triplehorn and Johnson (2005), and I followed the taxonomic nomenclature of this source. I collected 3,840 total samples each year in 2007, 2008, and 2009. Because of logistical constraints only 2,240 samples were sorted for each year resulting in 35 traps sorted for each trapping arrangement.

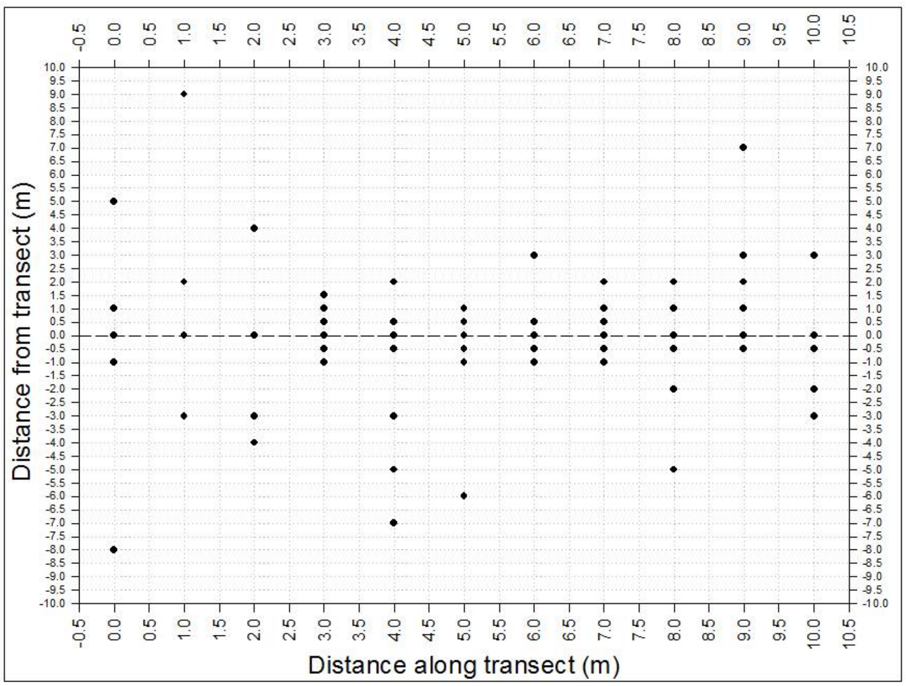

Figure 2.4. Arrangement of pitfall traps at each terrestrial arthropod sampling plot of the study, San Juan County, Utah, 2007-2009 (Graham et al. 2008).

Sage-grouse Pellet Counts

Pellet counts were used to survey sage-grouse use of the treatment and control sites in each habitat (Dahlgren 2005). I established 20 transects two meters apart in each plot. Information collected included pellet type (cecal or regular pellet) and number of pellets or cecal droppings per cluster. Roost piles were counted separately and equaled one cluster occurrence. Once a pellet was counted it was removed from the site to prevent double counting.

Dormant Season Cattle Grazing

Both CRP and sagebrush plots were grazed in November of 2006, 2007, and 2008. Utilization was measured using a paired-plot design. Utilization cages were randomly placed on the grazed portion of each plot, resulting in 32 cages (USDI-BLM 1996). After grazing, forage within each cage was clipped and weighed. A random uncaged plot was identified on the grazed side of the plot and forage within this plot was also clipped and weighed. The difference between the 2 weights equaled the amount of forage consumed. Random plots were also identified on the ungrazed sides of each site and forage was clipped and weighed to determine the amount of forage production. Annually, 60% utilization was achieved each fall grazing occurred.

DATA ANALYSIS

I used a 3-way factorial split-split plot design with whole plots arranged in randomized complete blocks with repeated measures to analyze habitat metrics. The whole block unit included 4 plots, one of each 3 irrigation treatments and a control. The whole block factor was whether the plot was CRP or sagebrush. The split plot unit was the individual plot. The split plot factor was the irrigation assignment. The split-split plot unit was half of each plot. The split-split plot factor was whether the half was grazed or un-grazed. The repeated measure unit was the individual plot. The repeated measure factor was the month the vegetation was measured (June or July).

I addressed the question: Did the vegetation and arthropod communities change in relation to water and grazing treatments in CRP and sagebrush plots in 2007, 2008, and 2009? The model I used compared the means among treatments and controls for the percent cover, height and forage production of perennial grasses, annual grasses, forbs and sagebrush, and arthropods observed in 2007, 2008, and 2009. I used a mixed model with an arcsine-square root scale (SAS Institute, Cary, NC). In essence, the statistical model was three-way in a randomized spatial block design, with plots grouped into spatial blocks to control for spatial heterogeneity in the landscape. Data analyses were conducted using the Mixed Procedure in SAS/STAT for Windows Version 9.1.3 (SAS Institute, Cary, NC).

RESULTS

Vegetation and arthropod responses to irrigation and grazing treatments are presented in a series of Tables in Appendix 3.

Vegetation Response

Vegetation cover results for 2007 are presented in Table A.3.1. Annual grass cover differed (p=<0.01) when analyzed by time, with more cover in June than July. When comparing habitat by time there was a difference (p=<0.01) between habitats. The CRP plots had visibly more cheatgrass than the sagebrush plots, but there was no difference between months within CRP over time with June cover of 5.0 % (SE=2.8) and

July having 4.9 % (SE=2.8). There was a slight difference over time within the sagebrush plots with an annual grass cover of 1.8% in June (SE=1.8) and 0.7 % in July (SE=1.3). Cover of forbs differed when analyzed by time (*p*=<0.01) with a greater cover of forbs in June (3.3 %, SE=0.7) than in July (0.8 %, SE=0.3). Forb cover also differed when analyzed by habitat and time (*p*=<0.01) with a greater cover of forbs in June than in July in both habitats. The sagebrush plots had a greater cover of forbs than CRP in both June (5.4 %, SE=1.2) and July (0.8 %, SE=0.7). The CRP plots had a forb cover of 1.7 % (SE=0.7) in June and 0.6 % (SE=0.4) in July. Cover of Wyoming big sagebrush differed by habitats (*p*=<0.01) with CRP plots having a cover of 0.2 % (SE=0.3) and a cover of 13.3 % in sagebrush plots (SE=2.0).

Vegetation cover results for 2008 are presented in Table A.3.2. Perennial grass cover was greater in the CRP plots (32.7 %, SE=3.7) than in the sagebrush plots (17.1 %, SE=3.0) (*p*=0.02). When analyzed by time, annual grass had a greater cover in June (6.3 %, SE=2.9) than in July (3.1 %, SE=2.2) (*p*=<0.01). Forb cover differed when analyzing habitat (*p*=<0.01) and habitat by time (*p*=<0.01) with more forb cover in the sagebrush plots (4.4 %, SE 0.8) than the CRP plots (0.2 %, SE=0.2) and a greater forb cover in the sagebrush plots in June 5.8 % (SE=1.0) than in July 3.2 % (SE=0.8). But within the CRP plots there was little difference between the June (0.2 %, SE=0.2) and July (0.3 %, SE=0.3) forb cover. When analyzed by habitat, Wyoming big sagebrush cover was greater in the sagebrush plots (9.6 %, SE=1.3) than the CRP plots (0.1 %, SE=0.2) (*p*=<0.01).

Vegetation cover results for 2009 are presented in Table A.3.3. Perennial grass cover was greater in the CRP (32.7 %, SE=3.7) than the sagebrush plots (17.1 %, SE=3.0) (*p*=0.02). When analyzed by time, annual grass had a greater cover in July (6.3 %, SE=2.9) than in June (3.1 %, SE=2.2) (*p*=<0.01). Forbs had a greater cover in the sagebrush (4.4 %, SE=0.8) than the CRP plots (0.2 %, SE=0.2) when analyzed by habitat (*p*=<0.01). When analyzing habitat by time forbs had a greater cover in the sagebrush plots in June (5.8 %, SE=1.0) than July (3.2 %, SE=0.8), and greater cover than the CRP plots in both months. Wyoming big sagebrush cover was greater in sagebrush (9.6 %, SE=1.3) than CRP plots (0.1 %, SE=0.2) (*p*=<0.01).

Vegetation height results for 2007 are presented in Table A.3.4. Perennial grass height differed by time (*p*=<0.01) and habitat (*p*=<0.01). Perennial grass was taller in June (13.2 cm, SE=1.1) than in July (9.0 cm, SE=0.9) and was taller in the CRP plots (16.4 cm, SE=1.4) than in the sagebrush plots (6.6 cm, SE=1.0). Annual grass was taller in June (4.0 cm, SE=0.5) than in July (2.3 cm, SE=1.2) (*p*=0.01). Forbs were taller in June (4.0 cm, SE=0.7) than in July (1.3 cm, SE=0.4) (*p*=<0.01). Wyoming big sagebrush was taller in the sagebrush plots (4.2 cm, SE=4.1) than in the CRP plots (3.6 cm, SE=1.7) (*p*=<0.01).

Vegetation height results for 2008 are presented in Table A.3.5. Perennial grass was taller in the CRP (24.7 cm, SE=2.5) than in the sagebrush plots (13.5 cm, SE=2.0) (*p*=0.01). Perennial grass was taller in the CRP plots that were not grazed (27.7 cm, SE=3.0) than in the CRP plots that were grazed (21.9 cm, SE=2.8) (*p*=0.02). Perennial grass taller in both habitats in July (21.2 cm, SE=1.7) than in June (16.5 cm, SE=1.6)

(p=<0.01). Annual grass was taller in June (3.6 cm, SE=1.2) than in July (2.0 cm, SE=0.9) (p=<0.01). Forbs were taller in the sagebrush plots (3.8 cm, SE=0.5) than in the CRP plots (0.4 cm, SE=0.2) (p=<0.01). Wyoming big sagebrush was taller in the sagebrush plots (43.1 cm, SE=5.0) than in the CRP plots (4.0 cm, SE=2.2) (p=<0.01) and was taller in June (21.5 cm, SE=3.0) than in July (17.9 cm, SE=3.0) (p=<0.01).

Vegetation height results for 2009 are presented in Table A.3.6. Perennial grass was taller in CRP (24.7 cm, SE=2.5) than sagebrush plots (13.5 cm, SE=2.0) (p=0.012) and was taller in July (21.2 cm, SE=1.7) than in June (21.2 cm, SE=1.6) (p=<0.01). Annual grass was taller in June (3.6 cm, SE=1.2) than in July (2.0 cm, SE=0.9) (p=<0.01). Forbs were taller in sagebrush plots (3.8 cm, SE= 0.5) than the CRP plots (0.4 cm, SE=0.2) (p= <0.01). Wyoming big sagebrush was taller in the sagebrush (43.1 cm, SE=5.0) than the CRP plots (4.0 cm, SE=2.2) (p=<0.01) and was taller in July (21.5 cm, SE=3.0) than in June (17.9 cm, SE=2.8) (p=<0.01).

Forage production results for 2007 are presented in Table A.3.7. Annual grass forage production differed (p=<0.01) when I compared habitat by water treatment by grazing treatment. The results suggested that in both habitats annual grass produces more forage in the once a week and every two weeks watering treatments, except in the CRP grazed plots. The result could merely be noise because of a higher order interaction of the three-way comparison. Forb forage production was found to be significant when I compared habitats (p=0.01). There was more forb production in the sagebrush plots (1.6 g, SE=0.4)] than the CRP plots (0.2 g, SE=0.2).

Forage production results for 2008 are presented in Table A.3.8. Perennial grass forage production differed by habitats (p=<0.01). The CRP plots produced 17.6 g (SE=2.9), while the sagebrush plots produced 3.1 g (SE=1.4). Forb forage production was greater in the sagebrush plots (1.1 g, SE=0.3) than the CRP plots (0.1 g, SE=0.1) when analyzed in terms of habitat (p=0.01).

Forage production results for 2009 are presented in Table A.3.9. Perennial grass forage production was greater in the CRP (17.6 g, SE=2.9) than sagebrush plots (3.1 g, SE=1.4) (p=<0.01). Forb production was greater in the sagebrush (1.1 g, SE=0.3) than CRP plots (0.1 g, SE=0.1) (p=0.01).

Arthropod Response

Arthropod results for 2007 are presented in Table A.3.10. Differences were found when comparing habitats, but not when comparing grazing and watering treatments. The orders Aranae (p=0.03), Diptera (p=<0.01), and Orthoptera (p=<0.01) were more abundant in CRP plots with means of 237.3 (SE=29.2), 502 (SE=53.0), and 331.4 (SE=32.0) individuals, respectively, than the sagebrush plots with means of 136.8 (SE=22.4), 211.3 (SE=34.8), and 100.3 (SE=17.9) individuals, respectively.

Arthropod results for 2008 are presetned in Table A.3.11. Again, differences were found when comparing between habitats, but not between grazing and watering treatments. Hemiptera (p=0.02) and Orthoptera (p=<0.01) were more abundant in CRP plots with means of 938.5 (SE=271.7) and 330.4 (SE=18.0) individuals, respectively, than in the sagebrush plots with means of 131.5 (SE=112.5) and 121.3 (SE=11.0) individuals. Homoptera (p=<0.01) and Lepidoptera (p=0.01) were more abundant in sagebrush plots with means of 727.0 (SE=66.8) and 53.2 (SE=11.9) individuals, respectively, than in the CRP plots with means of 345.6 (SE=46.5) and 9.8 (SE=5.4) individuals, respectively. When analyzed by habitat and water treatment, Hemiptera differed (p=0.01), but when analyzed further this result did not follow the same pattern.

Arthropod results for 2009 are presented in Table A.3.12. Orthoptera were more abundant in the CRP plots with a mean of 341.7 (SE=23.9) individuals than the sagebrush plots with a mean of 105.7 (SE=13.5) individuals (p=<0.01).

Most of the individuals captured in the CRP plots belonged, in decreasing order, to Hymenoptera (ants, 22%), Hemiptera (21%), Homoptera (19%), Orthoptera (10%), Diptera (9%), Coleoptera (8%), Araneae (7%), Hymenoptera (bees and wasps, 5%), and Lepidoptera (0.3%). The majority of individuals captured in sagebrush plots belonged, in decreasing order, to Homoptera (34%), Hymenoptera (ants, 28%), Coleoptera (10%), Diptera (6%), Araneae (5%), Hemiptera (5%), Hymenoptera (bees and wasps, 5%), Orthoptera (4%), and Lepidoptera (1%). All orders occurred in both habitats.

Sage-grouse Use

Pellet count transects were conducted in May and July of 2007, 2008, and 2009. Pellets were only found during the counts in May 2007. These pellets were found on four adjacent plots in the sagebrush habitat. After examination it was determined that these pellets were left during the winter months and not during the nesting or brood-rearing period. Because of heavy snowfall that winter, I believe grouse used this area because it was located on a windswept ridge, leaving more sagebrush exposed (Ward 2006). In conclusion, I found no evidence of grouse finding and using the brood-rearing areas created by my study.

DISCUSSION

A combination of factors contributed to and continues to exacerbate sage-grouse population declines. Declining populations have been characterized as exhibiting poor recruitment attributed to loss or fragmentation of brood-rearing habitats (Connelly et al. 2004). Concomitantly, the creation or restoration of mesic brood-rearing habitats in xeric environments has been identified as a conservation priority by regional and local sage-grouse working groups. These areas typically provide a higher abundance and diversity of forbs and arthropods essential to the diets of young chicks (Peterson 1970, Wallestad 1971, Klott and Lindzey 1990, Johnson and Boyce 1990, Sveum et al. 1998, Connelly et al. 2000, Crawford et al. 2004). The availability of forbs and arthropods has been positively associated with survival and recruitment of sage-grouse chicks.

The two most important habitats for Gunnison sage-grouse in San Juan County, Utah are CRP fields and areas of native sagebrush (Lupis 2005, Ward 2006). Over time wet meadow areas in each of these habitats have been reduced through changes in land use, therefore decreasing the habitat available to grouse during the brood-rearing season.

My study evaluated the role of irrigation and dormant season cattle grazing as practical management tools to create brood-rearing habitat in CRP and sagebrush. Although irrigated study plots retained their greenness longer in the growing season, I did not record any differences in vegetation or arthropod abundance and diversity because of irrigation or grazing. I did, however, note differences in vegetation and arthropod composition between habitat types. The CRP plots studied contained greater arthropod diversity and abundance than native sagebrush plots. However, CRP plots were not equivalent to the native sagebrush plots in terms of providing vegetation essential for brood-rearing.

Arthropod abundance and diversity was higher in the CRP plots than in the sagebrush plots possibly because of the different vegetation communities the two habitats supported. The highest overall arthropod diversity values were obtained from the non-native CRP grassland habitat even though it had less vegetation diversity than the native sagebrush. This difference was not anticipated but could have occurred because the perennial grass (crested wheatgrass) of the CRP plots better suited the diets and feeding methods of the arthropods. CRP fields have been shown to support a high invertebrate biomass, even after losing their forb component, and have been proven to be an important habitat for songbirds and game birds that feed on arthropods (Hull et al. 1996, McIntyre and Thompson 2003, Doxon and Carroll 2007).

Sagebrush contains secondary metabolites as an antiherbivore defense that may act as toxins or digestion inhibitors with increasing concentration during the growing season of spring and summer (Wallestad and Eng 1975, Shipley et al. 2006, Wiens et al. 1991). Wiens et al. (1991) examined the secondary metabolites of sagebrush leaf tissue, and its effects on the abundance and diversity of arthropods. The study found that after an herbivorous attack by arthropods, sagebrush increased their level of toxins and the number of arthropods on the shrubs decreased. Sap and phloem feeding insects recovered more quickly than chewing insects. The feeding methods of sap and phloem feeders may permit them to be highly discriminatory and avoid plant tissues containing secondary metabolites. Herbivorous leaf chewers were less likely to be able to discriminate among cell and tissue types within leaves and will therefore encounter more chemical compounds.

The differences I observed in the vegetation between the two habitats were expected. Land enrolled in CRP was once plowed agricultural land. This practice eliminated most of the sagebrush from the system and probably most of the seed bank supporting native forbs and grasses, and potentially changed the nutrient content of the soil. The seed mixture used in CRP fields was designed to establish a perennial grass cover, therefore it was expected that the CRP plots would have a greater occurrence of perennial grasses and little sagebrush. It was also expected that what sagebrush had begun to re-establish in the CRP plots would be smaller than those in the native sagebrush plots that had never been cultivated.

After the original seeding of the CRP fields, little if no sagebrush successfully established from seed (G. Wallace, Utah Division of Wildlife Resources, personal communication). Forbs successfully established from the seed mixture and remained in the system for a few years and then began to disappear. For this reason, few of those forbs still remained in the plots. Forbs that did occur in the CRP plots were invasive weeds, such as Russian knapweed (*Centaurea repens*), African mustard (*Malcomia africana*), and Russian thistle (*Salsola pestifer*). Crested wheatgrass was the one plant from the original mixture that remained in the system and was found to dominate the CRP plots.

Crested wheatgrass has been shown to develop monoculture stands and dominate plant communities for decades following establishment (Hull and Klomp 1967, Dormaar et al. 1995). The species has been shown to thicken and spread into adjacent areas (Hull and Klomp 1967). Its rapid dispersal rate and long-term dominance over and exclusion of native species have resulted in it being called an invader (Schuman et al. 1982, Henderson and Naeth 2005). Dormaar et al. (1995) found that altering the plant community from native mixed prairie to sequences of cropping followed by introduced grass monocultures significantly reduced the chemical quality of the soils by decreasing the root mass and organic matter evident in the top 7.5 cm of the soil, therefore reducing the energy flow into the soil system. Crested wheatgrass has been to shown to have less live root biomass and a high accumulation of aboveground dead material (Redente et al. 1989). The species allocated nearly twice the amount of carbon to aboveground photosynthetic tissue than plants in the blue grama ecosystem.

Stands of crested wheatgrass also tend to be very stable (Marlette and Anderson 1986). Stand stability was found to be largely a consequence of its dominance in the seed bank (Marlette and Anderson 1986, Henderson and Naeth 2005). Seed banks in crested wheatgrass stands support little diversity. There is little evidence that propagules from native communities are widely dispersed into adjacent crested wheatgrass stands and accumulate to form a diverse seed bank.

The results of this study support previous studies conducted on crested wheatgrass. It appeared that the native seed bank within the CRP plots had been lost. This probably occurred during the decades the land was under cultivation, time dominated by crested wheatgrass, and lack of native seed dispersal from nearby areas (Marlette and Anderson 1986). The seeds from the original seed mixture seem to have also been lost. This could have occurred because of competition from crested wheatgrass, the effect of the species on the soil, and its dominance of the seed bank (Hull and Klomp 1967, Marlette and Anderson 1986, Redente et al. 1989, Dormaar et al. 1995, Henderson and Naeth 2005).

Seeding crested wheatgrass may inhibit or even preclude the development of a diverse plant community by retarding the recovery of native vegetation (Hull and Klomp 1967, Marlette and Anderson 1986). Monoculture stands have resisted the reintroduction of native species and maintained low species diversity (Marlette and Anderson 1986, Dormaar et al. 1995). A monoculture cannot be restored to a diverse plant community simply by removing some crested wheatgrass plants. If an increase in species diversity is desired, existing crested wheatgrass and their propagules in the soil must be destroyed and other species deliberately introduced. To improve the chances of creating brood-rearing habitat in CRP fields it might be necessary to physically remove or reduce the number of crested wheatgrass plants in the treatment areas and re-seed the plots with a mixture of native annual and perennial grasses and forbs. It may also be necessary to invest in proper seed bank preparation techniques and irrigation to ensure seed germination, seedling survival, and species persistence.

Irrigation of plots within each habitat resulted in the lengthening of the growing season for vegetation, but did not result in the anticipated increase in abundance and diversity of forbs, grasses, and arthropods. Vegetation in watered plots remained green throughout the entire watering season (June to July) while vegetation in the control plots desiccated by the end of July. If a consistent watering pattern is continued over large areas and long periods of time, it might be possible for an abundant and diverse arthropod community to develop.

I found that using a sprinkler irrigation system was not efficient enough to produce the desired results. Using sprinklers was time consuming and required considerable maintenance. In the undulating landscape of the study area it was difficult to maintain water pressure in the pipes. The sprinklers were inefficient in the windy environment and because of strong daily afternoon winds I was forced to split the watering schedule in two, with a morning watering period and an evening watering period. In order for an irrigation method to be developed into a land management practice for creating brood-rearing habitat a different water delivery system will be necessary.

Although, this study did not provide the anticipated results, it did reveal important information about the vegetation and arthropod communities in both CRP and sagebrush that will affect the development of future management techniques, especially when managing CRP. During the study, I recorded an increase in vegetation growth and diversity in areas where leaks occurred in the irrigation system and water kept the soil saturated throughout the summer. This has led me to the conclusion that it is necessary to keep the soil saturated throughout the summer through flood irrigation. Solar panel powered pumps can be used to easily distribute water to certain areas. The use of the solar panel will allow the pump to run under its own power throughout the day while the sun is shining. Distributing the water to low lying areas through a network of pipes will allow a large area of soil to be saturated throughout the summer. This method would reduce maintenance costs and the amount of labor required, while promoting perennial grass, forb, and sagebrush growth.

Future management techniques will also need to control crested wheatgrass. During this study, even under heavy dormant season cattle grazing, crested wheatgrass continued to dominate the CRP plots at the expense of forbs. Techniques should also address possible invasion by cheatgrass and other invasive weeds, while promoting the growth of native perennial grasses, forbs, and sagebrush. Possible techniques to accomplish this are a combination of mechanical disking, grazing, re-seeding of native perennial grasses and forbs, planting of sagebrush seedlings, and irrigation. This information can be used by managers and private landowners to implement brood-rearing restoration projects.

MANAGEMENT IMPLICATIONS

The creation of brood-rearing habitat is crucial for the recruitment of individuals into grouse populations. Techniques employed in these restoration projects could be developed into a cost-share program under EQIP. Restoration projects in CRP will require biological and mechanical treatments. The use of irrigation for the creation of brood-rearing habitat is essential to ensure the establishment and continued propagation of seeded perennial grasses and forbs, and sagebrush seedlings. Irrigation on public and private land is both a feasible and practical method when using solar powered groundwater pumps and flood irrigation. Control of crested wheatgrass will also be necessary. Cattle grazing and mechanical disking are methods that could be used to control crested wheatgrass. These methods should be used in combination with re-seeding and flood irrigation.

LITERATURE CITED

Buckland, S.T., D.R. Anderson, K.P. Burnham, J.L. Laake, D.L. Borchers, and L. Thomas, eds. 2001. Introduction to distance sampling—Estimating abundance of biological populations: Oxford University Press, Oxford, United Kingdom.

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Crawford, J.A., R.A. Olson, N.E. West, J.C. Mosley, M.A. Schroeder, T.D. Whitson, R.F. Miller, M.A. Gregg, and C.S. Boyd. 2004. Ecology and management of sage-grouse and sage-grouse habitat. Journal of Range Management 57:2-19.

Dahlgren, D. 2005. Greater sage-grouse brood-rearing habitat manipulation in mountain big sagebrush, use of treatments, and reproductive ecology on Parker Mountain, Utah. Thesis, Utah State University, Logan, USA.

Daubenmire, R. F. 1959. A canopy coverage method of vegetational analysis. Northwest Science 33:43-64.

Dormaar, J.F., M.A. Naeth, W.D. Willms, and D.S. Chanasyk. 1995. Effect of native prairie, crested wheatgrass (*Agropyron cristatum* (L.) Gareth.) and Russian wildrye (*Elymus junceus* Fish.) on soil chemical properties. Journal of Range Management 48:258-263.

Doxon, E.D., and J.P. Carroll. 2007. Vegetative and invertebrate community characteristics of Conservation Reserve Program fields relative to gamebirds in western Kansas. The American Midland Naturalist 158:243-259.

Drut, M.S., W.H. Pyle, and J.A. Crawford. 1994. Diets and food selection of sage-grouse chicks in Oregon. Journal of Range Management 47:90-93.

Graham, T.B., A.M.D. Brasher, and R.N. Close. 2008. Mormon cricket control in Utah's west desert; evaluation of impacts of the pesticide diflubenzuron on nontarget arthropod communities: U.S. Geological Survey Open-File Report 2008–1305, 82 p. [http://pubs.usgs.gov/of/2008/1305/].

Gunnison Sage-grouse Rangewide Steering Committee (GSRSC). 2005. Gunnison Sage-grouse rangewide conservation plan. Colorado Division of Wildlife, Denver, USA.

Gunnison Sage-grouse Rangewide Steering Committee (GSRSC). 2007. Minimum Structural Vegetation Collection Guidelines for the Gunnison Sage-grouse. Colorado Division of Wildlife, Denver, USA.

Henderson, D.C., and M.A. Naeth. 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. Biological Invasions 7:639-650.

Hull, A.C., and G.J. Klomp. 1967. Thickening and spread of crested wheatgrass stands on southern Idaho ranges. Journal of Range Management 20:222-227.

Hull, S.D., R.J. Robel, and K.E. Kemp. 1996. Summer avian abundance, invertebrate biomass, and forbs in Kansas CRP. Prairie Naturalist 28:1-12.

Johnson, G.D., and M.S. Boyce. 1990. Feeding trials with insect in the diet of sage-grouse chicks. Journal of Wildlife Management 54:88-91.

Klebenow, D.A., and G.M. Gray. 1968. Food habits of juvenile sage-grouse. Journal of Range Management 12:80-83.

Klott, J.H., and F.G. Lindzey. 1990. Brood habitat of sympatric sage grouse and Columbian Sharp-tailed grouse in Wyoming. Journal of Wildlife Management 54:84-88.

Lukacs, P.M. 2001. Estimating density of animal populations using trapping webs—Evaluation of web design and data analysis. Colorado State University, Fort Collins.

Lukacs, P.M. 2002. WebSim—Simulation software to assist in trapping web design. Wildlife Society Bulletin, v. 30, p. 1259−1261.

Lukacs, P.M., A.B. Franklin, and D.R. Anderson. 2004. Passive approaches to detection in distance sampling, Pages 260-280 *in* Buckland and others, editors. Advanced distance sampling—Estimating abundance of biological populations. Oxford University Press, Oxford, United Kingdom.

Lupis, S.G. 2005. Summer ecology of Gunnison Sage-grouse (*Centrocercus minimus*) in San Juan County, Utah. Thesis, Utah State University, Logan, USA.

Lupis, S.G., T.A. Messmer, and T. Black. 2006. Gunnison sage-grouse use of Conservation Reserve Program fields in Utah and response to emergency grazing: a preliminary evaluation. Wildlife Society Bulletin 34:957-962.

Marlette, G.M., and J.E. Anderson. 1986. Seed bank and propagule dispersal in crested wheatgrass stands. Journal of Applied Ecology 23:161-175.

McIntyre, N.E., and T.R. Thompson. 2003. A comparison of Conservation Reserve Program habitat plantings with respect to arthropod prey for grassland birds. The American Midland Naturalist 150:291-301.

New, T.R. 1998. Invertebrate surveys for conservation. Oxford University Press, Oxford, United Kingdom.

Olsen, M.E., L. Wilson, J.W. Metcalf, and T.B. Hutchings. 1962. Soil survey of the San Juan Area, Utah. Series 1945, No. 3. Utah Agricultural Experiment Station.

Peterson, J.G. 1970. The food habits and summer distribution of juvenile sage-grouse in central Montana. Journal of Wildlife Management 34:147-155.

Redente, E.F., M.E. Biondini, and J.C. Moore. 1989. Observations on biomass dynamics of a crested wheatgrass and native short grass ecosystem in southern Wyoming. Journal of Range Management 42:113-118.

San Juan County Gunnison Sage-grouse Working Group (SWOG). 2000. Gunnison Sage-grouse Conservation Plan, San Juan County, Utah.

San Juan County Gunnison Sage-grouse Working Group (SWOG). 2003. Gunnison Sage-grouse Conservation Plan Update San Juan County, Utah.

Schuman, G.E., F. Rauzi, and D.T. Booth. 1982. Production o Russian wildrye and crested wheatgrass and their effect on prairie soils. Agronomy Journal 74:23-26.

Shipley, L.A., T.B. Davila, N.J. Thines, and B.A. Elias. 2006. Nutritional requirements and diet choices of the pygmy rabbit (*Brachylagus idahoensis*): A sagebrush specialist. Journal of Chemical Ecology 32:2455-2474.

Sveum, C.M., J.A. Crawford, and W.D. Edge. 1998. Use and selection of brood-rearing habitats by sage-grouse in south central Washington. Great Basin Naturalist 58:344-351.

Triplehorn, C.A., and N.F. Johnson. 2005. Borror and DeLong's introduction to the study of insects. Seventh edition. Thomson Brooks/Cole, Belmont, Calif.

USDI-BLM, Interagency Technical Team. 1996. Utilization Studies and Residual Measurements. Technical Reference 4400-3, BLM/RS/ST-96/004+1730, USDI-BLM, National Applied Resource Sciences Center.

Wallestad, R.O. 1971. Summer movements and habitat use by sage grouse broods in central Montana. Journal of Wildlife Management 35:129-136.

Wallestad, R.O., and R.L. Eng. 1975. Foods of adult sage-grouse in central Montana. The Journal of Wildlife Management 39:628-630.

Ward, S. 2006. Gunnison Sage-grouse winter ecology and arthropod abundance diversity in relation to habitat use in San Juan County, Utah. Thesis, Utah State University, Logan, USA.

Wiens, J.A., R.G. Cates, J.T. Rotenberry, N. Cobb, B. Van Horne, and R.A. Redak. 1991. Arthropod dynamics on sagebrush (*Artemisia tridentata*): effects of plant chemistry and avian predation. Ecological Monographs 61:299-321.

CHAPTER 3

ASSESSMENT OF VEGETATION CONDITIONS OF SAGEBRUSH HABITATS WITHIN THE GUNNISON SAGE-GROUSE CONSERVATION AREA IN SAN JUAN COUNTY, UTAH

ABSTRACT San Juan County supports the only population of Gunnison sage-grouse (*Centrocercus minimus*) found in Utah. The current population estimates are below the minimum desired population objective established in the Gunnison Sage-grouse Rangewide Conservation Plan (RCP) and the San Juan County Gunnison Sage-grouse Conservation Plan (SJCCP). Both plans identified the need to complete periodic assessments of the existing vegetation conditions in occupied and potential (unoccupied) sage-grouse habitat to ensure compliance with recommended guidelines and guide management actions. In the summer of 2009 I completed a habitat assessment of the 1,392,812 ha Gunnison Sage-grouse Conservation Area (CA) in San Juan County using RCP protocols. Using randomly generated points I measured vegetation conditions at 93 sites within occupied and unoccupied sagebrush habitats within the CA. Occupied habitat was defined as use areas contained within the 24,177 ha Conservation Study Area (CSA). Unoccupied habitat was defined as historical areas that previously supported sage-grouse or were in close proximity to areas that were currently or historically inhabited. I compared the current vegetation conditions for breeding, summer/fall, and winter habitats to RCP recommended guidelines. Perennial grass cover and height met or exceeded guidelines for occupied and unoccupied areas for all habitat categories. This was attributed largely to the introduction of crested wheatgrass (*Agropyron cristatum*)

into the system via the Conservation Reserve Program. Forb cover in unoccupied areas for all habitats approximated guidelines. In occupied areas forb cover was below RCP recommendations for all habitats. Forb cover height met the lowest limits of the guidelines for occupied and unoccupied areas for summer/fall but not breeding habitats. Sagebrush cover met or exceeded recommended guidelines for occupied and unoccupied areas for breeding and summer/fall habitats, but not winter habitats. Sagebrush (*Artemisia tridentata*) height met or exceeded guidelines for unoccupied and occupied areas for all habitat categories. To maximize habitat benefits for Gunnison sage-grouse in San Juan County, managers should implement conservation actions that protect existing sagebrush habitats and increase forb and grass cover in currently occupied habitats. This information will assist the Monticello/Dove Creek Local Working Group in prioritizing conservation efforts.

INTRODUCTION

Connelly et al. (2004) suggested that of the factors contributing to range wide declines in sage-grouse (*Centrocercus* spp.), the loss, degradation, and fragmentation of the sagebrush (*Artemisia* spp.) ecosystem were paramount. As sagebrush obligates, sage-grouse require sagebrush habitats to complete their life cycle. Thus, structure and composition of plant communities within sagebrush ecosystems influence sage-grouse nesting, breeding, brood-rearing, fall, and winter habitat selection.

Gunnison sage-grouse (*C. minimus*) currently occupy 4,787 km^2 (8.5% of their original range) in Colorado and Utah. There is one known population in the state of Utah. The Gunnison Sage-grouse Rangewide Conservation Plan (RCP) and the San Juan

County Gunnison Sage-grouse Conservation Plan (SJCCP) recommended management strategies to conserve the species (SWOG 2000, GSRSC 2005). Both plans identified the need for periodic habitat assessments to determine if existing vegetation conditions meet the desired vegetation criteria stated in the RCP. Periodic habitat assessments can assist managers in developing and prioritizing habitat restoration projects (GSRSC 2005).

The RCP established vegetation condition goals for Gunnison sage-grouse seasonal habitats (GSRSC 2005). Breeding habitats include lek, nesting, and early brood-rearing habitat from mid-March through late-June. The RCP defined breeding habitat as sagebrush communities delineated within 6.4 km of a lek. The SJCCP identified a long-term goal of reestablishing desired vegetation conditions on 50-75% of the area within 6.4 km of occupied lek sites (SWOG 2003). The defined vegetation characteristics for breeding habitats included: total shrub canopy cover of 20-40% (15-25% sagebrush canopy cover) with an average sagebrush height of 25-50 cm, 10-30% grass canopy cover with a height of 10-15 cm, and 5-15% forb canopy cover with a height of 5-10 cm (GSRSC 2005).

The RCP defined summer/fall habitat as vegetation communities, including sagebrush, agricultural fields, and wet meadows that are within 6.4 km of lek sites (GSRSC 2005). The SJCCP recommended establishing these conditions on 50-75% of the area (SWOG 2003). The defined desired vegetation conditions identified were: 10-30% total shrub canopy cover (5-15% sagebrush canopy cover) with an average sagebrush height of 20-40 cm, 10-25% grass canopy cover with a height of 10-15 cm, and 5-15% forb canopy cover with a height of 3-10 cm (GSRSC 2005). Mesic areas should support a grass cover of 10-35% with a height of 10-15 cm and a forb cover of 15-35% with a height of 5-10 cm.

The SJCCP further identified the need to reestablish desired vegetation conditions of wintering habitats on 50% of the areas located within the Conservation Study Area (CSA) and 25% within the area buffering the CSA (SWOG 2003). Lupis (2005) and Ward (2007) previously defined the CSA based on location data obtained from radio-collared sage-grouse. The RCP defined winter habitat as sagebrush areas within currently occupied habitats that are available to sage-grouse in average winters (GSRSC 2005). The defined vegetation conditions for winter habitat include: sagebrush canopy cover of 30-40% with a height of 40-55 cm.

I completed a vegetation conditions assessment to determine the habitat conditions for Gunnison sage-grouse that inhabit San Juan County. This information will assist managers in quantifying the relative contribution of occupied and potential habitats to achieving overall SJCCP and RCP habitat and population goals. The results will be used by members of the Monticello/Dove Creek Local Working Group to update the current SJCCP, the RCP, and prioritize future conservation efforts.

STUDY AREA

The habitat assessment was conducted in San Juan County, Utah, during the summer of 2009. San Juan County is located in the extreme southeastern corner of Utah. The county is bordered by the Colorado River to the north and west, Arizona to the south, and Colorado to the east. The San Juan County Gunnison Sage-grouse Working Group (SWOG) previously designated an area northeast of the town of Monticello, Utah, as a

Gunnison sage-grouse priority conservation area (CA, Fig. 3.1, SWOG 2000). The CA consisted of 1,392,812 ha, 38% (127,170 ha) of which was privately owned. The CA was identified by encompassing historic and current lek sites, potentially suitable sage-grouse habitat, and sage-grouse observations. The CA was characterized by agricultural fields enrolled in the Conservation Reserve Program (CRP), active agricultural fields, and grazed interspersed with fragmented patches of Wyoming big sagebrush (*A. tridentata wyomingensis*), black sagebrush (*A. nova*), pinyon pine (*Pinus edulis*), juniper (*Juniperus osteosperma*), and oak (*Quercus gameblii*).

Within the CA, SWOG also identified the Conservation Core Area (CCA) that consisted of 136,249 ha, of which 89% (88,420 ha) was privately owned. Within the CCA, SWOG designated a priority study area, the CSA. The CSA consisted of 24,177 ha, of which 93% (22,556 ha) was privately owned. The CSA encompassed the current year round range of the population (Lupis 2005).

METHODS

In the summer of 2009, I measured vegetation parameters within Gunnison sage-grouse occupied and unoccupied habitats in San Juan County, Utah. I defined occupied habitat as areas located within the CSA. I defined unoccupied habitat as areas that were

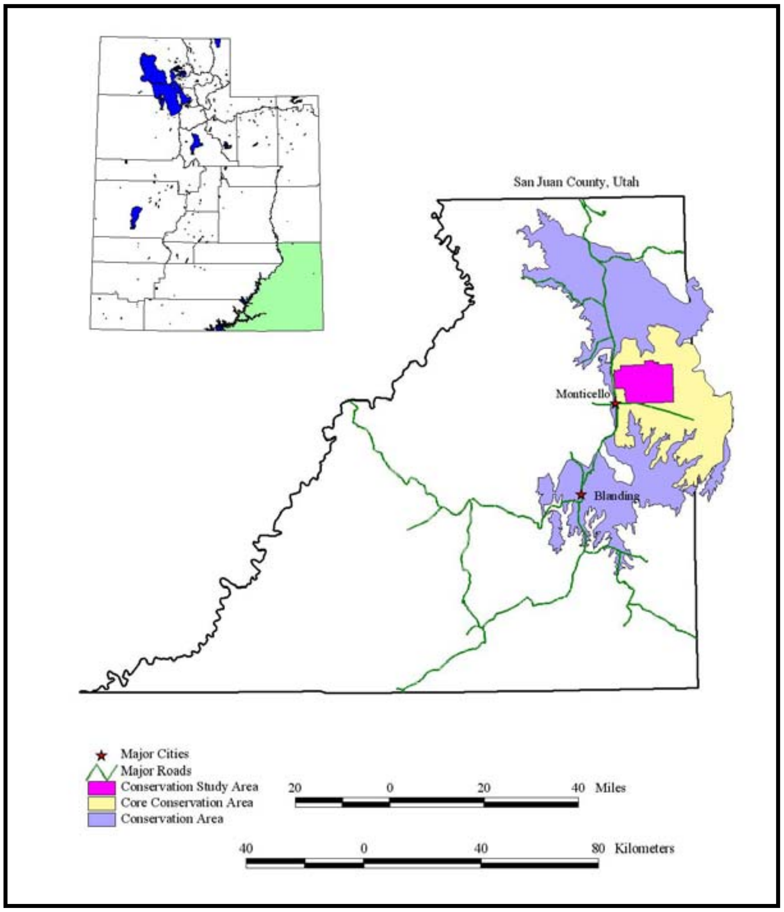

Figure 3.1. Gunnison Sage-grouse Conservation Area, San Juan County, Utah (Lupis 2005).

historically inhabited or were near an area currently or historically inhabited by sage-grouse. Unoccupied habitat largely fell within the CA and CCA. I compared these data to the habitat guidelines identified in the RCP to assess the status of existing and potential habitat in the CA, CCA, and CSA.

I conducted the habitat assessment by ground truthing LandSat imagery of the CA. I used ArcGIS (ArcMap version 9.3.1) to plot historic and current lek locations, which were located within the CSA. I created a polygon by buffering around the leks in

1,500 m increments to incorporate the distances the birds move from the leks throughout the year, such as from the lek to nesting areas, nesting areas to brood-rearing areas, and wintering areas (SWOG 2000, GSRSC 2005, Lupis 2005, Ward 2007). I further extended the buffer to include unoccupied habitat within the CCA and CA. After incorporating all possible movement distances and habitats the buffer totaled 7,500 m from lek sites (Fig. 3.2). Occupied habitats were confined to the CSA. Unoccupied habitats encompassed all other areas in the CA, excluding the CSA.

I generated 1,000 random points within the polygon and randomly selected 150 of these points (Fig. 3.2). Using satellite imagery I eliminated points that were located in agricultural fields, CRP, grazed rangelands, and pinyon-juniper and oak woodlands, focusing on points that encompassed sagebrush habitats, leaving 144 points. I visited each mapped point. Upon field visits some points were eliminated because they did not meet the established criteria. Points that fell within CRP, agricultural fields, woodlands, and grazed rangelands that did not support sagebrush were eliminated. Points that fell within private land posted as no trespassing were also eliminated. This left 93 points, 39 points in unoccupied and 54 points in occupied habitats, respectively.

At points that met the criteria, I measured the vegetation conditions using the Gunnison Sage-grouse Rangewide Steering Committee (GSRSC) Structural Vegetation Collection Guidelines (SVCG, GSRSC 2007). At each point, two 30-m transects were established. Cover of grasses, forbs, and shrubs was visually estimated by placing a Daubenmire frame every 3-m along each 30-m transect (Daubenmire 1959). The SVCG identified six cover classes based on the standardized Daubenmire method. The GSRSC believed the Daubenmire method lumped too much vegetation into the 5-25% class for the Gunnison sage-grouse vegetation variables. Thus, they split the 5-25% category into 2 cover classes. The canopy cover classes used in this study were: 0-5%, 5-15%, 15-25%, 25-50%, 50-75%, 75-100% (GSRSC 2007).

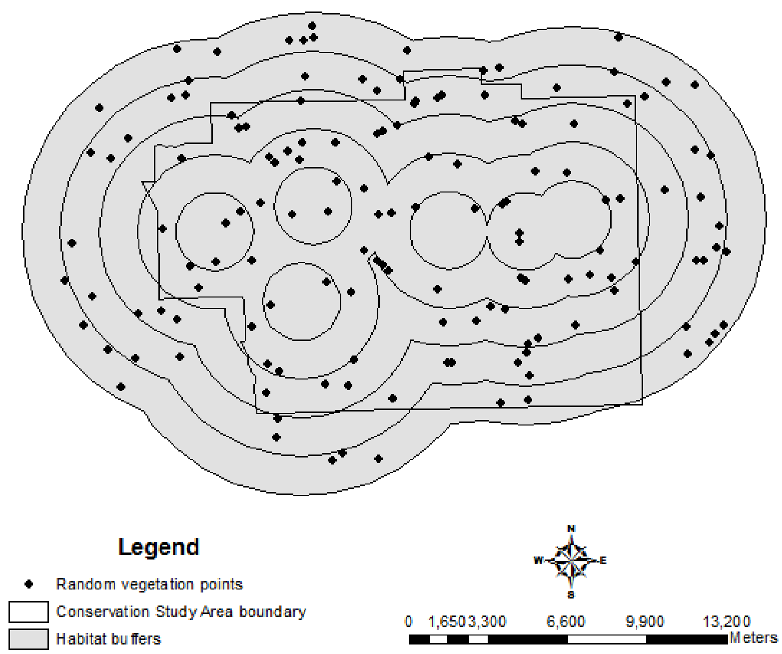

Figure 3.2. Buffer distances from active and historic leks to show seasonal movements of Gunnison sage-grouse (*Centrocercus minimus*) in and around the Conservation Study Area, San Juan County, Utah, 2009. Locations of randomly generated points used to assess habitat conditions in currently, historically, and potential habitat occupied by Gunnison sage-grouse. Occupied habitat is within the boundary of the Conservation Study Area. Unoccupied habitat is located outside of the Conservation Study Area boundary.

One height measurement of sagebrush, forb, annual grass, and perennial grass was taken at each Daubenmire frame by selecting the plant closest to the lower left hand corner of the frame (Daubenmire 1959, GSRSC 2007). If sagebrush was not found within the frame then the closest sagebrush within 10m of the frame was used. If no sagebrush was within 10m of the frame it was marked as not present. Only forbs and grasses within the frame were used to measure height. If no forb or grass was within the frame the plant group was marked as not being present.

DATA ANALYSIS

Vegetation data collected at each point were summed and averaged for each habitat area (occupied and unoccupied). Means for cover and height for each habitat area are reported with 95% confidence intervals. The results were then compared to the defined vegetation conditions recommended in the RCP for breeding, summer/fall, and winter habitat categories.

RESULTS

The RCP defines breeding and summer/fall habitat as the land within 6.4 km of lek sites (GSRSC 2005). This distance encompassed the entire CSA. As a result, all vegetation data collected at points within the CSA (n= 54) fell into the composite category encompassing breeding, summer/fall, and winter habitat.

Because sagebrush was the dominate shrub cover, I report total sagebrush cover in lieu of total shrub cover. The results for occupied and unoccupied habitat relative to RCP guidelines are presented in Table 3.1.

Occupied Habitat

In breeding habitats, perennial grass ($\bar{x}$=17%, CI=3.19) and sagebrush ($\bar{x}$=17%, CI=3.61) cover were within the RCP guidelines of 10-30% and 15-25%, respectively (Table 3.1). Forb cover ($\bar{x}$=3%, CI=1.40) did not meet the guidelines of 5-15%. Height of perennial grass ($\bar{x}$=23 cm, CI=4.39) was slightly above the guidelines of 10-15 cm. The height of sagebrush ($\bar{x}$=51 cm, CI=4.21) was within the upper limit of the guidelines of 25-51 cm. Forb height ($\bar{x}$=3 cm, CI=0.92) was below the guidelines of 5-10 cm.

In summer/fall habitats, cover of perennial grass ($\bar{x}$=17%, CI=3.19) was within the RCP guidelines of 10-25%. Sagebrush cover ($\bar{x}$=17%, CI=3.61) was within the upper limits of the guidelines of 5-15% cover. Cover of forbs ($\bar{x}$=3%, CI=1.40) was below the guidelines of 5-15%. Perennial grass height ($\bar{x}$=23 cm, CI=4.39) exceeded the guidelines of 10-15 cm. Forb height ($\bar{x}$=3 cm, CI=0.92) was at the lower limits of the guidelines of 3-10 cm. Sagebrush height ($\bar{x}$=51 cm, CI=4.21) exceeded the upper limits of 20-40 cm.

In winter habitats, cover of sagebrush ($\bar{x}$=17%, CI=3.6) was below the RCP guidelines of 30-40%. Sagebrush height ($\bar{x}$=51 cm, CI=4.21) exceeded the upper limits of 20-40 cm.

Unoccupied Habitat

In unoccupied breeding habitat, cover of perennial grass ($\bar{x}$=18%, CI=3.77), forbs ($\bar{x}$=6%, CI=1.64) and sagebrush ($\bar{x}$=17%, CI=4.40) were within the RCP guidelines of 10-30%, 5-15%, and 15-25%, respectively. Perennial grass height ($\bar{x}$=15 cm, CI=2.34) was also within the guidelines of 10-15 cm. Height of forbs ($\bar{x}$=4 cm, CI=0.72) was below the guidelines of 5-10 cm. Sagebrush height ($\bar{x}$=46 cm, CI=4.64) was within the guidelines of 25-50 cm.

In unoccupied summer/fall habitats, cover of perennial grass ($\bar{x}$=18%, CI=3.77) and forbs ($\bar{x}$=6%, CI=1.64) were within the RCP guidelines of 10-25% and 5-15%, respectively. Cover of sagebrush ($\bar{x}$=17%, CI=4.40) slightly exceeded the guidelines of 5-15%. Height of perennial grass ($\bar{x}$=15 cm, CI=2.34) and forbs ($\bar{x}$=4 cm, CI=0.72) were within the guidelines of 10-15 cm and 3-10 cm, respectively. Sagebrush height ($\bar{x}$=46 cm, CI=4.64) exceeded the guidelines of 20-40 cm.

In unoccupied winter habitats, cover of sagebrush ($\bar{x}$=17%, CI=4.40) was below the guidelines of 30-40%. Sagebrush height ($\bar{x}$=46 cm, CI=4.64) was within the guidelines of 40-55 cm.

DISCUSSION

Based on the results of my habitat assessment of the vegetation parameters within occupied Gunnison sage-grouse habitats in the CA, CCA and CSA, I recommend that managers focus their attention on protection of existing sagebrush canopy cover and the

Table 3.1. Average percent cover and height of vegetation in occupied and unoccupied habitat in the Gunnison Sage-grouse (*Centrocercus minimus*) Conservation Area in San Juan County, Utah, 2009, reported with 95% confidence intervals. Occupied habitat was defined as land within the Gunnison Sage-grouse Conservation Study Area. Unoccupied habitat was defined as areas that once supported sage-grouse or were in close proximity to areas that support or once-supported sage-grouse. Unoccupied habitat fell within the Conservation Area and Core Conservation Area. Current habitat conditions were compared to habitat guidelines stated in the Gunnison Sage-grouse Rangewide Conservation Plan (GSRSC 2005).

	Percent Cover						Height (cm)					
Habitat	Perennial Grass	CI	Forbs	CI	Sagebrush	CI	Perennial Grass	CI	Forbs	CI	Sagebrush	CI
Measurments												
Occupied	17	3.19	3	1.40	17	3.61	23	4.39	3	0.92	51	4.21
Unoccupied	18	3.77	6	1.64	17	4.40	15	2.34	4	0.72	46	4.64
Guidelines												
Breeding	10-30		5-15		15-25		10-15		5-10		25-50	
Summer/fall	10-25		5-15		5-15		10-15		3-10		20-40	
Winter					30-40						40-55	

restoration of the forb components in CRP and native sagebrush. These observations are in line with the conservation strategies currently outlined in both the SJCCP and RCP.

Although unoccupied habitat in the CA better approximated SJCCP and RCP habitat guidelines, this area is avoided by Gunnison sage-grouse. Gunnison sage-grouse evolved in a landscape free of vertical structures, such as trees, power poles, and fence posts (Connelly, 2000*a*). Because of this evolutionary trait, they will avoid certain areas and will not cross over vertical structures even if the habitat on the other side is of good quality. Much of the area surrounding the occupied habitat confined within the CSA was dominated by pinyon-juniper and oak woodlands, therefore the birds will not cross over the trees to utilize these areas.

While ground truthing the randomly generated points that fell within unoccupied habitats in the CA, I discovered mosaics of open areas among the pinyon-juniper and oak woodlands. In many of these areas sagebrush could be found in small isolated patches, surrounded by or located near woodlands. Upon searching these patches, I did not find any evidence (i.e., pellets) that the sites were used by sage-grouse (Connelly et al. 2000*a*). Furthermore, when I overlaid the random points with known bird locations from previous studies, the locations were concentrated in the CSA (Fig 3.3). The small patches of sagebrush within the woodlands in the CA and CSA were avoided. The CSA was preferred by Gunnison sage-grouse because it contains little vertical structure in terms of oak and pinyon-juniper (Connelly et al. 2000*a*, GSRSC 2005).

Sage-grouse evolved in habitats free of vertical structures, including trees (Connelly et al. 2000*a*). Raptors and corvids prey on sage-grouse adults, young, and

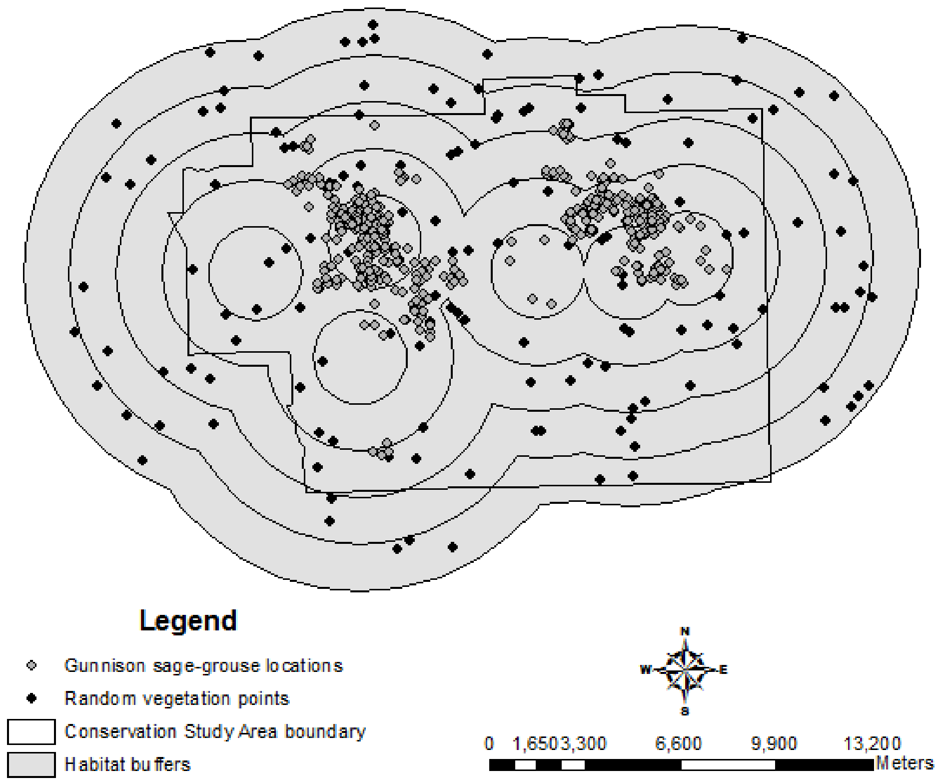

Figure 3.3. Location of vegetation monitoring points and known Gunnison sage-grouse locations within the Conservation Study Area, San Juan County, Utah, 2009 (Lupis 2005, Ward 2007).

nests. Previous research has shown that their presence increases with the presence of vertical structures (Hartzler 1974, Ellis 1984, Connelly et al. 2000*b*, Fletcher et al. 2003, Manzer and Hannon 2005). This not only increases possible predation of sage-grouse but also results in the fragmentation of habitat and populations by acting as a barrier and subdividing suitable habitat. The agricultural history of land use in the CSA may have contributed to the loss and fragmentation of sagebrush and corresponding reduction in grass and forb cover. Currently, the dominant perennial grass throughout the CSA is crested wheatgrass (*Agropyron cristatum*). Crested wheatgrass was established in the CSA when thousands of hectares of cropland were originally enrolled in CRP and planted with a seed mixture that contained the non-native perennial grass (SWOG 2000). Crested wheatgrass has the potential to effectively out-compete native forbs and grasses and spread to adjacent areas (Hull and Klomp 1967, Schuman et al. 1982, Henderson andhas invaded sagebrush areas throughout the entire CSA, and it was more dominant than native perennial grasses at the sites evaluated.

Even though habitat quality in the CSA did not meet SJCCP and RCP habitat guidelines, the Gunnison sage-grouse population has steadily rebounded after an initial drop in the 1980s and has held steady over the past 20 years with only minor increases and decreases in response to drought conditions (Fig. 3.4, SWOG 2000, Lupis 2005).

I believe this rebound can be attributed largely to the advent of the CRP program in the CSA. Although CRP fields do not achieve vegetation habitat guidelines, these areas constitute new permanent contiguous vegetation cover that has provided Gunnison sage-grouse important seasonal habitats (Lupis 2005, Ward 2007). Thus, the retention and habitat restoration of CRP fields in the CSA for Gunnison sage-grouse should remain the highest conservation priority in San Juan County.

Reestablishing sagebrush, grass, and forb cover in CRP fields to approximate SJCCP and RCP guidelines would provide missing components to the habitat. These restoration efforts would help connect native sagebrush areas throughout the CSA, reducing the effects of fragmentation on the population.

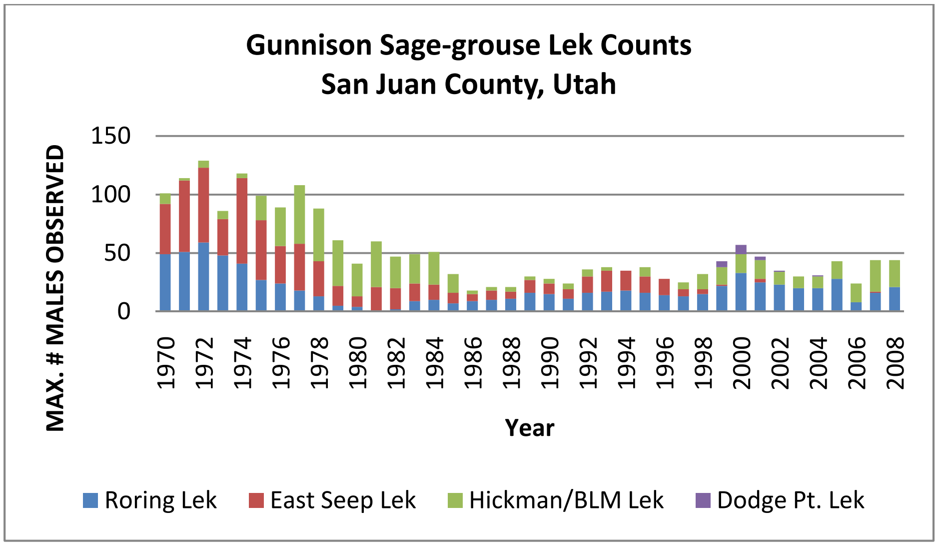

Figure 3.4. Gunnison Sage-grouse (*Centrocercus minimus*) lek counts from San Juan County, Utah. Maximum number of males observed is recorded. Data from Hickman and BLM leks have been combined because of daily movements of males between these 2 leks (SWOG 2004).

Later stages of habitat restoration efforts should focus on identifying areas outside of the CSA that hold promise for providing habitat for the sage-grouse. Restoration efforts designed to remove pinyon-juniper to open corridors would allow Gunnison sage-grouse access to areas exhibiting better habitats and facilitate population exchanges, which could increase genetic diversity. In the interim, managers should consider translocation of birds from both Colorado and Utah to mitigate concerns about low genetic diversity (GSRSC 2005).

MANAGEMENT IMPLICATIONS

The results of the habitat assessment illustrate that the sage-grouse are restricted to occupied habitats in the CSA by the presence of pinyon-juniper and oak woodlands. This exemplifies the need to improve the habitat within the CSA to maximize what little habitat the grouse have available to them. The habitat assessment also illustrated that forbs and grasses are lacking from much of the habitat within the CSA. Habitat improvement projects should be focused on the remaining sagebrush areas within the CSA. Efforts should also be made to re-establish sagebrush, forb, and grass patches within CRP fields throughout the CSA to expand the habitat available to the grouse.

LITERATURE CITED

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000*a*. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Connelly, J.W., A.D. Apa, R.B. Smith, and K.P. Reese. 2000*b*. Effects of predation and hunting on adult sage-grouse (*Centrocercus urophasianus*) in Idaho. Wildlife Biology 6: 227-232.

Connelly, J.W., S.T. Knick, M.A. Schroeder, and S.J. Stiver. 2004. Conservation Assessment of Greater Sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming.

Daubenmire, R. F. 1959. A canopy coverage method of vegetational analysis. Northwest Science 33:43-64.

Ellis, K.L. 1984. Behavior of leking greater sage-grouse in response to a perched golden eagle. Western Birds 15: 37-38.

Fletcher, Q.E., C.W. Dockrill, D.J. Saher, and C.L. Aldridge. 2003. Northern harrier, *Circus cyaneus*, attacks on greater sage-grouse, *Centrocercus urophasianus*, in southern Alberta. Canadian Field Naturalist 117: 479-480.

Gunnison Sage-grouse Rangewide Steering Committee (GSRSC). 2005. Gunnison Sage-grouse rangewide conservation plan. Colorado Division of Wildlife, Denver, USA.

Gunnison Sage-grouse Rangewide Steering Committee (GSRSC). 2007. Minimum Structural Vegetation Collection Guidelines for the Gunnison Sage-grouse. Colorado Division of Wildlife, Denver, USA.

Hartzler, J.E. 1974. Predation and the daily timing of greater sage-grouse leks. Auk 91: 532-536.

Henderson, D.C., and M.A. Naeth. 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. Biological Invasions 7:639-650.

Hull, A.C., and G.J. Klomp. 1967. Thickening and spread of crested wheatgrass stands on southern Idaho ranges. Journal of Range Management 20:222-227.

Lupis, S.G. 2005. Summer ecology of Gunnison Sage-grouse (*Centrocercus minimus*) in San Juan County, Utah. Thesis, Utah State University, Logan, USA.

Manzer, D.L., and S.J. Hannon. 2005. Relating grouse nest success and corvid density to habitat: A multi-scale approach. Journal of Wildlife Management 69: 110-123.

Schuman, G.E., F. Rauzi, and D.T. Booth. 1982. Production of Russian wildrye and crested wheatgrass and their effect on prairie soils. Agronomy Journal 74:23-26.

San Juan County Gunnison Sage-grouse Working Group (SWOG). 2000. Gunnison Sage-grouse Conservation Plan, San Juan County, Utah.

San Juan County Gunnison Sage-grouse Working Group (SWOG). 2003. Gunnison Sage-grouse Conservation Plan Update San Juan County, Utah.

Ward, S. 2007. Gunnison sage-grouse winter ecology and arthropod abundance diversity in relation to habitat use in San Juan County, Utah. Thesis, Utah State University, Logan, USA.

CHAPTER 4

RAPTOR AND CORVID RESPONSE TO POWER DISTRIBUITION LINE PERCH DETERRENTS IN UTAH

ABSTRACT Increased raptor and corvid abundance has been documented in landscapes fragmented by man-made structures, such as fence posts and power lines. These vertical structures may enhance raptor and corvid foraging and predation efficiency because of increased availability of perch, nesting, and roosting sites. Concomitantly, vertical structures, in particular power distribution lines, have been identified as a threat to sage-grouse (*Centrocercus* spp.) conservation. To mitigate potential impacts of power distribution lines on sage-grouse and other avian species, the electrical power industry has retrofitted support poles with perch deterrents to discourage raptor and corvid use. No published information is available regarding efficacy of contemporary perch deterrents on avian predator use of lower-voltage power distribution lines. We evaluated efficacy of 5 perch deterrents mounted on support poles of an 11-km section of a 12.5-kV distribution line that bisected occupied Gunnison sage-grouse (*C. minimus*) habitat in southeastern Utah, USA. Perch deterrents were mounted on the line in November–December 2006 following a random replicated block design that included controls. During 168 hours and 84 hours of direct observation in 2007 and 2008, respectively, we recorded 276 and 139 perching events of 7 potential avian predators of sage-grouse. Golden eagles (*Aquila chrysaetos*) were the dominant species we recorded during both years. We did not detect any difference in perching events by perch deterrent we evaluated and controls ($p > 0.05$). Perch deterrents we evaluated were not effective

because of inherent design and placement flaws. Additionally, previous pole modifications that mitigated avian electrocutions provided alternative perches. We did not record any raptor or corvid electrocutions or direct predation on Gunnison sage-grouse. The conclusions of this study can be applied by conservation groups and power companies to future management of power distribution lines within areas inhabited by species sensitive to man-made vertical structures.

INTRODUCTION

Transmission lines are defined as power lines designed and constructed to support voltages >60 kV (Avian Power Line Interaction Committee 2006). Distribution lines are defined as a circuit of low-voltage lines, energized at voltages from 2.4 kV to 60 kV and used to distribute electricity to residential, industrial, and commercial customers. The Western Association of Fish and Wildlife Agencies (J. W. Connelly, Western Association of Fish and Wildlife Agencies, unpublished report) reported ≥15,296 km^2 of current sage-grouse (*Centrocercus* spp.) range contained power transmission lines; however, the group was unable to map density of power distribution lines in rural areas.

Connelly et al. (2000b, Connelly, unpublished report) suggested that because of the potential for raptors and corvids to use transmission-line towers and distribution-line poles as new perches and nest sites, placement of these facilities in seasonal sage-grouse habitats could impact the species through increased predation of adults, juveniles, and nests or result in sage-grouse abandoning sites (Knight and Kawashima 1993, Knight et al.1995, Kochert and Olendorff 1999). Corvids and raptors prey on sage-grouse adults, young, and nests. Hartzler (1974), Ellis (1984), Connelly et al. (2000a), Fletcher et al.

(2003), and Manzer and Hannon (2005) reported the impact of avian predators on sage-grouse populations may be exacerbated in human-altered landscapes. Because of these concerns, the Gunnison Sage-Grouse Rangewide Steering Committee (2005) identified retrofitting of distribution line poles with perch deterrents to discourage raptors and corvids from perching as a priority species conservation strategy.

Previous studies have evaluated perch deterrents' effectiveness on transmission-line towers and towers associated with air traffic control (Michener 1928, Janss and Ferrer 1999, Kochert and Olendorff 1999, Avery and Genchin 2004). Lammers and Collopy (2007) studied effectiveness of perch deterrents on towers of a high-voltage (345 kV) transmission line that bisected habitats occupied by greater sage-grouse (*Centrocercus urophasianus*). However, no studies have been published that evaluate efficacy of perch deterrents on distribution lines. We studied raptor and corvid response to 5 types of perch deterrents mounted on a power distribution line that traversed occupied Gunnison sage-grouse habitat in southeastern Utah, USA. Our objective was to determine if raptor or corvid use of the distribution line differed by perch deterrent type or control.

STUDY AREA

We conducted our study during winters of 2007 and 2008 in the Gunnison sage-grouse Conservation Study Area (CSA) located in San Juan County, Utah, USA. The San Juan County Gunnison Sage-grouse Working Group (SWOG) previously identified the CSA. The CSA was located east of the town of Monticello, Utah, USA (SWOG 2005). The CSA contained the primary breeding and wintering complexes of the San

Juan County Gunnison sage-grouse population. The habitat within the CSA consisted of sagebrush (*Artemisia* spp.), grazed rangelands, agriculture fields, and croplands enrolled in the Conservation Reserve Program (CRP).

The study distribution line we selected for study was the longest continuous line located within the CSA. This line paralleled the northern edge of the CSA, which provided Gunnison sage-grouse winter habitat, and was located within 1 km of active leks (Lupis 2005, Ward 2007). The distribution line had a voltage rating of 12.5 kV and paralleled a well-maintained county road. The road allowed access during winter and across private land. The distribution line traversed an undulating landscape and a variety of habitats that included CRP fields, agriculture fields, grazed rangelands, and sagebrush.

METHODS

With the cooperation of PacifiCorp field crews, we established an experimental randomized block design for installation of perch deterrents, which controlled for differences in vegetation and landscape topography that could affect raptor and corvid pole preferences. This design eliminated sampling bias by ensuring that we evaluated each type of deterrent and control relative to habitat types and topography present throughout the length of the distribution line. We considered each pole an experimental unit. We divided the line into 14 blocks consisting of 6 poles each. Within a block, we randomly assigned each pole to one of the 5 treatments or control. The result was multiple replications of each treatment and the control across all habitat types and topographies present. Poles assigned as controls were not fitted with a deterrent.

In November–December 2006 an 11-km section of the selected distribution line, consisting of 84 poles, was modified by PacifiCorp field crews with 5 types of perch deterrents following manufacturer recommendations and in accordance with the established experimental design (Fig. 4.1a–f). Physical deterrents consisted of cones and triangles (Kaddas Enterprises Inc., Salt Lake City, UT), and mini-zenas (Prommel Enterprises Inc., Odenville, AL). The reflective hazing deterrent consisted of displaying single or paired FireFlies™ (P and R Technologies Inc., Portland, OR) suspended on the top and cross arm of the distribution pole. Because of differences in construction, some poles could not support the assigned deterrent, which resulted in incomplete blocks with 14 replications of control poles, mini-zenas, and the 1-FireFly and 2-FireFly arrangements; 16 replications of cones; and 12 replications of triangles.

We conducted perching surveys in 2007 and 2008. We initiated surveys in January and concluded them in April. We selected this survey period because it coincided with the peak number of wintering and migrating raptors and corvids in the area of the distribution line (G. Wallace, Utah Division of Wildlife Resources, personal communication). During this time period raptor and corvid numbers are increased by presence of migrant winter raptor species, including bald eagles (*Haliaeetus leucocephalus*) and rough-legged hawks (*Buteo lagopus*).

Fig. 4.1a.

Fig. 4.1b.

Fig. 4.1c.

Fig. 4.1d.

Fig. 4.1e.

Fig. 4.1f.

Figures 4.1 a-f. Five types of commercially available perch deterrents we evaluated included: a) single and b) paired arrangement of the FireFly™ (P and R Technologies Inc., Portland, OR) hazing deterrent; c) cones (Kaddas Enterprises Inc., Salt Lake City, UT); d) triangles (Kaddas Enterprises Inc., Salt Lake City, UT); e) spikes (Prommel Enterprises Inc., Odenville, AL); and f) control; San Juan County, Utah, 2007-2008.

We surveyed the distribution line twice a day, 5 days a week, weather permitting. We conducted surveys at 0800–1100 hr and 1400–1700 hr (Stahlecker 1978, Fuller and Mosher 1987). We randomly selected the starting point (west or east end) for each survey. We used alternative routes to arrive at the starting points to avoid disturbing any birds already perched. We spent 5 minutes at the starting point and at each mile point thereafter observing and recording any birds seen. While driving to the mile points we maintained speed of the vehicle at 15–25 km/hour.

We recorded all birds perched on the distribution poles. We defined a perching event as an observation of a raptor or corvid perched on a pole. This number of perching events was not a reflection of the density of birds inhabiting the study area, as we could record one bird more than once if it continued down the line perching on different poles. Observations included species, numbers, and perch locations. We recorded exact positions of birds perched on individual poles within the study distribution line.

In our data analysis, we addressed the following questions: 1) did total count of perching events recorded by treatment and control in 2007 and 2008 differ by perch deterrent type and year, and 2) did total count for each species on each type of deterrent and control in 2007 and 2008 differ? The model we used compared means among treatments for total count of perching events and total species counts observed in 2007 and 2008. We used a generalized linear-mixed model with Poisson distribution and log link (SAS Institute, Cary, NC). We made pairwise comparisons among treatment means where necessary. Thus, the statistical model was 1-way in a randomized spatial block design, with poles grouped into spatial blocks to control for spatial heterogeneity in the landscape. We conducted data analyses using the GLIMMIX procedure in SAS–STAT for Windows Version 9.1.3 (SAS Institute).

RESULTS

During 168 hours and 84 hours of direct observation in 2007 and 2008, respectively, we recorded 253 and 136 perching events, respectively, of 7 potential avian predator species of sage-grouse (J. W. Connelly, Western Association of Fish and Wildlife Agencies, unpublished report). The most common perching events by species were golden eagles (*Aquila chrysaetos*), common ravens (*Corvus corax*), and rough-legged hawks. Other species included red-tailed hawks (*B. jamaicensis*), bald eagles, black-billed magpies (*Pica hudsonia*), and ferruginous hawks (*B. regalis*). For analysis we used golden eagle, common raven, and rough-legged hawk counts. Because of the small sample sizes for the other species they were excluded from our statistical analysis.

In 2007, we conducted 112 surveys and recorded 172 (68%) perching events on poles fitted with perch deterrents. Perching events recorded did not differ ($p > 0.05$) for controls (32%), triangles (25%), cones (22%), and minizenas (21%, p=0.31, Table 1). Number of perching events also did not differ by control and perch deterrent type for golden eagles (p=0.07), common ravens (p=0.67), or rough-legged hawks (p=0.71, Table 2). Golden eagles were the most common with 195 (77%) perching events, of which 128 (74%) were on poles fitted with perch deterrents.

In 2008, winter snow conditions periodically closed the survey road and reduced the number of surveys completed. We conducted 56 surveys and recorded 136 avian predator perching events with 91 (67%) events on poles fitted with perch deterrents. Perching events recorded did not differ (p>0.05) for controls (33%), cones (26%), mini-zenas (24%), or triangles (17%, p=0.15, Table 4.1). Number of perching events did not differ by control and perch deterrent type for golden eagles (*p*=0.33), common ravens (*p*= 0.22), and rough-legged hawks (*p*=0.91, Table 4.2). Golden eagles were also most common in 2008 with 110 (81%) perching events, of which 76 (84%) were on poles fitted with perch deterrents. In both survey years, avian predators avoided deterrents, opting for alternative perch sites on the same pole such as insulators, bird guards, and deterrent-free parts of the cross arm, which allowed the birds to perch next to deterrents (Figs. 4.2a–f).

The structural design of the FireFly hazing deterrent could not withstand weather conditions. The FireFly was designed to spin in the wind, creating a reflective strobe effect intended to deter birds from perching. Average wind speed during the 2007 winter surveys was 19 km/hour, with gusts up to 74 km/hour. By the end of the 2007 survey period 10 of the 14 single Firefly arrangements and 11 of 14 double FireFly arrangements were damaged as a result of weather conditions and were largely inoperable, preventing us from evaluating their effectiveness as perch deterrents. Because part of the study design was to assess cost-effectiveness, including maintenance, we did not replace damaged FireFly arrangements prior to 2008 surveys. Thus, we did not analyze these data. Problems included 1) cracking at the site of the swiveling connector causing the reflector to break off of the unit, 2) support arms bending or breaking off under prevailing winds, and 3) swiveling connectors separating from their support base.

Table 4.1. Number of perching events (n) and percentage of perching events (%) by golden eagles, common ravens, and rough-legged hawks recorded documented on each perch deterrent tested and control power poles, and the estimated treatment mean ($\bar{x}$) with standard error (SE). San Juan County, Utah 2007 and 2008. The perch deterrents tested included: a) cones (Kaddas Enterprises Inc., Salt Lake City, UT), b) mini zenas (Prommel Enterprises Inc., Odenville, AL), and c) triangles (Kaddas Enterprises Inc., Salt Lake City, UT).

	2007				2008			
Perch deterrent	n	%	$\bar{x}$	SE	n	%	$\bar{x}$	SE
Cones	56	22	3.9	0.87	36	26	2.0	0.47
Mini zena	54	21	3.9	0.83	32	24	2.1	0.47
Triangle	62	25	4.2	0.92	23	17	2.0	0.49
Control	81	32	5.1	1.05	45	33	3.2	0.62

DISCUSSION

Our study was the first to evaluate commercially available perch deterrents as a means to prevent perching on poles of distribution lines by avian predators that pose a threat to sage-grouse. Perch deterrents we evaluated were ineffective. Our results support those reported by Lammers and Callopy (2007) for 345-kV towers within a transmission line in occupied sage-grouse habitat. Lammers and Callopy (2007) reported that deterrents did not prevent perching but did reduce raptor perching duration. However, the transmission towers in their study were 23–40 m tall and spaced in 366-m intervals.

Table 4.2. Number of perching events (n) documented for golden eagles, common ravens, and rough-legged hawks by perch deterrent type and control, and *F*-statistics (*F*) and *P*-values (*P*), San Juan County, Utah 2007 and 2008. The perch deterrents tested included: a) cones (Kaddas Enterprises Inc., Salt Lake City, UT), b) mini zenas (Prommel Enterprises Inc., Odenville, AL), and c) triangles (Kaddas Enterprises Inc., Salt Lake City, UT).

	2007						2008					
Species	Cones	Mini zena	Triangles	Control			Cones	Mini zena	Triangles	Control		
	n	n	n	n	*F*	*p*	n	n	n	n	*F*	*p*
Golden eagle	42	35	51	67	2.6	0.07	33	24	19	34	1.2	0.33
Common raven	9	13	8	7	0.5	0.46	1	6	3	8	1.5	0.22
Rough-legged hawk	5	6	3	7	0.5	0.71	2	2	1	3	0.2	0.91
Total	56	54	62	81	1.2	0.31	36	32	23	45	1.9	0.15

Fig. 4.2a.

Fig. 4.2b.

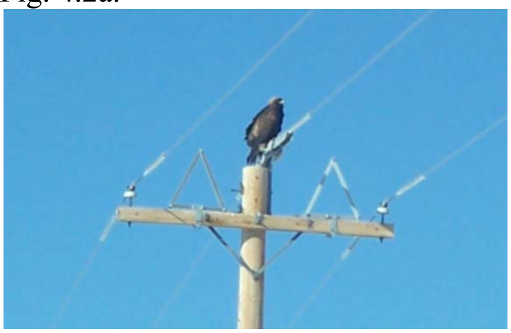

Fig. 4.2c.

Fig. 4.2d

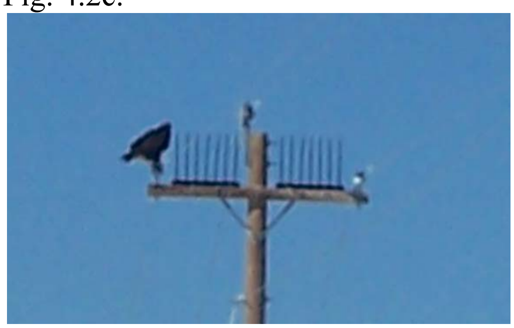

Fig. 4.2e.

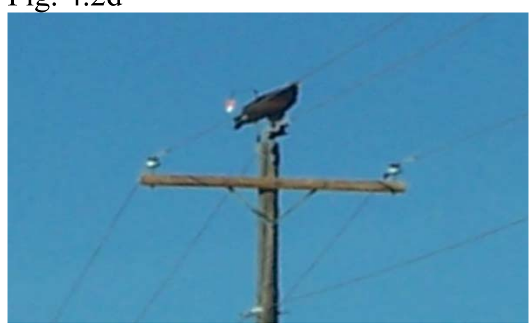

Fig. 4.2f.

Figures 4.2a-f. Typical golden eagle perching events documented relative to perch deterrent type on power distribution poles: a) cones (Kaddas Enterprises Inc., Salt Lake City, UT), b) mini zenas (Prommel Enterprises Inc., Odenville, AL), c) triangles (Kaddas Enterprises Inc., Salt Lake City, UT), d) 2-FireFly™ (P and R Technologies Inc., Portland, OR) arrangement, e) mini zenas, and f) 1-FireFly™ arrangement, San Juan County, Utah, 2007 and 2008.

The deterrent they tested was designed for discomfort and placed on parts of towers where avian predators would most likely perch.

Effectiveness of perch deterrents we evaluated may have been affected by the structure of power poles and the basic design and placement of deterrents. Perch deterrents we tested were partially successful in that they had the ability to prevent perching on parts of the poles. However, birds continued to perch on parts of the poles without deterrents, such as insulators. A perch deterrent that covers insulators, in combination with physical deterrents we tested, has potential to prevent perching of avian predators on power poles of distribution lines.

Before any further evaluation of FireFly as a perch deterrent we recommend the current design be modified. Modifications should include increased durability of plastic reflectors, stronger support bases, and swivel connections that can better withstand weather extremes.

MANAGEMENT IMPLICATIONS

We found that current commercially available perch deterrents used to prevent avian species electrocutions did not mitigate potential avian predators of sage-grouse from perching on poles of a distribution line. For the perch deterrents we evaluated to be successful, they would need to be redesigned to retrofit all parts of the pole, including insulators, rather than just the cross arm. Deterrents must also be designed to better withstand weather extremes.

LITERATURE CITED

Avian Power Line Interaction Committee. 2006. Suggested practices for avian protection on power lines—the state of the art in 2006. Edison Electric Institute, Washington, D.C., USA.

Avery, M.L., and A.C. Genchi. 2004. Avian perching deterrents on ultrasonic sensors at airport wind-shear alert systems. Wildlife Society Bulletin 32:718–725.

Connelly, J.W., A.D. Apa, R.B. Smith, and K.P. Reese. 2000a. Effects of predation and hunting on adult sage-grouse (*Centrocercus urophasianus*) in Idaho. Wildlife Biology 6:227–232.

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000b. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967–985.

Ellis, K.L. 1984. Behavior of lekking greater sage-grouse in response to a perched golden eagle. Western Birds 15:37–38.

Fletcher, Q.E., C.W. Dockrill, D.J. Saher, and C.L. Aldridge. 2003. Northern harrier, *Circus cyaneus*, attacks on greater sage-grouse, *Centrocercus urophasianus*, in southern Alberta. Canadian Field Naturalist 117:479–480.

Fuller, M.R., and J.A. Mosher. 1987. Raptor survey techniques. Pages 37–65 in B. G. Pendleton, B. A. Millsap, K. W. Cline, and D. M. Bird, editors. Raptor management techniques manual. National Wildlife Federation Scientific and Technical Series 10, Washington, D.C., USA.

Hartzler, J.E. 1974. Predation and the daily timing of greater sage-grouse leks. Auk

91:532–536.

Janss, G.F.E., and M. Ferrer. 1999. Mitigation of raptor electrocution on steel power poles. Wildlife Society Bulletin 27:263273.

Knight, R.L., and J.Y. Kawashima. 1993. Responses of raven and red-tailed hawk populations to linear rights-of-ways. Journal of Wildlife Management 57:266–271.

Knight, R.L., H.A.L. Knight, and R.J. Camp. 1995. Common ravens and number and types of linear rights-of-way. Biological Conservation 4:65–67.

Kochert, M.N., and R.R. Olendorff. 1999. Creating raptor benefits from power line problems. Journal of Raptor Research 33:39–42.

Lammers, W.M., and M.W. Collopy. 2007. Effectiveness of avian predator perch deterrents on electric transmission lines. Journal of Wildlife Management 71:2752–2758.

Lupis, S.G. 2005. Summer ecology of Gunnison sage-grouse (*Centrocercus minimus*) in San Juan County, Utah. Thesis, Utah State University, Logan, USA.

Manzer, D.L., and S.J. Hannon. 2005. Relating grouse nest success and corvid density to habitat: a multi-scale approach. Journal of Wildlife Management 69:110–123.

Michener, H. 1928. Where engineer and ornithologist meet: transmission line troubles caused by birds. Condor 30:169–175. San Juan County Gunnison Sage-Grouse Working Group [SWOG]. 2005. Gunnison sage-grouse annual report. San Juan County, Utah, USA.

Stahlecker, D.W. 1978. Effect of a new transmission line on wintering raptors. Condor

80:444–446.

Ward, S. 2007. Gunnison sage-grouse winter ecology and arthropod abundance and diversity in relation to habitat use in San Juan County, Utah. Thesis, Utah State University, Logan, USA.

CHAPTER 5

CONCLUSIONS

Gunnison sage-grouse (*Centrocercus minimus*) currently occupy 8.5% of their presumed historical range (Schroeder et al. 2004). The decline has been attributed to the loss or conversion of sagebrush (*Artemisia* spp.) to other land uses. The quality of the remaining habitat has been impacted by urbanization, grazing, agriculture and fragmentation. As a result, the Gunnison sage-grouse is limited to seven known populations in Colorado and one population in southeastern Utah (GSRSC 2005). The only known Gunnison sage-grouse population in Utah occurs in San Juan County, Utah, near the town of Monticello.

The San Juan County Gunnison Sage-grouse Working Group (SWOG) previously designated an area northeast of the town of Monticello as a sage-grouse priority conservation area (SWOG 2000). The Conservation Area (CA) consisted of 1,392,812 ha, 38% (127,170 ha) of which is privately owned. The CA was identified by encompassing historic and current lek sites, potentially suitable sage-grouse habitat, and sage-grouse observations. The CA is characterized by large fields enrolled in the Conservation Reserve Program (CRP), agricultural fields, and grazed rangelands interspersed with fragmented patches of Wyoming big sagebrush (*A. tridentata wyomingensis*), black sagebrush (*A. nova*), pinyon pine (*Pinus edulis*), juniper (*Juniperus osteosperma*), and oak (*Quercus gambelii*). Within the CA, SWOG also identified a Core Conservation Area (CCA) that consisted of 136,249 ha, of which 89% (88,420 ha) was privately owned. Within the CCA, a Conservation Study Area (CSA) was also identified.

The CSA consisted of 24,177 ha, of which 93% (22,556 ha) was privately owned. The CSA encompassed currently occupied habitat (Lupis 2005, Ward 2007).

The Gunnison Sage-grouse Rangewide Conservation Plan (RCP) and the San Juan County Gunnison Sage-grouse Conservation Plan (SJCCP) recommend management strategies to address identified conservation threats to the San Juan County population (SWOG 2000, GSRSC 2005). Current management of Gunnison sage-grouse in San Juan County, Utah, was based on studies that gathered information regarding the population's life history, habitat use, and movement patterns (Lupis 2005, Ward 2007). This information was used by the Monticello/Dove Creek Local Working Group to guide conservation and management strategies stated within the SJCCP. The research I conducted addressed three conservation strategies identified in the SJCCP: 1) the creation and enhancement of brood-rearing areas; 2) the assessment of habitat conditions within the CA; and 3) the prevention or reduction of perching events by raptors and corvids on distribution line power poles.

The RCP and the SJCCP identified protection and enhancement of mesic brood-rearing habitats as a priority conservation strategy. Increased availability of forbs and arthropods in brood-rearing habitats has been positively associated with survival and recruitment of sage-grouse chicks (Peterson 1970, Wallestad 1971, Klott and Lindzey 1990, Johnson and Boyce 1990, Sveum et al. 1998, Connelly et al. 2000, Crawford et al. 2004). From 2007-2009, I evaluated the role of irrigation in creating mesic or wet meadow environments and dormant season grazing by cattle on habitat quality as measured by changes in vegetation structure and composition, arthropod abundance and diversity, and sage-grouse use. I conducted the experiment on 24 randomly selected 0.1 ha plots located in agricultural lands enrolled in CRP and native sagebrush.

Observationally, the vegetation in the irrigated plots remained greener longer through the season than in the non-irrigated plots, but vegetation diversity did not differ ($p>0.01$). The CRP plots exhibited greater arthropod abundance and cover of perennial grass than the native sagebrush plots, but lower diversity of perennial grasses and abundance and diversity of forbs ($p<0.01$). Crested wheatgrass (*Agropyron cristatum*) was the dominant species in the CRP plots and may have out-competed native forbs and grasses (Hull and Klomp 1967, Schuman et al. 1982, Henderson and Naeth 2005). Dormant season grazing of the CRP plots did not reduce crested wheatgrass cover but did eventually remove residual growth from previous seasons. Lastly, I did not detect any increased sage-grouse use of the treatment plots. This observation may be an artifact of the small plot size and isolated locations.

The increased arthropod abundance in CRP plots relative to the native sagebrush plots and the increased greenness of vegetation because of irrigation suggests a role for irrigation in managing these areas as brood-rearing habitats. The sprinkler irrigation system used in this study allowed quantification of water application rates. However, because of frequent winds, this system did not provide uniform plot coverage and may have resulted in increased evaporation. Thus, creation of mesic areas in brood-rearing habitats may best be accomplished by a system of terraces, ditch plugs or small check dams that retain moisture longer, and by providing flood irrigation. To increase forb and grass diversity in CRP, managers should evaluate the use of mechanical treatments, coupled with spring grazing and reseeding to mitigate the potential competitive effects of crested wheatgrass.

The second conservation strategy I addressed was the assessment of habitat conditions within the CA. In the summer of 2009 I used randomly generated points to measure vegetation conditions within habitat unoccupied and currently occupied by Gunnison sage-grouse. I compared the measured vegetation characteristics with the criteria for desired vegetation conditions outlined within the RCP. The results of the habitat assessment showed that sage-grouse movement and habitat use may be restricted to the CSA by the presence of pinyon-juniper and oak woodlands that surround the area. Because the woodlands occupy larger areas surrounding the CSA and do not provide sagebrush habitats, they may impede population exchanges between Utah and Colorado. These wooded areas are also avoided by the grouse because they provide perch sites for avian predators. These observations highlight the need to improve the habitat within the CSA to maximize the benefits of the habitat the grouse have available to them. Once habitat quality in the CSA approaches SJCCP and RCP guidelines, management actions should focus on opening corridors through these woodlands to facilitate population interchange. In the meantime, managers should consider species translocation between both Colorado and Utah to increase the genetic diversity in both populations.

The habitat assessment verified that forb and grass cover in the CSA is below SJCCP and RCP recommendations. Habitat improvement projects should be focused on retaining and enhancing the habitat quality of remaining sagebrush areas within the CSA. In particular, management efforts should be renewed to re-establish sagebrush, forb, and grass cover within CRP fields throughout the CSA to expand the habitat available to the grouse.

Connelly et al. (2000, Connelly et al., Western Association of Fish and Wildlife Agencies, unpublished report) suggested that because of the potential for raptors and corvids to use transmission line towers and distribution line poles as new perches and nest sites, placement of these facilities in seasonal sage-grouse habitats could impact the species through increased predation of adults, juveniles, and nests or result in sage-grouse abandoning sites (Knight and Kawashima 1993, Knight et al. 1995, Kochert and Olendorff 1999). The RCP identified as a priority conservation strategy the retrofitting of distribution line poles with perch deterrents to discourage raptors and corvids from perching. I evaluated the efficacy of five perch deterrents mounted on support poles of an 11-km section of a 12.5-kV distribution line that bisected the CA and habitat occupied by the sage-grouse population. Perch deterrents were mounted on the line in November-December 2006 following a random replicated block design that included controls. During 168 hours and 84 hours of direct observation in 2007 and 2008, respectively, I recorded 276 and 139 perching events of 7 potential avian predators of sage-grouse. Golden eagles (*Aquila chrysaetos*) were the dominant species recorded during both years. I did not detect any difference in perching events by perch deterrent we evaluated and controls ($p > 0.05$).

The effectiveness of perch deterrents evaluated may have been compromised by the structure of power poles and the basic design and placement of deterrents. The perch deterrents tested were partially successful in that they had the ability to prevent perching on parts of the poles. However birds continued to perch on parts of the poles without deterrents, such as insulators. A perch deterrent that covers insulators, in combination with the physical deterrents tested, may increase the potential to prevent perching of avian predators on power poles of distribution lines.

The results of these studies will help update the information within the RCP and the SJCCP. The results can also be used by the Monticello/Dove Creek Local Working Group to plan future conservation activities within the CA. These studies provided a sound first step that can be built upon to improve habitat conditions within the CA and to reduce the threat of avian predation. Future work should take these results and expand them to larger scale projects.

LITERATURE CITED

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Crawford, J.A., R.A. Olson, N.E. West, J.C. Mosley, M.A. Schroeder, T.D. Whitson, R.F. Miller, M.A. Gregg, and C.S. Boyd. 2004. Ecology and management of sage-grouse and sage-grouse habitat. Journal of Range Management 57:2-19.

Gunnison Sage-grouse Rangewide Steering Committee. 2005. Gunnison Sage-grouse rangewide conservation plan. Colorado Division of Wildlife, Denver, USA.

Henderson, D.C., and M.A. Naeth. 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. Biological Invasions 7:639-650.

Hull, A.C., and G.J. Klomp. 1967. Thickening and spread of crested wheatgrass stands on southern Idaho ranges. Journal of Range Management 20:222-227.

Johnson, G.D., and M.S. Boyce. 1990. Feeding trials with insect in the diet of sage-grouse chicks. Journal of Wildlife Management 54:88-91.

Klott, J.H., and F.G. Lindzey. 1990. Brood habitat of sympatric sage grouse and Columbian Sharp-tailed grouse in Wyoming. Journal of Wildlife Management 54:84-88.

Knight, R.L., and J.Y. Kawashima. 1993. Responses of raven and red-tailed hawk populations to linear rights-of-ways. Journal of Wildlife Management 57: 266-271.

Knight, R.L., H.A.L. Knight, and R.J. Camp. 1995. Common Ravens and number and types of linear rights-of-way. Biological Conservation 4: 65-67.

Kochert, M.N., and R.R. Olendorff. 1999. Creating raptor benefits from power line problems. Journal of Raptor Research 33: 39-42.

Lupis, S.G. 2005. Summer ecology of Gunnison Sage-grouse (*Centrocercus minimus*) in San Juan County, Utah. Thesis, Utah State University, Logan, USA.

Peterson, J.G. 1970. The food habits and summer distribution of juvenile sage-grouse in central Montana. Journal of Wildlife Management 34:147-155.

San Juan County Gunnison Sage-grouse Working Group (SWOG). 2000. Gunnison Sage-grouse Conservation Plan, San Juan County, Utah.

Schroeder, M.A., C.L. Aldridge, A.D. Apa, J.R. Bohne, C.E. Braun, S.D. Bunnell, J.W. Connelly, P.A. Deibert, S.C. Gardner, MA. Hilliard, G.D. Kobriger, S.M. McAdam, C.W. McCarthy, J.J. McCarthy, D.L. Mitchell, E.V. Rickerson, and

S.J. Stiver. 2004. Distribution of sage-grouse in North America. The Condor 106:363-376.

Schuman, G.E., F. Rauzi, and D.T. Booth. 1982. Production o Russian wildrye and crested wheatgrass and their effect on prairie soils. Agronomy Journal 74:23-26.

Sveum, C.M., J.A. Crawford, and W.D. Edge. 1998. Use and selection of brood-rearing habitats by sage-grouse in south central Washington. Great Basin Naturalist 58:344-351.

Wallestad, R.O. 1971. Summer movements and habitat use by sage grouse broods in central Montana. Journal of Wildlife Management 35:129-136.

Ward, S. 2007. Gunnison sage-grouse winter ecology and arthropod abundance diversity in relation to habitat use in San Juan County, Utah. Thesis, Utah State University, Logan, USA.

APPENDICES

Table A.1. Vegetation mixture seeded on Conservation Reserve Program (CRP) lands within the Gunnison Sage-grouse Conservation Area in San Juan County, Utah (SWOG 2000).

Species	PLS lbs/acre
Grasses	
Bluebunch wheatgrass	1.0
Thickspike wheatgrass	1.0
Western wheatgrass	1.5
Crested wheatgrass	0.5
Pubescent wheatgrass	1.0
Legumes/Forbs	
Alfalfa (Rambler)	1.0
Alfalfa (Ladak, Normad)	1.5
Western yarrow	0.12
Lewis flax	0.25
Sainfoin	0.5
Small burnet	2.0
Shrubs	
Wyoming big sagebrush	0.5
Forage kochia	0.5
Total	11.37

Table A.2. Shrubs, perennial grasses, annual grasses, and forbs measured in Conservation Reserve Program (CRP) and native sagebrush within the Gunnison Sage-grouse Conservation Study Area during the summers of 2007, 2008, and 2009. San Juan County, Utah.

		CRP plots	Sagebrush patches within CRP	Sagebrush plots
Perennial grass				
Crested wheatgrass	*Agropyron cristatum*	x	x	x
Blue grama	*Bouteloua gracilis*			x
Foxtail barley	*Hordeum jubatum*			x
Annual grass				
Cheat grass	*Bromus tectorum*	x	x	x
Forbs				
Russian knapweed	*Centaurea repens*	x		
Scaly globemallow	*Sphaeralcea leptophylla*		x	x
Goatsbeard	*Tragopogon dubius*	x		
Basin daisy	*Erigeron pulcherrimus*		x	x
Pale evening primrose	*Oenothera pallida*		x	x
Spreading daisy	*Erigeron divergens*		x	x
Cisco woody aster	*Xylorhia venusta*		x	x
African mustard	*Malcomia africana*	x	x	
Sulphur buckwheat	*Eriogonum umbellatum*		x	x
Vetch	*Astragalus* sp.			x
Heronsbill	*Erodium cicutarium*		x	x
Uinta groundsel	*Senecio multilobatus*			x
Russian thistle	*Salsola pestifer*	x		x
Common purslane	*Portulaca oleracea*			x
Hairy golden aster	*Heterotheca villosa*		x	x
Alfalfa	*Medicago polymorpha*	x		
Foothill deathcamas	*Zigadenus paniculatus*			x
Cryptantha	*Cryptantha* sp.		x	x
Rose-heath	*Leucelene ericoides*		x	x
Sub-shrub				
Broom snakeweed	*Gutierrezia sarothrae*		x	x
Shrubs				
Rubber rabbitbrush	*Chrysothamnus nauseosus*	x	x	x
Fringed sage	*Artemisia frigida*			x
Spineless horsebush	*Tetradymia canescens*			x
Wyoming big sage	*Artemisia tridentata* spp. wyomingensis		x	x

Table A.3. Tables reporting Type 3 tests of fixed effects and covariance parameter estimates of percent cover, height, and forage production of sagebrush, perennial grass, annual grass and forbs, and arthropod abundance in Conservation Reserve Program and native sagebrush plots for each water treatment (once a week, every 2 weeks, every 3 weeks) and grazing treatment (grazed, not grazed) in 2007, 2008, 2009, San Juan County, Utah (p<0.001).

A.3.1: Type 3 tests of fixed effects and covariance parameter estimates of forage production of perennial grasses, annual grasses, and forbs by Conservation Reserve Program and native sagebrush plots), water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2007, San Juan County, Utah (p<0.01).

		2007					
		Perennial grass		Annual grass		Forbs	
Effect	DF	F	p	F	p	F	p
Habitat	1,6	0.03	0.87	0.51	0.50	11.64	0.01
Water	3,18	0.08	0.97	0.23	0.88	0.21	0.89
Habitat x water	3,18	0.23	0.87	0.26	0.85	0.42	0.74
Grazing	1,24	0.59	0.45	0.91	0.35	2.66	0.12
Habitat x grazing	1,24	4.48	0.05	0.91	0.35	0.46	0.50
Water x grazing	3,24	0.13	0.94	1.31	0.30	0.62	0.61
Habitat x water x grazing	3,24	0.42	0.74	4.14	0.02	0.3	0.83
		Estimate	SE	Estimate	SE	Estimate	SE
Habitat		0.000	.	0.001	0.001	0.000	.
Water x habitat		0.032	0.012	0.003	0.001	0.002	0.001
Water x grazing x habitat		0.000	.	0.000	.	0.000	.
Residual		0.018	0.005	0.002	0.001	0.004	0.001

A.3.2: Type 3 tests of fixed effects and covariance parameter estimates of forage production of perennial grasses, annual grasses, and forbs by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2008, San Juan County, Utah (p<0.01).

		2008					
		Perennial grass		Annual grass		Forbs	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*
Habitat	1,6	23.82	<0.01	2.17	0.19	12.03	0.01
Water	3,18	0.23	0.88	1.86	0.17	0.86	0.48
Habitat x water	3,18	1.38	0.28	4.1	0.02	0.1	0.96
Grazing	1,24	3.43	0.08	4.6	0.04	0.08	0.78
Habitat x grazing	1,24	0.12	0.73	1.79	0.19	0.44	0.51
Water x grazing	3,24	0.5	0.69	0.53	0.67	1.51	0.24
Habitat x water x grazing	3,24	0.55	0.65	0.65	0.59	0.33	0.81
		Estimate	SE	Estimate	SE	Estimate	SE
Habitat		0.004	0.003	0.003	0.002	0.000	.
Water x habitat		0.000	.	0.000	.	0.000	.
Water x grazing x habitat		0.000	.	0.000	.	0.000	0.001
Residual		0.013	0.003	0.007	0.002	0.006	0.000

A.3.3: Type 3 tests of fixed effects and covariance parameter estimates of forage production of perennial grasses, annual grasses, and forbs by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2009, San Juan County, Utah (p<0.01).

		2009					
		Perennial grass		Annual grass		Forbs	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*
Habitat	1,6	12.37	0.01	0.49	0.51	13.12	0.01
Water	3,18	5.39	0.01	0.21	0.89	0.28	0.84
Habitat x water	3,18	0.55	0.65	1.96	0.16	1.74	0.20
Grazing	1,24	0.09	0.77	2.09	0.16	0.02	0.90
Habitat x grazing	1,24	12.51	0.00	0.67	0.42	0.76	0.39
Water x grazing	3,24	0.47	0.70	0.5	0.69	0.19	0.90
Habitat x water x grazing	3,24	0.55	0.65	0.55	0.65	0.57	0.64
		Estimate	SE	Estimate	SE	Estimate	SE
Habitat		0.007	0.006	0.002	0.002	0.002	0.002
Water x habitat		0.000	0.004	0.000	.	0.000	.
Water x grazing x habitat		0.000	.	0.000	.	0.000	.
Residual		0.018	0.005	0.005	0.001	0.008	0.002

A.3.4: Type 3 tests of fixed effects and covariance parameter estimates of percent cover of perennial grasses, annual grasses, forbs, and sagebrush by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2007, San Juan County, Utah (p<0.01).

		2007							
		Perennial grass		Annual grass		Forbs		Sagebrush	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*
Habitat	1,6	3.06	0.13	1.99	0.21	3.6	0.11	67.67	<0.01
Water	3,18	0.86	0.48	1.12	0.37	0.47	0.71	0.41	0.75
Habitat x water	3,18	0.65	0.59	2.35	0.11	0.51	0.68	2.17	0.13
Grazing	1,24	0.58	0.45	2.19	0.15	0.6	0.45	6.6	0.02
Habitat x grazing	1,24	1.58	0.22	1.4	0.25	0.3	0.59	0.77	0.39
Water x grazing	3,24	0.33	0.80	0.74	0.54	0.39	0.76	0.67	0.58
Habitat x water x grazing	3,24	2.16	0.12	0.17	0.92	1.01	0.41	0.34	0.80
Time	1,48	1.77	0.19	8.67	0.01	72.59	<0.01	0.36	0.55
Habitat x time	1,48	0.71	0.40	7.72	0.01	11.97	<0.01	0.33	0.57
Water x time	3,48	1.11	0.35	0.93	0.43	2.16	0.11	0.5	0.68
Habitat x water x time	3,48	1.08	0.37	0.42	0.74	1.91	0.14	0.63	0.60
Grazing x time	1,48	0.66	0.42	1.96	0.17	0.75	0.39	0.34	0.56
Habitat x grazing x time	1,48	4.09	0.05	0.02	0.88	0.81	0.37	0.31	0.58
Water x grazing x time	3,48	0.57	0.64	0.12	0.95	0.32	0.81	0.46	0.71
Habitat x water x grazing x time	3,48	0.09	0.97	0.21	0.89	0.53	0.66	0.65	0.59
		Estimate	SE	Estimate	SE	Estimate	SE	Estimate	SE
Habitat		0.007	0.006	0.009	0.008	0.000	0.001	0.001	0.002
Water x habitat		0.002	0.005	0.004	0.006	0.001	0.003	0.004	0.004
Water x grazing x habitat		0.017	0.006	0.022	0.007	0.012	0.004	0.007	0.004
Residual		0.010	0.002	0.002	0.000	0.004	0.001	0.010	0.002

A.3.5: Type 3 tests of fixed effects and covariance parameter estimates of percent cover of perennial grasses, annual grasses, forbs, and sagebrush by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2008, San Juan County, Utah (p<0.01).

		2008							
		Perennial grass		Annual grass		Forbs		Sagebrush	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*
Habitat	1,6	10.82	0.02	1.12	0.33	37.71	<0.01	86.6	<0.01
Water	3,18	0.58	0.64	2.26	0.12	0.33	0.81	0.06	0.98
Habitat x water	3,18	1.39	0.28	4.07	0.02	1.82	0.18	1.38	0.28
Grazing	1,24	0.3	0.59	4.38	0.05	1.34	0.26	1.44	0.24
Habitat x grazing	1,24	1.96	0.17	0.46	0.51	0.19	0.67	0.04	0.85
Water x grazing	3,24	0.13	0.94	0.12	0.95	0.43	0.74	0.23	0.87
Habitat x water x grazing	3,24	3.15	0.04	0.12	0.95	0.67	0.58	1.41	0.26
Time	1,48	0.06	0.80	11.88	<0.01	4.16	0.05	0.41	0.53
Habitat x time	1,48	1.32	0.26	0.06	0.81	7.97	0.01	0.12	0.73
Water x time	3,48	0.56	0.64	0.97	0.42	0.63	0.60	1.09	0.36
Habitat x water x time	3,48	0.2	0.89	2.3	0.09	0.53	0.67	0.93	0.43
Grazing x time	1,48	0.44	0.51	0.86	0.36	1.68	0.20	0.23	0.64
Habitat x grazing x time	1,48	0.03	0.87	0.01	0.91	0.23	0.63	0.11	0.74
Water x grazing x time	3,48	2.43	0.08	1.04	0.38	1.6	0.20	0.37	0.78
Habitat x water x grazing x time	3,48	0.53	0.67	0.21	0.89	0.87	0.46	0.59	0.62
		Estimate	SE	Estimate	SE	Estimate	SE	Estimate	SE
Habitat		0.003	0.004	0.017	0.014	0.000	0.001	0.000	.
Water x habitat		0.000	.	0.009	0.009	0.000	.	0.001	0.003
Water x grazing x habitat		0.016	0.005	0.022	0.009	0.006	0.002	0.009	0.004
Residual		0.010	0.002	0.016	0.003	0.006	0.001	0.006	0.001

A.3.6: Type 3 tests of fixed effects and covariance parameter estimates of percent cover of perennial grasses, annual grasses, forbs, and sagebrush by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2009, San Juan County, Utah (p<0.01).

		2009							
		Perennial grass		Annual grass		Forbs		Sagebrush	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*
Habitat	1,6	24.8	<0.01	0.25	0.63	15.72	<0.01	43.73	<0.01
Water	3,18	0.15	0.93	3.93	0.03	0.17	0.91	1.31	0.30
Habitat x water	3,18	0.4	0.75	3.35	0.04	1.01	0.41	2.4	0.10
Grazing	1,24	0.11	0.74	1.55	0.23	1.56	0.22	2.25	0.15
Habitat x grazing	1,24	1.62	0.22	0	0.97	0.51	0.48	0.38	0.54
Water x grazing	3,24	1.52	0.24	2.39	0.09	0.7	0.56	0.68	0.57
Habitat x water x grazing	3,24	4.59	0.01	1.87	0.16	0.75	0.53	0.15	0.93
Time	1,48	4.29	0.04	0.75	0.39	0.5	0.48	2.9	0.10
Habitat x time	1,48	0.05	0.82	0.94	0.34	0.4	0.53	2.06	0.16
Water x time	3,48	1.01	0.40	0.4	0.75	1.04	0.38	2.63	0.06
Habitat x water x time	3,48	0.52	0.67	1.2	0.32	2.73	0.05	2.17	0.10
Grazing x time	1,48	0.43	0.52	0	0.95	0.17	0.68	0.25	0.62
Habitat x grazing x time	1,48	0.75	0.39	0	0.95	1.49	0.23	0.58	0.45
Water x grazing x time	3,48	1.18	0.33	2.03	0.12	1.25	0.30	0.06	0.98
Habitat x water x grazing x time	3,48	2.26	0.09	1.35	0.27	0.82	0.49	0.16	0.92
		Estimate	SE	Estimate	SE	Estimate	SE	Estimate	SE
Habitat		0.000	.	0.003	0.003	0.000	0.001	0.004	0.004
Water x habitat		0.004	0.002	0.003	0.004	0.002	0.003	0.002	0.004
Water x grazing x habitat		0.000	.	0.006	0.004	0.007	0.003	0.009	0.005
Residual		0.017	0.003	0.011	0.002	0.005	0.001	0.015	0.003

A.3.7: Type 3 tests of fixed effects and covariance parameter estimates of height of perennial grasses, annual grasses, forbs, and sagebrush by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2007, San Juan County, Utah (p<0.01).

		2007							
		Perennial grass		Annual grass		Forbs		Sagebrush	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*
Habitat	1,6	34.98	<0.01	1.54	0.26	1.06	0.34	77.38	<0.01
Water	3,18	1.72	0.20	2.58	0.09	0.38	0.77	0.49	0.69
Habitat x water	3,18	0.42	0.74	2.07	0.14	1.37	0.28	0.43	0.74
Grazing	1,24	1.34	0.26	1.07	0.31	2.75	0.11	2.97	0.10
Habitat x grazing	1,24	1.50	0.23	0.03	0.85	3.56	0.07	2.92	0.10
Water x grazing	3,24	0.10	0.96	0.48	0.70	0.24	0.87	0.81	0.50
Habitat x water x grazing	3,24	1.21	0.32	0.12	0.95	0.22	0.88	0.08	0.97
Time	1,48	19.38	<0.01	7.78	0.01	23.11	<0.01	0.72	0.40
Habitat x time	1,48	0.10	0.75	1.97	0.17	4.71	0.04	1.18	0.28
Water x time	3,48	0.25	0.86	2.30	0.09	1.08	0.37	0.82	0.49
Habitat x water x time	3,48	0.60	0.62	0.54	0.66	0.47	0.70	2.67	0.06
Grazing x time	1,48	1.57	0.22	1.08	0.30	0.34	0.56	0	0.95
Habitat x grazing x time	1,48	0.31	0.58	1.99	0.17	0.08	0.78	0.13	0.72
Water x grazing x time	3,48	2.02	0.12	1.14	0.34	1.18	0.33	0.35	0.79
Habitat x water x grazing x time	3,48	0.37	0.77	0.30	0.83	0.79	0.51	0.48	0.70
		Estimate	SE	Estimate	SE	Estimate	SE	Estimate	SE
Habitat		0.001	0.001	0.007	0.006	0.001	0.001	0.005	0.004
Water x habitat		0.000	.	0.004	0.003	0.000	.	0.000	.
Water x grazing x habitat		0.000	.	0.005	0.003	0.000	.	0.003	0.004
Residual		0.007	0.001	0.010	0.002	0.010	0.002	0.025	0.005

A.3.8: Type 3 tests of fixed effects and covariance parameter estimates of height of perennial grasses, annual grasses, forbs, and sagebrush by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2008, San Juan County, Utah (p<0.01).

		2008							
		Perennial grass		Annual grass		Forbs		Sagebrush	
Effect	DF	F	p	F	p	F	p	F	p
Habitat	1,6	12.56	0.01	0.38	0.56	48.33	<0.01	52.86	<0.01
Water	3,18	0.47	0.70	2.51	0.09	1.1	0.37	0.52	0.67
Habitat x water	3,18	0.48	0.70	2.91	0.06	0.29	0.83	0.25	0.86
Grazing	1,24	0.22	0.64	3.36	0.08	1.5	0.23	0.65	0.44
Habitat x grazing	1,24	6.37	0.02	0.08	0.78	4.37	0.05	2.85	0.10
Water x grazing	3,24	1.27	0.31	0.14	0.94	0.25	0.86	0.98	0.42
Habitat x water x grazing	3,24	1.55	0.23	0.03	0.99	1.14	0.35	0.39	0.76
Time	1,48	31.36	<0.01	17.45	<0.01	0.38	0.54	9.99	<0.01
Habitat x time	1,48	22.31	<0.01	0.57	0.45	0.12	0.73	0.6	0.44
Water x time	3,48	0.36	0.79	1.81	0.16	0.1	0.96	0.19	0.90
Habitat x water x time	3,48	1.79	0.16	1.63	0.19	0.48	0.70	1.12	0.35
Grazing x time	1,48	0.24	0.63	0.83	0.37	1.7	0.20	0.07	0.79
Habitat x grazing x time	1,48	1.39	0.24	1.08	0.30	0	0.98	0.39	0.53
Water x grazing x time	3,48	0.86	0.47	2.51	0.07	2.09	0.11	0.77	0.51
Habitat x water x grazing x time	3,48	2.04	0.12	0.08	0.97	3.24	0.03	0.93	0.43
		Estimate	SE	Estimate	SE	Estimate	SE	Estimate	SE
Habitat		0.002	0.002	0.005	0.004	0.000	0.000	0.004	0.006
Water x habitat		0.000	.	0.004	0.004	0.000	.	0.011	0.008
Water x grazing x habitat		0.006	0.002	0.010	0.004	0.002	0.001	0.018	0.006
Residual		0.004	0.001	0.005	0.001	0.004	0.001	0.006	0.001

A.3.9: Type 3 tests of fixed effects and covariance parameter estimates of height of perennial grasses, annual grasses, forbs, and sagebrush by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2009, San Juan County, Utah (p<0.01).

		2009							
		Perennial grass		Annual grass		Forbs		Sagebrush	
Effect	DF	*F*	*P*	*F*	*p*	*F*	*p*	*F*	*p*
Habitat	1,6	14.66	0.01	0	0.98	8.82	0.03	31.12	<0.01
Water	3,18	1.4	0.27	6.29	<0.01	0.38	0.77	0.18	0.91
Habitat x water	3,18	0.09	0.97	4.04	0.02	2.47	0.10	0.33	0.81
Grazing	1,24	1.12	0.30	1.93	0.18	0.2	0.66	4.63	0.04
Habitat x grazing	1,24	12.61	<0.01	0.62	0.44	0.46	0.50	7.91	0.01
Water x grazing	3,24	0.64	0.60	0.67	0.58	0.04	0.99	1.91	0.16
Habitat x water x grazing	3,24	0.67	0.58	2.3	0.10	0.68	0.58	1.97	0.15
Time	1,48	17.16	<0.01	26.31	<0.01	0.21	0.65	11.00	<0.01
Habitat x time	1,48	0.21	0.65	1.47	0.23	1.16	0.29	11.00	<0.01
Water x time	3,48	1.23	0.31	0.6	0.62	2.14	0.11	1.03	0.39
Habitat x water x time	3,48	1.61	0.20	1.81	0.16	0.57	0.64	1.03	0.39
Grazing x time	1,48	0.08	0.78	1.03	0.32	0.33	0.57	0.00	0.99
Habitat x grazing x time	1,48	0.03	0.87	0.01	0.94	0.13	0.73	0.00	0.99
Water x grazing x time	3,48	0.49	0.69	1.46	0.24	1.07	0.37	0.43	0.74
Habitat x water x grazing x time	3,48	0.13	0.94	0.23	0.88	0.79	0.50	0.43	0.74
		Estimate	SE	Estimate	SE	Estimate	SE	Estimate	SE
Habitat		0.002	0.002	0.003	0.002	0.000	0.001	0.013	0.010
Water x habitat		0.002	0.002	0.002	0.001	0.002	0.001	0.008	0.007
Water x grazing x habitat		0.000	0.001	0.002	0.001	0.001	0.001	0.021	0.006
Residual		0.007	0.001	0.005	0.001	0.003	0.001	0.002	0.000

A.3.10: Type 3 tests of fixed effects arthropod orders by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2007, San Juan County, Utah (p<0.01).

		2007									
		Aranae		Coleoptera		Diptera		Hemiptera		Homoptera	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*P*	*F*	*p*
Habitat	1,6	8.10	0.03	1.46	0.27	23.40	<0.01	2.33	0.18	0.20	0.67
Water	3,18	1.12	0.37	0.22	0.88	0.17	0.91	1.29	0.31	0.35	0.79
Habitat x water	3,18	0.06	0.98	0.38	0.77	1.73	0.20	0.67	0.58	2.29	0.11
Grazing	1,24	0.11	0.75	6.80	0.04	0.07	0.81	1.12	0.33	2.17	0.19
Habitat x grazing	1,24	0.00	0.95	0.00	0.97	0.71	0.43	1.38	0.28	0.00	0.97
Water x grazing	3,24	0.31	0.82	0.76	0.53	1.75	0.19	1.16	0.35	0.58	0.64
Habitat x water x grazing	3,24	0.07	0.98	0.86	0.48	0.24	0.87	2.03	0.15	1.10	0.37

		Hymenoptera (ants)		Hymenoptera (bees and wasps)		Lepidoptera		Orthoptera	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*P*
Habitat	1,6	0.17	0.70	2.85	0.14	1.02	0.35	45.4	<0.01
Water	3,18	1.03	0.40	0.79	0.52	0.21	0.89	0.25	0.86
Habitat x water	3,18	3.44	0.04	1.86	0.17	0.37	0.78	0.17	0.92
Grazing	1,24	0.59	0.48	0.16	0.70	0.47	0.52	0.06	0.82
Habitat x grazing	1,24	0.52	0.50	0.48	0.51	3.31	0.12	0.11	0.75
Water x grazing	3,24	1.39	0.28	0.53	0.67	0.68	0.57	0.52	0.67
Habitat x water x grazing	3,24	0.02	0.99	1.79	0.19	1.20	0.34	1.61	0.22

A.3.11: Type 3 tests of fixed effects arthropod orders by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2008, San Juan County, Utah (p<0.01).

2008											
		Aranae		Coleoptera		Diptera		Hemiptera		Homoptera	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*P*	*F*	*p*
Habitat	1,6	1.49	0.27	3.61	0.11	0.04	0.85	10.7	0.02	23.9	<0.01
Water	3,18	0.49	0.69	0.68	0.58	2.01	0.15	3.26	0.05	3.36	0.04
Habitat x water	3,18	0.55	0.66	0.75	0.54	0.71	0.56	4.75	0.01	2.26	0.12
Grazing	1,24	0.23	0.65	7.22	0.04	0.55	0.49	0.96	0.37	6.52	0.04
Habitat x grazing	1,24	1.61	0.25	2.08	0.20	1.88	0.22	1.27	0.30	7.65	0.03
Water x grazing	3,24	1.06	0.39	2.20	0.12	1.37	0.29	1.60	0.22	3.94	0.03
Habitat x water x grazing	3,24	1.85	0.17	2.22	0.12	2.01	0.15	2.52	0.09	2.75	0.07
		Hymenoptera (ants)		Hymenoptera (bees and wasps)		Lepidoptera		Orthoptera			
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*P*		
Habitat	1,6	0.09	0.78	0.01	0.94	14.4	0.01	108	<0.01		
Water	3,18	0.76	0.53	0.35	0.79	0.42	0.74	0.28	0.84		
Habitat x water	3,18	0.77	0.53	3.59	0.03	1.31	0.30	2.85	0.07		
Grazing	1,24	3.50	0.11	0.80	0.40	0.45	0.53	0.32	0.59		
Habitat x grazing	1,24	1.26	0.30	0.02	0.91	0.22	0.65	0.04	0.84		
Water x grazing	3,24	1.70	0.20	2.07	0.14	3.48	0.04	0.72	0.55		
Habitat x water x grazing	3,24	0.92	0.45	2.97	0.06	1.05	0.39	0.92	0.45		

A.3.12: Type 3 tests of fixed effects arthropod orders by Conservation Reserve Program and native sagebrush plots, water treatment (once a week, every 2 weeks, every 3 weeks), and grazing treatment (grazed, not grazed) in 2009, San Juan County, Utah (p<0.01).

		2009									
		Aranae		Coleoptera		Diptera		Hemiptera		Homoptera	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*P*	*F*	*p*
Habitat	1,6	4.23	0.09	0.00	0.99	1.04	0.35	6.10	0.05	0.60	0.47
Water	3,18	0.46	0.72	1.28	0.31	0.60	0.63	2.50	0.09	2.49	0.09
Habitat x water	3,18	1.00	0.42	1.11	0.37	1.31	0.30	2.10	0.14	3.12	0.05
Grazing	1,24	0.94	0.37	0.45	0.53	0.03	0.88	0.13	0.73	2.57	0.16
Habitat x grazing	1,24	1.29	0.30	0.42	0.54	0.05	0.83	0.00	0.97	0.08	0.79
Water x grazing	3,24	1.16	0.35	0.89	0.47	0.75	0.54	0.58	0.64	0.99	0.42
Habitat x water x grazing	3,24	1.62	0.22	0.48	0.70	1.25	0.32	0.66	0.59	3.15	0.05

		Hymenoptera (ants)		Hymenoptera (bees and wasps)		Lepidoptera		Orthoptera	
Effect	DF	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*P*
Habitat	1,6	0.15	0.71	0.48	0.52	3.95	0.09	83.6	<0.01
Water	3,18	0.82	0.50	0.27	0.84	1.18	0.35	0.55	0.65
Habitat x water	3,18	0.70	0.56	3.66	0.03	0.09	0.96	0.24	0.87
Grazing	1,24	1.11	0.33	0.47	0.52	1.20	0.32	0.32	0.59
Habitat x grazing	1,24	3.57	0.11	0.10	0.76	1.20	0.32	0.03	0.87
Water x grazing	3,24	0.59	0.63	0.87	0.48	2.20	0.12	3.34	0.04
Habitat x water x grazing	3,24	0.40	0.76	0.36	0.78	0.20	0.90	0.28	0.84

A.4. Copyright release letters of the *Journal of Wildlife Management* and Allen Press allowing the republication of: Prather, P.R. and T.A. Messmer. 2010. Raptor and Corvid Response to Power Distribution Line Perch Deterrents in Utah. *Journal of Wildlife Management* 74(4): 796-800.

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I am preparing my dissertation in the Department of Wildland Resources at Utah State University. I hope to complete my degree in the summer of 2010.

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CURRICULUM VITAE
Phoebe R. Prather
(May 2010)

CARREER OBJECTIVE:

To obtain a position with an organization that requires technical expertise and communication skills. Special areas of interest: habitat management, habitat restoration, and avian, wetland and riparian ecology and management.

EDUCATION:

B.A. in Environmental Studies, University of California at Santa Cruz, Santa Cruz, California. (06/2002). Emphasis in environmental policy and land management. Received honors on undergraduate thesis. Ph.D. in Ecology, Utah State University, Logan, Utah. (expected 07/2010). Grad GPA: 3.63. Dissertation research in Utah 2006-2009.

EXPERIENCE:

GRADUATE RESEARCH ASSISTANT, Utah State University, Ph.D. Candidate in Ecology, Logan, Utah (06/2006-05/2010).

- Designed, implemented, and analyzed statistically valid field experiments to address conservation strategies identified in the San Juan County, Utah Gunnison Sage-grouse Conservation Plan. Collected, compiled, organized and analyzed field data, and presented the results in both written and oral formats.
- Communicated, collaborated and formed relationships with state and federal agencies, non-profit organizations, and private landowners.
- Supervised field technicians and Youth Conservation Corp work crews.

- Surveyed raptor and corvid use of distribution line power poles fitted with perch deterrents.
- Created wet meadow brood-rearing habitats in sagebrush habitats and Conservation Reserve Program fields through the use of irrigation.
- Proficient in diverse vegetation measurement techniques and methodologies.
- Trapped arthropods using pit-fall traps and sorted them to family using a dissecting scope and dichotomous key.
- Prepared and edited annual reports.
- Presented results at stakeholder conferences and professional meetings to diverse audiences.

RESEARCH ASSISTANT, Delta Waterfowl Foundation, Minnedosa Field Station, Manitoba, Canada (04/2005-08/2005).

- Conducted research on nesting mallard hens. Trapped hens using decoy traps, Weller traps, and nesting tunnel traps. Banded, took measurements, collected feathers, and placed nasal markers on captured hens. Maintained a field notebook and field datasheets.
- Nest searched using ATV's and a drag chain.
- Monitored nests of several upland nesting waterfowl species.
- Used the candling technique to age eggs.
- Banded mallard ducklings using plastacine bands.
- Re-sighted Mallard hens with nasal markers using a spotting scope.

- Assisted other graduate student crews based out of the field station conducting research on American Coots, Ruddy Ducks, and Stripped Skunks. Became familiar with radio-telemetry techniques while assisting the skunk crew.

ENDANGERED SHOREBIRD MONITOR AND NATURALIST, Massachusetts Audubon Society, Cummaquid, MA (04/2004-09/2004).

- Worked at the Mass Audubon Sampson's Island/Dead Neck Wildlife Sanctuary, Cape Cod.
- Monitored nesting Piping Plovers. Visited island daily to determine breeding territories. Fenced off territories and posted signs. Constructed predator exclosures around nests. Intensely monitored broods, keeping a record of number of chicks hatched and fledged. Completed daily observation forms.
- Monitored Least and Common Tern nesting colonies. Fenced off nesting colonies. Conducted weekly counts of nests. Constructed and maintained a solar powered electric predator fence.
- Researched, prepared, and conducted weekend nature walks for groups of 2 to 20 people of all ages.
- Conversed with visitors, checked Audubon memberships, sold memberships, collected day fees, and provided information on nesting birds and the methods taken to protect them.
- Used a 13 foot Boston Whaler to reach and patrol the island.

- Volunteered with Massachusetts Fish and Wildlife to conduct Roseate Tern nesting surveys and assist in banding and weighing Common and Roseate Tern chicks.
- Volunteered with Monomoy National Wildlife Refuge to conduct Herring and Great Black-backed Gull nesting survey and to conduct Horseshoe Crab surveys twice a month on night full and new moon tides.
- Supervised high school student volunteers.

RESOURCE ASSISTANT, Student Conservation Association, Arches National Park, Interpretation Division, Moab, Utah (07/2003-10/2003).

- Researched, composed, and presented an evening slideshow program on the natural histories of raptors for an audience of up to 70 people.
- Researched, prepared, and conducted guided hikes and nature walks for groups of up to 30 people.
- Staffed park visitor desk, provided park and area information, aided visitors in trip planning, responsible for the handling of fees, selling tour tickets, reconciling funds, and completing associated paperwork.
- Educated visitors on the desert ecosystems and the challenges that they face.
- Aided in search and rescue incidents.

RESOURCE ASSISTANT, Student Conservation Association, Arches National Park, Resource Management Division, Moab, Utah (03/2003-07/2003).

- Monitored raptor nests of eight species of birds prey, including determining activity and counting number of young and fledged young at 70 historic nesting sites.
- Monitored a Great-Blue Heron Rookery on the Colorado River throughout the nesting season.
- Conducted a weekly Breeding Bird Survey identifying songbirds of the pinyon-juniper habitat by sight and song.
- Assisted researchers in vegetation transects and riparian bird point count surveys.
- Assisted in boundary fencing projects and rehabilitation of areas damaged by off-road vehicle use.
- Researched and composed the life histories for the raptor and songbird species surveyed.
- Used a four-wheel drive vehicle and hiking in extreme summer desert conditions to reach the nest sites.
- Prepared and edited seasonal reports.
- Aided in search and rescue incidents.

INTERNSHIP, Alaska Audubon Society, Anchorage, Alaska (10/2002-12/2002).

- Aided in the development of the Alaska State Important Bird Area (IBA) program.
- Entered data into the World Bird Database for over 200 Important Bird Areas in the Cook Inlet watershed and along the Bering Sea coast.

- Designed criteria for passerine Important Bird Areas with U.S. Fish and Wildlife biologists.
- Researched and prepared a pamphlet aiming to educate the public on the IBA program.
- Researched and composed the life histories for bird species on the Audubon Alaska Watch List.

BIRD BANDER, Denali Institute, Denali National Park, Alaska (07/2002-09/2002).

- Operated a passerine bird banding station in the heart of Denali National Park at the Denali Institute Migration Station.
- Utilized and maintained 10 nets; banded 7 days a week; recorded specific data on each individual.
- Presented an evening program to 20 to 40 lodge guests and conducted hands-on demonstrations for the guests at the banding station.

AWARDS AND HONORS:

- Received Honors on undergraduate thesis entitled, "What the Heck in as IBA: A case study of the Important Bird Area Program in the Owens Valley, California," from the University of California at Santa Cruz in 2002.
- Recipient of the Utah Chapter of the Wildlife Society Annual Scholarship in 2007.

PROFESSIONAL PRESENTATIONS:

- Prather, Phoebe R. and T.A. Messmer. 2006. "Use of Artificial Wet Meadow Areas by Gunnison Sage-grouse (*Centrocercus minimus*) in San Juan County, Utah." Presented at The Utah Chapter of the Wildlife Society Annual Conference in Moab, UT.

- Prather, Phoebe R. and T.A. Messmer. 2007. "Raptor and Corvid Use of Distribution Line Power Poles: An Assessment of the Efficacy of Perch Deterrents." Presented at The Utah Chapter of the Wildlife Society Annual Conference in Moab, UT.
- Prather, Phoebe R. and T.A. Messmer. 2007. "Raptor and Corvid Use of Distribution Line Power Poles: An Assessment of the Efficacy of Perch Deterrents." Presented at The Utah Sage-grouse Summit in Salt Lake City, UT.
- Prather, Phoebe R. and T.A. Messmer. 2008. "Raptor and Corvid Use of Distribution Line Power Poles: An Assessment of the Efficacy of Perch Deterrents." Presented at The Gunnison Sage-grouse Summit in Montrose, CO.
- Prather, Phoebe R. and T.A. Messmer. 2008. "Raptor and Corvid Use of Distribution Line Power Poles: An Assessment of the Efficacy of Perch Deterrents." Presented at The Western Association of Fish and Wildlife Agencies Sage-grouse and Sharp-tailed Grouse Conference in Mammoth Lakes, CA.

RESEARCH PUBLICATIONS AND REPORTS:

- Prather, Phoebe R. and T.A. Messmer. 2007. Annual Report. "Gunnison Sage-grouse (*Centrocercus minimus*) Conservation in Utah: A summary of species and habitat response to conservation strategies identified in the Gunnison Sage-grouse Rangewide Conservation Plan." Department of Wildland Resources, Utah State University, Logan, UT.

- Prather, Phoebe R. and T.A. Messmer. 2008. Annual Report. "Gunnison Sage-grouse (*Centrocercus minimus*) Conservation in Utah: A summary of species and habitat response to conservation strategies identified in the Gunnison Sage-grouse Rangewide Conservation Plan." Department of Wildland Resources, Utah State University, Logan, UT.
- Prather, Phoebe R. and T.A. Messmer. 2009. Annual Report. "Gunnison Sage-grouse (*Centrocercus minimus*) Conservation in Utah: A summary of species and habitat response to conservation strategies identified in the Gunnison Sage-grouse Rangewide Conservation Plan." Department of Wildland Resources, Utah State University, Logan, UT.
- Prather, P.R. and T.A. Messmer. In press. "Raptor and Corvid Response to Power Distribution Line Perch Deterrents in Utah." *Journal of Wildlife Management.*

PROFESSIONAL MEMBERSHIPS:

The Wildlife Society

Utah Chapter of the Wildlife Society

Environmental Management (2016) 58:585–596
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RESEARCH

Does Wyoming's Core Area Policy Protect Winter Habitats for Greater Sage-Grouse?

Kurt T. Smith[1] · Jeffrey L. Beck[1] · Aaron C. Pratt[1]

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Abstract Conservation reserves established to protect important habitat for wildlife species are used world-wide as a wildlife conservation measure. Effective reserves must adequately protect year-round habitats to maintain wildlife populations. Wyoming's Sage-Grouse Core Area policy was established to protect breeding habitats for greater sage-grouse (*Centrocercus urophasianus*). Protecting only one important seasonal habitat could result in loss or degradation of other important habitats and potential declines in local populations. The purpose of our study was to identify the timing of winter habitat use, the extent which individuals breeding in Core Areas used winter habitats, and develop resource selection functions to assess effectiveness of Core Areas in conserving sage-grouse winter habitats in portions of 5 Core Areas in central and north-central Wyoming during winters 2011–2015. We found that use of winter habitats occured over a longer period than current Core Area winter timing stipulations and a substantial amount of winter habitat outside of Core Areas was used by individuals that bred in Core Areas, particularly in smaller Core Areas. Resource selection functions for each study area indicated that sage-grouse were selecting habitats in response to landscapes dominated by big sagebrush and flatter topography similar to other research on sage-grouse winter habitat selection. The substantial portion of sage-grouse locations and predicted probability of selection during winter outside small Core Areas illustrate that winter requirements for sage-grouse are not adequately met by existing Core Areas. Consequently, further considerations for identifying and managing important winter sage-grouse habitats under Wyoming's Core Area Policy are warranted.

Keywords *Centrocercus urophasianus* · Sage-grouse · Resource selection functions · Winter habitat selection · Wyoming sage-grouse core area policy

Introduction

Conservation reserves designed to protect habitats have been established to maintain viable wildlife populations and biodiversity in protected areas. Approximately 14.6 % of Earth's land surface is designated as protected areas for conservation (Butchart et al. 2015). Early advocates of conservation reserves generally regarded that reserve size would predict the reserves ability to maintain species abundance and diversity (e.g., Diamond 1975). However, regardless of size, protected areas may not sufficiently capture habitat needs of a species on a yearly basis. This is particularly the case for species with large home ranges that move between distinct seasonal habitats. Information regarding a species annual distribution and selection of habitats, within and outside of breeding seasons, is necessary when designating protection areas for conserving habitats (Johnson et al. 2004).

One analysis suggests greater sage-grouse (*Centrocercus urophasianus*; hereafter, sage-grouse) occupy approximately 56 % of their potential pre-settlement habitat in 11 states and 2 Canadian provinces and are closely linked to sagebrush (*Artemisia* spp.) habitats (Schroeder et al. 2004). Long-term declines of sage-grouse across much of the

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species range (Connelly and Braun, 1997) are largely attributed to landscape change resulting in direct loss and fragmentation of sagebrush habitats (Braun 1998; Connelly et al. 2004). Land changes including agricultural development (Swenson et al. 1987), energy development (Doherty et al. 2008, 2011; Harju et al. 2010; Gregory and Beck 2014; LeBeau et al. 2014), urban and exurban development (Braun 1998; Connelly et al. 2004), livestock grazing (Beck and Mitchell 2000; Boyd et al, 2014), and fire (Connelly et al. 2000a; Blomberg et al. 2012) have resulted in declining populations, with the effects of different disturbances acting synergistically to influence sage-grouse populations (Hess and Beck 2012).

Wyoming's Sage-Grouse Core Area Policy (hereafter, Core Area) was implemented to limit disturbance (activities that result in removal of sage-grouse habitat) in areas of high sage-grouse breeding population densities by setting disturbance limits and timing stipulations (Doherty et al. 2011; Kiesecker et al. 2011; State of Wyoming 2011). Core Areas were originally defined by Doherty et al. (2011) who delineated priority nesting areas based on proximity of surrounding leks and habitat within 6.4 km of leks. Breeding density areas were modeled by assigning an abundance-weighted density of male sage-grouse to each lek until 75 % (core75) of the population was included. The Wyoming Core Areas represent an adapted version of core75 areas defined by Doherty et al. (2011), modified to incorporate multiple land-use decisions such as leased oil and gas well sites or planned residential development. Build-out scenarios suggest that Core Areas focused on breeding habitats may reduce projected long term sage-grouse population declines (Copeland et al. 2013). However, because breeding habitats may not contain all habitats necessary for survival, protection of crucial habitats must focus on all seasonal requirements for effective sage-grouse conservation (Doherty et al. 2011; Fedy et al. 2012). Winter survival estimates for sage-grouse are generally higher (78–97 %; Beck et al. 2006; Baxter et al. 2013) than annual breeding-age survival rates (58 %; Taylor et al. 2012), but winter survival may be depressed during severe winter conditions (Moynahan et al. 2006; Anthony and Willis 2009). Adult female survival is of critical importance for sage-grouse population viability (Taylor et al, 2012; Dahlgren et al. 2016); consequently, winter survival of females may represent a significant vital rate for population persistence (Moynahan et al. 2006). The effectiveness of Core Areas hinge on their ability to not only protect high quality breeding habitats used by sage-grouse, but also habitats necessary for survival during other seasons.

The Core Area Policy suggests that the majority of winter habitat likely occurs inside Core Areas (State of Wyoming 2011). Approximately 90 % of sage-grouse yearlong habitat use was within 5 km of lek sites in the Bi-State Population in eastern California and western Nevada (Coates et al. 2013), suggesting that breeding habitats include a large portion of year-round habitats for sage-grouse in that region. Conversely, Fedy et al. (2012) found that the average movement of sage-grouse from late summer to winter areas averaged 17.3 km, with 31 to 100 % of winter locations occurring within 100 % Core Areas for 11 study populations distributed across Wyoming indicating that a substantial portion of winter habitat use by sage-grouse populations may occur outside Core Areas. Because habitat selection varies considerably across seasons (e.g., Fedy et al. 2014), Core Areas are unlikely to afford protection for sage-grouse outside of the breeding season unless winter areas are in close proximity to breeding habitats. Also, if winter habitats represent a limiting seasonal habitat within Core Areas, special conservation strategies must be implemented to create additional protection in critical wintering areas.

Seasonal use restrictions are in place to limit disturbance activities in identified winter concentration areas both in and out of Core Areas from 1 December to 15 March (State of Wyoming 2011; BLM 2012). The Wyoming Sage-grouse Executive Order (SGEO) suggests that disturbance in non-Core Areas should be minimized in mature sagebrush habitats in winter concentration areas (State of Wyoming 2011); however, no regulation has been established for these areas explicitly regulating the amount of allowable disturbance.

We used data collected from Global Positioning System (GPS)-marked female sage-grouse across two study areas that overlap portions of the Greater South Pass, Shell, Hyattville, Oregon Basin, and Washakie Core Areas to evaluate how well the Core Area policy protects sage-grouse winter habitats. Specifically, our objectives were to evaluate the effectiveness of Core Areas to protect sage-grouse winter habitats by (1) evaluating the timing of winter habitat use relative to current winter seasonal timing stipulations, (2) determining the portion of winter habitat use of individuals that use breeding habitats within Core Areas, and (3) developing winter resource selection functions (RSFs) for female sage-grouse to determine the amount and arrangement of winter habitats in relation to Core Areas.

Methods

Study Area

The Bighorn Basin study area (3834 km^2) was associated with the Hyattville, Oregon Basin, Shell, and Washakie Core Areas in eastern Big Horn and Washakie counties, and northeastern Hot Springs County, Wyoming (Figs. 1–3).

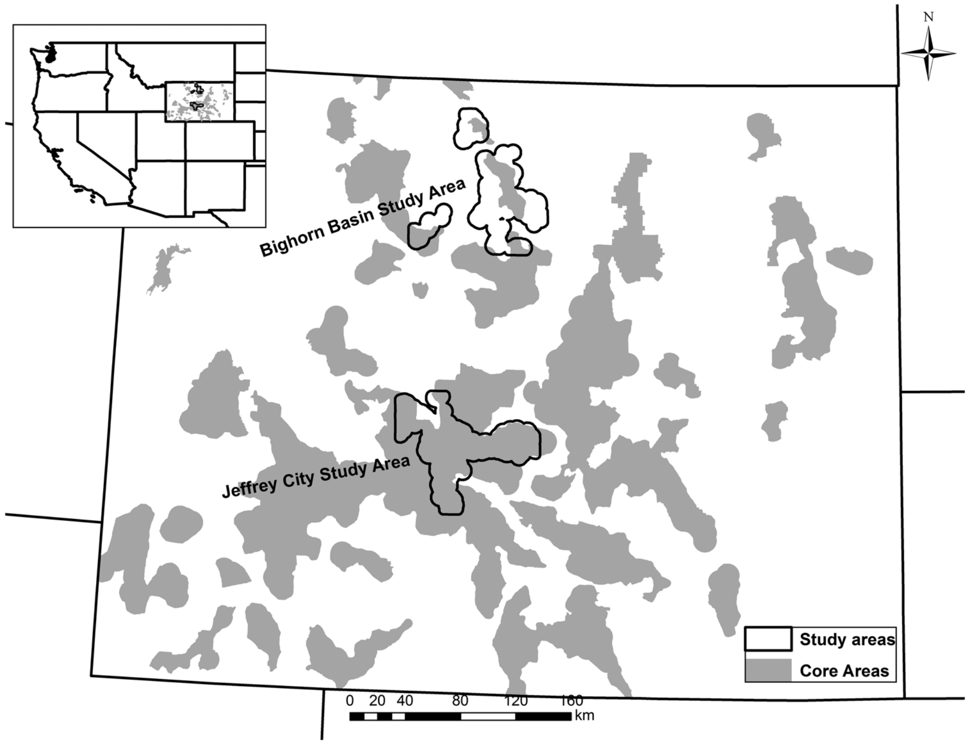

Fig. 1 Map of the two study areas based on 100 % kernel density estimates encompassing winter sage-grouse use locations in central and north-central Wyoming, USA, winters 2011–2015

The area included approximately 78.9 % Federal, 5.2 % State, and 15.9 % privately administered lands. The 30 year normal monthly precipitation averaged from November through March was 8.1 cm and ranged from 8.2 to 10.8 cm during 2011 to 2015 (Prism Climate Group 2016). Elevation ranged from 1157 to 2976 m. Major land uses in this area included bentonite mining, livestock grazing, and a variety of recreational activities. The 4144-km^2 Jeffrey City study area occurred in portions of Fremont, Natrona, and Sweetwater counties, Wyoming, within the Greater South Pass Core Area (Figs. 1 and 4). The area included approximately 82.4 % Federal, 7.3 % State, and 10.3 % privately administered lands. The 30 year normal monthly precipitation averaged from November through March was 6.6 cm and ranged from 5.1 to 7.8 cm during the study period (Prism Climate Group 2016). Elevation ranged from 1529 to 2524 m. Major land uses during the study included livestock grazing. There is interest to resume uranium ore mining that historically occurred in this area. Dominant shrub species that composed the shrub-steppe in both areas include Wyoming big sagebrush (*Artemisia tridentata wyomingensis*), with communities of mountain big sagebrush (*A.t. vaseyana*) at higher elevations. Other shrub species occurring in each area included black sagebrush (*A. nova*), silver sagebrush (*A. cana*), rabbitbrush (*Ericameria nauseosa* and *Chrysothamnus viscidiflorus*), Gardner's saltbush (*Atriplex gardneri*), shadscale saltbush (*Atriplex confertifolia*), and greasewood (*Sarcobatus vermiculatus*).

Field Procedures and Monitoring

We captured and radio-marked female sage-grouse around leks during spring or at roost sites during summer in 2011–2014 by spot-lighting and hoop-netting (Giesen et al, 1982; Wakkinen et al., 1992). We attached GPS transmitters (22-g PTT-100 Solar Argos/GPS PTT [Microwave Telemetry, Columbia, MD, USA] or Model 22 GPS PTT [North Star Science and Technology, King George, VA, USA]) via rump mount. GPS transmitters were solar-powered and uploaded their GPS locations (±~20-m error) to satellites used by the Argos system (CLS America, Largo, MD, USA) every 3 days. Transmitters were programmed to acquire 3 locations per day from 1 November to 14 March (at 0900, 1200, and 1500 local time ignoring Daylight Savings Time), 4 locations per day from 15 March to 30 April and 25 August to 30 October (at 0700, 1000, 1300, 1600), 5 locations per day from 1 May to 24 August (at 0600, 0900, 1200, 1500, 1800), and included an additional location every night at midnight (2400). All applicable international, national, and/or institutional guidelines for the care and use of animals were followed. Sage-grouse were captured, marked, processed, and monitored in adherence with approved protocols (Bighorn Basin study [Wyoming Game and Fish Department Chapter 33–800 permit and University of Wyoming Institutional Animal Care and Use Committee protocols 03142011 and 20140228JB00065]; Jeffrey City study [Wyoming Game and Fish Department

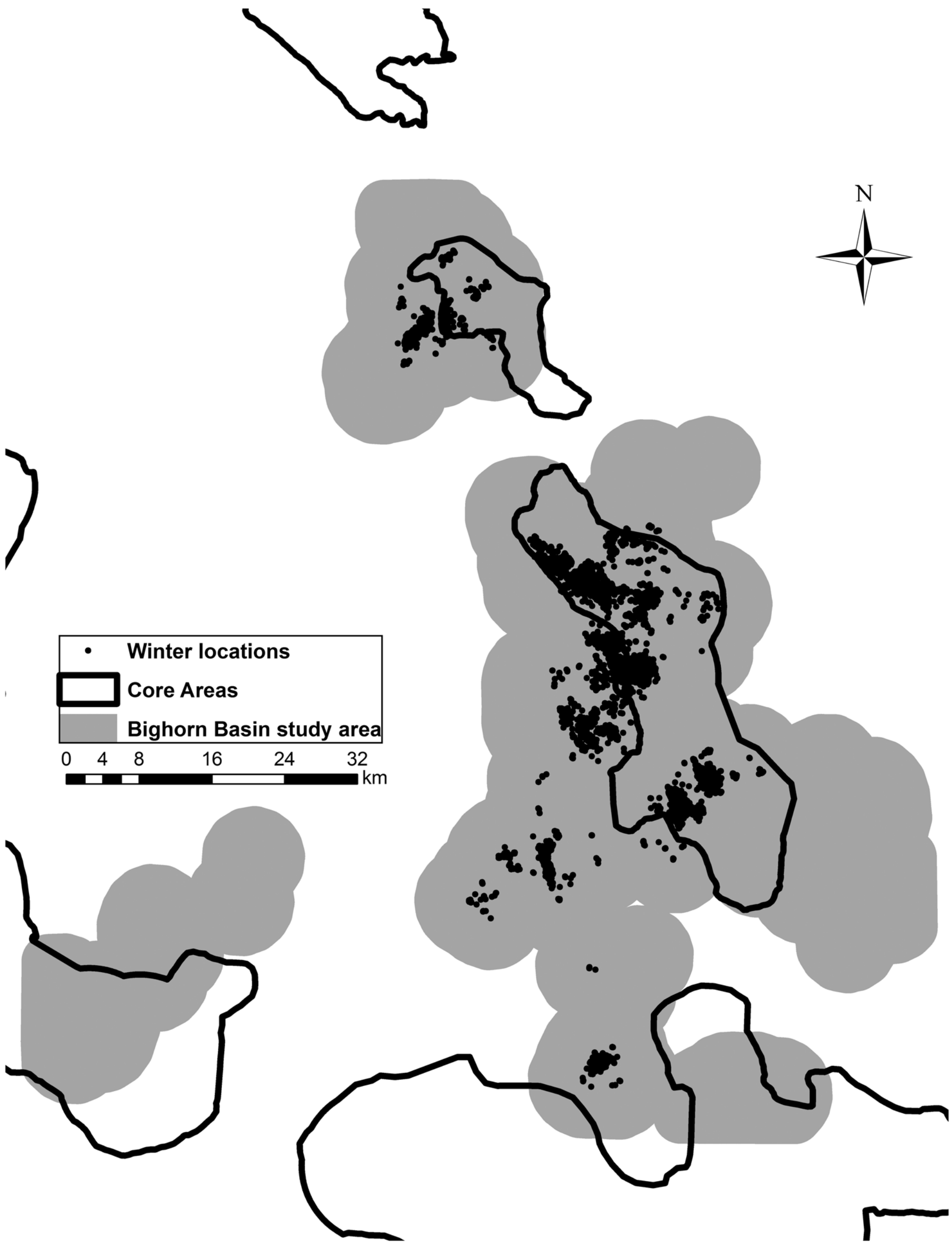

Fig. 2 Winter locations for 17 female greater sage-grouse that nested in Core Areas in the Bighorn Basin study area (3834-km^2), Wyoming, 2011–2015. Winter use locations were based on seasonal movement timing (26 Oct to 21 Mar; $n = 24,311$)

Chapter 33–801 permit and University of Wyoming Institutional Animal Care and Use Committee protocols 03132011 and 20140128JB0059]).

Timing of Winter Habitat Use and Proportion of Winter Use in Core Areas

We defined the winter season based on distinct movements of migratory individuals (≥10 km; Connelly et al. 2000b) between fall and winter, and winter and spring ranges. If individuals did not exhibit distinct movement to winter ranges, we used the average movement timing of migratory sage-grouse in each study area to delineate winter locations for that individual. We defined a female as a Core Area inhabitant if it nested within a Core Area. For those individuals, we determined the proportion of their locations in Core Areas during the winter season.

Sage-Grouse Resource Selection

Landscape Predictor Variables

We used a suite of remotely sensed vegetation, topography, and anthropogenic predictor variables that have been shown

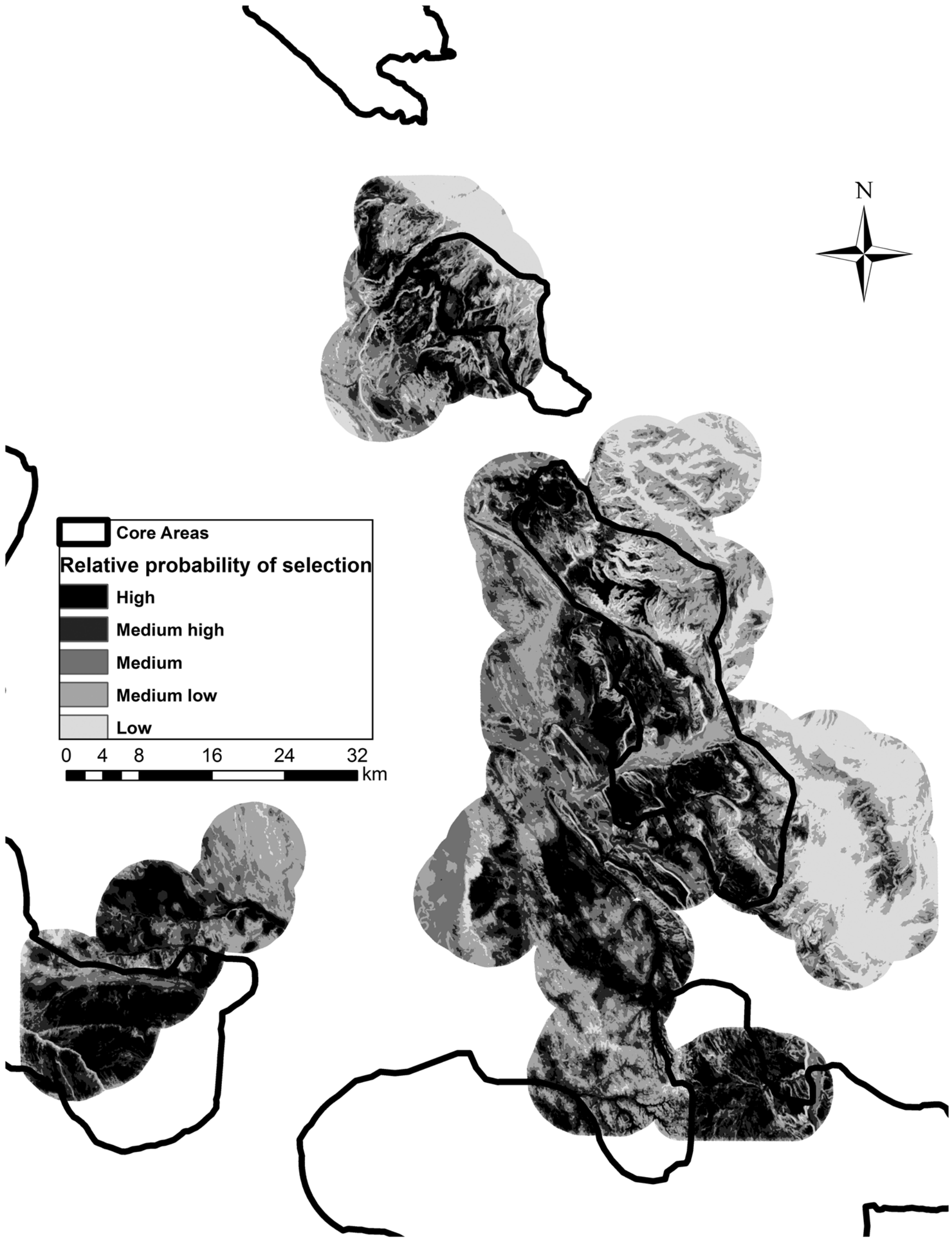

Fig. 3 Predicted probability of sage-grouse winter habitat selection in the 3834-km^2 Bighorn Basin study area during winters 2011–2015. This map spatially depicts a resource selection function that was binned into five quantiles of predicted relative probability of occurrence

to influence sage-grouse winter habitat selection in other studies (Homer et al. 1993; Doherty et al. 2008; Carpenter et al. 2010; Fedy et al. 2014; Smith et al. 2014; Table 1). We evaluated variables at six analysis scales: 0.1-km radii (0.03 km^2), 0.25-km radii (0.20 km^2), 0.5-km radii (0.79 km^2), 1.0-km radii (3.14 km^2), 2.0-km radii (12.56 km^2), and 3.2-km radii (32.15 km^2). Scales were similar to other studies evaluating sage-grouse winter habitat selection (Doherty et al. 2008; Carpenter et al. 2010; Dzialak et al. 2013; Smith et al. 2014; Walker et al. 2016), and are relevant to sage-grouse management (*sensu* Walker et al. 2016).

We derived land cover and vegetation variables from the U.S. Department of Agriculture Forest Service LANDFIRE Existing Vegetation Type raster dataset to estimate land cover type for big sagebrush, shrub, and forest land cover (i.e., dominant land cover within a 30 × 30 m pixel; LANDFIRE 2013). We used LANDFIRE products because they were readily available and spatial coverage included both study areas as well as most western range land systems. We used a 30-m digital elevation map (DEM; U.S. Geological Survey 2011) to calculate slope, a Topographic Ruggedness Index (TRI), and a Topographic Wetness Index (TWI). TRI is a measure of the difference between local elevation and the mean of the elevation at the surrounding 8 raster cells; higher values correspond to increasing ruggedness (Riley et al. 1999). TWI measured wetness

Table 1 Variables used for model selection evaluating greater sage-grouse winter habitat selection in central and north-central Wyoming, USA, winters 2011–2015

Variable name	Description
Environmental	
Bsage	Proportion of big sagebrush land cover (*Artemisia* spp.; LANDFIRE 2013)
Shrub	Proportion of shrub land cover (LANDFIRE 2013)
Forest	Proportion of forest land cover (LANDFIRE 2013)
Slope	Mean slope (%) derived from 30-m digital elevation map (DEM; USGS 2011)
NDVI	Mean normalized difference vegetation index derived from NAIP imagery (USDA 2012)
TWI	Mean topographic wetness index (TWI; high values = increased soil moisture; Theobald 2007)
TRI	Mean topographic ruggedness (TRI; high values = increased ruggedness; Riley et al. 1999).
Anthropogenic	
TDstbarea	Total surface disturbance (ha); any bare ground resulting from vegetation removal
MajRd	Surface disturbance (ha); bare ground resulting from vegetation removal for improved roads
MinRd	Surface disturbance (ha); bare ground resulting from vegetation removal for minor roads
GenD	Surface disturbance (ha); bare ground resulting from vegetation removal, excluding major and minor roads
DistTDstbarea	Average Euclidean distance (km) to TDstbarea
DistMajRd	Average Euclidean distance (km) to MajRd
DistMinRd	Average Euclidean distance (km) to MinRd
DistGenD	Average Euclidean distance (km) to GenD

potential based on drainage of the local slope and upslope (integration of slope and aspect; Sorensen et al. 2006; Theobald 2007). TWI incorporates solar insulation to identify differences in north- and south-facing aspects to predict soil moisture (Zinko et al. 2005).

We followed the Wyoming Density Disturbance Calculation Tool (DDCT) protocol to create time-stamped disturbance layers that quantified areas of bare ground resulting from removal of vegetation (Wyoming Geographic Information Science Center 2016). Disturbances included energy infrastructure, roads, and non-energy related disturbance such as human structures. We obtained road data for Wyoming from the U.S. Geological Survey (O'Donnell et al. 2014). We separated roads into major roads (i.e., improved gravel or paved roads) and minor roads (i.e., high-clearance four-wheel drive or two tracks). Major and minor roads were buffered by 10 m and 3 m, respectively. We inspected the accuracy, validated, and manually digitized remaining disturbances using 2012 and 2015 NAIP imagery (USDA 2012; USDA 2015).

Experimental Design and Statistical Analysis

We evaluated sage-grouse winter resource selection with a use-availability framework in each study area at the population level by pooling locations across individuals (Manley et al. 2002) and estimated the RSF with an exponential link function (Johnson et al. 2006, McDonald 2013). We identified use as locations of marked individuals during the winter season (defined above). Habitat availability was defined at the population level for each study area where we generated random locations at a rate of 20X grouse use locations within 100 % fixed kernels of GPS-marked sage-grouse winter locations using "adehabitat" package in R (default bivariate kernel smoothing parameter; Worton 1989). We modeled relative probability of selection using generalized estimating equations (GEE) with PROC GENMOD in SAS software 9.4 (SAS Institute Inc. 2012). GEE models provide robust standard error estimates, account for repeated observations of the same individual, and are appropriate for unbalanced designs while providing population averaged inference (Fieberg et al. 2009, 2010; Koper and Manseau 2009). Individuals and randomly assigned available locations in proportion to the number of used locations for each individual were assigned to clusters. We selected between independent and compound-symmetric correlation structures by comparing the ratio of empirical and model based standard error estimates and selected the working correlation structure with the lowest ratio (Koper and Manseau 2009). We used quasi-likelihood criteria (QIC) to assess model support (Pan 2001).

We performed a series of variable screening procedures to remove non-informative variables. We removed individual variables when 85 % confidence intervals for coefficients overlapped 0 (Arnold 2010). We determined the most predictive of the six analysis scales by comparing each variable scale individually and retained the scale with the lowest QIC value. We tested remaining predictor variables for collinearity ($|r| > 0.6$) and did not allow correlated variables to be included in the same model. We also checked for stability and consistency of regression coefficient estimates when variables were moderately correlated ($0.3 \le |r| \le 0.6$). If coefficient sign switching occurred, we did not permit these variables to compete in the same model.

We used a sequential modeling approach (Arnold 2010) by evaluating predictor variables within environmental and anthropogenic subsets. In the first level of model selection, we explored all variable combinations within the environmental and anthropogenic variable subsets separately (Burnham and Anderson 2002). The model with the lowest QIC value was identified as being the best fit model; however, models within 4 QIC of the best fit model were considered competitive (Arnold 2010). Competitive models within each variable subset were then allowed to compete across the environmental and anthropogenic variable subsets to assess model improvement. We assessed model fit by the weight of evidence (w_i) and differences between QIC (ΔQIC; Burnham and Anderson 2002) for the top model and candidate models.

We evaluated the performance of our top RSF for each study area using 5-fold cross validation. We randomly retained locations from approximately 80 % of the individuals to develop five RSF models from the most supportive GEE model and tested each RSF with the withheld data. We binned RSF predictions from each fold into 5 quantile intervals and performed linear regression on the number of observed locations from the test dataset vs. the expected test locations generated from each RSF bin adjusted by the midpoint of the raw RSF values and area of each bin (Johnson et al. 2006).

We mapped our final models with 30-m pixel resolution across each study area. We distributed relative probabilities into 5 RSF bins based on quantile breaks in probabilities to classify areas as low, medium to low, medium, medium high, and high probability of selection (Sawyer et al. 2006) representing increasing relative probability of selection.

Results

We obtained 24,311 locations from 38 female sage-grouse during 4 winters (2011–2015) in the Bighorn Basin study area and 19,689 winter locations from 34 female sage-grouse across 3 winters (2012–2015) in the Jeffrey City study area. The mean winter season, based on population averaged movements by individual grouse to and from winter range, was delineated as 26 October to 21 March for Bighorn Basin and 7 October to 21 March for Jeffrey City. Average movement distance from fall to winter range was 8.2 ± 1.7 km (range: 0–80.3 km) and 5.1 ± 1.3 km (range: 0–37.4 km) in the Bighorn Basin and Jeffrey City, respectively.

Of the individuals with nesting location data, 17 of 30 (56.7 %) nested in Core Areas in the Bighorn Basin study area. The portion of winter locations in Core Areas for those individuals was 62.5 ± 9.5 % (SE; Fig. 2). Three individuals (17.6 %) wintered entirely outside and 2 (11.7 %) wintered entirely inside Core Areas. In the Jeffrey City study area, all individuals nested in Core and 98.0 ± 1.4 % (SE) of winter locations were in Core Areas. Only 6 (17.6 %) of 34 individuals occupied a portion of any seasonal range outside of Core Area in Jeffrey City.

Sage-Grouse Resource Selection

Bighorn Basin Study Area

The top model explaining sage-grouse winter habitat use in the Bighorn Basin study area included 6 predictor variables across 4 analysis scales (Table 2). Sage-grouse selected areas with lower slope and less total surface disturbance at the 0.1-km radii scale, greater proportion of big sagebrush habitats and closer to minor roads within 0.5-km, lower surface area of major roads within 1.0-km, and lower proportion of forest habitats within 2.0-km (Table 3). Variables with 95 % confidence intervals of coefficients overlapping 0 included proportion of forest habitats, surface area of major roads, total surface disturbance, and distance to minor roads. We considered these variables to be marginal predictors, but they were retained to develop the RSF

Table 2 Top and competing models best explaining sage-grouse winter habitat selection in the Bighorn Basin and Jeffrey City study areas, Wyoming, winters 2011–2015

Model	Model fit statistics		
	K	ΔQIC	w_i
Bighorn Basin study area			
$[Bsage_{0.5} + Forest_{2.0} + Slope_{0.1}]^{env} + [DistMinRd_{0.5} + MajRd_{1.0} + TDstbarea_{0.1}]^{anthro}$	7	0.0	1.0
$[Bsage_{0.5} + Forest_{2.0} + Slope_{0.1}]^{env}$	4	660.0	0.0
$[DistMinRd_{0.5} + MajRd_{1.0} + TDstbarea_{0.1}]^{anthro}$	4	25458.8	0.0
Jeffrey City study area			
$[Bsage_{0.25} + Slope_{0.25}]^{env}$	3	0.0	1.0
$[Bsage_{0.25} + Slope_{0.25}]^{env} + [MajRd_{3.2}]^{anthro}$	4	833.8	0.0
$[MajRd_{3.2}]^{anthro}$	2	10584.1	0.0

surface because they influenced other variables in the model informing our RSF (e.g., Aldridge et al. 2012). Predicted high and medium-high areas of winter selection encompassed 37.7 % of the study area (1445 km^2; Fig. 3); 30.4 % of those areas were in Core Areas. Cross-validation indicated that the top model performed well at predicting winter habitat selection within the study area with high r^2 values from linear regression models of observed vs. expected locations in each RSF bin (average $r^2 = 0.92 \pm 0.04$ SE), intercept coefficients did not differ from 0, slope coefficients differed from 0 in all but 1 fold, and slope coefficients did not differ from 1.

Table 3 Estimated variable coefficients, standard errors (SE), and 95 % confidence intervals (CI) for variables that were included in top models depicting population-level sage-grouse winter habitat selection in Jeffrey City and Bighorn Basin study areas, Wyoming, winters 2011–2015

Parameter	Estimate	SE	95 % CI Lower	95 % CI Upper
Bighorn Basin study area				
$Bsage_{0.5}$	0.0037	0.0006	0.0025	0.0049
$Forest_{2.0}$	−0.0002	0.0002	−0.0005	0.0001
$Slope_{0.1}$	−0.2264	0.0287	−0.2827	−0.1701
$DistMinRd_{0.5}$	−0.0002	0.0001	−0.0005	0.0000
$MajRd_{1.0}$	−0.0000	0.0000	−0.0001	0.0000
$TDstbarea_{0.1}$	−0.0000	0.0000	−0.0000	0.0000
Jeffrey City study area				
$Bsage_{0.25}$	0.0115	0.0036	0.0044	0.0186
$Slope_{0.25}$	−0.3090	0.0593	−0.4253	−0.1927

Jeffrey City Study Area

The model that best explained sage-grouse winter habitat selection in the Jeffrey City study area included 2 predictor variables at the 0.25-km radii scale (Table 2); greater proportion of big sagebrush and lower slopes within 0.25-km (Table 3). We predicted high or medium-high winter habitat selection across 39.6 % of the Jeffrey City study area (1643 km^2; Fig. 4). Our top model was a strong predictor of selection. Linear regressions of observed vs. expected winter locations produced high r^2 values (average $r^2 = 0.94 \pm 0.03$ SE). Intercept coefficients did not differ from 0, and slope coefficients differed from 0 and did not differ from 1 with the exception of 1 fold.

Discussion

The ability of a conservation area to maintain wildlife populations is a function of the reserves ability to meet seasonal habitat requirements. We found a meaningful portion of female sage-grouse occupying areas in winter

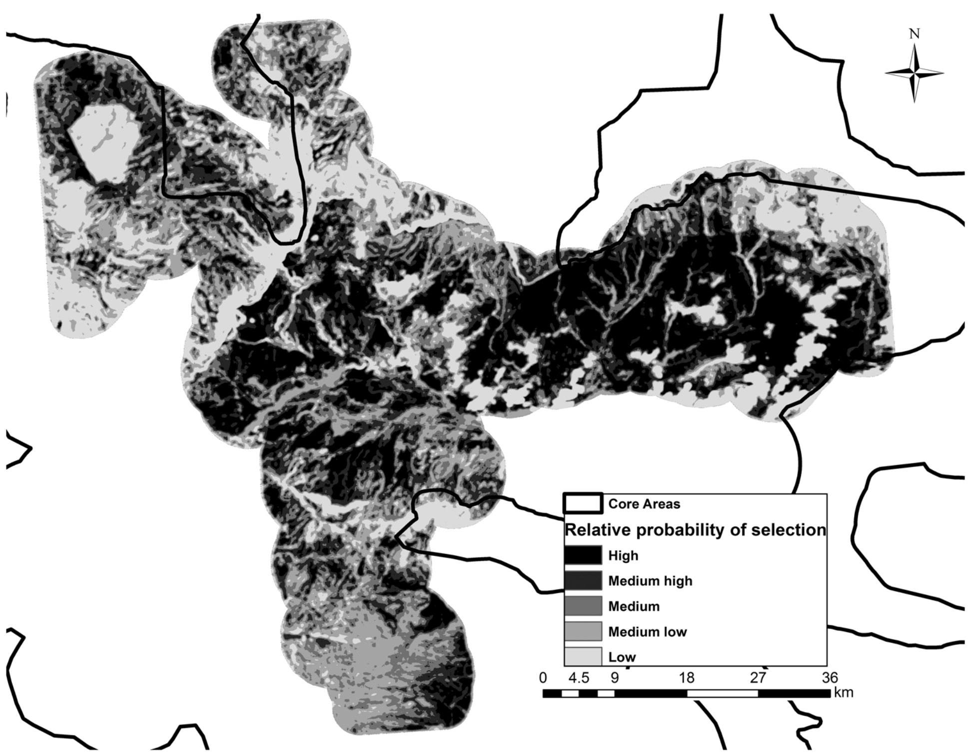

Fig. 4 Predicted probability of sage-grouse winter habitat selection in the 4144-km^2 Jeffrey City study area during winters 2012–2015. This map spatially depicts a resource selection function that was binned into five quantiles of predicted relative probability of occurrence

entirely outside of designated Core Areas in the Bighorn Basin. Because sage-grouse have a high fidelity to wintering areas (Connelly et al. 2004), highly used winter habitats that are compromised by development activities could negatively influence sage-grouse populations. This is supported by studies that documented sage-grouse avoidance of energy development and associated infrastructure during winter (Doherty et al, 2008; Carpenter et al. 2010; Smith et al. 2014; Holloran et al. 2015) and could result in indirect loss of otherwise suitable habitats (functional habitat loss; Aldridge and Boyce 2007; Smith et al. 2014).

Seasonal use restrictions in known winter concentration areas afford some level of protection during winter months. However, allowing disturbance outside of this period in known wintering areas may result in loss or subsequent avoidance of winter habitats. The timing of seasonal restrictions in winter concentration areas (1 Dec to 15 Mar; State of Wyoming 2011) must also align with the duration that sage-grouse spend on winter range. We found that the date of average movement from fall to winter habitat was earlier and that movement from winter to breeding habitat was later than current seasonal restrictions in both study areas. Minimal differences in the distribution of winter locations relative to our definition of the winter season and the Wyoming Core Area Policy stipulation of a 1 December to 15 March seasonal use restriction suggest that the distribution of winter locations (and presumably habitat use) was similar under these two definitions, yet winter habitats were used for considerably longer than the SGEO designation of the winter season. The Greater South Pass Core Area is the largest Core Area in Wyoming (~18,588 km^2) and likely contains a significant proportion of winter habitat for sage-grouse that occupy that region during breeding seasons. In the Jeffrey City study area, only 17.6 % of radio-marked individuals spent a portion of time in habitats outside of Core Areas. Individuals occupying smaller Core Areas likely relied on seasonal habitats outside of Core Areas to meet their annual life history requirements. Over one-third of the winter locations of GPS-marked females that nested in Core Areas in the Bighorn Basin study area occurred outside of Core Areas.

In both study areas, sage-grouse selected areas dominated by big sagebrush habitats and gentle slopes. Selection for landscapes dominated by big sagebrush is consistent with other studies that report sage-grouse selection of continuous sagebrush cover in winter. For example, Doherty et al. (2008) found that sage-grouse were more likely to occur in areas of greater sagebrush cover within 4-km^2 of winter grouse locations in northeast Wyoming. Sage-grouse selected less rugged areas with lower slopes in both study areas. Selection of areas with low topographic relief is consistent with findings of other studies evaluating sage-grouse winter habitat selection (Doherty et al. 2008; Carpenter et al. 2010, Dzialak et al. 2012, Walker et al. 2016).

Our models showed little support for anthropogenic variables being predictive of winter habitat selection. This was generally expected given the relatively low levels of disturbance in areas occupied by sage-grouse during winter in both study areas. We caution that while even low levels of disturbance may lead to habitat avoidance by sage-grouse, our estimates represent total surface disturbance that may not result in avoidance behaviors during the winter. For example, minor roads contributed to a significant portion of estimated surface disturbance across both study areas. However, minor roads are not counted in Wyoming's DDCT process. We found that grouse were selecting areas closer to minor roads in the Bighorn Basin study area, although this was considered a marginal predictor. Carpenter et al. (2010) found the opposite relationship for sage-grouse wintering in Alberta. However, the relative probability of selection did not increase greatly after habitat was greater than 1.2 km from a two track truck trail (Fig. 2 in Carpenter et al. 2010). Aldridge and Boyce (2007) and Kirol et al. (2015) found that brood-rearing females also selected areas closer to two track roads. Minor roads in both of our winter study areas were generally located in gentle topography and likely received little traffic volume, particularly in winter when snow precludes vehicle use in many areas.

We estimated that only one-third of predicted high and medium-high use winter habitat in the Bighorn Basin study area was in Core Areas, leaving a significant portion of predicted high and medium-high selected habitats outside of Core Area protection. We did not collect information regarding flock sizes of female sage-grouse in winter. Therefore, we did not explicitly model numbers of birds using areas in winter with our RSF models. However, sage-grouse generally exhibit flocking behaviors during winter (e.g. Beck 1977) and we assume that radio-marked individuals were representative of each population. It is likely that many more individual grouse were exhibiting similar patterns of winter habitat use and occupying these areas. Significant use by sage-grouse outside of Core Areas warrants further consideration for managing winter sage-grouse habitats in relation to Wyoming's Core Area Policy.

Land-use decisions that influenced Core Area boundaries resulted in removing some areas used by female sage-grouse from Core Area protection. Many areas outside of Core Areas identified as winter habitats contain breeding habitats, but were not included in Core Area designations to avoid existing development. The size and shape of constrained Core Areas relative to available sage-grouse breeding habitat in these areas resulted in more grouse locations falling outside Core Area protection during the breeding (15 Mar to 30 Jun) and winter (1 Dec to 15 Mar) seasons. This suggests seasonal use restrictions and potentially other means to avoid impacts should be afforded to winter habitats outside designated Core Areas, particularly in the Bighorn Basin where 17.6 % of sage-grouse did not winter in designated Core Areas and only 62.5 % of their winter locations fell within Core Areas. The amount and arrangement of winter habitats that fall outside of Core Areas dictates a need to assess Wyoming's Core Area Policy for future sage-grouse conservation. While Core Areas function as protection areas across a significant portion of sage-grouse breeding and nesting habitats throughout Wyoming, limited protection during other seasons does not support comprehensive sage-grouse conservation.

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Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

References

Aldridge CL, Boyce MS (2007) Linking occurrence and fitness to persistence: habitat-based approach for endangered greater sage-grouse. Ecol Appl 17:508–526

Aldridge CL, Saher DJ, Childers TM, Stahlnecker KE, Bowen ZH (2012) Crucial nesting habitat for Gunnison sage-grouse: a spatially explicit hierarchical approach. J Wildlife Manage 76:391–406

Anthony RG, Willis MJ (2009) Survival rates of female greater sage-grouse in autumn and winter in southeastern Oregon. J Wildlife Manage 73:538–545

Arnold TW (2010) Uninformative parameters and model selection using Akaike's information criterion. J Wildlife Manage 74:1175–1178

Baxter RJ, Larsen RT, Flinders JT (2013) Survival of resident and translocated greater sage-grouse in Strawberry Valley, Utah: a 13-year study. J Wildlife Manage 77:802–881

Beck TD (1977) Sage grouse flock characteristics and habitat selection in winter. J Wildlife Manage 41:18–26

Beck JL, Mitchell DL (2000) Influences of livestock grazing on sage grouse habitat. Wildlife Society B 28:993–1002

Beck JL, Reese KP, Connelly JW, Lucia MB (2006) Movements and survival of juvenile greater sage-grouse in southeastern Idaho. Wildlife Soc B 34:1070–1078

[BLM] Bureau of Land Management (2012) Greater sage-grouse habitat management policy on Wyoming Bureau of Land Management (BLM) administered public lands including the Federal Mineral Estate. Instruction Memorandum No. WY-2012-019

Blomberg EJ, Sedinger JS, Atamian MT, Nonne DV (2012) Characteristics of climate and landscape disturbance influence the dynamics of greater sage-grouse populations. Ecosphere 3(6):art55

Boyd CS, Beck JL, Tanaka JA (2014) Livestock grazing and sage-grouse habitat: impacts and opportunities. J Rangeland Appl 1:58–77

Braun CE (1998) Sage-grouse declines in western North America: what are the problems? Proc West Assoc State Fish Wildlife Agencies 78:139–156

Burnham KP, Anderson DR (2002) Model selection and multimodal inference: a practical information-theoretic approach, 2nd edn. New York, NY

Butchart SHM, Clarke M, Smith RJ, Sykes RE, Scharlemann JPW, Harfoot M, Buchanan GM, Angulo A, Balmford A, Bertzky B, Brooks TM, Carpenter KE, Comeros-Raynal MT, Cornell J, Francesco Ficetola G, Fishpool LDC, Fuller RA, Geldmann J, Harwell H, Hilton-Taylor C, Hoffmann M, Joolia A, Joppa L, Kingston N, May I, Milam A, Polidoro B, Ralph G, Richman N, Rondini C, Segan DB, Skolnik B, Spalding MD, Stuart SN, Symes A, Taylor J, Visconti P, Watson JEM, Wood L, Burgess ND (2015) Shortfalls and solutions for meeting national global conservation area targets. Conserv Lett 8:329–337

Carpenter J, Aldridge CL, Boyce MS (2010) Sage-grouse habitat selection during winter in Alberta. J Wildlife Manage 74: 1806–1814

Coates PS, Casazza ML, Blomberg EJ, Gardner SC, Espinosa SP, Yee JL, Wiechman L, Halstead BJ (2013) Evaluating greater sage-grouse seasonal space us relative to leks: Implications for surface use designations in sagebrush ecosystems. J Wildlife Manage 77:1598–1609

Connelly JW, Braun CE (1997) Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. Wildlife Biol 3:229–234

Connelly JW, Knick ST, Schroeder MA, Stiver SJ (2004) Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies, Cheyenne, WY

Connelly JW, Reese RP, Fischer RA, Wakkinen WL (2000a) Response of a sage grouse breeding population to fire in southeastern Idaho. Wildlife Soc B 28:90–96

Connelly JW, Schroeder MA, Sands AR, Braun CE (2000b) Guidelines to manage sage grouse populations and their habitats. Wildlife Soc B 28:967–985

Copeland HE, Pocewicz A, Naugle DE, Griffiths T, Keinath D, Evans J, Platt J (2013) Measuring the effectiveness of conservation: a novel framework to quantify the benefits of sage-grouse conservation policy and easements in Wyoming. PLoS One 8:e67261

Dahlgren DK, Guttery MR, Messmer TA, Caudill D, Elmore RD, Chi RY, Koons DN (2016) Evaluating vital rate contributions to greater sage-grouse population dynamics to inform conservation. Ecosphere 7:e01249

Diamond JM (1975) The island dilemma: lessons of modern biogeographic studies for the design of natural reserves. Biol Conserv 7:129–146

Doherty KE, Naugle DE, Copeland H, Pocewicz A, Kiesecker J (2011) Energy development and conservation tradeoffs: systematic planning for sage-grouse in their eastern range. In: Knick ST, Connelly JW (eds) Greater sage-grouse; ecology and conservation of a landscape species and its habitats, Studies in Avian

Biology vol. 38. University of California Press, Berkeley, California, pp 505-516
Doherty KE, Naugle DE, Walker BL, Graham JM (2008) Greater sage-grouse winter habitat selection and energy development. J Wildlife Manage 72:187–195
Dzialak MR, Olson CV, Harju SM, Webb SL, Winstead JB (2012) Temporal and hierarchical spatial components of animal occurrence: conserving seasonal habitat for greater sage-grouse. Ecosphere 3:art30
Dzialak MR, Webb SL, Harju SM, Olson CV, Winstead JB, Hayden-Wing LD (2013) Greater sage-grouse and severe winter conditions: identifying habitat for conservation. Rangeland Ecol Manag 66:10–18
Fedy BC, Aldridge CL, Doherty KE, O'Donnell M, Beck JL, Bedrosian B, Holloran MJ, Johnson GD, Kaczor NW, Kirol CP, Mandich CA, Marshall D, McKee G, Olson C, Swanson CC, Walker BL (2012) Interseasonal movements of greater sage-grouse, migratory behavior, and an assessment of the core regions concept in Wyoming. J Wildlife Manage 76: 1062–1071
Fedy BC, Doherty KE, Aldridge CL, O'Donnell M, Beck JL, Bedrosian B, Gummer D, Holloran MJ, Johnson GD, Kaczor NW, Kirol CP, Mandich CA, Marshall D, McKee G, Olson C, Pratt AC, Swanson CC, Walker BL (2014) Habitat prioritization across large landscapes, multiple seasons, and novel areas: an example using greater sage-grouse in Wyoming. Wildlife Monogr 190:1–39
Fieberg J, Matthiopoulus J, Hebblewhite M, Boyce MS, Frair JL (2010) Correlation and studies of habitat selection: problem, red herring or opportunity? Philos T Roy Soc B 365: 2233–2244
Fieberg J, Rieger RH, Zicus MC, Schildcrout JS (2009) Regression modeling of correlated data in ecology: subject specific and population averaged response. J Appl Ecol 46: 1018–1025
Giesen KM, Schoenberg TJ, Braun CE (1982) Methods for trapping sage grouse in Colorado. Wildlife Soc B 10:224–231
Gregory AJ, Beck JL (2014) Spatial heterogeneity in response of male greater sage-grouse lek attendance to energy development. PloS One 9:e97132. doi:10.1371
Harju SM, Dzialak MR, Taylor RC, Hayden-Wing LD, Winstead JB (2010) Thresholds and time lags in effects of energy development on greater sage-grouse populations. J Wildlife Manage 74: 437–448
Hess JE, Beck JL (2012) Disturbance factors influencing greater sage-grouse lek abandonment in north-central, Wyoming. J Wildlife Manage 76:1625–1634
Holloran MJ, Fedy BC, Dahlke J (2015) Winter habitat use of greater sage-grouse relative to activity levels at natural gas well pads. J Wildlife Manage 79:630–640
Homer CJ, Edwards TC, Ramsey DR, Price KP (1993) Use of remote sensing methods in modeling sage grouse winter habitat. J Wildlife Manage 57:78–84
Johnson CJ, Nielson SE, Merrill EH, McDonald TL, Boyce MS (2006) Resource selection functions based on use-availability data: theoretical motivation and evaluation methods. J Wildlife Manage 70:347–357
Johnson CJ, Seip DR, Boyce MS (2004) A quantitative approach to conservation planning: using resource selection functions to map the distribution of mountain caribou at multiple spatial scales. J Appl Ecol 41:238–251
Kiesecker JM, Copeland HE, McKenny BA, Pocewicz A, Doherty KE (2011) Energy by design: making mitigation work for conservation and development. In: Naugle D (ed) Energy development and wildlife conservation in western North America. Island Press, Washington DC, pp 159-181
Kirol CP, Beck JL, Huzurbazar SV, Holloran MJ, Miller SN (2015) Identifying greater sage-grouse source and sink habitats for conservation planning in an energy development landscape. Ecol Appl 25:968–990
Koper M, Manseau M (2009) Generalized estimating equations and generalized linear mixed-effects models for modeling resource selection. J Appl Ecol 46:590–599
LANDFIRE: Department of Agriculture, Forest Service; U.S. Department of Interior (2013) http://landfire.gov/index.php
LeBeau CW, Beck JL, Johnson GD, Holloran MJ (2014) Short-term impacts of wind energy development on greater sage-grouse fitness. J Wildlife Manage 78:522–553
Manley BF, McDonald LL, Thomas DL, McDonald TL (2002) Resource selection by animals: statistical design and analysis for field studies. 2nd edn. Boston, MA
McDonald TC (2013) The point process use-availability or presence-only likelihood and comments on analysis. J Anim Ecol 82: 1174–1182
Moynahan BJ, Lindberg MS, Thomas JW (2006) Factors contributing to process variance in annual survival of female greater sage-grouse in Montana. Ecol Appl 16:1529–1538
O'Donnell MS, Fancher TS, Freeman AT, Ziegler AE, Bowen ZH, Aldridge CL (2014) Large scale Wyoming transportation data-a resource planning tool: U.S. Geological Survey Data Series 821, p21, http//dx.doi.org/10.3133/ds821
Pan W (2001) Akaike's Information Criterion in generalized estimating equations. Biometrics 57:120–125
PRISM Climate Group (2016) Oregon State University, http://prism.oregonstate.edu
Riley SJ, DeGloria SD, Elliot R (1999) A terrain ruggedness index that quantifies topographic heterogeneity. Intermt J Sci 5:1–4
Sawyer H, Nielson RM, Lindzey F, McDonald LL (2006) Winter habitat selection of mule deer before and during development of a natural gas field. J Wildlife Manage 70:396–403
SAS Institute Inc (2012) The SAS system for windows version 9.4. Cary, NC, USA
Schroeder MA, Aldridge CL, Apa AD, Bohne JR, Braun CE, Bunnell SD, Connelly JW, Deibert PA, Gardner SC, Hilliard MA, Kobriger GD, McAdam SM, McCarthy CW, McCarthy JJ, Mitchell DL, Rickerson EV, Stiver SJ (2004) Distribution of sage-grouse in North America. Condor 106:363–376
Smith KT, Kirol CP, Beck JL, Blomquist FC (2014) Prioritizing winter habitat quality for greater sage-grouse in a landscape influenced by energy development. Ecosphere 5:art15
Sorensen R, Zinko U, Seibert J (2006) On the calculation of the topographic wetness index: evaluation of different methods based on field observations. Hydrol Earth Syst Sc 10:101–112
State of Wyoming (2011) Greater sage-grouse core area protection. Office of the Governor, Executive Order Number 2011-5, http://wgfd.wyo.gov/web2011/Departments/Wildlife/pdfs/SAGEGROUSE_EO_COREPROTECTION0000651.pdf
Swenson JE, Simmons CA, Eustace CD (1987) Decrease of sage grouse *Centrocercus urophasianus* after ploughing of sagebrush steppe. Biological Conserv 41:125–132
Taylor RL, Walker BL, Naugle DE, Mills LS (2012) Managing multiple vital rates to maximize greater sage-grouse population growth. J Wildlife Manage 76:336–347
Theobald DM (2007) LCap v1.0: Landscape connectivity and pattern tools for ArcGIS. Colorado State University, Fort Collins, CO
USDA (U.S. Department of Agriculture) 2012 NAIP imagery-1m (RGBIR)-Wyoming (2012) USDA farm service agency aerial photography field office, Salt Lake City, UT
USDA (U.S. Department of Agriculture) 2015 NAIP imagery-1m (RGBIR)-Wyoming (2015) USDA Farm Service Agency Aerial Photography Field Office, Salt Lake City, UT

USGS (U.S. Geological Survey) (2011) Seamless data warehouse. http://seamless.usgs.gov/

Wakkinen WL, Reese KP, Connelly JW, Fischer RA (1992) An improved spotlighting technique for capturing sage grouse. Wildlife Soc B 20:425–426

Walker BL, Apa AD, Eichhoff K (2016) Mapping and prioritizing seasonal habitats for greater sage-grouse in northwestern Colorado. J Wildlife Manage 80:63–77

Worton BJ (1989) Kernel Methods for estimating the utilization distribution in home-range studies. Ecology 70:164–168

Wyoming Geographic Information Science Center. 2016. Wyoming density and disturbance calculation tool. Version 2.0. http://ddct.wygisc.org/Data/Sites/24/files/DDCT_Procedures.pdf

Zinko U, Seibert J, Dynesius M, Nilsson C (2005) Plant species numbers predicted by a topography-based groundwater flow index. Ecosystems 8:430–441

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ORIGINAL RESEARCH

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Phenology largely explains taller grass at successful nests in greater sage-grouse

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Abstract
Much interest lies in the identification of manageable habitat variables that affect key vital rates for species of concern. For ground-nesting birds, vegetation surrounding the nest may play an important role in mediating nest success by providing concealment from predators. Height of grasses surrounding the nest is thought to be a driver of nest survival in greater sage-grouse (*Centrocercus urophasianus*; sage-grouse), a species that has experienced widespread population declines throughout their range. However, a growing body of the literature has found that widely used field methods can produce misleading inference on the relationship between grass height and nest success. Specifically, it has been demonstrated that measuring concealment following nest fate (failure or hatch) introduces a temporal bias whereby successful nests are measured later in the season, on average, than failed nests. This sampling bias can produce inference suggesting a positive effect of grass height on nest survival, though the relationship arises due to the confounding effect of plant phenology, not an effect on predation risk. To test the generality of this finding for sage-grouse, we reanalyzed existing datasets comprising >800 sage-grouse nests from three independent studies across the range where there was a positive relationship found between grass height and nest survival, including two using methods now known to be biased. Correcting for phenology produced equivocal relationships between grass height and sage-grouse nest survival. Viewed in total, evidence for a ubiquitous biological effect of grass height on sage-grouse nest success across time and space is lacking. In light of these findings, a reevaluation of land management guidelines emphasizing specific grass height targets to promote nest success may be merited.

KEYWORDS
Centrocercus urophasianus, concealment, greater sage-grouse, nest survival, phenology

1 | INTRODUCTION

Environmental factors affecting influential demographic parameters are appropriate targets of management to promote habitat quality for species of conservation concern (Mills, 2007). For many birds, characteristics of nest sites that influence nest predation are of interest, as nest success is a key driver of population growth and predation is the primary cause of nest failure (Martin, 1993; Ricklefs, 1969). According to the nest concealment hypothesis, nests surrounded by dense vegetation should be more successful because they are more difficult for predators to detect or access (Martin, 1992; Martin & Roper, 1988). Furthermore, vegetative concealment may represent an attractive target for conservation action because it can often be managed, for example, through manipulation of herbivory by livestock.

Support for the nest concealment hypothesis is mixed. In a recent review and comparative analysis, 26% of 114 reviewed studies in open-cup-nesting songbirds supported an effect (Borgmann & Conway, 2015). Effects of concealment on nest survival may be difficult to detect if strong selection for concealed nest sites canalizes variation among nests such that most occur in "adaptive peaks" providing adequate concealment (Latif, Heath, & Rotenberry, 2012; Remeš, 2005). However, even studies employing experimental removal of vegetation have returned mixed support for the nest concealment hypothesis (e.g., Bengtson, 1972; Howlett & Stutchbury, 1996; Latif et al., 2012; Peak, 2003). Numerous intrinsic and extrinsic factors may influence the effect of concealment on nest success. For example, birds with more brightly colored plumage appear more dependent on vegetation to conceal the nest from predators (Borgmann & Conway, 2015), and the benefits of visual concealment may depend on the composition of the local predator community (Clark & Nudds, 1991; Colombelli-Negrel & Kleindorfer, 2009; Dion, Hobson, & Lariviere, 2000). More problematic, however, are methodological aspects of studies that produce biased inference with regard to effects of concealment on nest survival (Borgmann & Conway, 2015; Burhans & Thompson, 1998; Gibson, Blomberg, & Sedinger, 2016; McConnell, Monroe, Burger, & Martin, 2017). Here, we focus on a recently highlighted methodological bias pervasive in research regarding habitat–fitness relationships in greater sage-grouse (*Centrocercus urophasianus*).

The greater sage-grouse (hereafter, sage-grouse) is a precocial, ground-nesting species of conservation concern inhabiting sagebrush ecosystems of western North America. Although sage-grouse nest beneath shrubs—primarily sagebrush—perennial grasses and forbs in the interspaces between shrubs have long been thought to provide critical concealment of nests from potential predators (Connelly, Schroeder, Sands, & Braun, 2000). This hypothesis is supported by studies reporting positive associations between height and/or cover of herbaceous vegetation surrounding nest sites and nest survival (Coates & Delehanty, 2008; DeLong, Crawford, & DeLong, 1995; Doherty et al., 2014; Gregg, Crawford, Drut, & DeLong, 1994; Sveum, Edge, & Crawford, 1998). Consequently, sage-grouse conservation efforts and land management policy have focused on increasing herbaceous hiding cover in suitable nesting habitat throughout the range of the species. Although direct links between livestock grazing and sage-grouse demography are lacking, studies indicating positive effects of herbaceous vegetation height and/or cover on nest survival provide a plausible mechanism linking livestock grazing and nest success (Connelly & Braun, 1997; Connelly et al., 2000), a key demographic rate for sage-grouse (Taylor, Walker, Naugle, & Mills, 2012). Thus, the validity of inference about the importance of herbaceous hiding cover for sage-grouse nest success has major implications for the management of sagebrush ecosystems, where livestock grazing is a ubiquitous land use (Knick et al., 2003).

Recent evidence has demonstrated that the positive association between grass height, a commonly used metric of herbaceous concealing cover among sage-grouse nesting studies, and nest survival may be indicative of biased methods rather than a causal relationship (Gibson, Blomberg, et al., 2016; McConnell et al., 2017). Using both empirical and simulation approaches, it has been shown that measuring grass height at nests following nest fate (i.e., hatch or failure) produces inflated or even spurious statistical relationships between grass height and nest survival. Because successful nests persist and are therefore measured later in the season than failed nests, measured concealment is greater at successful nests due to concurrent plant growth rather than a presumed reduction in predation. Despite knowledge of this sampling issue dating back decades (e.g., Burhans & Thompson, 1998), this sampling bias remains pervasive in sage-grouse and other ground-nesting bird literature, with a majority of sage-grouse studies sampling vegetation following nest fate (Gibson, Blomberg, et al., 2016).

Given the far-reaching implications derived from inference about grass height and sage-grouse demography, we were interested in exploring the generality of recent findings reported by Gibson, Blomberg, et al. (2016), and McConnell et al. (2017). Using field data from four geographically distinct study sites representative of the diversity of vegetation communities, predator communities, precipitation regimes, and evolutionary history of grazing found across the range of sage-grouse, we tested the hypothesis that studies using biased field methods that had previously supported a positive association between grass height measured around the nest and nest survival would fail to support such an association after accounting for phenology.

2 | METHODS

We employed the model-based methods presented in Gibson, Blomberg, et al. (2016) to correct for phenology in a reanalysis of three datasets from Montana, Utah, and Wyoming (Table 1). In a dataset from Eureka County, Nevada, analyzed by Gibson, Blomberg, et al. (2016), vegetation measurements were made at predicted hatch date and a linear regression relating vegetation height to the date of measurement was used to predict vegetation height at fate date, thereby demonstrating the potential bias arising from such a sampling scheme. We employed this concept in reverse fashion, that is, we regressed vegetation height on date of measurement to predict grass height at hatch date, as although it had been sampled using unbiased methods.

2.1 | Datasets

Reanalyzed datasets included a previously published study that found a significant positive influence of live grass height on sage-grouse nest survival across two study areas in the Powder River Basin (PRB) in southeast Montana (hereafter PRB North, n = 209) and northeast

TABLE 1 We used predictions from five studies across the range of greater sage-grouse, representing n = 1204 nests over a total of 24 study site-years

Study area	n	Years	Transect length (m)	Samples per nest	Data source
Eureka County	396	2004-2012	10	10	Gibson, Blomberg, et al. (2016);
PRB North	209	2003-2006	30	20	Doherty et al. (2014)
PRB South	174	2004-2006	30	20	Doherty et al. (2014)
Roundup	320	2012-2015	12	8	J. Smith, Unpublished Data
NE Utah	105	2012-2015	30	20	S. Dettenmaier, Unpublished Data
Total	1204				

Each study sampled grass height similarly, using measurements of the nearest grass height to various points along two intersecting transects centered at the nesting shrub. However, total transect length and the number of samples per nest varied by study.

Wyoming (hereafter PRB South, n = 164; Doherty et al., 2014); preliminary data from an ongoing evaluation of grazing treatments on sage-grouse ecology in central Montana (Joseph Smith, University of Montana, Unpublished Data, n = 320); and the first 4 years of a study comparing sage-grouse demography across two study areas in northern Utah (Seth Dettenmaier, Utah State University, Unpublished Data, n = 105). Including findings from Gibson, Blomberg, et al. (2016), these studies encompassed 1204 sage-grouse nests over 24 study site-years from across the range of sage-grouse (Table 1). Each study used similar methodologies to sample herbaceous vegetation surrounding nest sites by taking multiple measurements of grass height along intersecting transects centered on the nesting shrub and using the mean of replicated measurements to represent grass height-surrounding nests (Table 1).

2.2 | Statistical analyses

We assumed hatch date was 27 days after the estimated nest initiation date and applied a correction to measured grass height covariates following Gibson, Blomberg, et al. (2016):

$$\text{GrassHeight}_{\text{Hatch}} = \text{GrassHeight}_{\text{Fate}} - \left(\text{SurveyDate}_{\text{Fate}} - \text{SurveyDate}_{\text{Hatch}}\right) \times \beta_{\text{grass}}$$

where, for each study area and year, we fit a linear regression of measured grass height ($\text{GrassHeight}_{\text{Fate}}$) on day of nesting season ($\text{SurveyDate}_{\text{Fate}}$) to estimate β_{grass}. This simple correction provided a standardized measurement for grass height across nests regardless of fate. We estimated the effect of grass height on nest success using both corrected and uncorrected covariate measurements by fitting Bayesian daily nest survival models to each dataset (Schmidt, Walker, Lindberg, Johnson, & Stephens, 2010) with the exception of data from Gibson, Blomberg, et al. (2016), who provided estimates from their published analysis. In this approach, we estimated nest survival (S) for each nest (i) on each day of the nesting season (t) via a logit-linear model, which at minimum included an intercept (β_0) and coefficient for grass height, while also including coefficients that respective authors deemed supportive in top models. Nest encounter histories consisted of observed nest states (y) for each day of observation, where $y_{i,t}$ = 1 if nest i was observed alive on day t, $y_{i,t}$ = 0 if nest i was observed to have failed (female absent and some or all eggs destroyed), and $y_{i,t}$ = NA on days when nest state was not observed. Beginning on the first day after the nest was detected,

$$y_{i,t} \sim \text{Bern}(y_{i,t-1} S_{i,t})$$

and

$$\text{logit}\,(S_{i,t}) = \beta_0 + x_i{}'\beta$$

Specifically, Doherty et al. (2014), following the original population analyses in Walker (2008), modeled nest survival using covariates including a main and quadratic effect for nest age, and categorical variables for a particularly harsh spring nesting season with major snow events that caused nest abandonment (2003) and the two study regions (PRB North and PRB South). Although the PRB datasets were collected independently, they were combined in the analysis presented in Doherty et al. (2014), and we combine them here for consistency. Although it appears this study was mistakenly recorded as having used a fate date protocol in Gibson, Blomberg, et al. (2016; Table 1), the investigators did attempt to control for phenology by sampling vegetation near the predicted hatch date regardless of nest fate. Nonetheless, close examination of the dataset revealed that a temporal bias in measurement date existed across all study site-year combinations, such that successful nests were measured from 2 to 10 days later than failed nests, on average. To attempt to correct this persistent bias and maintain consistency among reanalyzed datasets, we corrected grass heights to predicted hatch date in the PRB North and PRB South datasets, but these corrections were generally smaller than corrections in the other reanalyzed datasets. Unpublished data from J. Smith included covariates for the log of distance to major roads and a measure of 4-day cumulative rainfall, as well as a random effect for year. Data from Gibson, Blomberg, et al. (2016), and models fit to Utah data included only an intercept and coefficient for measurements of grass height. Our estimates of daily nest survival and nest success are only reflective of the incubation period, as sage-grouse nests are typically found after the onset of incubation, and thus overestimate true nest success from initiation to hatch (Blomberg, Gibson, & Sedinger, 2015). Moreover, as monitoring intensity of prenesting females may have varied among datasets, incubation success may be more or less biased relative to true nest success and overall success rates are therefore not directly comparable among studies.

We fit daily nest survival models in JAGS 4.0 (Plummer, 2003) with the package rjags (Plummer 2016) in R 3.3.0 (R Core Team 2016), estimating posterior distributions with a total of 90,000 samples from 3 independent Markov chain Monte Carlo (MCMC) chains (30,000 per chain) after discarding the first 20,000 iterations from each chain for burn-in. We placed vague normal prior distributions on all coefficients (μ=0; σ=1000). Using coefficient posterior distributions, we generated predictions for the mean influence of grass height on nest success, the product of daily nest survival over a 27-day incubation period, and 95% credible intervals over the range of grass height values observed within each respective dataset. We held additional covariates at their mean value where applicable.

We performed an additional analysis to provide a comprehensive assessment of the influence of grass height on nest survival across datasets, excluding nests from Eureka County for which we only had data on the predicted response. Here, we pooled datasets and used generalized linear mixed models to test whether grass surrounding successful nests was taller than grass surrounding failed nests after accounting for phenology. Under the null hypothesis, grass heights (GH) measured at nests are a linear function of ordinal date of measurement (DAY; days since January 1), with normally distributed errors and no difference between successful and failed nests. Our alternative hypothesis was that grass is taller at successful nests than at failed nests after accounting for the linear function of ordinal date. We first used AIC_C model selection (Burnham & Anderson, 2002) to determine the best structure for a null (i.e., phenology) model. We considered a phenology model with a random intercept for each study area-year (1|STUDY:YEAR) combination to allow for variation in grass height inherent among geographically distant study areas and in different years, and a random intercepts and slopes phenology model (DAY|STUDY:YEAR) to allow for different rates of grass growth among years and study areas. To aid in model convergence, we centered the independent variable DAY by subtracting the median day of measurement from all observations. After we determined the best structure for the phenology model using AIC_C, we used a likelihood ratio test to assess support for our alternative hypothesis, which was represented with a model following the structure of the most supported phenology model and including a categorical fixed effect for nest fate (FATE; failed = 0, hatched = 1). Linear mixed models were fit using the lme4 package (Bates, Maechler, Bolker, & Walker, 2015) in R. Using these datasets, we also tabulated all corrected grass height measurements at successful and failed nests and performed a one-sided Kolmogorov–Smirnov test to examine if distributions of measurements differed between pooled data sets. A one-sided test was chosen to increase statistical power given our a priori expectation that grass would be taller surrounding successful nests than failed nests.

3 | RESULTS

Uncorrected, each of the three reanalyzed datasets revealed a strong, positive association between grass height and daily nest survival (Figure 1; dotted lines). Estimated coefficients for grass height using uncorrected grass heights were 0.063 (95% CI from 0.037 to 0.092) for PRB North and PRB South, 0.099 (95% CI from 0.063 to 0.137) for Roundup, and 0.058 (95% CI from 0.002 to 0.118) for NE Utah. Corrections to measured grass heights averaged—1.32 cm and mean absolute correction (|corrected–uncorrected|) was 2.08 cm, with a standard deviation of 2.31 cm. Following adjustment of measured grass heights to remove temporal bias, we found no association between grass height and nest survival in two of the three datasets (Roundup and NE Utah), and a weakened but persistent association in the PRB dataset (Figure 1; solid lines). Estimated coefficients for grass height

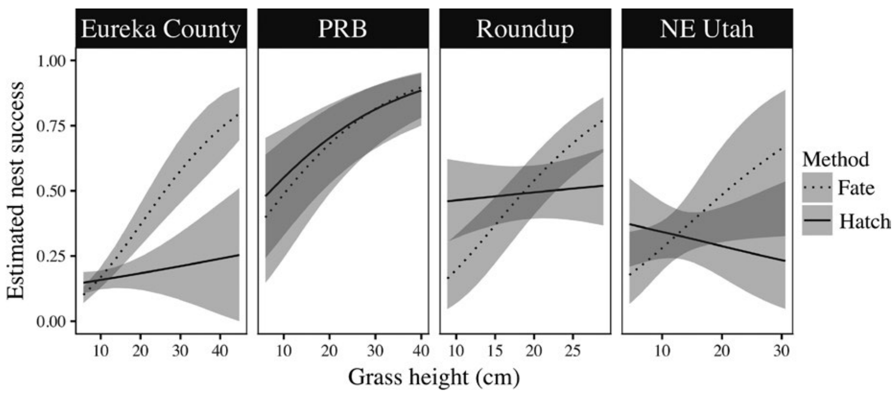

FIGURE 1 Predicted response of sage-grouse nest success (and 95% CI [Eureka County] or CRI [other studies]) to live grass height using measurements collected with a biased method following determination of nest fate (dotted lines), and those measured or corrected to the predicted hatch date of nests (solid lines). Nest data includes studies from the powder river basin (PRB) in southeastern Montana (PRB North, Doherty et al., 2014, n = 209, 2003–2006) and northeast Wyoming (PRB South, Doherty et al., 2014, n = 174, 2004–2006); Eureka County, Nevada (Gibson, Blomberg, et al., 2016, n = 396, 2004–2012); central Montana near the town of Roundup (J. Smith, University of Montana, unpublished data, n = 320, 2012–2015), and northeast Utah (Dettenmaier, Utah State University, unpublished data; n = 105, 2012–2015). Note that limits of x-axes change to reflect the range of grass heights observed within respective studies

using corrected grass heights were 0.053 (95% CI from 0.025 to 0.081) for PRB North and PRB South, 0.008 (95% CI from -0.027 to 0.042) for Roundup, and −0.015 (95% CI from −0.060 to 0.032) for NE Utah.

The random intercept and slope phenology model (conditional R^2 = 0.51 [Nakagawa & Schielzeth, 2013]) received the most support with an AIC_C score 9.64 units lower than the constant slope model (conditional R^2 = .46) and was used as the null model (Figure 2). The alternative hypothesis, that grass height surrounding successful nests was greater than that surrounding failed nests after accounting for phenology, was not supported (χ^2 = 2.74, df = 1, p = .098). Overall, median height of live grasses, corrected to hatch date, was 15.3 cm at successful nests (n = 336) and 15.1 cm at failed nests (n = 472; Figure 3). A one-sided Kolmogorov–Smirnov test provided no evidence that the distributions of phenology-corrected grass heights differed between successful and failed nests when pooling across sites and years (p = .307).

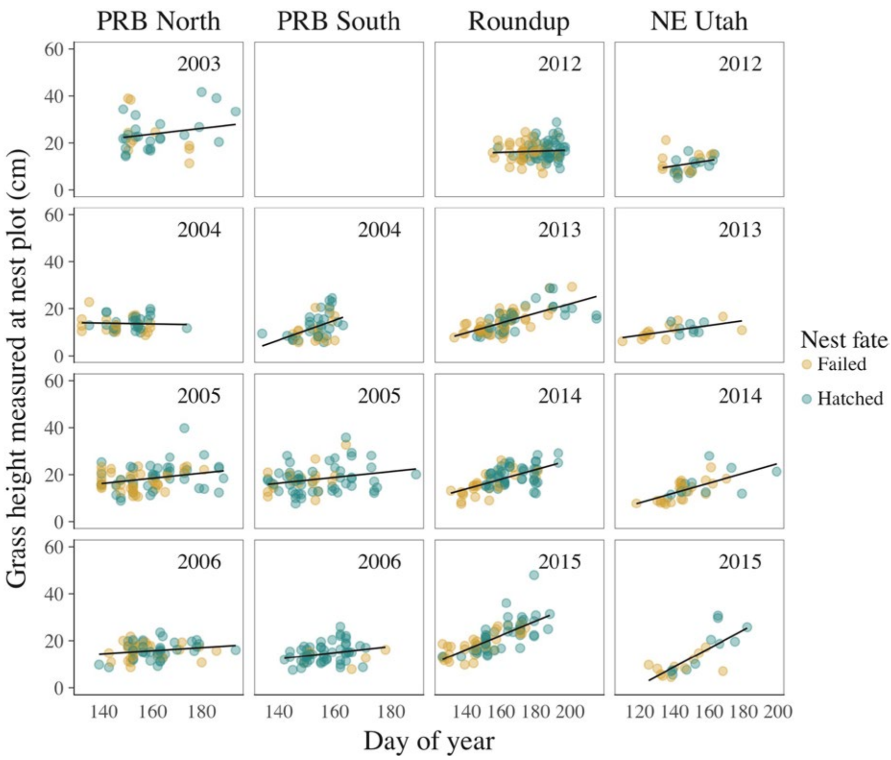

FIGURE 2 Average grass height surrounding successful and failed sage-grouse nests (n = 808) at the ordinal date of measurement by year (rows) and study area (columns). After accounting for phenology, a difference in grass height between successful and failed nests was not supported

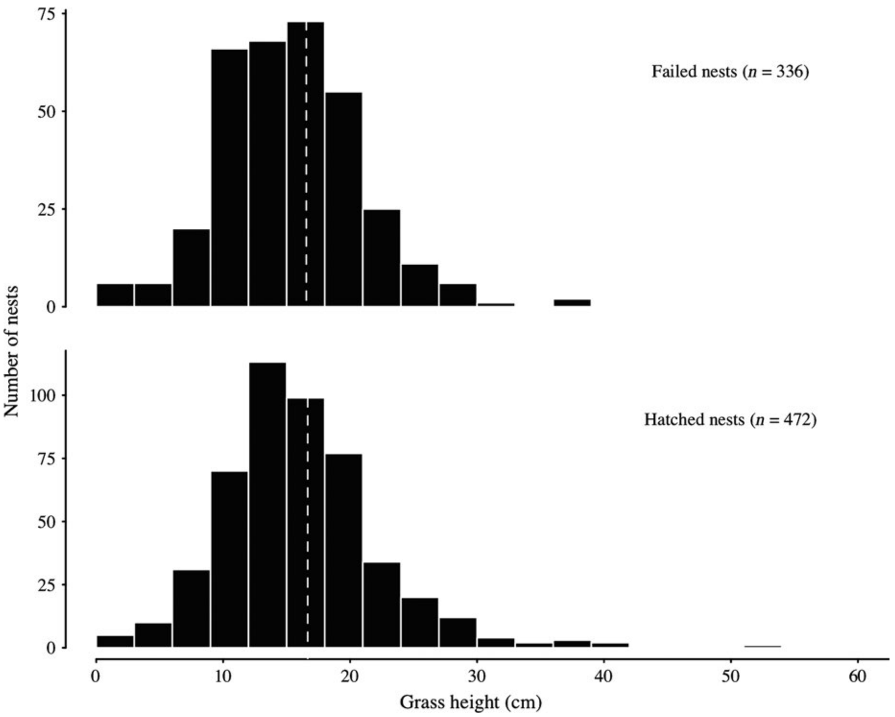

FIGURE 3 Grass heights surrounding greater sage-grouse nests (n = 808) corrected to hatch date. Median height of grass-surrounding nests (dashed vertical lines) was 15.26 cm at successful nests and 15.14 cm at failed nests. A one-sided Kolmogorov–Smirnov test provided no evidence that the distributions of grass heights differed between successful and failed nests (ground-nesting p = .307)

4 | DISCUSSION

While our analyses revealed mixed support for relationships between grass height and nest survival in sage-grouse, they confirmed recent findings that associations between herbaceous vegetation structure and nest success are frequently byproducts of temporally biased sampling rather than indicative of effect of concealing cover on detectability by predators (Gibson, Blomberg, et al., 2016; McConnell et al., 2017). Sampling vegetation following nest fate, a pervasive practice in studies of sage-grouse and other ground-nesting birds, consistently produces spurious relationships between grass height and nest survival and should, therefore, be avoided. As field crews are rarely able to strictly adhere to a schedule due to weather or other logistic constraints, even studies using field protocols intended to control for phenology may be affected by some degree of temporal bias between failed and successful nests, producing inflated effect sizes (e.g., the PRB dataset reanalyzed here; Doherty et al., 2014).

Taller grass may be associated with reduced nest predation under some conditions, such as in the context of particular predator communities or in years with particularly tall grass. However, grass height does not appear to be a universal indicator of nesting habitat quality for sage-grouse. Including the PRB dataset, we are aware of only three published studies using unbiased methods that support a positive association between grass height and nest survival (Doherty et al., 2014; Gregg et al., 1994; Sveum et al., 1998) among the 11 published studies testing for such an effect (Table 1 in Gibson, Blomberg, et al., 2016). Although the results have generally been interpreted to support the hypothesis that taller grass promotes greater nest survival (Connelly et al., 2000; Crawford et al., 2004), data presented by Sveum et al. (1998; Table 2) merely indicated that cover of short grasses (<18 cm) was lower at successful nests than failed nests in 1 out of 2 years (n = 32 nests), while cover of tall grasses (≥18 cm) did not differ between successful and failed nests in any year, even using a liberal α level of 0.1. Positive relationships between grass height and nest survival may, in fact, be uncommon. It is telling that, when analyzed together, data from the four study areas examined here provided no evidence for a difference in herbaceous vegetation height between successful and failed nests after accounting for plant phenology and timing of sampling (Figures 2 and 3).

The research and management communities must guard against uncritical acceptance of intuitive but untested mechanistic explanations for correlative patterns emerging from observational studies of habitat–fitness relationships. Within the sagebrush ecosystem, the broad acceptance that taller grass causes greater nest success by concealing nests from predators is an example of this type of untested logical connection, as equally plausible alternative hypotheses exist. For example, in multiyear studies, annual variation in precipitation and temperature in the prenesting and nesting periods may simultaneously affect female body condition, incubation behavior, and plant phenology. If conditions favorable to increased body condition or nest attentiveness have coincident positive effects on grass growth, nest success may be positively correlated with grass height absent any causal relationship between the two variables.

An experimental approach involving manipulation of vegetation height-surrounding nests could circumvent these issues, but would be fraught with its own set of difficulties. Sage-grouse females display a propensity toward abandoning reproductive efforts following disturbance by investigators (e.g., Gibson, Blomberg, Atamian, & Sedinger, 2015; Moynahan, Lindberg, Rotella, & Thomas, 2007). Disturbance from experimental manipulation at treatment nests would, therefore, need to be simulated at control nests such that observer-induced abandonment rates would be equal among nests in both groups. This may present an ethical dilemma for a species of conservation concern, or may simply yield sample sizes with inappropriately low statistical power. Furthermore, results of such an experiment would be of questionable relevance to management if manipulations bore little resemblance to defoliation patterns arising via herbivory (France, Ganskopp, & Boyd, 2008). Thus, experimental research is unlikely to provide an easy resolution to the problem. A critical examination of past evidence and careful consideration of alternative mechanistic hypotheses are warranted when considering the observational evidence at hand.

Habitat–fitness relationships are often context-dependent, and therefore variable across a species' range. Effects of concealment on nest survival, for example, may be more likely where cover is sparse. If that were the case, we might expect effects of grass height on nest survival to be more common in study sites characterized by low-shrub cover-surrounding nests. Indeed, the positive association between grass height and nest survival in the PRB study site reanalyzed here occurred in the eastern portion of the range, characterized by high spring precipitation and herbaceous vegetation cover compared to the rest of the sage-grouse range (Doherty, Evans, Coates, Juliusson, & Fedy, 2016). However, there was no relationship between grass height and nest survival in the Roundup study area, which had the lowest average shrub cover (18%) among datasets we considered. Selection of nest sites surrounded by tall grasses (Hagen, Connelly, & Schroeder, 2007) may result in a truncated covariate space such that nests surrounded by very short vegetation are rarely observed, thereby precluding the ability to detect an effect on survival (Chalfoun & Schmidt, 2012; Latif et al., 2012). However, with data from 15 study site-year combinations, we are confident we have surveyed a representative range of conditions chosen by nesting females. The lack of difference in grass height between successful and failed nests across these datasets strongly suggests that height of grasses was not a limiting resource (Figure 3).

The absence of support for an effect of grass height does not imply concealment is wholly unrelated to nest survival in sage-grouse. Selection for larger, taller sagebrush for nest substrates and preference for nesting in areas with greater areal cover of shrubs are well documented (reviewed in Hagen et al., 2007). In preferred sites, grasses and forbs may simply provide little additional visual or olfactory obstruction between a nest and a potential predator beyond that already provided by shrubs (see France, Ganskopp, & Boyd, 2008). Furthermore, while grasses and forbs afford mostly lateral cover, shrubs may provide more effective cover from aerial visual predators such as common ravens (*Corvus corax*), a primary nest predator for sage-grouse (Coates, Connelly, & Delehanty, 2008; Coates & Delehanty, 2008). Previous research indicates nest site selection in sage-grouse is driven by avian predators at broad scales (Dinkins, Conover, Kirol, & Beck, 2012) and characteristics of nest sites at small scales are more consistent with avoidance of visual (i.e., avian) predators than olfactory (i.e., mammalian) predators (Conover, Borgo, Dritz, Dinkins, & Dahlgren, 2010; Fogarty, Elmore, Fuhlendorf, & Loss, 2017). The lack of association between height of grasses and survival may also indicate a trade-off between nest concealment and the ability of incubating females to detect predators from a distance and alter their behavior in such a way as to reduce detection (Götmark, Blomqvist, Johansson, & Bergkvist, 1995).

Nest success is only one among several influential vital rates affecting sage-grouse population growth, and further research is needed to address how structure of grasses and forbs affects other life stages in sage-grouse. Studies of other grouse suggest vegetation height may be an important driver of brood survival. For example, increased vegetation height and/or greater insect abundance resulting from reduced grazing intensity positively affected production in black grouse (*Tetrao tetrix*) in Britain (Baines, 1996; Calladine, Baines, & Warren, 2002). The positive effect on production was, however, diminished or even reversed when grazing reduction treatments covered larger areas (Calladine et al., 2002), suggesting mosaics of vegetation height may confer greater benefits than uniformly tall vegetation (also see Baines, Richardson, & Warren, 2017; Jahren, Storaas, Willebrand, Moa, & Hagen, 2016). Taller vegetation may also moderate thermal extremes experienced by grouse, a function which may take on increased importance under climate change (Hovick, Elmore, Allred, Fuhlendorf, & Dahlgren, 2014). Although selection of sites with greater visual concealment by brood-rearing sage-grouse has been documented (Kaczor, Herman-Brunson, & Jensen, 2011; Schreiber et al., 2015), studies testing effects of herbaceous vegetation structure on sage-grouse chick survival are few and have produced mixed results (Aldridge, 2005; Gregg & Crawford, 2009). Recently, Gibson, Blomberg, et al. (2016) found survival of sage-grouse chicks to 2 weeks of age was positively associated with height of grasses surrounding the nest, presumably because structure of vegetation at the nest site is assumed to be correlated with structure of vegetation encountered by the precocial chicks during the first weeks of life. Again, however, a causal relationship between grass height and chick survival cannot be inferred. Positive relationships between herbaceous plant height and chick survival could implicate concealment from predators, but it is also plausible that taller grass at the nest is associated with some unmeasured factor—for example, site productivity, precipitation, or soil moisture—which in turn influences factors causally related to chick survival.

While the herbaceous understory is a key component of sagebrush ecosystems and sage-grouse habitat (e.g., Chambers et al., 2014), its role in concealing nests from predators has been overstated in management guidelines and land management documents. For example, the habitat assessment framework (HAF; Stiver et al., 2015), a tool used by the US Bureau of Land Management and US Forest Service to evaluate whether public lands are meeting habitat requirements of sage-grouse, included guidelines for maintaining a minimum height of perennial grasses and forbs in upland nesting habitat (18 cm) based largely on studies suggesting positive effects of vegetation height on nest success. There is, however, little evidence for the existence of the causal relationship between grass height and nest survival on which these guidelines were predicated. While it appears these "fourth order" guidelines may place unwarranted emphasis on the importance of maintaining herbaceous hiding cover for nesting, it should be noted that the HAF appropriately lays out a hierarchical management approach which suggests policies be set at the rangewide and regional scales to limit habitat loss and fragmentation—known causes of population declines among prairie grouse—but emphasizes that significant flexibility should be granted to local managers applying finer scale guidelines (see Chapter 1, Stiver et al., 2015). Persistent, broad-scale threats to sagebrush ecosystems including oil and gas development (Naugle, Doherty, Walker, Holloran, & Copeland, 2011), wildfire and invasive annual grasses (Coates et al., 2016), cropland conversion (Smith et al., 2016), and conifer encroachment (Miller, Naugle, Maestas, Hagen, & Hall, 2017) are well-documented drivers of sage-grouse population declines and should therefore be the highest priority for managers. Maintenance of tall grasses and forbs for nesting cover should not distract managers from addressing these larger threats or preclude the use of management tools that could otherwise improve sage-grouse habitat.

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AUTHOR CONTRIBUTIONS

JTS conceptualized the study, collected field data in central Montana, compiled and quality checked data from all study sites, analyzed data, produced figures, and wrote the manuscript. JDT analyzed data, produced figures, and assisted in writing the manuscript. KED collected field data in PRB and assisted in writing the manuscript. BWA, JDM, and DEN assisted with study conceptualization, interpretation of results, and manuscript writing, and revised several early versions of the manuscript. LIB and TAM contributed field data in central Montana and Northern Utah, respectively, and critically revised the final manuscript. SJD collected field data in Northern Utah. All authors critically revised and approved the final version of the manuscript.

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REFERENCES

Aldridge, C. (2005). Identifying habitats for persistence of greater sage-grouse (*Centrocercus urophasianus*) in Alberta, Canada. Dissertation, University of Alberta.

Baines, D. (1996). The Implications of grazing and predator management on the habitats and breeding success of black grouse *Tetrao tetrix*. *The Journal of Applied Ecology*, *33*(1), 54–62. https://doi.org/10.2307/2405015

Baines, D., Richardson, M., & Warren, P. (2017). The invertebrate diet of Black Grouse *Tetrao tetrix* chicks: A comparison between northern England and the Scottish Highlands. *Bird Study*, *64*(2), 117–124. https://doi.org/10.1080/00063657.2017.1295018

Bates, D., Maechler, M., Bolker, B., & Walker, S. (2015). Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, *67*(1), 1–48.

Bengtson, S.-A. (1972). Reproduction and fluctuations in the size of duck populations at Lake Mývatn, Iceland. *Oikos*, *23*(1), 35–58. https://doi.org/10.2307/3543925

Blomberg, E. J., Gibson, D., & Sedinger, J. S. (2015). Biases in nest survival associated with choice of exposure period: A case study in North American upland game birds. *The Condor, American Ornithological Society*, *117*(4), 577–588.

Borgmann, K. L., & Conway, C. J. (2015). The nest-concealment hypothesis: New insights from a comparative analysis. *The Wilson Journal of Ornithology*, *127*(4), 646–660. https://doi.org/10.1676/14-162.1

Burhans, D. E., & Thompson, F. R. (1998). Effects of time and nest-site characteristics on concealment of songbird nests. *The Condor*, *100*(4), 663–672. https://doi.org/10.2307/1369747

Burnham, K. P., & Anderson, D. R. (2002). *Model selection and multimodel inference: A practical information theoretic approach*. New York, NY, USA: Springer-Verlag.

Calladine, J., Baines, D., & Warren, P. (2002). Effects of reduced grazing on population density and breeding success of black grouse in northern England. *The Journal of Applied Ecology*, *39*(5), 772–780. https://doi.org/10.1046/j.1365-2664.2002.00753.x

Chalfoun, A. D., & Schmidt, K. A. (2012). Adaptive breeding-habitat selection: Is it for the birds? *The Auk*, *129*(4), 589–599. https://doi.org/10.1525/auk.2012.129.4.589

Chambers, J. C., Bradley, B. A., Brown, C. S., D'Antonio, C., Germino, M. J., Grace, J. B., … Pyke, D. A.. (2014). Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. Invasion in Cold Desert Shrublands of Western North America. *Ecosystems*, *17*(2), 360–375. https://doi.org/10.1007/s10021-013-9725-5

Clark, R. G., & Nudds, T. D. (1991). Habitat patch size and duck nesting success: The crucial experiments have not been performed. *Wildlife Society Bulletin*, *19*(4), 534–543.

Coates, P. S., Connelly, J. W., & Delehanty, D. J. (2008). Predators of greater sage-grouse nests identified by video monitoring. *Journal of Field Ornithology*, *79*(4), 421–428. https://doi.org/10.1111/j.1557-9263.2008.00189.x

Coates, P. S., & Delehanty, D. J. (2008). Effects of environmental factors on incubation patterns of greater sage-grouse. *The Condor*, *110*(4), 627–638. https://doi.org/10.1525/cond.2008.8579

Coates, P. S., Ricca, M. A., Prochazka, B. G., Brooks, M. L., Doherty, K. E., Kroger, T., … Casazza, M. L. (2016). Wildfire, climate, and invasive grass interactions negatively impact an indicator species by reshaping sagebrush ecosystems. *Proceedings of the National Academy of Sciences*, *113*(45), 12745–12750. https://doi.org/10.1073/pnas.1606898113

Colombelli-Negrel, D., & Kleindorfer, S. (2009). Nest height, nest concealment, and predator type predict nest predation in superb fairy-wrens (*Malurus cyaneus*). *Ecological Research*, *24*(4), 921–928. https://doi.org/10.1007/s11284-008-0569-y

Connelly, J. W., & Braun, C. E. (1997). Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. *Wildlife Biology*, *3*(3–4), 229–234.

Connelly, J. W., Schroeder, M. A., Sands, A. R., & Braun, C. E. (2000). Guidelines to manage sage grouse populations and their habitats. *Wildlife Society Bulletin*, *28*(4), 967–985.

Conover, M. R., Borgo, J. S., Dritz, R. E., Dinkins, J. B., & Dahlgren, D. K. (2010). Greater sage-grouse select nest sites to avoid visual predators but not olfactory predators. *The Condor*, *112*(2), 331–336. https://doi.org/10.1525/cond.2010.090172

Crawford, J. A., Olson, R. A., West, N. E., Mosley, J. C., Schroeder, M. A., Whitson, T. D., … Boyd, C. S. (2004). Ecology and management of sage-grouse and sage-grouse habitat. *Journal of Range Management*, *57*(1), 2–19. https://doi.org/10.2307/4003949

DeLong, A. K., Crawford, J. A., & DeLong, D. C. (1995). Relationships between vegetational structure and predation of artificial sage grouse nests. *The Journal of Wildlife Management*, *59*(1), 88–92. https://doi.org/10.2307/3809119

Dinkins, J. B., Conover, M. R., Kirol, C. P., & Beck, J. L. (2012). Greater sage-grouse (*Centrocercus urophasianus*) select nest sites and brood sites away from avian predators. *The Auk*, *129*(4), 600–610. https://doi.org/10.1525/auk.2012.12009

Dion, N., Hobson, K. A., & Lariviere, S. (2000). Interactive effects of vegetation and predators on the success of natural and simulated nests of grassland songbirds. *The Condor*, *102*(3), 629–634. https://doi.org/10.1650/0010-5422(2000)102[0629:IEOVAP]2.0.CO;2

Doherty, K. E., Evans, J. S., Coates, P. S., Juliusson, L. M., & Fedy, B. C. (2016). Importance of regional variation in conservation planning: A rangewide example of the Greater Sage-Grouse. *Ecosphere*, *7*(10), e01462. Available from:https://doi.org/10.1002/ecs2.1462

Doherty, K. E., Naugle, D. E., Tack, J. D., Walker, B. L., Graham, J. M., & Beck, J. L. (2014). Linking conservation actions to demography: Grass height explains variation in greater sage-grouse nest survival. *Wildlife Biology*, *20*(6), 320–325. https://doi.org/10.2981/wlb.00004

Fogarty, D. T., Elmore, R. D., Fuhlendorf, S. D., & Loss, S. R. (2017). Influence of olfactory and visual cover on nest site selection and nest success for grassland-nesting birds. *Ecology and Evolution*, *7*, 6247–6258. https://doi.org/10.1002/ece3.3195

France, K. A., Ganskopp, D. C., & Boyd, C. S. (2008). Interspace/undercanopy foraging patterns of beef cattle in sagebrush habitats. *Rangeland Ecology & Management*, *61*(4), 389–393. https://doi.org/10.2111/06-072.1

Gibson, D., Blomberg, E. J., Atamian, M. T., & Sedinger, J. S. (2015). Observer effects strongly influence estimates of daily nest survival probability but do not substantially increase rates of nest failure in Greater Sage-Grouse. *The Auk*, *132*(2), 397–407. https://doi.org/10.1642/AUK-14-197.1

Gibson, D., Blomberg, E. J., Atamian, M. T., & Sedinger, J. S. (2016). Nesting habitat selection influences nest and early offspring survival in Greater Sage-Grouse. *The Condor*, *118*(4), 689–702. https://doi.org/10.1650/CONDOR-16-62.1

Gibson, D., Blomberg, E. J., & Sedinger, J. S. (2016). Evaluating vegetation effects on animal demographics: The role of plant phenology and sampling bias. *Ecology and Evolution*, *6*(11), 3621–3631. Available from:https://doi.org/10.1002/ece3.2148

Götmark, F., Blomqvist, D., Johansson, O. C., & Bergkvist, J. (1995). Nest site selection: A trade-off between concealment and view of the

surroundings? *Journal of Avian Biology*, *26*(4), 305–312. https://doi.org/10.2307/3677045
Gregg, M. A., & Crawford, J. A. (2009). Survival of greater sage-grouse chicks and broods in the Northern Great Basin. *The Journal of Wildlife Management*, *73*(6), 904–913. https://doi.org/10.2193/2007-410
Gregg, M. A., Crawford, J. A., Drut, M. S., & DeLong, A. K. (1994). Vegetational cover and predation of sage grouse nests in Oregon. *The Journal of Wildlife Management*, *58*(1), 162–166. https://doi.org/10.2307/3809563
Hagen, C. A., Connelly, J. W., & Schroeder, M. A. (2007). A meta-analysis of greater sage-grouse *Centrocercus urophasianus* nesting and brood-rearing habitats. *Wildlife Biology*, *13*, 42–50. https://doi.org/10.2981/0909-6396(2007)13[42:AMOGSC]2.0.CO;2
Hovick, T. J., Elmore, R. D., Allred, B. W., Fuhlendorf, S. D., & Dahlgren, D. K. (2014). Landscapes as a moderator of thermal extremes: A case study from an imperiled grouse. *Ecosphere*, *5*(3), 1–12.
Howlett, J. S., & Stutchbury, B. J. (1996). Nest concealment and predation in hooded warblers: Experimental removal of nest cover. *The Auk*, *113*(1), 1–9. https://doi.org/10.2307/4088930
Jahren, T., Storaas, T., Willebrand, T., Moa, P. F., & Hagen, B. R. (2016). Declining reproductive output in capercaillie and black grouse – 16 countries and 80 years. *Animal Biology*, *66*(3–4), 363–400. https://doi.org/10.1163/15707563-00002514
Kaczor, N. W., Herman-Brunson, K. M., & Jensen, K. C. (2011). Resource selection during brood-rearing by greater sage-grouse. In B. K. Sandercock, K. Martin, & G. Segelbacher (Eds.), *Ecology, conservation, and management of grouse, studies in avian biology* (pp. 169–177). Berkeley, CA, USA: University of California Press.
Knick, S. T., Dobkin, D. S., Rotenberry, J. T., Schroeder, M. A., Vander Haegen, W. M., & Van Riper, C. III (2003). Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats *The Condor*, *105*(4), 611–634. https://doi.org/10.1650/7329
Latif, Q. S., Heath, S. K., & Rotenberry, J. T. (2012). How avian nest site selection responds to predation risk: Testing an 'adaptive peak hypothesis'. *The Journal of Animal Ecology*, *81*(1), 127–138. https://doi.org/10.1111/j.1365-2656.2011.01895.x
Martin, T. E. (1992). Breeding productivity considerations: What are the appropriate habitat features for management? In J. M. Hagan, & D. W. Johnston (Eds.), *Ecology and conservation of neotropical migrant land birds*. Washington, DC, USA: Smithsonian Institute Press.
Martin, T. E. (1993). Nest predation and nest sites - new perspectives on old patterns. *BioScience*, *43*(8), 523–532. https://doi.org/10.2307/1311947
Martin, T. E., & Roper, J. J. (1988). Nest predation and nest-site selection of a western population of the hermit thrush. *The Condor*, *90*(1), 51–57. https://doi.org/10.2307/1368432
McConnell, M. D., Monroe, A. P., Burger, L. W., & Martin, J. A. (2017). Timing of nest vegetation measurement may obscure adaptive significance of nest-site characteristics: A simulation study. *Ecology and Evolution*, *7*(4), 1259–1270. Available from: http://dx.doi.org/10.1002/ece3.2767.
Miller, R. F., Naugle, D. E., Maestas, J. D., Hagen, C. A., & Hall, G. (2017). Special Issue: Targeted Woodland Removal to Recover At-Risk Grouse and Their Sagebrush-Steppe and Prairie Ecosystems. *Rangeland Ecology & Management*, *70*(1), 1–8. https://doi.org/10.1016/j.rama.2016.10.004
Mills, L. S. (2007). *Conservation of wildlife population: Demography, genetics and management*. Malden, MA, USA: Blackwell Publishing.
Moynahan, B. J., Lindberg, M. S., Rotella, J. J., & Thomas, J. W. (2007). Factors affecting nest survival of greater sage-grouse in northcentral montana. *The Journal of Wildlife Management*, *71*(6), 1773–1783. https://doi.org/10.2193/2005-386
Nakagawa, S., & Schielzeth, H. (2013). A general and simple method for obtaining R^2 from generalized linear mixed-effects models. *Methods in Ecology and Evolution*, *4*(2), 133–142. https://doi.org/10.1111/j.2041-210x.2012.00261.x
Naugle, D. E., Doherty, K. E., Walker, B. L., Holloran, M. J., & Copeland, H. E. (2011). Energy development and greater sage-grouse. In J. W. Connelly & S. T. Knick (Eds.), *Greater sage-grouse: Ecology and conservation of a landscape species and its habitats, studies in avian biology* (pp. 489–503). Berkeley, CA, USA: University of California Press
Peak, R. G. (2003). An experimental test of the concealment hypothesis using American goldfinch nests. *The Wilson Bulletin*, *115*(4), 403–408. https://doi.org/10.1676/03-024
Plummer, M. 2003. JAGS: A Program for Analysis of Bayesian Graphical Models Using Gibbs Sampling. mcmc-jags.sourceforge.net.
Plummer, M. 2016. rjags: Bayesian Graphical Models using MCMC. R package version 4-6. https://CRAN.R-project.org/package=rjags
R Core Team. 2016. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/
Remeš, V. (2005). Birds and rodents destroy different nests: A study of Blackcap *Sylvia atricapilla* using the removal of nest concealment. *Ibis*, *147*(1), 213–216.
Ricklefs, R. E. (1969). An analysis of nesting mortality in birds. *Smithsonian Contributions to Zoology*, *9*, 1–48. https://doi.org/10.5479/si.00810282.9
Schmidt, J. H., Walker, J. A., Lindberg, M. S., Johnson, D. S., & Stephens, S. E. (2010). A general Bayesian hierarchical model for estimating survival of nests and young. *The Auk*, *127*(2), 379–386. https://doi.org/10.1525/auk.2009.09015
Schreiber, L. A., Hansen, C. P., Rumble, M. A., Millspaugh, J. J., Gamo, R. S., Kehmeier, J. W., & Wojcik, N. (2015). Microhabitat selection of brood-rearing sites by greater sage-grouse in Carbon County, Wyoming. *Western North American Naturalist*, *75*(3), 348–363. https://doi.org/10.3398/064.075.0312
Smith, J. T., Evans, J. S., Martin, B. H., Baruch-Mordo, S., Kiesecker, J. M., & Naugle, D. E. (2016). Reducing cultivation risk for at-risk species: Predicting outcomes of conservation easements for sage-grouse. *Biological Conservation*, *201*, 10–19. https://doi.org/10.1016/j.biocon.2016.06.006
Stiver, S. J., Rinkes, E. T., Naugle, D. E., Makela, P. D., Nance, D. A., & Karl, J. W. (2015). *Sage-grouse habitat assessment framework: a multi-scale assessment tool*. Bureau of Land Management. Retrieved from: https://www.blm.gov/style/medialib/blm/wo/blm_library/tech_refs.Par.34086.File.dat/TR_6710-01.pdf.
Sveum, C. M., Edge, W. D., & Crawford, J. A. (1998). Nesting habitat selection by sage grouse in south-central Washington. *Journal of Range Management*, *51*(3), 265–269. https://doi.org/10.2307/4003409
Taylor, R. L., Walker, B. L., Naugle, D. E., & Mills, L. S. (2012). Managing multiple vital rates to maximize greater sage-grouse population growth. *The Journal of Wildlife Management*, *76*(2), 336–347. https://doi.org/10.1002/jwmg.267
Walker, B. L. 2008. Greater sage-grouse response to coal-bed natural gas development and West Nile virus in the Powder River Basin, Montana and Wyoming, USA. Dissertation, The University of Montana.

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ORIGINAL RESEARCH

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Phenology largely explains taller grass at successful nests in greater sage-grouse

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Abstract

Much interest lies in the identification of manageable habitat variables that affect key vital rates for species of concern. For ground-nesting birds, vegetation surrounding the nest may play an important role in mediating nest success by providing concealment from predators. Height of grasses surrounding the nest is thought to be a driver of nest survival in greater sage-grouse (*Centrocercus urophasianus*; sage-grouse), a species that has experienced widespread population declines throughout their range. However, a growing body of the literature has found that widely used field methods can produce misleading inference on the relationship between grass height and nest success. Specifically, it has been demonstrated that measuring concealment following nest fate (failure or hatch) introduces a temporal bias whereby successful nests are measured later in the season, on average, than failed nests. This sampling bias can produce inference suggesting a positive effect of grass height on nest survival, though the relationship arises due to the confounding effect of plant phenology, not an effect on predation risk. To test the generality of this finding for sage-grouse, we reanalyzed existing datasets comprising >800 sage-grouse nests from three independent studies across the range where there was a positive relationship found between grass height and nest survival, including two using methods now known to be biased. Correcting for phenology produced equivocal relationships between grass height and sage-grouse nest survival. Viewed in total, evidence for a ubiquitous biological effect of grass height on sage-grouse nest success across time and space is lacking. In light of these findings, a reevaluation of land management guidelines emphasizing specific grass height targets to promote nest success may be merited.

KEYWORDS
Centrocercus urophasianus, concealment, greater sage-grouse, nest survival, phenology

1 | INTRODUCTION

Environmental factors affecting influential demographic parameters are appropriate targets of management to promote habitat quality for species of conservation concern (Mills, 2007). For many birds, characteristics of nest sites that influence nest predation are of interest, as nest success is a key driver of population growth and predation is the primary cause of nest failure (Martin, 1993; Ricklefs, 1969). According to the nest concealment hypothesis, nests surrounded by dense vegetation should be more successful because they are more difficult for predators to detect or access (Martin, 1992; Martin & Roper, 1988). Furthermore, vegetative concealment may represent an attractive target for conservation action because it can often be managed, for example, through manipulation of herbivory by livestock.

Support for the nest concealment hypothesis is mixed. In a recent review and comparative analysis, 26% of 114 reviewed studies in open-cup-nesting songbirds supported an effect (Borgmann & Conway, 2015). Effects of concealment on nest survival may be difficult to detect if strong selection for concealed nest sites canalizes variation among nests such that most occur in “adaptive peaks” providing adequate concealment (Latif, Heath, & Rotenberry, 2012; Remeš, 2005). However, even studies employing experimental removal of vegetation have returned mixed support for the nest concealment hypothesis (e.g., Bengtson, 1972; Howlett & Stutchbury, 1996; Latif et al., 2012; Peak, 2003). Numerous intrinsic and extrinsic factors may influence the effect of concealment on nest success. For example, birds with more brightly colored plumage appear more dependent on vegetation to conceal the nest from predators (Borgmann & Conway, 2015), and the benefits of visual concealment may depend on the composition of the local predator community (Clark & Nudds, 1991; Colombelli-Negrel & Kleindorfer, 2009; Dion, Hobson, & Lariviere, 2000). More problematic, however, are methodological aspects of studies that produce biased inference with regard to effects of concealment on nest survival (Borgmann & Conway, 2015; Burhans & Thompson, 1998; Gibson, Blomberg, & Sedinger, 2016; McConnell, Monroe, Burger, & Martin, 2017). Here, we focus on a recently highlighted methodological bias pervasive in research regarding habitat–fitness relationships in greater sage-grouse (*Centrocercus urophasianus*).

The greater sage-grouse (hereafter, sage-grouse) is a precocial, ground-nesting species of conservation concern inhabiting sagebrush ecosystems of western North America. Although sage-grouse nest beneath shrubs—primarily sagebrush—perennial grasses and forbs in the interspaces between shrubs have long been thought to provide critical concealment of nests from potential predators (Connelly, Schroeder, Sands, & Braun, 2000). This hypothesis is supported by studies reporting positive associations between height and/or cover of herbaceous vegetation surrounding nest sites and nest survival (Coates & Delehanty, 2008; DeLong, Crawford, & DeLong, 1995; Doherty et al., 2014; Gregg, Crawford, Drut, & DeLong, 1994; Sveum, Edge, & Crawford, 1998). Consequently, sage-grouse conservation efforts and land management policy have focused on increasing herbaceous hiding cover in suitable nesting habitat throughout the range of the species. Although direct links between livestock grazing and sage-grouse demography are lacking, studies indicating positive effects of herbaceous vegetation height and/or cover on nest survival provide a plausible mechanism linking livestock grazing and nest success (Connelly & Braun, 1997; Connelly et al., 2000), a key demographic rate for sage-grouse (Taylor, Walker, Naugle, & Mills, 2012). Thus, the validity of inference about the importance of herbaceous hiding cover for sage-grouse nest success has major implications for the management of sagebrush ecosystems, where livestock grazing is a ubiquitous land use (Knick et al., 2003).

Recent evidence has demonstrated that the positive association between grass height, a commonly used metric of herbaceous concealing cover among sage-grouse nesting studies, and nest survival may be indicative of biased methods rather than a causal relationship (Gibson, Blomberg, et al., 2016; McConnell et al., 2017). Using both empirical and simulation approaches, it has been shown that measuring grass height at nests following nest fate (i.e., hatch or failure) produces inflated or even spurious statistical relationships between grass height and nest survival. Because successful nests persist and are therefore measured later in the season than failed nests, measured concealment is greater at successful nests due to concurrent plant growth rather than a presumed reduction in predation. Despite knowledge of this sampling issue dating back decades (e.g., Burhans & Thompson, 1998), this sampling bias remains pervasive in sage-grouse and other ground-nesting bird literature, with a majority of sage-grouse studies sampling vegetation following nest fate (Gibson, Blomberg, et al., 2016).

Given the far-reaching implications derived from inference about grass height and sage-grouse demography, we were interested in exploring the generality of recent findings reported by Gibson, Blomberg, et al. (2016), and McConnell et al. (2017). Using field data from four geographically distinct study sites representative of the diversity of vegetation communities, predator communities, precipitation regimes, and evolutionary history of grazing found across the range of sage-grouse, we tested the hypothesis that studies using biased field methods that had previously supported a positive association between grass height measured around the nest and nest survival would fail to support such an association after accounting for phenology.

2 | METHODS

We employed the model-based methods presented in Gibson, Blomberg, et al. (2016) to correct for phenology in a reanalysis of three datasets from Montana, Utah, and Wyoming (Table 1). In a dataset from Eureka County, Nevada, analyzed by Gibson, Blomberg, et al. (2016), vegetation measurements were made at predicted hatch date and a linear regression relating vegetation height to the date of measurement was used to predict vegetation height at fate date, thereby demonstrating the potential bias arising from such a sampling scheme. We employed this concept in reverse fashion, that is, we regressed vegetation height on date of measurement to predict grass height at hatch date, as although it had been sampled using unbiased methods.

2.1 | Datasets

Reanalyzed datasets included a previously published study that found a significant positive influence of live grass height on sage-grouse nest survival across two study areas in the Powder River Basin (PRB) in southeast Montana (hereafter PRB North, $n = 209$) and northeast

TABLE 1 We used predictions from five studies across the range of greater sage-grouse, representing n = 1204 nests over a total of 24 study site-years

Study area	*n*	Years	Transect length (m)	Samples per nest	Data source
Eureka County	396	2004-2012	10	10	Gibson, Blomberg, et al. (2016);
PRB North	209	2003-2006	30	20	Doherty et al. (2014)
PRB South	174	2004-2006	30	20	Doherty et al. (2014)
Roundup	320	2012-2015	12	8	J. Smith, Unpublished Data
NE Utah	105	2012-2015	30	20	S. Dettenmaier, Unpublished Data
Total	1204				

Each study sampled grass height similarly, using measurements of the nearest grass height to various points along two intersecting transects centered at the nesting shrub. However, total transect length and the number of samples per nest varied by study.

Wyoming (hereafter PRB South, n = 164; Doherty et al., 2014); preliminary data from an ongoing evaluation of grazing treatments on sage-grouse ecology in central Montana (Joseph Smith, University of Montana, Unpublished Data, n = 320); and the first 4 years of a study comparing sage-grouse demography across two study areas in northern Utah (Seth Dettenmaier, Utah State University, Unpublished Data, n = 105). Including findings from Gibson, Blomberg, et al. (2016), these studies encompassed 1204 sage-grouse nests over 24 study site-years from across the range of sage-grouse (Table 1). Each study used similar methodologies to sample herbaceous vegetation surrounding nest sites by taking multiple measurements of grass height along intersecting transects centered on the nesting shrub and using the mean of replicated measurements to represent grass height-surrounding nests (Table 1).

2.2 | Statistical analyses

We assumed hatch date was 27 days after the estimated nest initiation date and applied a correction to measured grass height covariates following Gibson, Blomberg, et al. (2016):

$$\text{GrassHeight}_{\text{Hatch}} = \text{GrassHeight}_{\text{Fate}} - (\text{SurveyDate}_{\text{Fate}} - \text{SurveyDate}_{\text{Hatch}}) \times \beta_{\text{grass}}$$

where, for each study area and year, we fit a linear regression of measured grass height ($\text{GrassHeight}_{\text{Fate}}$) on day of nesting season ($\text{SurveyDate}_{\text{Fate}}$) to estimate β_{grass}. This simple correction provided a standardized measurement for grass height across nests regardless of fate. We estimated the effect of grass height on nest success using both corrected and uncorrected covariate measurements by fitting Bayesian daily nest survival models to each dataset (Schmidt, Walker, Lindberg, Johnson, & Stephens, 2010) with the exception of data from Gibson, Blomberg, et al. (2016), who provided estimates from their published analysis. In this approach, we estimated nest survival (S) for each nest (i) on each day of the nesting season (t) via a logit-linear model, which at minimum included an intercept (β_0) and coefficient for grass height, while also including coefficients that respective authors deemed supportive in top models. Nest encounter histories consisted of observed nest states (y) for each day of observation, where $y_{i,t}$ = 1 if nest i was observed alive on day t, $y_{i,t}$ = 0 if nest i was observed to have failed (female absent and some or all eggs destroyed), and $y_{i,t}$ = NA on days when nest state was not observed. Beginning on the first day after the nest was detected,

$$y_{i,t} \sim \text{Bern}(y_{i,t-1} S_{i,t})$$

and

$$\text{logit}\,(S_{i,t}) = \beta_0 + x_i{}'\beta$$

Specifically, Doherty et al. (2014), following the original population analyses in Walker (2008), modeled nest survival using covariates including a main and quadratic effect for nest age, and categorical variables for a particularly harsh spring nesting season with major snow events that caused nest abandonment (2003) and the two study regions (PRB North and PRB South). Although the PRB datasets were collected independently, they were combined in the analysis presented in Doherty et al. (2014), and we combine them here for consistency. Although it appears this study was mistakenly recorded as having used a fate date protocol in Gibson, Blomberg, et al. (2016; Table 1), the investigators did attempt to control for phenology by sampling vegetation near the predicted hatch date regardless of nest fate. Nonetheless, close examination of the dataset revealed that a temporal bias in measurement date existed across all study site-year combinations, such that successful nests were measured from 2 to 10 days later than failed nests, on average. To attempt to correct this persistent bias and maintain consistency among reanalyzed datasets, we corrected grass heights to predicted hatch date in the PRB North and PRB South datasets, but these corrections were generally smaller than corrections in the other reanalyzed datasets. Unpublished data from J. Smith included covariates for the log of distance to major roads and a measure of 4-day cumulative rainfall, as well as a random effect for year. Data from Gibson, Blomberg, et al. (2016), and models fit to Utah data included only an intercept and coefficient for measurements of grass height. Our estimates of daily nest survival and nest success are only reflective of the incubation period, as sage-grouse nests are typically found after the onset of incubation, and thus overestimate true nest success from initiation to hatch (Blomberg, Gibson, & Sedinger, 2015). Moreover, as monitoring intensity of prenesting females may have varied among datasets, incubation success may be more or less biased relative to true nest success and overall success rates are therefore not directly comparable among studies.

We fit daily nest survival models in JAGS 4.0 (Plummer, 2003) with the package rjags (Plummer 2016) in R 3.3.0 (R Core Team 2016), estimating posterior distributions with a total of 90,000 samples from 3 independent Markov chain Monte Carlo (MCMC) chains (30,000 per chain) after discarding the first 20,000 iterations from each chain for burn-in. We placed vague normal prior distributions on all coefficients (μ=0; σ=1000). Using coefficient posterior distributions, we generated predictions for the mean influence of grass height on nest success, the product of daily nest survival over a 27-day incubation period, and 95% credible intervals over the range of grass height values observed within each respective dataset. We held additional covariates at their mean value where applicable.

We performed an additional analysis to provide a comprehensive assessment of the influence of grass height on nest survival across datasets, excluding nests from Eureka County for which we only had data on the predicted response. Here, we pooled datasets and used generalized linear mixed models to test whether grass surrounding successful nests was taller than grass surrounding failed nests after accounting for phenology. Under the null hypothesis, grass heights (GH) measured at nests are a linear function of ordinal date of measurement (DAY; days since January 1), with normally distributed errors and no difference between successful and failed nests. Our alternative hypothesis was that grass is taller at successful nests than at failed nests after accounting for the linear function of ordinal date. We first used AIC_C model selection (Burnham & Anderson, 2002) to determine the best structure for a null (i.e., phenology) model. We considered a phenology model with a random intercept for each study area-year (1|STUDY:YEAR) combination to allow for variation in grass height inherent among geographically distant study areas and in different years, and a random intercepts and slopes phenology model (DAY|STUDY:YEAR) to allow for different rates of grass growth among years and study areas. To aid in model convergence, we centered the independent variable DAY by subtracting the median day of measurement from all observations. After we determined the best structure for the phenology model using AIC_C, we used a likelihood ratio test to assess support for our alternative hypothesis, which was represented with a model following the structure of the most supported phenology model and including a categorical fixed effect for nest fate (FATE; failed = 0, hatched = 1). Linear mixed models were fit using the lme4 package (Bates, Maechler, Bolker, & Walker, 2015) in R. Using these datasets, we also tabulated all corrected grass height measurements at successful and failed nests and performed a one-sided Kolmogorov–Smirnov test to examine if distributions of measurements differed between pooled data sets. A one-sided test was chosen to increase statistical power given our a priori expectation that grass would be taller surrounding successful nests than failed nests.

3 | RESULTS

Uncorrected, each of the three reanalyzed datasets revealed a strong, positive association between grass height and daily nest survival (Figure 1; dotted lines). Estimated coefficients for grass height using uncorrected grass heights were 0.063 (95% CI from 0.037 to 0.092) for PRB North and PRB South, 0.099 (95% CI from 0.063 to 0.137) for Roundup, and 0.058 (95% CI from 0.002 to 0.118) for NE Utah. Corrections to measured grass heights averaged—1.32 cm and mean absolute correction (|corrected–uncorrected|) was 2.08 cm, with a standard deviation of 2.31 cm. Following adjustment of measured grass heights to remove temporal bias, we found no association between grass height and nest survival in two of the three datasets (Roundup and NE Utah), and a weakened but persistent association in the PRB dataset (Figure 1; solid lines). Estimated coefficients for grass height

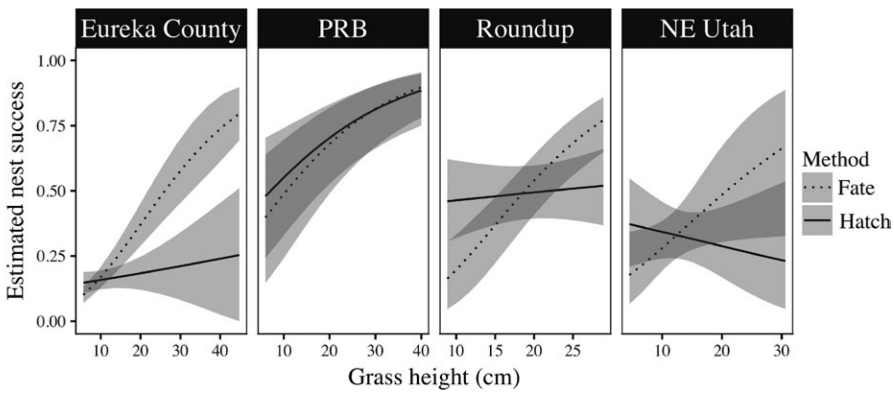

FIGURE 1 Predicted response of sage-grouse nest success (and 95% CI [Eureka County] or CRI [other studies]) to live grass height using measurements collected with a biased method following determination of nest fate (dotted lines), and those measured or corrected to the predicted hatch date of nests (solid lines). Nest data includes studies from the powder river basin (PRB) in southeastern Montana (PRB North, Doherty et al., 2014, *n* = 209, 2003–2006) and northeast Wyoming (PRB South, Doherty et al., 2014, *n* = 174, 2004–2006); Eureka County, Nevada (Gibson, Blomberg, et al., 2016, *n* = 396, 2004–2012); central Montana near the town of Roundup (J. Smith, University of Montana, unpublished data, *n* = 320, 2012–2015), and northeast Utah (Dettenmaier, Utah State University, unpublished data; *n* = 105, 2012–2015). Note that limits of x-axes change to reflect the range of grass heights observed within respective studies

using corrected grass heights were 0.053 (95% CI from 0.025 to 0.081) for PRB North and PRB South, 0.008 (95% CI from -0.027 to 0.042) for Roundup, and −0.015 (95% CI from −0.060 to 0.032) for NE Utah.

The random intercept and slope phenology model (conditional R^2 = 0.51 [Nakagawa & Schielzeth, 2013]) received the most support with an AIC_C score 9.64 units lower than the constant slope model (conditional R^2 = .46) and was used as the null model (Figure 2). The alternative hypothesis, that grass height surrounding successful nests was greater than that surrounding failed nests after accounting for phenology, was not supported (χ^2 = 2.74, *df* = 1, *p* = .098). Overall, median height of live grasses, corrected to hatch date, was 15.3 cm at successful nests (*n* = 336) and 15.1 cm at failed nests (*n* = 472; Figure 3). A one-sided Kolmogorov–Smirnov test provided no evidence that the distributions of phenology-corrected grass heights differed between successful and failed nests when pooling across sites and years (*p* = .307).

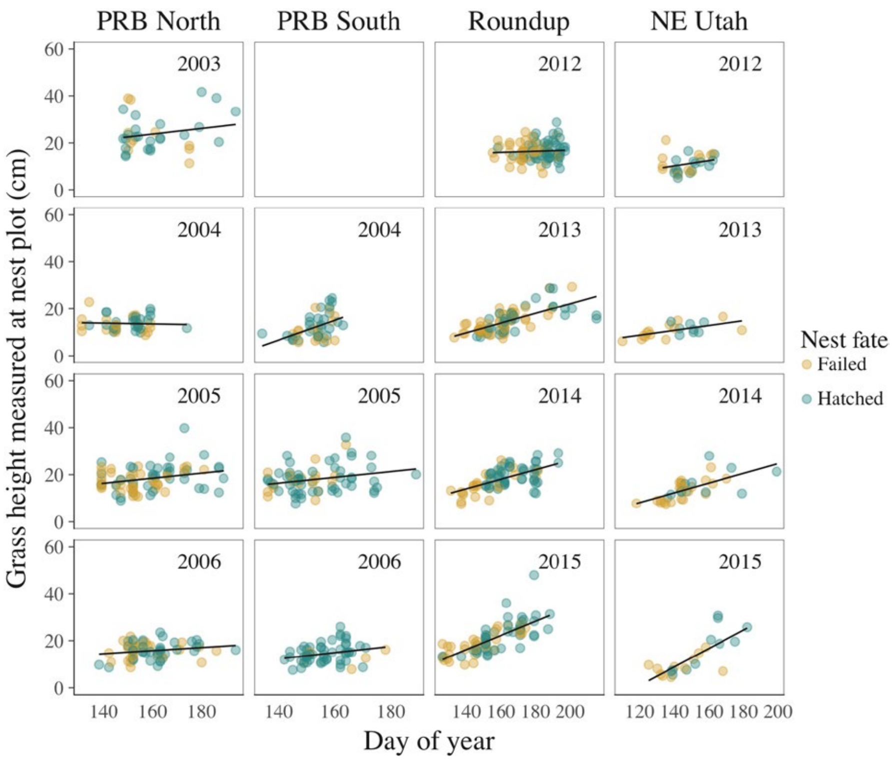

FIGURE 2 Average grass height surrounding successful and failed sage-grouse nests (*n* = 808) at the ordinal date of measurement by year (rows) and study area (columns). After accounting for phenology, a difference in grass height between successful and failed nests was not supported

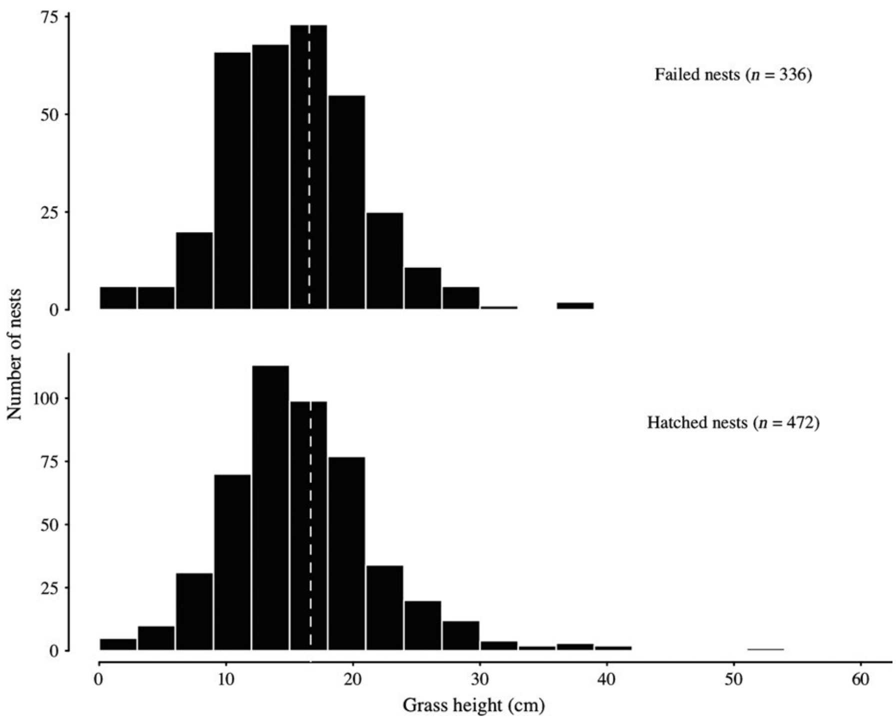

FIGURE 3 Grass heights surrounding greater sage-grouse nests (*n* = 808) corrected to hatch date. Median height of grass-surrounding nests (dashed vertical lines) was 15.26 cm at successful nests and 15.14 cm at failed nests. A one-sided Kolmogorov–Smirnov test provided no evidence that the distributions of grass heights differed between successful and failed nests (ground-nesting *p* = .307)

4 | DISCUSSION

While our analyses revealed mixed support for relationships between grass height and nest survival in sage-grouse, they confirmed recent findings that associations between herbaceous vegetation structure and nest success are frequently byproducts of temporally biased sampling rather than indicative of effect of concealing cover on detectability by predators (Gibson, Blomberg, et al., 2016; McConnell et al., 2017). Sampling vegetation following nest fate, a pervasive practice in studies of sage-grouse and other ground-nesting birds, consistently produces spurious relationships between grass height and nest survival and should, therefore, be avoided. As field crews are rarely able to strictly adhere to a schedule due to weather or other logistic constraints, even studies using field protocols intended to control for phenology may be affected by some degree of temporal bias between failed and successful nests, producing inflated effect sizes (e.g., the PRB dataset reanalyzed here; Doherty et al., 2014).

Taller grass may be associated with reduced nest predation under some conditions, such as in the context of particular predator communities or in years with particularly tall grass. However, grass height does not appear to be a universal indicator of nesting habitat quality for sage-grouse. Including the PRB dataset, we are aware of only three published studies using unbiased methods that support a positive association between grass height and nest survival (Doherty et al., 2014; Gregg et al., 1994; Sveum et al., 1998) among the 11 published studies testing for such an effect (Table 1 in Gibson, Blomberg, et al., 2016). Although the results have generally been interpreted to support the hypothesis that taller grass promotes greater nest survival (Connelly et al., 2000; Crawford et al., 2004), data presented by Sveum et al. (1998; Table 2) merely indicated that cover of short grasses (<18 cm) was lower at successful nests than failed nests in 1 out of 2 years (n = 32 nests), while cover of tall grasses (≥18 cm) did not differ between successful and failed nests in any year, even using a liberal α level of 0.1. Positive relationships between grass height and nest survival may, in fact, be uncommon. It is telling that, when analyzed together, data from the four study areas examined here provided no evidence for a difference in herbaceous vegetation height between successful and failed nests after accounting for plant phenology and timing of sampling (Figures 2 and 3).

The research and management communities must guard against uncritical acceptance of intuitive but untested mechanistic explanations for correlative patterns emerging from observational studies of habitat–fitness relationships. Within the sagebrush ecosystem, the broad acceptance that taller grass causes greater nest success by concealing nests from predators is an example of this type of untested logical connection, as equally plausible alternative hypotheses exist. For example, in multiyear studies, annual variation in precipitation and temperature in the prenesting and nesting periods may simultaneously affect female body condition, incubation behavior, and plant phenology. If conditions favorable to increased body condition or nest attentiveness have coincident positive effects on grass growth, nest success may be positively correlated with grass height absent any causal relationship between the two variables.

An experimental approach involving manipulation of vegetation height-surrounding nests could circumvent these issues, but would be fraught with its own set of difficulties. Sage-grouse females display a propensity toward abandoning reproductive efforts following disturbance by investigators (e.g., Gibson, Blomberg, Atamian, & Sedinger, 2015; Moynahan, Lindberg, Rotella, & Thomas, 2007). Disturbance from experimental manipulation at treatment nests would, therefore, need to be simulated at control nests such that observer-induced abandonment rates would be equal among nests in both groups. This may present an ethical dilemma for a species of conservation concern, or may simply yield sample sizes with inappropriately low statistical power. Furthermore, results of such an experiment would be of questionable relevance to management if manipulations bore little resemblance to defoliation patterns arising via herbivory (France, Ganskopp, & Boyd, 2008). Thus, experimental research is unlikely to provide an easy resolution to the problem. A critical examination of past evidence and careful consideration of alternative mechanistic hypotheses are warranted when considering the observational evidence at hand.

Habitat–fitness relationships are often context-dependent, and therefore variable across a species' range. Effects of concealment on nest survival, for example, may be more likely where cover is sparse. If that were the case, we might expect effects of grass height on nest survival to be more common in study sites characterized by low-shrub cover-surrounding nests. Indeed, the positive association between grass height and nest survival in the PRB study site reanalyzed here occurred in the eastern portion of the range, characterized by high spring precipitation and herbaceous vegetation cover compared to the rest of the sage-grouse range (Doherty, Evans, Coates, Juliusson, & Fedy, 2016). However, there was no relationship between grass height and nest survival in the Roundup study area, which had the lowest average shrub cover (18%) among datasets we considered. Selection of nest sites surrounded by tall grasses (Hagen, Connelly, & Schroeder, 2007) may result in a truncated covariate space such that nests surrounded by very short vegetation are rarely observed, thereby precluding the ability to detect an effect on survival (Chalfoun & Schmidt, 2012; Latif et al., 2012). However, with data from 15 study site-year combinations, we are confident we have surveyed a representative range of conditions chosen by nesting females. The lack of difference in grass height between successful and failed nests across these datasets strongly suggests that height of grasses was not a limiting resource (Figure 3).

The absence of support for an effect of grass height does not imply concealment is wholly unrelated to nest survival in sage-grouse. Selection for larger, taller sagebrush for nest substrates and preference for nesting in areas with greater areal cover of shrubs are well documented (reviewed in Hagen et al., 2007). In preferred sites, grasses and forbs may simply provide little additional visual or olfactory obstruction between a nest and a potential predator beyond that already provided by shrubs (see France, Ganskopp, & Boyd, 2008). Furthermore, while grasses and forbs afford mostly lateral cover, shrubs may provide more effective cover from aerial visual predators such as common ravens (*Corvus corax*), a primary nest predator for sage-grouse (Coates, Connelly, & Delehanty, 2008; Coates & Delehanty, 2008). Previous research indicates nest site selection in sage-grouse is driven by avian predators at broad scales (Dinkins, Conover, Kirol, & Beck, 2012) and characteristics of nest sites at small scales are more consistent with avoidance of visual (i.e., avian) predators than olfactory (i.e., mammalian) predators (Conover, Borgo, Dritz, Dinkins, & Dahlgren, 2010; Fogarty, Elmore, Fuhlendorf, & Loss, 2017). The lack of association between height of grasses and survival may also indicate a trade-off between nest concealment and the ability of incubating females to detect predators from a distance and alter their behavior in such a way as to reduce detection (Götmark, Blomqvist, Johansson, & Bergkvist, 1995).

Nest success is only one among several influential vital rates affecting sage-grouse population growth, and further research is needed to address how structure of grasses and forbs affects other life stages in sage-grouse. Studies of other grouse suggest vegetation height may be an important driver of brood survival. For example, increased vegetation height and/or greater insect abundance resulting from reduced grazing intensity positively affected production in black grouse (*Tetrao tetrix*) in Britain (Baines, 1996; Calladine, Baines, & Warren, 2002). The positive effect on production was, however, diminished or even reversed when grazing reduction treatments covered larger areas (Calladine et al., 2002), suggesting mosaics of vegetation height may confer greater benefits than uniformly tall vegetation (also see Baines, Richardson, & Warren, 2017; Jahren, Storaas, Willebrand, Moa, & Hagen, 2016). Taller vegetation may also moderate thermal extremes experienced by grouse, a function which may take on increased importance under climate change (Hovick, Elmore, Allred, Fuhlendorf, & Dahlgren, 2014). Although selection of sites with greater visual concealment by brood-rearing sage-grouse has been documented (Kaczor, Herman-Brunson, & Jensen, 2011; Schreiber et al., 2015), studies testing effects of herbaceous vegetation structure on sage-grouse chick survival are few and have produced mixed results (Aldridge, 2005; Gregg & Crawford, 2009). Recently, Gibson, Blomberg, et al. (2016) found survival of sage-grouse chicks to 2 weeks of age was positively associated with height of grasses surrounding the nest, presumably because structure of vegetation at the nest site is assumed to be correlated with structure of vegetation encountered by the precocial chicks during the first weeks of life. Again, however, a causal relationship between grass height and chick survival cannot be inferred. Positive relationships between herbaceous plant height and chick survival could implicate concealment from predators, but it is also plausible that taller grass at the nest is associated with some unmeasured factor—for example, site productivity, precipitation, or soil moisture—which in turn influences factors causally related to chick survival.

While the herbaceous understory is a key component of sagebrush ecosystems and sage-grouse habitat (e.g., Chambers et al., 2014), its role in concealing nests from predators has been overstated in management guidelines and land management documents. For example, the habitat assessment framework (HAF; Stiver et al., 2015), a tool used by the US Bureau of Land Management and US Forest Service to evaluate whether public lands are meeting habitat requirements of sage-grouse, included guidelines for maintaining a minimum height of perennial grasses and forbs in upland nesting habitat (18 cm) based largely on studies suggesting positive effects of vegetation height on nest success. There is, however, little evidence for the existence of the causal relationship between grass height and nest survival on which these guidelines were predicated. While it appears these "fourth order" guidelines may place unwarranted emphasis on the importance of maintaining herbaceous hiding cover for nesting, it should be noted that the HAF appropriately lays out a hierarchical management approach which suggests policies be set at the rangewide and regional scales to limit habitat loss and fragmentation—known causes of population declines among prairie grouse—but emphasizes that significant flexibility should be granted to local managers applying finer scale guidelines (see Chapter 1, Stiver et al., 2015). Persistent, broad-scale threats to sagebrush ecosystems including oil and gas development (Naugle, Doherty, Walker, Holloran, & Copeland, 2011), wildfire and invasive annual grasses (Coates et al., 2016), cropland conversion (Smith et al., 2016), and conifer encroachment (Miller, Naugle, Maestas, Hagen, & Hall, 2017) are well-documented drivers of sage-grouse population declines and should therefore be the highest priority for managers. Maintenance of tall grasses and forbs for nesting cover should not distract managers from addressing these larger threats or preclude the use of management tools that could otherwise improve sage-grouse habitat.

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AUTHOR CONTRIBUTIONS

JTS conceptualized the study, collected field data in central Montana, compiled and quality checked data from all study sites, analyzed data, produced figures, and wrote the manuscript. JDT analyzed data, produced figures, and assisted in writing the manuscript. KED collected field data in PRB and assisted in writing the manuscript. BWA, JDM, and DEN assisted with study conceptualization, interpretation of results, and manuscript writing, and revised several early versions of the manuscript. LIB and TAM contributed field data in central Montana and Northern Utah, respectively, and critically revised the final manuscript. SJD collected field data in Northern Utah. All authors critically revised and approved the final version of the manuscript.

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REFERENCES

Aldridge, C. (2005). Identifying habitats for persistence of greater sage-grouse (*Centrocercus urophasianus*) in Alberta, Canada. Dissertation, University of Alberta.

Baines, D. (1996). The Implications of grazing and predator management on the habitats and breeding success of black grouse *Tetrao tetrix*. *The Journal of Applied Ecology*, *33*(1), 54–62. https://doi.org/10.2307/2405015

Baines, D., Richardson, M., & Warren, P. (2017). The invertebrate diet of Black Grouse *Tetrao tetrix* chicks: A comparison between northern England and the Scottish Highlands. *Bird Study*, *64*(2), 117–124. https://doi.org/10.1080/00063657.2017.1295018

Bates, D., Maechler, M., Bolker, B., & Walker, S. (2015). Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, *67*(1), 1–48.

Bengtson, S.-A. (1972). Reproduction and fluctuations in the size of duck populations at Lake Mývatn, Iceland. *Oikos*, *23*(1), 35–58. https://doi.org/10.2307/3543925

Blomberg, E. J., Gibson, D., & Sedinger, J. S. (2015). Biases in nest survival associated with choice of exposure period: A case study in North American upland game birds. *The Condor, American Ornithological Society*, *117*(4), 577–588.

Borgmann, K. L., & Conway, C. J. (2015). The nest-concealment hypothesis: New insights from a comparative analysis. *The Wilson Journal of Ornithology*, *127*(4), 646–660. https://doi.org/10.1676/14-162.1

Burhans, D. E., & Thompson, F. R. (1998). Effects of time and nest-site characteristics on concealment of songbird nests. *The Condor*, *100*(4), 663–672. https://doi.org/10.2307/1369747

Burnham, K. P., & Anderson, D. R. (2002). *Model selection and multimodel inference: A practical information theoretic approach*. New York, NY, USA: Springer-Verlag.

Calladine, J., Baines, D., & Warren, P. (2002). Effects of reduced grazing on population density and breeding success of black grouse in northern England. *The Journal of Applied Ecology*, *39*(5), 772–780. https://doi.org/10.1046/j.1365-2664.2002.00753.x

Chalfoun, A. D., & Schmidt, K. A. (2012). Adaptive breeding-habitat selection: Is it for the birds? *The Auk*, *129*(4), 589–599. https://doi.org/10.1525/auk.2012.129.4.589

Chambers, J. C., Bradley, B. A., Brown, C. S., D'Antonio, C., Germino, M. J., Grace, J. B., … Pyke, D. A.. (2014). Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. Invasion in Cold Desert Shrublands of Western North America. *Ecosystems*, *17*(2), 360–375. https://doi.org/10.1007/s10021-013-9725-5

Clark, R. G., & Nudds, T. D. (1991). Habitat patch size and duck nesting success: The crucial experiments have not been performed. *Wildlife Society Bulletin*, *19*(4), 534–543.

Coates, P. S., Connelly, J. W., & Delehanty, D. J. (2008). Predators of greater sage-grouse nests identified by video monitoring. *Journal of Field Ornithology*, *79*(4), 421–428. https://doi.org/10.1111/j.1557-9263.2008.00189.x

Coates, P. S., & Delehanty, D. J. (2008). Effects of environmental factors on incubation patterns of greater sage-grouse. *The Condor*, *110*(4), 627–638. https://doi.org/10.1525/cond.2008.8579

Coates, P. S., Ricca, M. A., Prochazka, B. G., Brooks, M. L., Doherty, K. E., Kroger, T., … Casazza, M. L. (2016). Wildfire, climate, and invasive grass interactions negatively impact an indicator species by reshaping sagebrush ecosystems. *Proceedings of the National Academy of Sciences*, *113*(45), 12745–12750. https://doi.org/10.1073/pnas.1606898113

Colombelli-Negrel, D., & Kleindorfer, S. (2009). Nest height, nest concealment, and predator type predict nest predation in superb fairy-wrens (*Malurus cyaneus*). *Ecological Research*, *24*(4), 921–928. https://doi.org/10.1007/s11284-008-0569-y

Connelly, J. W., & Braun, C. E. (1997). Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. *Wildlife Biology*, *3*(3–4), 229–234.

Connelly, J. W., Schroeder, M. A., Sands, A. R., & Braun, C. E. (2000). Guidelines to manage sage grouse populations and their habitats. *Wildlife Society Bulletin*, *28*(4), 967–985.

Conover, M. R., Borgo, J. S., Dritz, R. E., Dinkins, J. B., & Dahlgren, D. K. (2010). Greater sage-grouse select nest sites to avoid visual predators but not olfactory predators. *The Condor*, *112*(2), 331–336. https://doi.org/10.1525/cond.2010.090172

Crawford, J. A., Olson, R. A., West, N. E., Mosley, J. C., Schroeder, M. A., Whitson, T. D., … Boyd, C. S. (2004). Ecology and management of sage-grouse and sage-grouse habitat. *Journal of Range Management*, *57*(1), 2–19. https://doi.org/10.2307/4003949

DeLong, A. K., Crawford, J. A., & DeLong, D. C. (1995). Relationships between vegetational structure and predation of artificial sage grouse nests. *The Journal of Wildlife Management*, *59*(1), 88–92. https://doi.org/10.2307/3809119

Dinkins, J. B., Conover, M. R., Kirol, C. P., & Beck, J. L. (2012). Greater sage-grouse (*Centrocercus urophasianus*) select nest sites and brood sites away from avian predators. *The Auk*, *129*(4), 600–610. https://doi.org/10.1525/auk.2012.12009

Dion, N., Hobson, K. A., & Lariviere, S. (2000). Interactive effects of vegetation and predators on the success of natural and simulated nests of grassland songbirds. *The Condor*, *102*(3), 629–634. https://doi.org/10.1650/0010-5422(2000)102[0629:IEOVAP]2.0.CO;2

Doherty, K. E., Evans, J. S., Coates, P. S., Juliusson, L. M., & Fedy, B. C. (2016). Importance of regional variation in conservation planning: A rangewide example of the Greater Sage-Grouse. *Ecosphere*, *7*(10), e01462. Available from:https://doi.org/10.1002/ecs2.1462

Doherty, K. E., Naugle, D. E., Tack, J. D., Walker, B. L., Graham, J. M., & Beck, J. L. (2014). Linking conservation actions to demography: Grass height explains variation in greater sage-grouse nest survival. *Wildlife Biology*, *20*(6), 320–325. https://doi.org/10.2981/wlb.00004

Fogarty, D. T., Elmore, R. D., Fuhlendorf, S. D., & Loss, S. R. (2017). Influence of olfactory and visual cover on nest site selection and nest success for grassland-nesting birds. *Ecology and Evolution*, *7*, 6247–6258. https://doi.org/10.1002/ece3.3195

France, K. A., Ganskopp, D. C., & Boyd, C. S. (2008). Interspace/undercanopy foraging patterns of beef cattle in sagebrush habitats. *Rangeland Ecology & Management*, *61*(4), 389–393. https://doi.org/10.2111/06-072.1

Gibson, D., Blomberg, E. J., Atamian, M. T., & Sedinger, J. S. (2015). Observer effects strongly influence estimates of daily nest survival probability but do not substantially increase rates of nest failure in Greater Sage-Grouse. *The Auk*, *132*(2), 397–407. https://doi.org/10.1642/AUK-14-197.1

Gibson, D., Blomberg, E. J., Atamian, M. T., & Sedinger, J. S. (2016). Nesting habitat selection influences nest and early offspring survival in Greater Sage-Grouse. *The Condor*, *118*(4), 689–702. https://doi.org/10.1650/CONDOR-16-62.1

Gibson, D., Blomberg, E. J., & Sedinger, J. S. (2016). Evaluating vegetation effects on animal demographics: The role of plant phenology and sampling bias. *Ecology and Evolution*, *6*(11), 3621–3631. Available from:https://doi.org/10.1002/ece3.2148

Götmark, F., Blomqvist, D., Johansson, O. C., & Bergkvist, J. (1995). Nest site selection: A trade-off between concealment and view of the surroundings? *Journal of Avian Biology*, *26*(4), 305–312. https://doi.org/10.2307/3677045

Gregg, M. A., & Crawford, J. A. (2009). Survival of greater sage-grouse chicks and broods in the Northern Great Basin. *The Journal of Wildlife Management*, *73*(6), 904–913. https://doi.org/10.2193/2007-410

Gregg, M. A., Crawford, J. A., Drut, M. S., & DeLong, A. K. (1994). Vegetational cover and predation of sage grouse nests in Oregon. *The Journal of Wildlife Management*, *58*(1), 162–166. https://doi.org/10.2307/3809563

Hagen, C. A., Connelly, J. W., & Schroeder, M. A. (2007). A meta-analysis of greater sage-grouse *Centrocercus urophasianus* nesting and brood-rearing habitats. *Wildlife Biology*, *13*, 42–50. https://doi.org/10.2981/0909-6396(2007)13[42:AMOGSC]2.0.CO;2

Hovick, T. J., Elmore, R. D., Allred, B. W., Fuhlendorf, S. D., & Dahlgren, D. K. (2014). Landscapes as a moderator of thermal extremes: A case study from an imperiled grouse. *Ecosphere*, *5*(3), 1–12.

Howlett, J. S., & Stutchbury, B. J. (1996). Nest concealment and predation in hooded warblers: Experimental removal of nest cover. *The Auk*, *113*(1), 1–9. https://doi.org/10.2307/4088930

Jahren, T., Storaas, T., Willebrand, T., Moa, P. F., & Hagen, B. R. (2016). Declining reproductive output in capercaillie and black grouse – 16 countries and 80 years. *Animal Biology*, *66*(3–4), 363–400. https://doi.org/10.1163/15707563-00002514

Kaczor, N. W., Herman-Brunson, K. M., & Jensen, K. C. (2011). Resource selection during brood-rearing by greater sage-grouse. In B. K. Sandercock, K. Martin, & G. Segelbacher (Eds.), *Ecology, conservation, and management of grouse, studies in avian biology* (pp. 169–177). Berkeley, CA, USA: University of California Press.

Knick, S. T., Dobkin, D. S., Rotenberry, J. T., Schroeder, M. A., Vander Haegen, W. M., & Van Riper, C. III (2003). Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats *The Condor*, *105*(4), 611–634. https://doi.org/10.1650/7329

Latif, Q. S., Heath, S. K., & Rotenberry, J. T. (2012). How avian nest site selection responds to predation risk: Testing an 'adaptive peak hypothesis'. *The Journal of Animal Ecology*, *81*(1), 127–138. https://doi.org/10.1111/j.1365-2656.2011.01895.x

Martin, T. E. (1992). Breeding productivity considerations: What are the appropriate habitat features for management? In J. M. Hagan, & D. W. Johnston (Eds.), *Ecology and conservation of neotropical migrant land birds*. Washington, DC, USA: Smithsonian Institute Press.

Martin, T. E. (1993). Nest predation and nest sites - new perspectives on old patterns. *BioScience*, *43*(8), 523–532. https://doi.org/10.2307/1311947

Martin, T. E., & Roper, J. J. (1988). Nest predation and nest-site selection of a western population of the hermit thrush. *The Condor*, *90*(1), 51–57. https://doi.org/10.2307/1368432

McConnell, M. D., Monroe, A. P., Burger, L. W., & Martin, J. A. (2017). Timing of nest vegetation measurement may obscure adaptive significance of nest-site characteristics: A simulation study. *Ecology and Evolution*, *7*(4), 1259–1270. Available from: http://dx.doi.org/10.1002/ece3.2767.

Miller, R. F., Naugle, D. E., Maestas, J. D., Hagen, C. A., & Hall, G. (2017). Special Issue: Targeted Woodland Removal to Recover At-Risk Grouse and Their Sagebrush-Steppe and Prairie Ecosystems. *Rangeland Ecology & Management*, *70*(1), 1–8. https://doi.org/10.1016/j.rama.2016.10.004

Mills, L. S. (2007). *Conservation of wildlife population: Demography, genetics and management*. Malden, MA, USA: Blackwell Publishing.

Moynahan, B. J., Lindberg, M. S., Rotella, J. J., & Thomas, J. W. (2007). Factors affecting nest survival of greater sage-grouse in northcentral montana. *The Journal of Wildlife Management*, *71*(6), 1773–1783. https://doi.org/10.2193/2005-386

Nakagawa, S., & Schielzeth, H. (2013). A general and simple method for obtaining R^2 from generalized linear mixed-effects models. *Methods in Ecology and Evolution*, *4*(2), 133–142. https://doi.org/10.1111/j.2041-210x.2012.00261.x

Naugle, D. E., Doherty, K. E., Walker, B. L., Holloran, M. J., & Copeland, H. E. (2011). Energy development and greater sage-grouse. In J. W. Connelly & S. T. Knick (Eds.), *Greater sage-grouse: Ecology and conservation of a landscape species and its habitats, studies in avian biology* (pp. 489–503). Berkeley, CA, USA: University of California Press

Peak, R. G. (2003). An experimental test of the concealment hypothesis using American goldfinch nests. *The Wilson Bulletin*, *115*(4), 403–408. https://doi.org/10.1676/03-024

Plummer, M. 2003. JAGS: A Program for Analysis of Bayesian Graphical Models Using Gibbs Sampling. mcmc-jags.sourceforge.net.

Plummer, M. 2016. rjags: Bayesian Graphical Models using MCMC. R package version 4-6. https://CRAN.R-project.org/package=rjags

R Core Team. 2016. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Remeš, V. (2005). Birds and rodents destroy different nests: A study of Blackcap *Sylvia atricapilla* using the removal of nest concealment. *Ibis*, *147*(1), 213–216.

Ricklefs, R. E. (1969). An analysis of nesting mortality in birds. *Smithsonian Contributions to Zoology*, *9*, 1–48. https://doi.org/10.5479/si.00810282.9

Schmidt, J. H., Walker, J. A., Lindberg, M. S., Johnson, D. S., & Stephens, S. E. (2010). A general Bayesian hierarchical model for estimating survival of nests and young. *The Auk*, *127*(2), 379–386. https://doi.org/10.1525/auk.2009.09015

Schreiber, L. A., Hansen, C. P., Rumble, M. A., Millspaugh, J. J., Gamo, R. S., Kehmeier, J. W., & Wojcik, N. (2015). Microhabitat selection of brood-rearing sites by greater sage-grouse in Carbon County, Wyoming. *Western North American Naturalist*, *75*(3), 348–363. https://doi.org/10.3398/064.075.0312

Smith, J. T., Evans, J. S., Martin, B. H., Baruch-Mordo, S., Kiesecker, J. M., & Naugle, D. E. (2016). Reducing cultivation risk for at-risk species: Predicting outcomes of conservation easements for sage-grouse. *Biological Conservation*, *201*, 10–19. https://doi.org/10.1016/j.biocon.2016.06.006

Stiver, S. J., Rinkes, E. T., Naugle, D. E., Makela, P. D., Nance, D. A., & Karl, J. W. (2015). *Sage-grouse habitat assessment framework: a multiscale assessment tool*. Bureau of Land Management. Retrieved from: https://www.blm.gov/style/medialib/blm/wo/blm_library/tech_refs.Par.34086.File.dat/TR_6710-01.pdf.

Sveum, C. M., Edge, W. D., & Crawford, J. A. (1998). Nesting habitat selection by sage grouse in south-central Washington. *Journal of Range Management*, *51*(3), 265–269. https://doi.org/10.2307/4003409

Taylor, R. L., Walker, B. L., Naugle, D. E., & Mills, L. S. (2012). Managing multiple vital rates to maximize greater sage-grouse population growth. *The Journal of Wildlife Management*, *76*(2), 336–347. https://doi.org/10.1002/jwmg.267

Walker, B. L. 2008. Greater sage-grouse response to coal-bed natural gas development and West Nile virus in the Powder River Basin, Montana and Wyoming, USA. Dissertation, The University of Montana.

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Tools and Technology

Mapping Sage-Grouse Fence-Collision Risk: Spatially Explicit Models for Targeting Conservation Implementation

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ABSTRACT Recent research suggested greater sage-grouse (*Centrocercus urophasianus;* hereafter, sage-grouse) fence collision may be widespread, and fence-marking methods have been developed for reducing prairie-grouse collision in sagebrush-steppe habitats. However, research also suggested sage-grouse collision was highly variable, and managers implementing mitigation desire targeting tools to prioritize mitigation efforts as a function of risk. We fit collision-risk models using widely available covariates to a sage-grouse fence-collision data set from Idaho, USA, and developed spatially explicit versions of the top model for all known sage-grouse breeding habitats (i.e., within 3 km of leks) in 10 of 11 western states where sage-grouse are found. Our models prioritize breeding habitats for mitigation as a function of terrain ruggedness and distance to nearest lek, and suggest that a relatively small proportion of the total landscape (6–14%) in each state would result in >1 collision over a lekking season. Managers can use resulting models to prioritize fence-marking by focusing efforts on high risk landscapes. Moreover, our models provide a spatially explicit tool to efficiently target conservation investments, and exemplify the way that researchers and managers can work together to turn scientific understanding into effective conservation solutions.

KEY WORDS avian collision, *Centrocercus urophasianus*, collision mitigation, fence collision, fence markers, infrastructure marking, sage-grouse.

Collision with elevated structures is a common phenomenon for many species of grouse (Catt et al. 1994, Baines and Summers 1997, Wolfe et al. 2007, Stevens et al. 2012*a*). Early research from Europe reported grouse among the most common infrastructure-collision victims, and suggested tetraonid collision susceptibility may be a function of morphology (e.g., heavy body wt, high wing loading; Baines and Summers 1997, Bevanger 1998, Bevanger and Brøseth 2000, Janss 2000). More recently, research in North America suggested prairie-grouse are susceptible to collision with fences (Patten et al. 2005, Wolfe et al. 2007, Stevens et al. 2012*a*). Fence collision was attributed to 39.8% of mortality for lesser prairie chickens (*Tympanuchus pallidicinctus*) in Oklahoma, USA (Wolfe et al. 2007), and uncorrected mean fence-collision rates of 0.38–0.41 strikes/km were reported for greater sage-grouse (*Centrocercus urophasianus;* hereafter, sage-grouse) during the breeding season in Idaho, USA (Stevens 2011). Fences and other anthropogenic structures are ubiquitous across western North America (Braun 1998, Knick et al. 2011); however, population-level impacts of prairie-grouse collision are poorly understood.

Infrastructure marking is a commonly suggested conservation strategy for reducing avian–infrastructure collision (Baines and Andrew 2003, Wolfe et al. 2009, Stevens et al. 2012*b*). Power-line markers appear to reduce collision for a variety of avian species (Morkill and Anderson 1991, Brown and Drewien 1995, Savereno et al. 1996, Barrientos et al. 2011), but assessments of fence-markers are less common. However, orange barrier netting reduced woodland grouse fence-collision in Scotland (Baines and Andrew 2003). Moreover, fence-marking methods have been developed for North American prairie grouse (Wolfe et al. 2009; Fig. 1), and evidence from Idaho suggested marking reduced the count of sage-grouse collisions by 83% during the breeding season (Stevens et al. 2012*b*).

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Figure 1. Male greater sage-grouse displaying on a lek directly beside a marked fence on an Idaho, USA, study site. Reflective fence markers were shown to reduce sage-grouse collision counts by approximately 83% in high-risk breeding habitats (Stevens et al. 2012*b*).

Managers are forced to make decisions with incomplete information and constrained budgets, and efficient allocation of resources promotes the greatest return on conservation investments (Bottrill et al. 2008). Targeting conservation to ensure that funds are allocated efficiently is often referred to as triage, a process that provides transparency and forces managers to consider opportunity costs of management actions (Bottrill et al. 2008). Sage-grouse collision appears highly variable within and between regions (Stevens et al. 2012*a*, *b*). Variation in collision risk suggests mitigation is unnecessary at many sites and prioritizing mitigation as a function of risk may enable cost-effective implementation of mitigation efforts (Stevens et al. 2012*a*, *b*). Thus, small but targeted investments could potentially alleviate much of the fence-collision risk in breeding habitats, freeing up resources for other conservation efforts.

The science behind conservation planning is often not conducted in partnership with managers, further complicating management decisions and resource allocation. Instead, researchers often conduct studies with little input from end users and hope the conservation community finds it useful (Knight et al. 2008). Steps to alleviate this research-implementation gap include sourcing research questions directly from managers, fostering relationships between researchers and managers, and linking research to implementation of conservation actions. Research showing that fence marking can reduce sage-grouse collisions (Stevens et al. 2012*b*) has spurred fence-marking efforts on public and private lands across 11 western states. However, sage-grouse occupy vast areas of western North America (Schroeder et al. 2004), and wildlife managers desire spatially explicit targeting tools to maximize their return on conservation investments. Therefore, the objective of this study was to bridge the research-implementation gap by developing spatially explicit fence-collision-risk models for sage-grouse in breeding areas across the western United States. Specifically, we developed models by re-analyzing landscape factors influencing collision risk from Stevens et al. (2012*a*), and applied resulting models to spatially predict and map fence-collision risk for all known sage-grouse breeding habitats in 10 of 11 western states.

STUDY AREA

We developed raster-regression models for areas within 3 km of all known and active sage-grouse leks ($n = 4{,}684$) in 10 of 11 states currently supporting sage-grouse. We used the most recently developed range-wide lek database for this analysis. The database was originally developed by Connelly et al. (2004), but has since been updated to reflect lek locations discovered and leks lost from 2004 to 2007 (Garton et al. 2011, Knick and Hanser 2011). Therefore, our analyses included all known and active sage-grouse leks as of 2007, although two states (ID and NV) provided lek location data updated through 2011.

METHODS

Stevens (2011) described a cluster sampling design used to survey fences in sage-grouse breeding areas of southern Idaho (2009: $n = 16$ sites; 2010: $n = 14$ sites), where 1×1-km sampling units were randomly selected and surveyed during the breeding season at each site (Mar–May; 2009: $n = 60$ clusters; 2010: $n = 80$ clusters). The number of sage-grouse collisions per square km was recorded for each sampled cluster, and clusters were sampled on >1 occasion when possible, resulting in 224 collision-count observations (Stevens 2011). Stevens et al. (2012*a*) modeled these collision counts as a function of covariates, including distance from each 1×1-km cluster's centroid to the nearest active lek, lek size (i.e., max. count) at the nearest lek, and a terrain ruggedness index (TRI; Riley et al. 1999). However, Stevens et al. (2012*a*) did not account for potential bias caused by removal of collision remains by scavengers, and only used a subset of collision-count observations representing the first sampling event at each site ($n = 123$). Therefore, we extended the analyses of Stevens et al. (2012*a*) and 1) used all 224 collision-count observations, 2) incorporated field-experiment data used to measure removal of collision evidence by scavengers, 3) used newly developed statistical models to combine collision-count data with removal-experiment data using joint-likelihood principles to estimate collision and removal process parameters, and 4) developed spatially explicit raster models to extrapolate estimated collision risk to all known sage-grouse breeding areas in 10 of 11 currently occupied states.

We modeled sage-grouse fence-collision counts from Idaho as a function of lek size, distance to lek, and TRI using a stochastic-process model for collision-count data developed by Stevens and Dennis (2013). Stevens et al. (2011) showed that removal of collision evidence prior to fence-collision sampling (i.e., evidence-removal bias) can be large, and removal of collision remains varied across regions of southern Idaho. The model used for our analyses predicts collision-count data with a generalized-regression approach that accounts for removal of collision evidence and accommodates covariates on collision- and removal-process parameters (Stevens and Dennis 2013). The model treats instantaneous collision counts as a stochastic-linear-immigration-death (SLID) process (Matis and Kiffe 2000), whereby Poisson arrivals represent addition of collisions to the system (immigration) and proportional deaths remove evidence from a site. The SLID model combines collision-count and removal-experiment data sets to estimate collision (θ) and removal (ψ) rate parameters using joint likelihood. Stevens and Dennis (2013) showed that regional variation in evidence removal can result in order-of-magnitude differences in expected collision counts between regions with identical collision rates. Thus, the removal rate (ψ) is, in effect, a nuisance parameter, and failing to account for evidence removal when modeling avian-collision counts results in parameter estimates that are difficult to interpret (Stevens and Dennis 2013).

We combined data from collision-count surveys (Stevens 2011) with carcass-removal-experiment data (Stevens et al. 2011) to estimate parameters of the SLID model. We fit 14 total models and compared models using Akaike's Information Criterion (hereafter, AIC; Akaike 1973). We fit models using the log link function and seven different covariate combinations, where collision (θ) was modeled as a function of distance to lek, lek size, and TRI, and removal (ψ) was modeled as a function of a binary variable indicating study region (i.e., region of ID where removal experiments were conducted; 1 = southeast Idaho, 0 = Magic Valley region). For the region-specific removal, fences west of Craters of the Moon National Monument were considered the Magic Valley, whereas fences east of this location were located in southeast Idaho. We fit each of the seven covariate combinations using the transient and stationary versions of the model, by numerically maximizing the transition (i.e., time dependent) and stationary (i.e., equilibrium and time-independent) distribution joint likelihoods (Stevens and Dennis 2013). We generated profile-likelihood confidence intervals for all model parameters and conducted goodness-of-fit testing for the most supported model (Stevens and Dennis 2013). We used leave-one-out cross-validation and root-mean-squared error to evaluate prediction success, calculating square root of the average squared error between predicted and observed collision counts for each model. We used the R statistical computing language for all model fitting and analyses (R Core Development Team 2006).

We developed spatially explicit models to predict collision as a function of covariates from the top SLID model. Because fence sampling in Idaho focused on areas within approximately 3 km of leks, we buffered all range-wide lek locations by 3 km in a Geographic Information System (GIS; ArcMap 10.0) and focused spatial analyses in these areas. We downloaded U.S. Geological Survey 30-m digital elevation models for each state (www.seamless.usgs.gov; accessed 7–9 Feb 2012), and calculated TRI for each 30-m pixel using ArcInfo. We calculated distance from each 30-m pixel to the nearest sage-grouse lek in GIS using the Euclidean distance function. Lastly, we used the raster calculator in GIS to extrapolate maximum-likelihood estimates of the total number of sage-grouse collisions over a lekking season for each 30-m pixel as a function of distance to lek and TRI, assuming a 78-day lekking season (15 March to 31 May; $\hat{y} = 78 \times \exp(\beta_0 + \beta_1 \times \mathrm{TRI} + \beta_2 \times \mathrm{distance})$). The SLID model explicitly accounts for evidence-removal bias in collision-count data, but does not account for detection error. Thus, our spatially-explicit models portray relative collision risk rather than absolute risk. Moreover, the predicted number of collisions for each 30-m pixel is entirely dependent on fence presence; obviously, not all pixels across the landscape have fences present. Lastly, we used an example collision-risk threshold of >1 collision/lekking season, and calculated the proportion of the 30-m pixels with a collision risk above this value for each state.

RESULTS

Modeling identified TRI and distance to lek effects on collision rates, and regional differences in removal of collision evidence (ΔAIC = 0; Table 1). The top model suggested collision decreased with increasing TRI ($\beta = -0.25$; 95% CI = -0.48 to -0.10; Fig. 2) and increasing distance from the nearest sage-grouse lek ($\beta = -0.0006$; 95% CI = -0.00115 to -0.00008; Fig. 2). Thus, an increase in topographic variation at a site and moving farther from a lek location strongly reduced the number of collisions predicted over a lekking season (Fig. 2), and sites predicted to be high risk were concentrated on flat areas in relatively close proximity to leks (Fig. 3). Goodness-of-fit testing failed to

Table 1. Model rankings for the stochastic linear-immigration-death model fit to the greater sage-grouse fence-collision data set from southern Idaho, USA. Covariates were size of nearest lek (lsize), distance to nearest lek (dist), terrain ruggedness index (TRI), and region (SE ID = 1, Magic Valley = 0; Stevens et al. 2011). Models were ranked and compared using Akaike's Information Criterion (AIC; Akaike 1973).

Model[a,b]	K[c]	ΔAIC	AIC
θ(TRI + distance) ψ(region)	5	0	403.505
θ(TRI + lsize + distance) ψ(region)	6	1.582	405.086
θ(TRI) ψ(region)	4	3.153	406.658
θ(TRI + lsize) ψ(region)	5	4.581	408.086
θ(distance) ψ(region)	4	12.210	415.715

[a] Model form is $\log(\theta) = \beta_0 + \beta_1 Y_1 + \ldots + \beta_k Y_k$ and $\log(\psi) = \gamma_0 + \gamma_1 Y_1 + \ldots + \gamma_k Y_k$, where θ = daily collision rate and ψ = per capita daily removal rate (Stevens and Dennis 2013).

[b] All top models were fit using the transient joint likelihood for collision-count observations after the first sampling occasion (Stevens and Dennis 2013). No models fit using the stationary joint likelihood for all count observations were supported by the data (ΔAIC > 19).

[c] K = no. of model parameters.

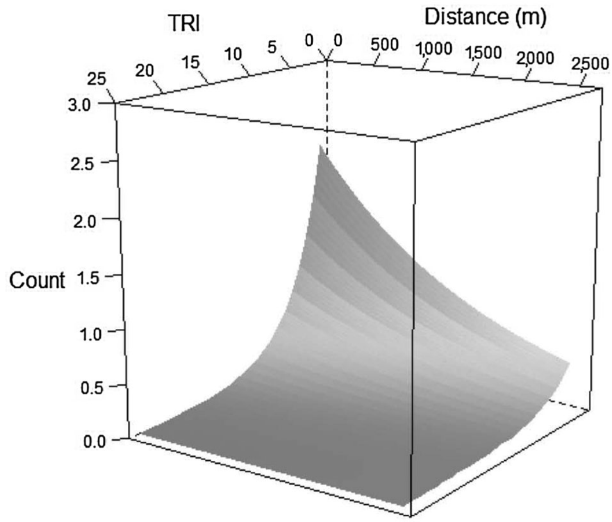

Figure 2. Maximum-likelihood estimates of total number of greater sage-grouse fence collisions over the 78-day lekking season from the top stochastic-linear-immigration-death model fit to data from southern Idaho, USA. Collision was a function of terrain ruggedness (TRI) and distance to the nearest lek. Maximum-likelihood estimates of total collisions from the top model = 78 × exp{$\beta_0 + \beta_1 \times$ TRI + $\beta_2 \times$ distance}.

reject the hypothesis that the top model fit the data ($P = 0.16$, $\chi^2_{249} = 271.22$), and cross-validated prediction error was similar among top three models (range = 0.634–0.648). The raster regression models demonstrated the large variability of predicted collisions per 30-m pixel across the landscape, and suggested that a relatively small proportion of the total landscape (6–14%) in each state would result in >1 collision over a lekking season (Fig. 3; Table 2). Despite spatial variation in collision risk, Idaho, South Dakota, California, Montana, and Oregon all had >10% of their area within 3 km of active leks with >1 predicted collision over a lekking season (Table 2). Montana (465,631 ha), Wyoming (295,770 ha), and Idaho (214,184 ha) had the greatest total area with >1 predicted collision over a lekking season (Table 2). In contrast, Utah (6.3%), North Dakota (7.3%), and Washington (7.5%) had the lowest percentage of pixels within 3 km of leks with >1 predicted collision over a lekking season due to increased terrain ruggedness near lek locations (Table 2).

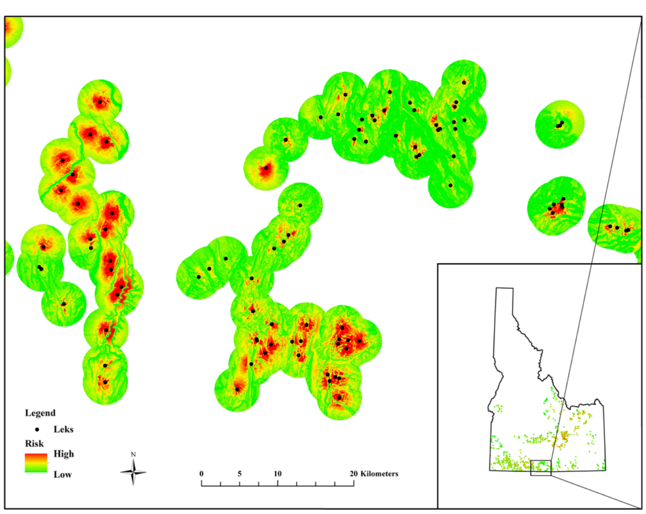

Figure 3. Example of spatially explicit fence-collision-risk maps from greater sage-grouse breeding habitats of southern Idaho, USA. Collision risk was a function of terrain ruggedness (TRI) and distance to the nearest lek. Maximum-likelihood estimates of total collisions (i.e., risk) from the top stochastic-linear-immigration-death model = 78 × exp{$\beta_0 + \beta_1 \times$ TRI + $\beta_2 \times$ distance}.

DISCUSSION

We created spatially explicit decision-support tools for wildlife and habitat managers who are marking fences to reduce sage-grouse collisions. Many previous avian-collision studies focused on known high-risk sites or used convenience-sampling methods to measure collision frequency, limiting generality of results and inferences. Moreover, rapid removal of collision remains can decrease accuracy of collision counts and bias estimates of collision totals (Smallwood 2007, Huso 2011, Stevens et al. 2011). We attempted to avoid pitfalls in study design by randomly sampling fences from sites spread across southern Idaho (n = 14–16 sites; Stevens et al. 2012*a*), measuring evidence removal with field experimentation (Stevens et al. 2011), and combining these data sets to model collision (θ) and removal (ψ) as a function of covariates using joint likelihood and generalized regression (Table 1). The models identified terrain ruggedness and distance from the lek metrics as drivers of fence-collision risk (Fig. 2; Stevens et al. 2012*a*). We hypothesize that collision risk is ultimately influenced by grouse flight behavior in flat terrain, where grouse fly low into leks before dawn and are thus vulnerable to colliding with fences. We found some evidence for the effect of lek size on collision (ΔAIC = 1.5; Table 1). However, our analyses suggested topography and distance were better predictors of collision than counts of displaying males on leks. This does not necessarily mean that local abundance does not influence collision risk, and measurement error in lek count indices may have attenuated the estimated effect on collision. Moreover, other covariates influencing sage-grouse collision were intentionally excluded from our analyses because they were not available at the range-wide extent (e.g., fence density; Stevens et al. 2012*a*). Regardless, terrain ruggedness attenuated other covariate effects and drove collision risk to nearly zero at moderate–high values (Fig. 2).

This study bridges the research-implementation gap by working in partnership with managers implementing mitigation measures to design user-friendly maps that suggest where targeted investments could alleviate much of the breeding season collision risk, freeing up resources for more pressing conservation concerns (Knight et al. 2008, Black and Groombridge 2010). Our models suggest that most of the breeding-area landscape across the West has low collision risk. As such, these models facilitate appropriate regional-scale resource allocation, by suggesting that targeted marking efforts may be beneficial to sage-grouse but that marking efforts are not necessary near all leks. We developed these maps at broad scales using covariate data that are widely available (e.g., terrain ruggedness); additional

Table 2. Summary statistics from spatially explicit fence-collision models in sage-grouse breeding habitats across the western United States. Statistics are: mean and standard deviation (SD) of predicted collision count per 30-m pixel, percent of the landscape (i.e., percent of total pixels) with >1 predicted collision over the lekking season (% >1 collision), and the number of hectares within 3 km of known leks (i.e., no. of pixels $\times$ 0.09 ha/pixel) with >1 predicted collision over the lekking season for each state. Both the percent of landscape and total area (ha) with >1 predicted collision over the lekking season are predicated on the presence of fence in each 30-m pixel.

State	$\bar{x}$	SD	% > 1 collision[a]	Area (ha) > 1 collision
ID	0.509	0.472	14.413	214,184
SD	0.563	0.413	13.107	6,933
CA	0.426	0.450	11.381	15,303
MT	0.477	0.415	11.157	465,631
OR	0.435	0.436	10.886	91,305
WY	0.422	0.403	9.239	295,770
NV	0.393	0.399	8.544	107,758
WA	0.397	0.375	7.531	4,715
ND	0.394	0.376	7.330	3,964
UT	0.319	0.369	6.264	28,380

[a] Max. of the predicted no. of collisions per 30-m pixel over a breeding season = 3.027 birds.

information at local scales (e.g., fence locations or densities, local space use) can be used to further inform management actions. Thus, our models can be used for local-scale planning by managers working in conjunction with local working groups and private landowners. Moreover, these models enable the linkage of management action to collision risk, which promotes effective resource use and minimizes the inefficient strategies of mitigating collision risk randomly or everywhere (Black and Groombridge 2010). Lastly, our example threshold of >1 collision/season was somewhat arbitrary, and maps with any desired risk threshold could be constructed in a GIS to delineate areas for fence marking or moving.

Our models provide a useful tool but they should also serve as testable hypotheses, and model validation is a valuable next step because spatial extrapolation and simplifying assumptions can lead to erroneous predictions (Miller et al. 2004). A model predicting blue crane (*Anthropoides paradiseus*) power-line collision in South Africa did not successfully predict high-risk sites (Shaw et al. 2010), but the model was based on expert opinion instead of a designed field study. Our model projected predictions at the 1 $\times$ 1-km scale onto 30-m pixels across sage-grouse breeding habitats, and with the exception of distance to lek, we assumed collision risk was independent of each pixel's position on the landscape, both of which could induce error in spatial extrapolation (Miller et al. 2004). Our models also extrapolated collision risk observed in Idaho to other western states, implicitly assuming the relationship observed between collision risk, terrain ruggedness, and lek location remains similar in other regions (Miller et al. 2004). However, prioritizing management actions using the best available science is better than proceeding with mitigation in an unorganized fashion (Miller et al. 2004). Moreover, our results are predicated on the presence of fences at each 30-m pixel. Thus, the true total area (i.e., no. of ha) of high collision risk in sage-grouse breeding areas will likely be considerably less than our models predicted because fences are not present at all sites. Lastly, our spatially-explicit models do account for removal error, but do not account for detection error and thus produce predictions of relative collision frequency over a breeding season. Predictions of relative collision frequency and cross-scale extrapolation of predictions complicate the assessment of demographic effects on grouse populations. Hierarchical statistical models for avian-collision data incorporating both detection and evidence-removal error are a necessary next step that should facilitate predictions of the absolute number of collisions over time as a function of covariates.

We caution readers against making direct inferences to population-level benefits resulting from reduced sage-grouse collision risk. We cannot say, for example, how many sage-grouse would be added to a population by reducing collisions because we lack demographic data to know whether populations can compensate for mortality via increased productivity. Population-level impacts of sage-grouse fence collision also likely depend on proportional mortality of male and female grouse, which is currently unknown (Stevens et al. 2012*a*). Moreover, the ability to compensate for collision mortality probably varies spatially, further complicating our ability to predict the number of birds added to a population as a result of fence-marking efforts. Future work addressing demographic consequences of sage-grouse collision and the conditions under which we would expect additive collision mortality should be a research priority.

MANAGEMENT IMPLICATIONS

These findings help guide implementation of the Natural Resources Conservation Service's Sage Grouse Initiative and provide decision support to others working in sage-grouse conservation. We attempted to bridge the research-implementation gap by applying our model to 4,684 known lek sites across 10 western states, and provided our GIS-based tool to Natural Resources Conservation Service practitioners and the state wildlife managers responsible for management of sage-grouse populations. Managers can use this tool to identify high-risk fences and to build new fences away from high-risk areas while still accomplishing grazing objectives. To facilitate use we also developed a how-to instructional guide and conducted multiple web-based training sessions. Lastly, we made our decision-support tool available to the Bureau of Land Management, the federal agency managing >50% of remaining sage-grouse habitats and currently revising their land-use plans for lands that include sage-grouse habitat. We encourage those interested in sage-grouse conservation to contact their state fish and wildlife agency to learn how to obtain a copy of the decision-support tool. Lastly, we remind managers that fence marking in other seasonal habitats, including areas of high sage-grouse concentration during winter, could potentially reduce fence strikes, but resulting benefits have not been measured.

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LITERATURE CITED

Akaike, H. 1973. Information theory and an extension of the maximum likelihood principle. Pages 267–281 *in* B. N. Petrov and F. Czáki, editors. Second international symposium on information theory. Akademiai Kiado, Budapest, Hungary.

Baines, D., and M. Andrew. 2003. Marking of deer fences to reduce frequency of collisions by woodland grouse. Biological Conservation 110:169–176.

Baines, D., and R. W. Summers. 1997. Assessment of bird collisions with deer fences in Scottish forests. Journal of Applied Ecology 34:941–948.

Barrientos, R., J. C. Alonso, C. Ponce, and C. Palacín. 2011. Meta-analysis of the effectiveness of marked wire in reducing avian collisions with power lines. Conservation Biology 25:893–903.

Bevanger, K. 1998. Biological and conservation aspects of bird mortality caused by electricity power lines: a review. Biological Conservation 86: 67–76.

Bevanger, K., and H. Brøseth. 2000. Reindeer *Rangifer tarandus* fences as a mortality factor for ptarmigan *Lagopus* spp. Wildlife Biology 6:121–127.

Black, S., and J. Groombridge. 2010. Use of a business excellence model to improve conservation programs. Conservation Biology 24:1448–1458.

Bottrill, M. C., L. N. Joseph, J. Carwardine, M. Bode, C. Cook, E. T. Game, H. Grantham, S. Kark, S. Linke, E. McDonald-Madden, R. L. Pressey, S. Walker, K. A. Wilson, and H. P. Possingham. 2008. Is conservation triage just smart decision making? Trends in Ecology and Evolution 23:649–654.

Braun, C. E. 1998. Sage grouse declines in western North America: what are the problems? Proceedings of the Western Association of State Fish and Wildlife Agencies 78:139–156.

Brown, W. M., and R. C. Drewien. 1995. Evaluation of two power line markers to reduce crane and waterfowl collision mortality. Wildlife Society Bulletin 23:217–227.

Catt, D. C., D. Dugan, R. E. Green, R. Moncrieff, R. Moss, N. Picozzi, R. W. Summers, and G. A. Tyler. 1994. Collisions against fences by woodland grouse in Scotland. Forestry 67:105–118.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of greater-sage grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies, unpublished report, Cheyenne, Wyoming, USA.

Garton, E. O., J. W. Connelly, J. S. Horne, C. A. Hagen, A. Moser, and M. A. Schroeder. 2011. Greater sage-grouse population dynamics and probability of persistence. Pages 293–381 *in* S. T. Knick, and J. W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology Series, Vol. 38. University of California Press, Berkeley, USA.

Huso, M. M. 2011. An estimator of wildlife fatality from observed carcasses. Environmetrics 22:318–329.

Janss, G. F. E. 2000. Avian mortality from power lines: a morphologic approach of a species-specific mortality. Biological Conservation 95:353–359.

Knick, S. T., and S. E. Hanser. 2011. Connecting pattern and process in greater sage-grouse populations and sagebrush landscapes. Pages 383–405 *in* S. T. Knick, and J. W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology, Vol. 38. University of California Press, Berkeley, USA.

Knick, S. T., S. E. Hanser, R. F. Miller, D. A. Pyke, M. J. Wisdom, S. P. Finn, E. T. Rinkes, and C. J. Henney. 2011. Ecological influence and pathways of land use in sagebrush. Pages 203–251 *in* S. T. Knick, and J. W. Connelly, editors. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology, Vol. 38. University of California Press, Berkeley, USA.

Knight, A. T., R. M. Cowling, M. Rouget, A. Balmford, A. T. Lombard, and B. M. Campbell. 2008. Knowing but not doing: selecting priority conservation areas and the research-implementation gap. Conservation Biology 22:610–617.

Matis, J. H., and T. R. Kiffe. 2000. Stochastic population models: a compartmental perspective. Springer lecture notes in statistics. Springer-Verlag, New York, New York, USA.

Miller, J. R., M. G. Turner, E. A. H. Smithwick, C. L. Dent, and E. H. Stanley. 2004. Spatial extrapolation: the science of predicting ecological patterns and processes. BioScience 54:310–320.

Morkill, A. E., and S. H. Anderson. 1991. Effectiveness of marking powerlines to reduce sandhill crane collisions. Wildlife Society Bulletin 19:442–449.

Patten, M. A., D. H. Wolfe, E. Shochat, and S. K. Sherrod. 2005. Habitat fragmentation, rapid evolution and population persistence. Evolutionary Ecology Research 7:235–249.

R Core Development Team. 2006. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Riley, S. J., S. D. DeGloria, and R. Elliot. 1999. A terrain ruggedness index that quantifies topographic heterogeneity. Intermountain Journal of Sciences 5:23–27.

Savereno, A. J., L. A. Savereno, R. Boettcher, and S. M. Haig. 1996. Avian behavior and mortality at power lines in coastal South Carolina. Wildlife Society Bulletin 24:636–648.

Schroeder, M. A., C. L. Aldridge, A. D. Apa, J. R. Bohne, C. E. Braun, S. D. Bunnell, J. W. Connelly, P. A. Deibert, S. C. Gardner, M. A. Hilliard, G. D. Kobriger, S. M. McAdam, C. W. McCarthy, J. J. McCarthy, D. L. Mitchell, E. V. Rickerson, and S. J. Stiver. 2004. Distribution of sage-grouse in North America. Condor 106:363–376.

Shaw, J. M., A. R. Jenkins, J. J. Smallie, and P. G. Ryan. 2010. Modelling power-line collision risk for the blue crane *Anthropoides paradiseus* in South Africa. Ibis 152:590–599.

Smallwood, K. S. 2007. Estimating wind turbine-caused bird mortality. Journal of Wildlife Management 71:2781–2791.

Stevens, B. S. 2011. Impacts of fences on greater sage-grouse in Idaho: collision, mitigation, and spatial ecology. Thesis, University of Idaho, Moscow, USA.

Stevens, B. S., J. W. Connelly, and K. P. Reese. 2012*a*. Multi-scale assessment of greater sage-grouse fence collision as a function of site and broad scale factors. Journal of Wildlife Management 76:1370–1380.

Stevens, B. S., and B. Dennis. 2013. Wildlife mortality from infrastructure collisions: statistical modeling of count data from carcass surveys. Ecology doi: 10.1890/12-1052.1

Stevens, B. S., K. P. Reese, and J. W. Connelly. 2011. Survival and detectability bias of avian fence collision surveys in sagebrush steppe. Journal of Wildlife Management 75:437–449.

Stevens, B. S., K. P. Reese, J. W. Connelly, and D. D. Musil. 2012*b*. Greater sage-grouse and fences: does marking reduce collisions? Wildlife Society Bulletin 36:297–303.

Wolfe, D. H., M. A. Patten, and S. K. Sherrod. 2009. Reducing grouse collision mortality by marking fences. Ecological Restoration 27: 141–143.

Wolfe, D. H., M. A. Patten, E. Shochat, C. L. Pruett, and S. K. Sherrod. 2007. Causes and patterns of mortality in lesser prairie-chickens *Tympanuchus pallidicinctus* and implications for management. Wildlife Biology 13(Supplement 1):95–104.

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Sage-Grouse Habitat Assessment Framework

A Multiscale Assessment Tool

Technical Reference 6710-1
June 2015

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Western Association of Fish & Wildlife Agencies

National System of Public Lands
U.S. Department of the Interior
Bureau of Land Management

Sage-Grouse Habitat Assessment Framework

A Multiscale Assessment Tool

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Foreword

The "Sage-Grouse Habitat Assessment Framework" (HAF) was conceived by several managers in the early 2000s. They assembled a diverse group of habitat specialists and sage-grouse experts from state, federal, and nongovernmental organizations to develop this habitat evaluation tool. In 2006, the "Greater Sage-grouse Comprehensive Conservation Strategy," published by the Western Association of Fish and Wildlife Agencies, highlighted the development and implementation of the HAF. That strategy outlined a number of objectives for the HAF, which included a temporal and spatial method for evaluating sagebrush habitats for sage-grouse suitability at various landscape scales. The HAF is a cornerstone of the habitat monitoring component of the sage-grouse conservation strategy.

Over the past several years, the BLM has developed a number of tools to help manage the public lands on a landscape basis. These tools include creating the capacity to synthesize large amounts of geospatial information to help the BLM and our partners develop a shared understanding of regional trends and identify conservation and development opportunities. The BLM is implementing this landscape approach in the Greater Sage-Grouse planning initiative, western solar plan, national cohesive wildland fire strategy, climate change strategy, regional mitigation, and other major initiatives. Incorporating the necessary adaptive management actions and understanding the success of these initiatives will require a coordinated approach to monitoring and assessments so information about multiple resources at multiple scales can be easily integrated. Thus, the HAF is timely as it fills the need for a multiple-scale, sage-grouse habitat assessment tool that can be easily integrated into the BLM landscape monitoring approach.

The HAF establishes indicators to determine the status of sage-grouse habitat needs at multiple scales and for seasonal habitats. The results of these assessments will provide the necessary information to evaluate whether the BLM-managed lands are meeting the sage-grouse land health habitat standard. Since the HAF assesses habitat needs at multiple scales, various datasets are needed for the analysis and assessment. To this end, the editors of the HAF coordinated with the BLM assessment, inventory, and monitoring (AIM) team to ensure the data required for the HAF indicator values are consistent with information currently being collected as described in "BLM Core Terrestrial Indicators and Methods," Westwide monitoring efforts, and grass-shrub stewardship efforts. This coordination between HAF and AIM efforts addresses one of the critical monitoring challenges in the BLM today—field capacity to complete the monitoring data collection.

To assess monitoring capacity and propose options to resolve this issue, the BLM initiated a review of its monitoring practices in 2006. The results of this survey, as discussed in "The Bureau of Land Management Assessment, Inventory, and Monitoring Strategy for Integrated Renewable Resources Management," indicated the need to coordinate and integrate monitoring activities and implement a data management strategy to eliminate redundant and duplicative data collection activities. The principles necessary to accomplish this integrated monitoring approach are described in BLM's "AIM-Monitoring: A Component of the BLM Assessment, Inventory, and Monitoring Strategy." When applying the principles of AIM monitoring to the HAF, field offices can minimize additional monitoring workloads. Applying these principles also creates opportunities to enhance national data layers and meet one of our primary goals of integrating monitoring activities: to collect data once and use it many times.

In summary, we commend the effort that has led to the development of the "Sage-Grouse Habitat Assessment Framework." The HAF will prove to be a valuable tool as the BLM and our partners implement the landscape approach for the management of our public lands. When the HAF is implemented using the principles outlined in "AIM-Monitoring: A Component of the BLM Assessment, Inventory, and Monitoring Strategy," the benefits to the BLM and our partners will be maximized and additional workloads will be minimized.

Ed Roberson
Assistant Director, Renewable Resources and Planning
Bureau of Land Management

Virgil Moore
Director
Western Association of Fish and Wildlife Agencies

Preface

This document provides policymakers, resource managers, and specialists with a comprehensive framework for assessing sage-grouse habitat in the sagebrush ecosystem. Four pillars form the foundation for the success of this approach: science, effective conservation policy, implementation, and adaptive management. Recent landscape evaluations indicate that conservation of sagebrush ecosystems has not been realized because large-scale mapping was not available to inform site-scale management actions. Advances in landscape ecology enable conservation planners to develop spatially explicit decision support tools that link populations with habitats for effective conservation planning, implementation, and evaluation at landscape scales. A shift from local to landscape conservation will empower decisionmakers to maximize the likelihood of achieving conservation by implementing site-scale actions within priority landscapes. Standardized methodologies provide consistency in terminology and techniques for site-scale assessments.

The habitat assessment framework (HAF) received progressive reviews during its development from 2000 to 2012. Those reviews focused and refocused the scope of the document, technical validity, and scientific rigor. The draft was edited for field use, and an outside peer review panel was contracted to evaluate the document. Appropriate comments, critiques, and suggestions were incorporated into the final document. In 2011, 2012, and 2013, the input matrix and outputs were field tested, and appropriate modifications were made in this current iteration of the HAF.

The HAF was developed for use by resource managers working closely with specialists in range management, landscape ecology, geographic information system (GIS), botany, wildlife biology, and other associated disciplines. To be fully functional, the HAF requires input from policy and operational staff. Some flexibility is incorporated into the suggested procedures, where appropriate, and professional judgment is required in its application, hence the need for experience. An increased capacity to deliver conservation will need to be addressed regionally because actions necessary to enhance populations vary widely across management zones. Quantity and quality of population and distribution data also vary widely for individual populations and across management zones; therefore, users of the HAF may be required to make certain assumptions concerning local populations. Shortcomings in existing datasets highlight the need to identify and subsequently collect additional datasets. Datasets that may aid in identifying important habitat areas and features include population and habitat information on seasonal use patterns, home ranges, migratory and dispersal movements, and fitness.

Acknowledgments

The habitat assessment framework is the culmination of many years of collaborative efforts. The editors participated in many of these efforts and would like to express our deep appreciation to the great number of individuals who helped develop this framework and who contributed to this document. This document would not have been possible without the forethought of Signe Sather-Blair, who led a group of Bureau of Land Management (BLM) wildlife biologists in Idaho (Paul Makela, Tim Carrigan, and Loren Anderson) in drafting the initial site-scale analysis methodology of this habitat assessment in 2000. A year later, the BLM revised the site-scale methodology to be applicable for habitats in Wyoming. George Soehn led this effort, with input from an interdisciplinary team that included Vicki Herren, Pat Karbs, Bruce Parker, Mary Read, Tom Rinkes, Dave Roberts, Dennis Saville, Andy Warren, Chet Wheeless, Don Whyde, and Jim Wolf.

In 2004, following the release of the "Conservation Assessment of Greater Sage-grouse and Sagebrush Habitats," the need to assess habitats at multiple scales became apparent because sage-grouse use large landscapes and occur over a large geographical range in western North America. In January 2005, the directors of the Western Association of Fish and Wildlife Agencies (WAFWA) passed a resolution to coordinate with the BLM on the development of the sage-grouse habitat assessment framework. Again, Signe Sather-Blair brought together a team composed of BLM and Idaho Department of Fish and Game biologists (Michelle Commons-Kemner, Tom Rinkes, and Alexis Carroll) to address sage-grouse at multiple habitat scales. An informal working group, with members from various disciplines, federal and state agencies, and universities, was convened to assist in addressing these issues, and their comments were valuable in the development of the final product. This working group was composed of the following individuals:

- Cameron Aldridge, Colorado State University
- Anthony Apa, Colorado Division of Wildlife
- Alexis Carroll, BLM
- Michelle Commons-Kemner, Idaho Department of Fish and Game
- Jack Connelly, Idaho Department of Fish and Game
- Mike Gregg, U.S. Fish and Wildlife Service
- Kristine Hasselblad, BLM
- Ann Hild, University of Wyoming
- Mark Hilliard, BLM
- Mike "Sherm" Karl, BLM
- Steve Knick, U.S. Geological Survey
- Don Major, BLM
- Dick Mayberry, BLM
- Clint McCarthy, U.S. Forest Service
- Richard Miller, Oregon State University
- Dave Naugle, University of Montana
- Mike Pellant, BLM
- Tom Rinkes, BLM
- Roger Rosentreter, BLM
- Paul Roush, BLM
- Signe Sather-Blair, BLM
- Pat Shaver, Natural Resources Conservation Service
- Jim Sedinger, University of Nevada, Reno

Others who provided valuable insight and comments during the development of this document include Brendan Moynahan, Mary Rowland, Cal McCluskey, Danielle Flynn, Barry Rose, Alan Sands, Jason Sutter, Susan Filkins, Paul Makela, Sarah McCord, and WAFWA's Western States Sage and Columbian Sharp-tailed Grouse Technical Committee (Tom Hemker, Christian Hagen, Joe Bohne, Shawn Espinosa, and Rick Northrup). Roger Rosentreter and Alan Sands significantly updated the list of forb species used by sage-grouse. Michael Schroeder, Fee Busby, Greg Neudecker, and David Pyke provided suggestions for improving the document in its final stages. Additional reviews and comments were incorporated by the BLM's Washington Office Resources and Planning staff and the BLM's National Operations Center staff.

Contents

Figures

Appendix B Figures

Tables

Appendix B Table

Chapter I: Conceptual Overview

Introduction

Sage-grouse provide resource managers with a unique impetus for conservation of the sagebrush ecosystem and species that depend upon that ecosystem. Sage-grouse select habitat at multiple scales and are sensitive to landscape change, making them an appropriate focal species, as defined by Mills (2013), for managing the sagebrush ecosystem (Wisdom et al. 2005; Rowland et al. 2006b; Hanser and Knick 2011). In 2004, scientists and managers remapped the current range of sage-grouse to evaluate change in presettlement distribution (figure 1; modified from Schroeder et al. 2004). The distribution of sage-grouse has declined by nearly half since presettlement, but they still occupy 668,400 km^2 of the sagebrush steppe in 11 western states and 2 Canadian provinces.

Loss and degradation of habitat from anthropogenic developments, fire, sodbusting, and invasive species are primary threats leading to isolation, reduction, and extirpation of populations (Connelly et al. 2000; Knick et al. 2013). These factors, combined with new constraints such as West Nile virus (Walker and Naugle 2011), climate change (Nielson et al. 2005) and genetic isolation (Knick and Hanser 2011; Oyler-McCance and Quinn 2011), require an integrated approach to landscape conservation to assess and effectively conserve sage-grouse populations and their habitats.

Conservation concerns will continue to exist until managers demonstrate the effectiveness of actions that maintain and restore habitats at scales that match the species' biological needs. Sage-grouse conservation can be daunting because the sagebrush sea is vast, threats to habitats are numerous and varied, and resources are limited. Maximizing return on conservation investment by targeting policy and implementation to the most biologically important places (Bottrill et al. 2008) for this conservation-reliant species (Scott et al. 2010) is a proactive yet fundamental shift occurring in management philosophy.

Policymakers and practitioners alike are now using broad-scale planning tools to help guide limited resources to the most biologically important places. In 2010, the BLM published a report that included a breeding bird density map (Doherty et al. 2010), providing the foundation for the delineation of core areas rangewide. Core areas are locations of high bird abundance containing a majority of sage-grouse. Figure 2 depicts the clumped distribution of males on leks within core areas that contain 25 percent, 50 percent, 75 percent, and 100 percent of the known breeding population. Approximately 75 percent of sage-grouse live within 25 percent of the occupied range.

Through time, 11 member states of the Western Association of Fish and Wildlife Agencies (WAFWA) improved the core area concept by delineating their boundaries to include all seasonal habitats instead of just breeding habitat. Many western states have incorporated newly approved core areas in their own state-based sage-grouse plans. In 2013, the U.S. Fish and Wildlife Service partnered with states to form the Conservation Objectives Team (COT). The team combined all the core areas across the range of the species into one new map (figure 3). This new map refers to core areas as priority areas for conservation (PACs) and the team's report identifies PAC-specific threats to be addressed through conservation (U.S. Fish and Wildlife Service 2013).

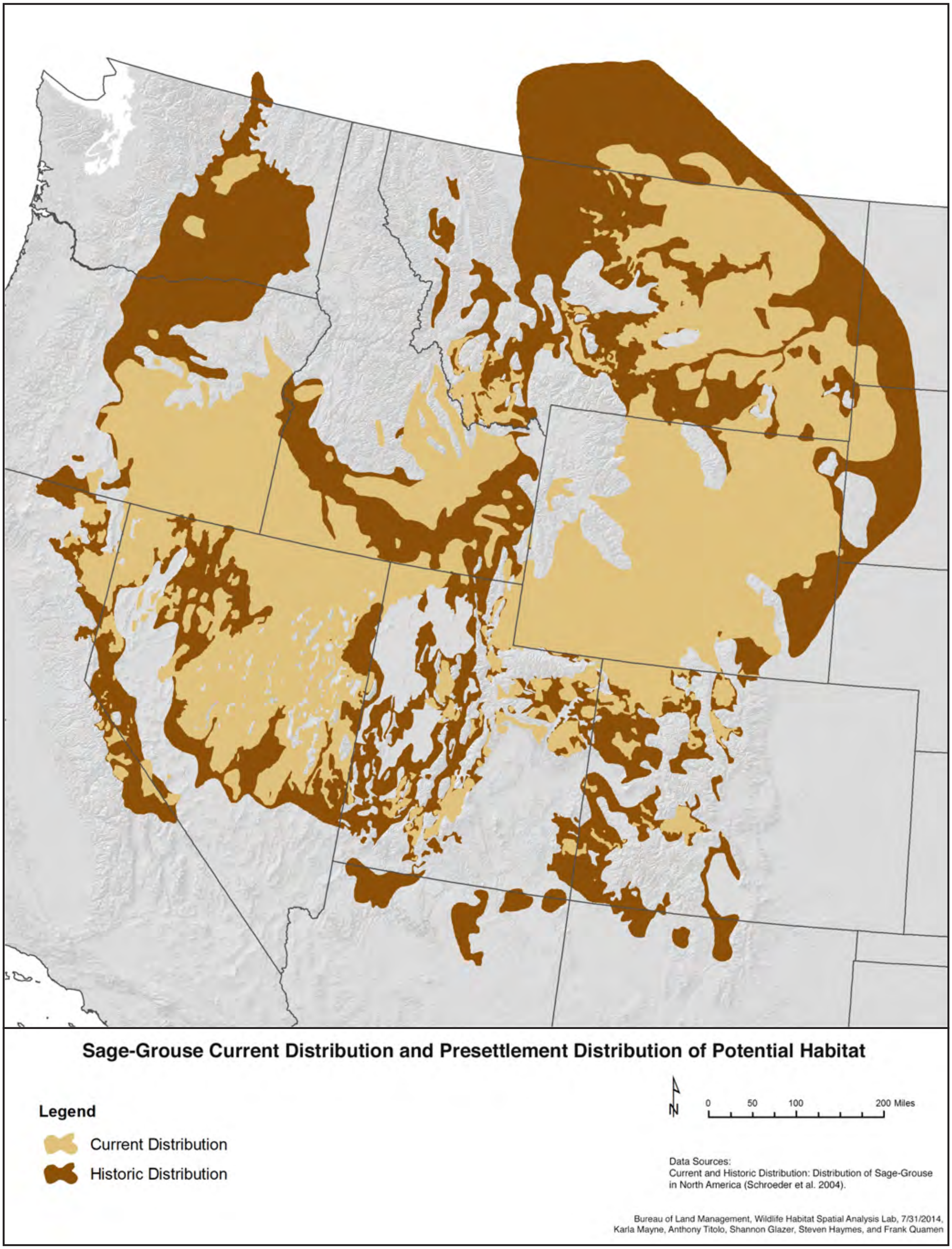


Figure 1. Current distribution and presettlement distribution of potential habitat of Greater and Gunnison Sage-Grouse in North America (as modified from Schroeder et al. 2004).

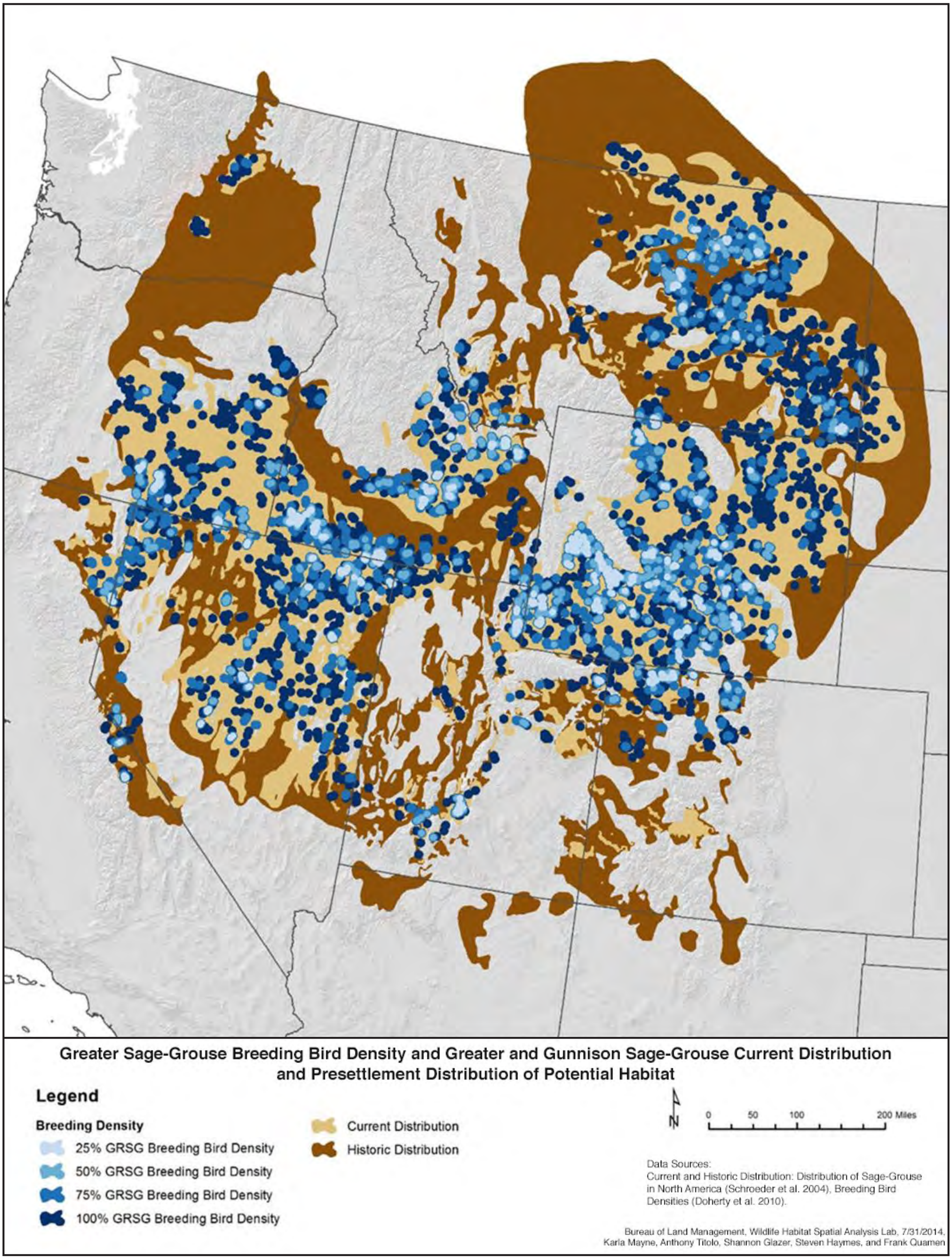


Figure 2. Greater Sage-Grouse (GRSG) population centers or "core areas" across the species' range. The lightest blue areas contain 25 percent of the breeding population, and each darker shade of blue indicates an additional 25 percent. Gunnison Sage-Grouse breeding bird density is not displayed.

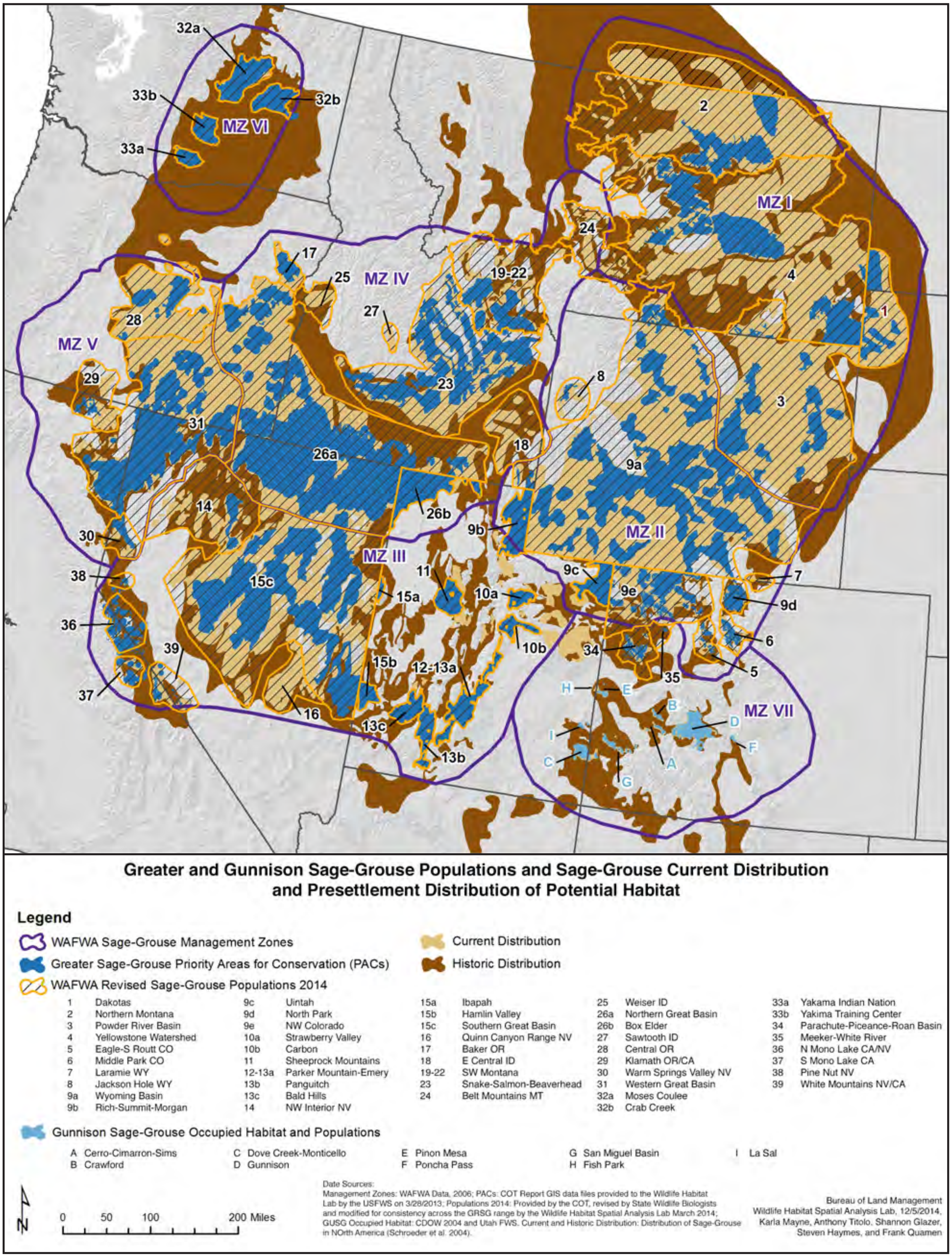


Figure 3. Priority areas for conservation (PACs) as identified by the Conservation Objectives Team (COT) and Gunnison Sage-Grouse occupied habitats. PACs are encircled by seven sage-grouse management zones established by the Western Association of Fish and Wildlife Agencies (WAFWA) based on populations within floristic provinces (Stiver et al. 2006).

A Landscape Vision for Implementing the Habitat Assessment Framework

Incorporating Scale into Sage-Grouse Policy and Implementation

The vision for this habitat assessment framework (HAF) is to empower managers to implement project-level actions that make sense at landscape scales. To achieve this vision, the HAF addresses two primary subjects: (1) applying the hierarchy for implementing landscape conservation, and (2) providing the inventory and outcome-based evaluation tools necessary for assessing effectiveness of resulting conservation actions. Sage-grouse habitats transcend jurisdictional boundaries and therefore require a coordinated approach to management. The HAF provides a blueprint for landscape conservation; success will be achieved through implementation with local stakeholder involvement.

The HAF's hierarchical approach begins with a policy vision for management of the sagebrush ecosystem (figure 4). Such policy changes are underway at federal and state levels in collaboration with major land users and the public. Emerging policies vary by agency and state, but all aim to reduce threats to sage-grouse by reducing disturbance and implementing beneficial actions primarily inside PACs. New policy direction and resources at the broad scale facilitate conservation and empower state and regional managers.

At the second level in the hierarchy, state and regional managers design the future landscape through mid-scale policy direction aimed at reducing specific threats facing sage-grouse in their jurisdiction. Threats vary geographically, but generally, policy will include actions to protect, manage, and restore seasonal habitats and to maintain connectivity of pathways that facilitate movement within and among populations. State and regional decisionmakers fulfill their place in the hierarchy by providing

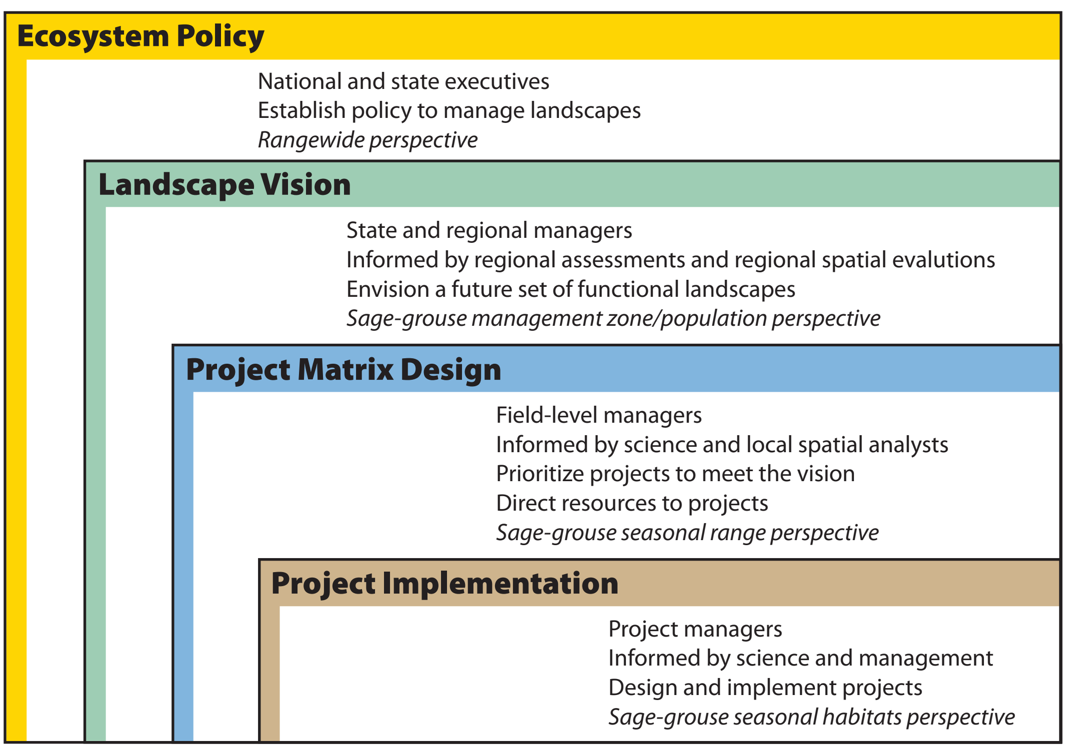


Figure 4. A hierarchical approach for implementing the habitat assessment framework.

their field managers with direction and resources to meet the landscape vision.

At the third level in the hierarchy, field managers design a matrix of fine-scale conservation actions that satisfy state and regional directives. Field managers develop specific actions and prioritize them by importance, timing for implementation, and cost. Field managers fulfill their place in the hierarchy by providing project managers with project implementation priorities.

At the fourth and equally important level in the hierarchy, project managers implement the specified site-scale conservation actions. Implementing the right actions in the right places at biologically relevant scales is the key to conserving and restoring the sagebrush ecosystem. Successfully implementing the HAF will initiate and foster a new era in landscape conservation of the sagebrush ecosystem.

Integrating Science into Habitat Assessment Framework Implementation

Inventory and monitoring are integral components of the HAF. Inventory provides baseline data and may provide projections of future condition. Together, these inputs provide for science-based evaluations to measure the biological response of sage-grouse populations to conservation actions, assess effectiveness, and adaptively improve delivery. The level of monitoring reflects the scales at which sage-grouse populations use habitat resources year-round and transcends that of an individual project to encompass the larger landscape. Rather than focusing on acres treated, the approach is biologically based and uses sage-grouse habitat and population responses at multiple scales to evaluate conservation benefits.

Outcome-based evaluations are vital to quantifying the success of past actions, informing future actions, and garnering additional social and financial support for conservation (e.g., Baruch-Mordo et al. 2013; Copeland et al. 2013). Such evaluations are a primary tool for applying effective adaptive management strategies in conservation and fulfilling the commitments in the "Greater Sage-Grouse Comprehensive Conservation Strategy" (Stiver et al. 2006) and the "Gunnison Sage-Grouse Rangewide Conservation Plan" (Gunnison Sage-Grouse Rangewide Steering Committee 2005). Shortcomings in existing datasets highlight the need to identify and subsequently collect additional information, including population and habitat information. For example, the HAF will be instrumental in assessing the effectiveness of a new management approach being implemented by the BLM Fire and Invasives Assessment Team (FIAT). The new management approach uses existing data to map soil temperature and moisture regimes along with the amount of sagebrush cover across landscapes to predict a sagebrush ecosystem's resilience to disturbance and resistance to invasive species (Chambers et al. 2013; Sage Grouse Initiative 2014). This tool helps prioritize and pinpoint management tactics across sagebrush landscapes, from fire and fuels management to restoration, and partners have already quickly engaged in implementation of this new strategic approach.

Biological Underpinnings of the Habitat Assessment Framework: Habitat Selection Processes

Landscape conservation is a scale-dependent process whereby priority landscapes are identified across the species range (broad scale) and appropriate conservation actions are implemented within seasonal habitats to benefit populations (site scale). The HAF has adopted the hierarchical orders of habitat selection as described by Johnson (1980). Johnson's orders of selection are widely accepted and provide the foundation for the HAF to discuss scale in common and consistent terms. Johnson (1980) described four orders of habitat selection in which each higher order is dependent on the previous order (figure 5). For example, a food item is nested within a feeding site, which is nested within a seasonal use area, which is nested within a home range, which is

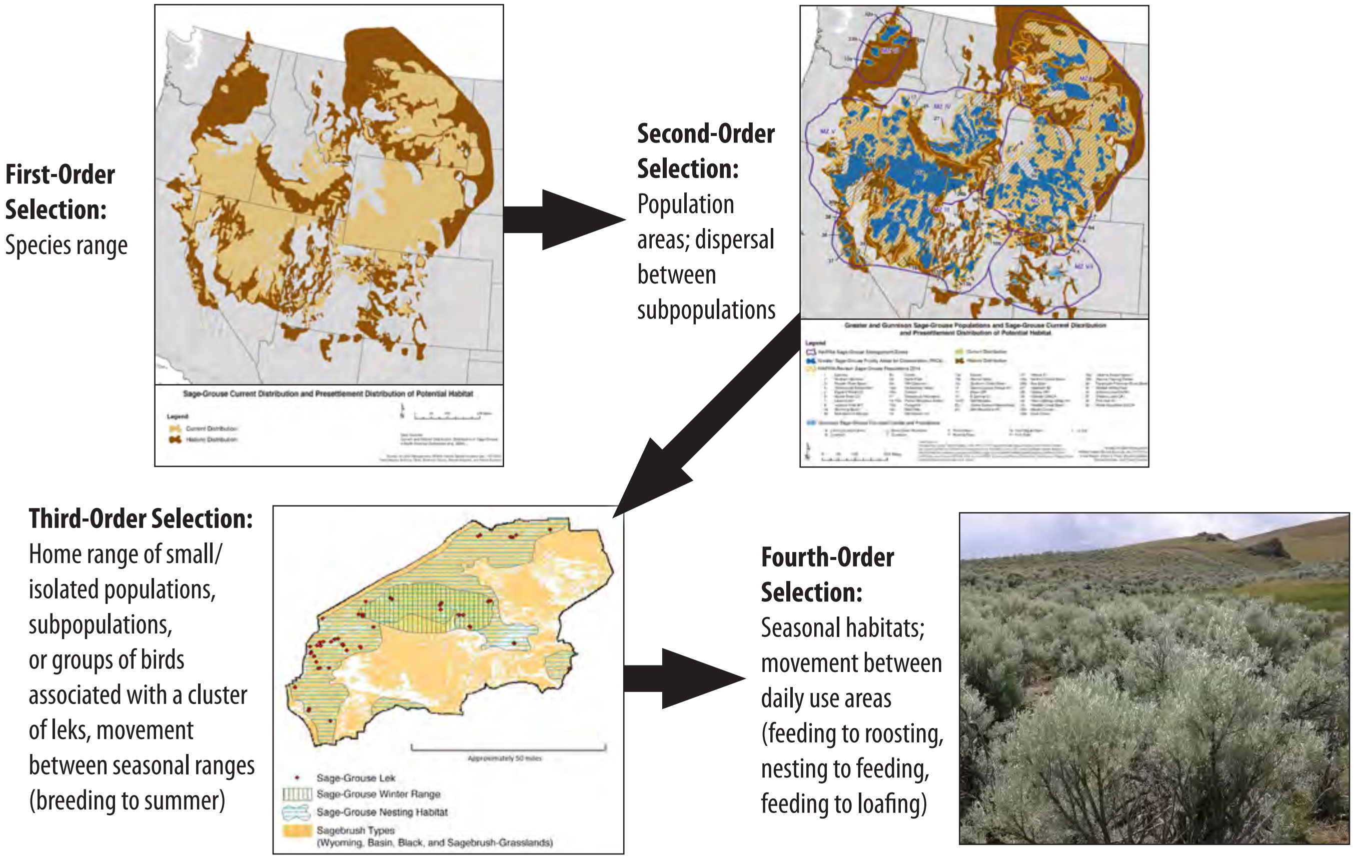


Figure 5. Habitat selection by sage-grouse based on Johnson's (1980) four orders.

nested within a population area, which is part of the species range. Sage-grouse select nesting and feeding areas within their seasonal range and that seasonal range is nested within their home range. An ecological or anthropogenic disturbance that changes their home range can affect nesting or feeding site selection.

First-order selection is described as "the selection of physical or geographical range of a species" (Johnson 1980). By definition, there is only one first-order habitat, the range of the species. For sage-grouse, the range is defined by populations of sage-grouse associated with sagebrush landscapes (Connelly et al. 2003). Populations or subpopulations within those populations are the second-order selection. The second-order selection habitats may include as many as 39 discrete populations (U.S. Fish and Wildlife Service 2013). Third-order selection is the home range of an individual bird. Location and size of a home range is determined in part by the quality and juxtaposition of resources within and between seasonal habitats. Fourth-order selection is the use of a particular nesting, feeding, or roosting site within one particular seasonal habitat. Spatial and temporal scales are evident throughout the selection process, becoming finer as orders of selection increase.

Orders of habitat selection provide a unifying framework in which to evaluate populations and their habitats. At the second order, state and regional planners and decisionmakers have the flexibility to design a future landscape and the location and types of actions necessary to achieve desired conditions. The resource manager has significant flexibility in evaluating third- and fourth-order habitat selection. The manager must provide an accurate estimate of populations, subpopulations, seasonal-use habitats, and ecological site potentials to effectively coordinate and design appropriate conservation actions.

Chapter II: Sage-Grouse Habitat and Data Descriptions

Habitat Suitability and Indicators

Sage-grouse habitat suitability is described at different spatial scales to address the ecological processes and population dynamics that occur at each scale. Although life requisites of space, food, water, and shelter are not easily segmented into spatial scales, they must be addressed for description and conservation planning purposes. The life requisite of space is significant at all scales though in different contexts. Pathways for movement within and between populations are critical for maintaining population viability. Having access to well-connected sagebrush patches that provide dispersal and movement among subpopulations is essential for sage-grouse population viability and persistence over the long term. However, a variety of natural or anthropogenic disturbances may interrupt or retard dispersal. Similarly, at the fine scale, habitat availability, security, and connectivity within home ranges are important for securing seasonal movements to shelter and food needs. Shelter and food availability at the site-scale within the seasonal ranges directly affects individual fitness, survival, and reproductive potential. Thus, the suitability of habitat at each scale has significant conservation implications on population health.

Biologists use measurable habitat characteristics, procedural steps, and habitat models to standardize techniques for preparing habitat descriptions that reflect life requisite needs (United States Department of the Interior 1980; Cooperrider et al. 1986; Gilbert and Dodds 1987; Morrison et al. 1998). Habitat indicators are often used to characterize the environment in terms of suitability for shelter, food, water, and space. The indicators must be sensitive to the ecological processes operating at the scale of interest. They are based on scientific research findings and should be quantitatively repeatable for data summarization and to avoid bias. A single habitat indicator does not necessarily define habitat suitability for an area or particular scale. Once measured or described, indicators must be collectively reviewed and put into context to correctly determine habitat suitability. In many cases, more than one scale with multiple indicators will be of interest. This chapter describes the important habitat indicators for each scale (table 1) and considerations for integrating information for within- and between-scale habitat descriptions. Habitat indicators for the mid and fine scales are generally evaluated based on trends of each of the scale indicators. Habitat indicators for the site scale are generally compared from the range, mean, proximity, shape, and stability of the various seasonal habitat components.

Table 1. Summary of habitat suitability indicators and descriptions for the mid, fine, and site scales. Suitability descriptions appropriate for each scale are based on the habitat indicator measurements for that scale.

Mid-Scale (Second-Order) Descriptions – Isolated/small population, subpopulation, or home range of group of leks	
Habitat Indicators	1. Habitat Availability 2. Patch Size and Number 3. Patch Connectivity 4. Linkage Area Characteristics 5. Landscape Matrix and Edge Effect 6. Anthropogenic Disturbances
General Suitability Descriptions	**Suitable:** Landscapes have connected mosaics of sagebrush shrublands that allow for bird dispersal and migration movements within the population or subpopulation area. Anthropogenic disturbances that can disrupt dispersal or cause mortality are generally not widespread or are absent. **Marginal:** Landscapes have patchy, fragmented sagebrush shrublands that are not well connected for dispersal and migration in portions of the population or subpopulation area. Anthropogenic disturbances that disrupt dispersal or cause mortality are present throughout all or portions of the landscape. Some lek groups or subpopulations are isolated or nearly isolated. **Unsuitable:** Landscapes were former shrubland habitat now converted to predominantly grassland or woodland cover or other unsuitable land cover or use. Remaining sagebrush patches are predominantly unoccupied or have few remaining birds. Portions of the population or subpopulation area may become occupied in the foreseeable future through succession or restoration.
Fine-Scale (Third-Order) Descriptions – Seasonal habitats within home ranges	
Habitat Indicators	1. Seasonal Habitat Availability 2. Seasonal Use Area Connectivity 3. Anthropogenic Disturbances
General Suitability Descriptions	**Suitable:** Home ranges have connected seasonal use areas. Anthropogenic features that can disrupt seasonal movements or cause mortality are generally absent or at least not widespread. **Marginal:** Home ranges have poorly connected or disjunct seasonal use areas. Anthropogenic features that can disrupt seasonal movements or cause mortality may occur within the home range. **Unsuitable:** Home ranges have seasonal use areas with predominantly grasslands, woodlands, or incompatible land uses (anthropogenic features) not conducive to sage-grouse seasonal movements or habitat use. Most leks have been abandoned or have few remaining birds.
Site-Scale (Fourth-Order) Descriptions – Use areas within seasonal habitats	
Habitat Indicators	1. Sagebrush Cover (all seasons) 2. Sagebrush Height (all seasons) 3. Predominant Sagebrush Shape (breeding only) 4. Perennial Grass and Forb Heights (breeding) 5. Perennial Grass Cover (breeding and summer/late brood-rearing) 6. Perennial Forb Cover (breeding and summer/late brood-rearing) 7. Preferred Forb Availability (breeding and summer/late brood-rearing) 8. Riparian Stability (summer/late brood-rearing) 9. Availability of Sagebrush Cover (leks and summer/late brood rearing – riparian/wet meadow) 10. Proximity of Detrimental Land Uses (leks) 11. Proximity of Trees or Other Tall Structures (leks)
General Suitability Descriptions	**Suitable:** Seasonal habitat has a preponderance of sagebrush cover types with sufficient shrub and herbaceous cover to protect sage-grouse from predators and weather and successfully raise young. Food resources are present or in close proximity to cover. **Marginal:** Seasonal habitat has a preponderance of sagebrush cover types with sparse shrub and/or herbaceous cover that does not provide the shelter needs for protection from predators and weather. Food resources are present but are either not at levels expected for ecological site potential or not in close proximity. **Unsuitable:** Seasonal habitat has a preponderance of land cover types that do not provide sufficient cover or food resources to meet the life requisite needs though there is potential to meet them in the future.

Broad Scale (First Order)

The broad-scale (first-order) habitat selection is the rangewide potential presettlement habitat of both species of sage-grouse (Schroeder et al. 2004) (figure 1). Connelly et al. (2004) provided figures that demonstrate the extent of the first order. Habitat suitability was demonstrated by evaluating sage-grouse numbers at leks distributed across the landscape (figure 2). This figure and its underlying dataset provide decisionmakers and conservation planners with a baseline from which they may begin the broad process of "visioning" the configuration of the landscape.

Connelly et al. (2004) discussed first-order sage-grouse habitat suitability in terms of characteristics such as availability of large expanses of sagebrush or grass/sagebrush habitat, presence of migration corridors, and juxtaposition of other habitats and land uses within these large expanses.

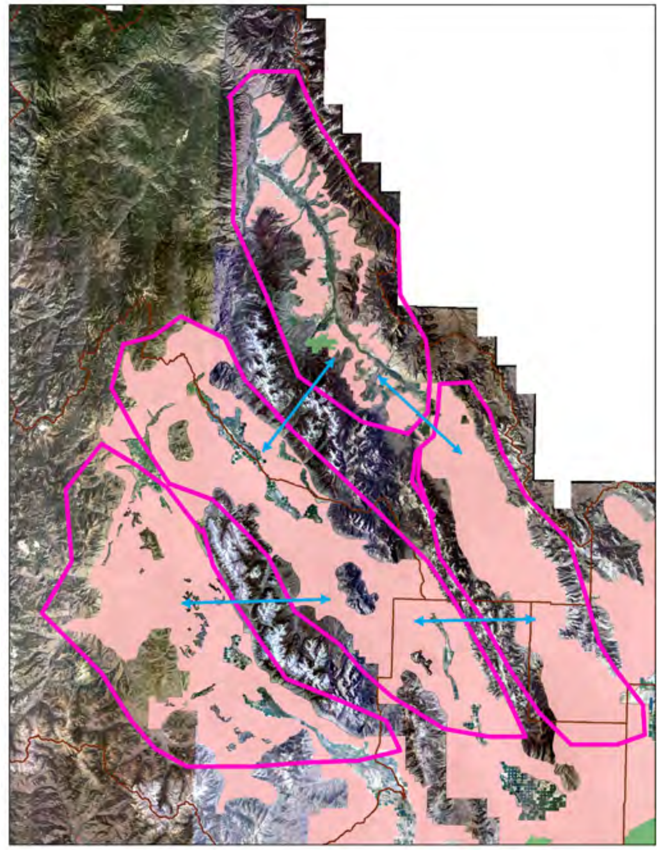


Figure 6. Mid-scale (second-order) habitat selection. The map demonstrates a series of interconnected subpopulations in mountain valleys.

Mid Scale (Second Order)

Second-order habitat descriptions are linked to bird dispersal capabilities in population and subpopulation areas (figure 6). These population areas have been geographically described in a general manner for the Greater Sage-Grouse (Connelly et al. 2004; figure 7) and Gunnison Sage-Grouse (Gunnison Sage-Grouse Rangewide Steering Committee 2005; figure 1). A detailed description of the distribution of Greater Sage-Grouse populations and subpopulations is provided by Connelly et al. (2004). Second-order descriptions are generally appropriate for subpopulations. However, some isolated populations may warrant second- or third-order habitat descriptions.

The mix of sagebrush or grassland/sagebrush patches on the landscape at the second order also provides the life requisite of space for sage-grouse dispersal needs. The configuration of sagebrush or grassland/sagebrush habitat patches and the land cover or land use between the habitat patches within a subpopulation defines suitability. Landscape suitability at the mid scale for populations and subpopulations can generally be described by the following scenarios:

- Suitable habitats within landscapes have connected mosaics of sagebrush shrublands that allow for bird dispersal and migration movements within the population and subpopulation area. Anthropogenic disturbances that can disrupt dispersal or cause mortality are generally not widespread or are absent.

- Marginal habitats within landscapes have patchy, fragmented, sagebrush shrublands or grasslands/sagebrush areas that are not well connected for dispersal and migration in portions of the population or subpopulation area. Marginal habitats could also include shrubland areas experiencing encroachment by junipers or other tree species. Anthropogenic

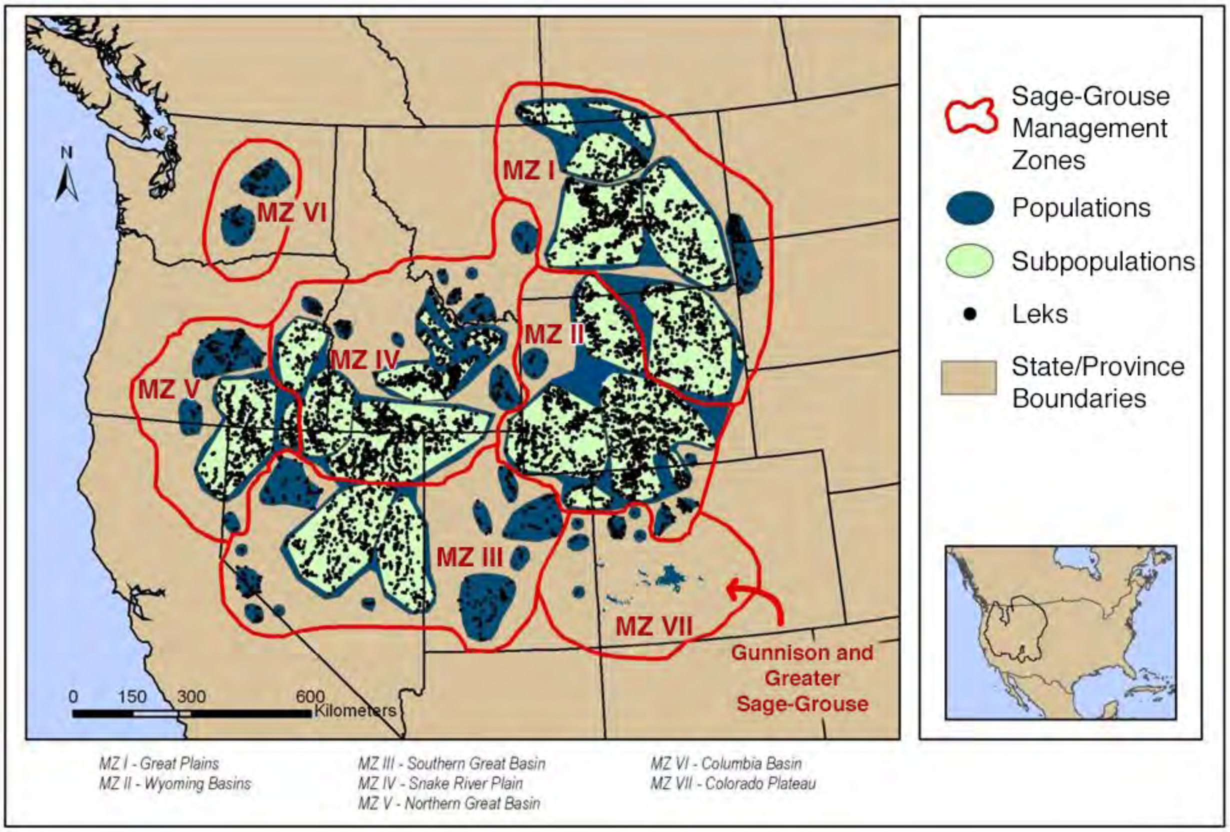


Figure 7. Sage-grouse management zones and populations (Stiver et al. 2006).

disturbances that disrupt dispersal or cause mortality may be common throughout all or portions of the landscape. Some lek groups or subpopulations are isolated or nearly isolated.

- Unsuitable habitats often include large areas of former shrublands that have been largely converted to annual grasslands or shrublands or other land uses. Remaining habitat patches are predominantly or nearly unoccupied by sage-grouse. The area may or may not have some potential to become occupied in the foreseeable future through succession or restoration.

At the second order, sage-grouse occupancy and dispersal are dependent on the extent and pattern of sagebrush shrublands within a landscape matrix of nonhabitat and unsuitable habitat. Other habitats such as grasslands, wet meadows, and riparian areas provide important habitat for sage-grouse but only when they are in close proximity to sagebrush habitat (Connelly et al. 2004). The importance of these habitats is more appropriately addressed with seasonal habitat needs at the site scale.

Six second-order habitat indicators influence habitat use, dispersal, and movement across population and subpopulation areas (table 2):

1. Availability of sagebrush habitat.
2. Size and number of habitat patches.
3. Connectivity of habitat patches.
4. Characteristics of linkage areas between patches.
5. Landscape matrix and edge effects.
6. Anthropogenic disturbances.

Habitat suitability thresholds are poorly understood at the second order of habitat selection (Connelly et al. 2004). The relationships among indicators likely confound thresholds. Consistently describing subpopulation areas using these indicators across the range of the species may provide insights important in conservation planning. Comparing changes in these second-order indicators over time (e.g., between existing

Table 2. Mid-scale (second-order) habitat indicators and suitability characteristics for sage-grouse habitats.

Habitat Indicators	Metric Description	Habitat Suitability Characteristics
1. Habitat Availability	The amount of sagebrush habitat in the area.	The more sagebrush habitat relative to potential habitat, the greater the area's suitability.
2. Patch Size and Number	The average size of habitat patches and the number of patches within the area.	Generally, the larger and more contiguous the habitat patches relative to the area, the greater the suitability of that area.
3. Patch Connectivity	The average distance from one habitat patch to the nearest similar patch within the area.	As the average distance between sage-grouse habitat patches in the area decreases, suitability increases.
4. Linkage Area Characteristics	Percent shrub cover in relation to tree or grass/forb cover of areas between habitat patches through which sage-grouse move.	As linkage areas between habitat patches increase in shrub cover rather than tree or grass/forb cover, habitat suitability increases. Presence of anthropogenic features between patches also decreases linkage area suitability.
5. Landscape Matrix and Edge Effect	The amount of edge in contact with plant communities or land uses with positive or negative influences on the habitat patch.	As the amount of sagebrush edge in contact with plant communities or land uses that positively influence shrubland patch habitat increases, the landscape matrix and edge suitability increase.
6. Anthropogenic Disturbances	The fragmentation of contiguous sagebrush patches in the area through land use changes and infrastructure development. Measured as the number, length, or area (or area of influence) of embedded anthropogenic features per unit patch area.	As the number and intensity of anthropogenic features within the habitat patches in the area decrease, suitability increases.

conditions and those of an earlier reference period) provides information on habitat trends.

Knick et al. (2013) have identified ecological minimums required by sage-grouse in the western portion of their range. Both land cover of sagebrush and anthropogenic features including human activity were the primary variables that defined those minimums. Taylor et al. (2013) reported on anthropogenic stressors from oil and gas development and West Nile virus and their effects on sage-grouse at this scale. Patch size, connectivity, habitat linkage, and landscape matrix thresholds for sage-grouse need further study.

Quantifying existing habitat conditions using the six indicators and population monitoring will help reveal habitat and population relationships, and comparing existing conditions over time or a reference period could be helpful for describing habitat trends associated with second-order indicators. However, the spatial analysis skills or tools and availability of adequate vegetation datasets needed for these types of analyses are limited in many cases, so agencies, academia, and other conservation partners are encouraged to work together to build capacity in this regard.

Habitat availability, patch size, and patch connectivity are major components of suitability in the second order. The amount of occupied habitat within the landscape matrix of nonhabitat and unsuitable habitat is important to describe (table 2, indicator 1). In some areas, the ratio of suitable to marginal to unsuitable habitat would be an important conservation statistic for measuring habitat restoration progress. The more sagebrush habitat relative to potential habitat, the greater the area's suitability. Whether the available habitat is contained in one large habitat patch or several patches (indicator 2) could influence sage-grouse use and dispersal between subpopulations (figure 8). Dispersal could be uninterrupted in large habitat patches, whereas movement between smaller patches may be disrupted, depending on the configuration of the patches and landscape matrix in which they are embedded. Generally, the larger and more contiguous the sagebrush patches of a population or subpopulation are, the greater the suitability of that area. The closer the

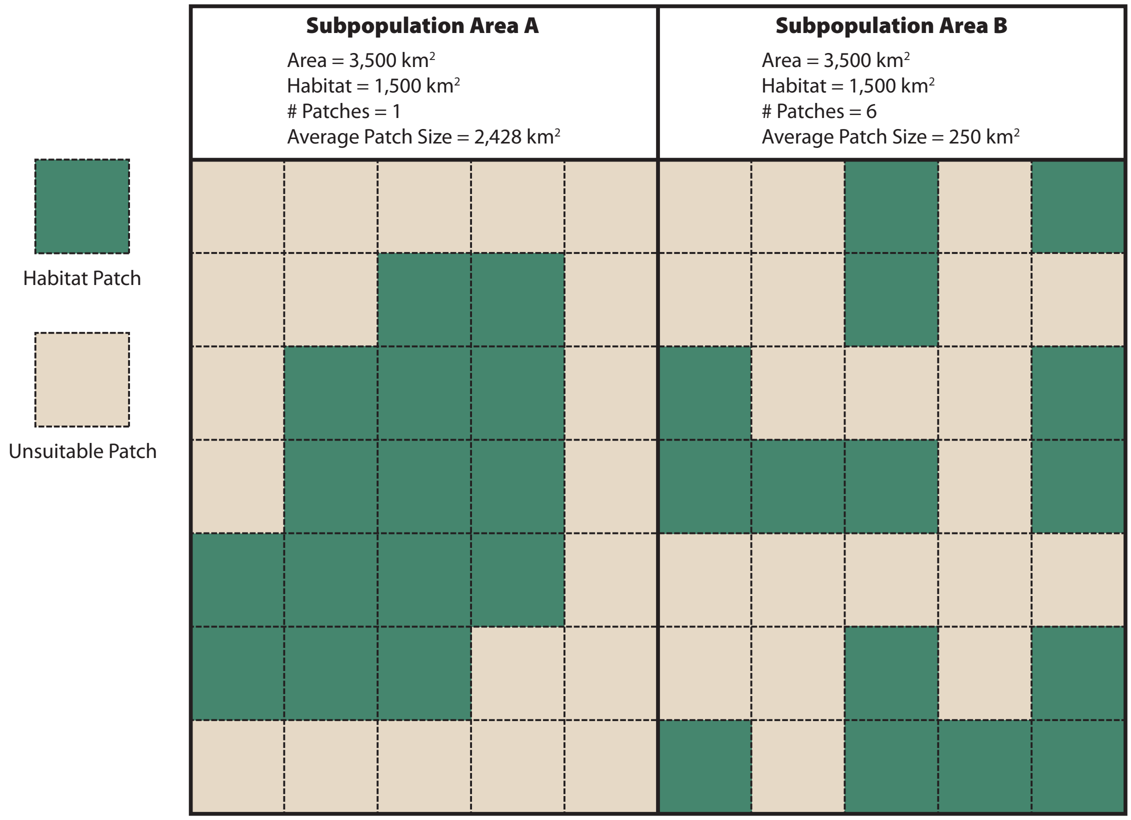


Figure 8. Habitat patches in two similar subpopulation areas. Areas A and B have similar total area and habitat quality, but area A has one large habitat patch while area B has several smaller ones. In area A, sage-grouse can freely disperse. The distance between patches in area B is great enough to limit sage-grouse movement between the patches, potentially affecting habitat suitability.

suitable habitat patches are to each other, the more likely sage-grouse can move freely between them (indicator 3).

Habitat linkage and patch edges forming a matrix on the landscape can greatly influence habitat use and dispersal within and between occupied areas. The landscape context in which patches are located has a bearing not only on habitat suitability for dispersal between patches but also on the likelihood that the habitat patches will persist into the future (Morrison et al. 1998). Resource managers, planners, and decisionmakers should evaluate existing or potential pathways from habitat patch to habitat patch. Barriers that compromise sage-grouse movements between habitat patches are not completely understood and are variable (Connelly et al. 1988; Leonard et al. 2000; Beck et al. 2006; Knick and Hanser 2011). Linkage area suitability is believed to improve as the percent of shrub cover (not necessarily sagebrush) increases relative to tree or grass cover in the areas between the habitat patches (indicator 4). The cover type or land use immediately adjacent to a habitat patch can positively or negatively affect the quality of that patch's suitability as sage-grouse habitat. Adjacent land cover types also differ in (1) mortality risks posed to birds occupying the habitat patch, (2) influence on existing patch quality, and (3) influence on patch and habitat persistence. As the amount of sagebrush edge in contact with plant communities or land uses that positively influence shrubland patch habitat increases, the landscape matrix and edge suitability increase (figure 9) (indicator 5). This is termed "positive edge" (Ries et al. 2004). Edge effects associated with roads and other linear anthropogenic features within habitat patches are discussed later as a component of fragmentation within the habitat patch.

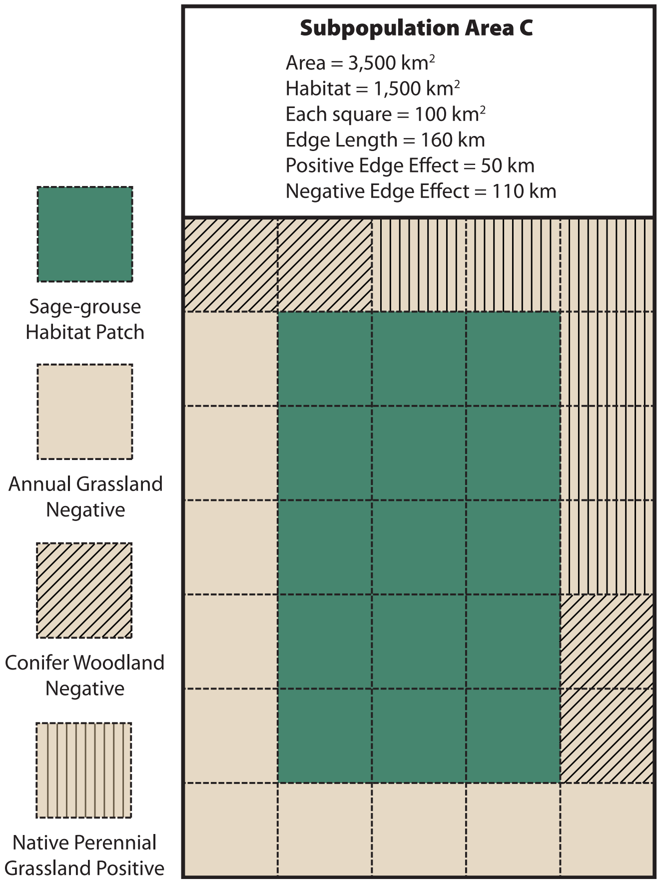


Figure 9. A habitat patch depicting a function of contrast and (dis)similarity. These communities greatly affect future risks to sage-grouse populations and habitat suitability.

Anthropogenic disturbances influence sage-grouse habitat, numbers, and distribution at each order of habitat selection (indicator 6). Anthropogenic features can affect sage-grouse demographics or habitat use in two significant ways:

- Anthropogenic features may directly and indirectly cause mortality, which can then affect the long-term sustainability of the population or subpopulation. The mortality significance of the features depends on their scope and intensity. However, an increase in anthropogenic features in otherwise suitable habitat increases the probability that the habitat will become a sink habitat rather than a source habitat (Aldridge 2005). Effects of the human footprint may not be readily apparent in the immediate population response, but over time, and if the scope and intensity of these features increase, there will likely be a negative impact on population trend (Connelly et al 2004; Aldridge 2005; Holloran 2005; Wisdom et al. 2005).

- Sage-grouse eventually avoid areas with a high density of anthropogenic features even if site-scale conditions are suitable (Connelly et al. 2004). While there is still much to learn about the dispersal and home range selection process, there is mounting evidence that sage-grouse are sensitive to human disturbances and will avoid areas they once used if those areas have been altered by anthropogenic features that exceed some threshold (Connelly et al. 2004; Aldridge 2005; Holloran 2005; Johnson et al. 2011; Knick et al. 2011; Knick et al. 2013). The anthropogenic feature thresholds that affect these selection processes likely vary depending on type of use, seasons of use, intensity of use, cumulative extent of features, topography, and other factors. However, if these changes occur quickly on the landscape, sage-grouse may not recognize the risks associated with these features and may not show an immediate avoidance response (Aldridge 2005; Aldridge and Boyce 2007).

Fine Scale (Third Order)

Sage-grouse select seasonal habitats (third-order habitats) within their home ranges, including breeding, summer, and winter habitats (figure 10) (Johnson 1980; Connelly et al. 2004). For many wildlife species with large home ranges, including sage-grouse, seasonal life requisite needs differ, and movement is required to meet seasonal shelter and food needs. Sage-grouse are generally traditional in their seasonal movement patterns (Schroeder et al. 1999; Connelly et al. 2004; Holloran 2005). Some sage-grouse may move long distances (>30 km) from breeding to summer and from summer and to winter habitats. Fedy et al. (2012) reported high variability of movement distances within and among seasonal habitats. Sage-grouse diets shift from insects and forbs during breeding and summer seasons to sagebrush during winter (Berry and Eng 1985; Schroeder et al. 1999; Connelly et al. 2004). The life requisite

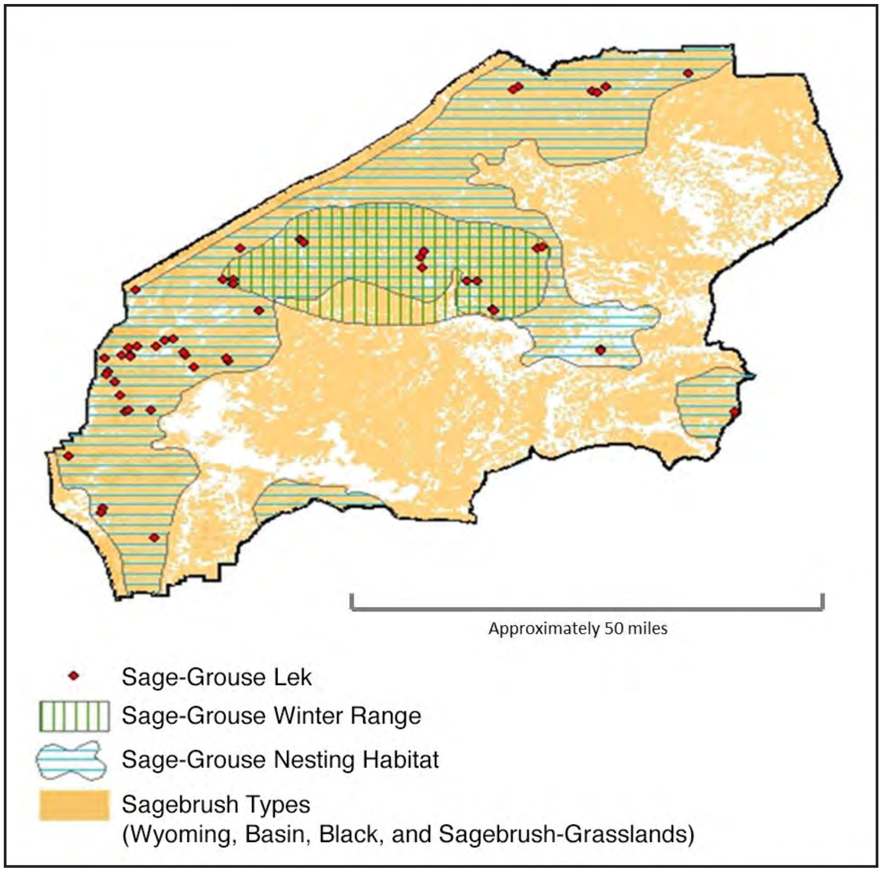


Figure 10. Fine-scale (third-order) habitat selection.

"space" is still a predominant need for sage-grouse to access their seasonal food and shelter needs at the fine scale.

Third-order habitat descriptions should address factors that affect sage-grouse use of, and movements between, seasonal use areas. Seasonal home ranges for sage-grouse associated with a lek or lek group within a population or subpopulation area should be the habitat focus. In some cases, small isolated populations or subpopulations may be the focus of fine-scale descriptions. Habitat suitability at the fine scale can generally be described as follows:

- Suitable habitats within home range areas have contiguous mosaics of sagebrush shrublands or grassland/sagebrush connecting seasonal use areas. Anthropogenic features within home ranges that can disrupt seasonal movements or cause mortality are generally absent or at least not widespread.
- Marginal habitats within home range areas have patchy, disjunct sagebrush shrublands or grassland/sagebrush between seasonal use areas or may exhibit some degree of tree/conifer encroachment. Anthropogenic features that can disrupt seasonal movements or cause mortality may occur within the home range.
- Unsuitable habitats within a home range area are potential shrublands currently dominated by perennial or annual grasses, invasive woodlands (e.g., western juniper), or incompatible land uses (some anthropogenic features) not conducive to sage-grouse seasonal movements or habitat use. Most leks have been abandoned or have few remaining birds. Other unsuitable habitat examples include conifer encroachment (>4 percent canopy cover); severe topographical features such as deep canyons; and lands converted to farmland, urban areas, reservoirs, etc.

At this scale, sage-grouse select seasonal ranges to meet their life requisite needs (Johnson 1980; Connelly et al. 2003). Sage-grouse generally inhabit large interconnected areas of sagebrush habitat, thus, there are three fine-scale (third-order) habitat indicators that influence sage-grouse use of and movements between seasonal use areas (table 3):

1. Seasonal habitat availability.
2. Seasonal use area connectivity.
3. Anthropogenic disturbances and habitat loss and fragmentation.

Seasonal habitat availability is the initial habitat indicator at this scale. Although sage-grouse are considered a landscape species, the amount of habitat required has not been determined due to the variability in quality and juxtaposition within the landscape (Connelly et al. 2011). Generally, the more sagebrush shrubland within seasonal use areas in the home range, the more suitable the habitat (indicator 1).

The availability and connectivity of sagebrush within seasonal use areas of sage-grouse home ranges can affect suitability. To address this, seasonal use areas need to be identified and mapped. Descriptions of the availability of other forb-rich habitats in summer and fall areas is also important at this scale, particularly if these habitats are in close proximity to sagebrush-dominated communities.

Following nesting, hens often move chicks to summer ranges for food. Connectivity between breeding and summer brood-rearing habitats is particularly important due to the restricted flight capability of chicks at this time. In general, the more contiguous the sagebrush cover between seasonal use areas, the more suitable the habitat (indicator 2). In some areas, other shrub communities may provide important connecting habitat between seasonal use areas.

There is increasing evidence that anthropogenic disturbances within a home range can cause local extirpations even if other habitat conditions appear suitable (Aldridge 2005; Holloran 2005; Aldridge et al. 2008). Anthropogenic features can affect sage-grouse in two significant ways at the fine scale. Anthropogenic features directly and indirectly increase mortality or decrease recruitment, and sage-grouse may eventually avoid seasonal use areas with a high density of anthropogenic features even if site-scale conditions are suitable (indicator 3). Anthropogenic features can also facilitate the intrusion of avian and mammalian species that directly depredate sage-grouse, or they may promote the spread of exotic plant species such as cheatgrass or noxious weeds that alter the

Table 3. Fine-scale (third-order) habitat indicators and suitability characteristics for sage-grouse habitat seasonal use areas within home ranges (in terms of potential barriers to movement, reproduction, and survival).

Habitat Indicators	Metric Description	Habitat Suitability Characteristics
1. Seasonal Habitat Availability	The amount of sagebrush shrubland in seasonal use areas. The amount of other forb-rich habitats in summer/fall seasonal use areas.	The more sagebrush shrubland within seasonal use areas in the home range, the greater the area's suitability. Other forb-rich habitats in summer/fall seasonal use areas are available.
2. Seasonal Use Area Connectivity	The extent of sagebrush connectivity between seasonal use areas.	As areas between seasonal use areas increase in sagebrush cover, habitat suitability increases.
3. Anthropogenic Disturbances	The disruption of movement between or use of seasonal use areas within a home range due to land use changes and infrastructure development. Measured as the number, length, or area of anthropogenic features within a home range area.	As the number and significance of anthropogenic features within a home range decrease, suitability increases.

suitability of habitats (Lyon 2000; Lyon and Anderson 2003; Holloran 2005; Aldridge 2005).

Site Scale (Fourth Order)

Habitat suitability at the site scale (fourth order) describes the more detailed vegetation indicators of seasonal habitat such as canopy cover and height of sagebrush (nesting and wintering); the associated understory vegetation (breeding, nesting, and early brood-rearing); and vegetation associated with riparian areas, wet meadows, and other mesic habitats adjacent to sagebrush (summer/late brood-rearing) (figure 11). Based on extensive research in many western states, Connelly et al. (2000) developed and Hagen et al. (2007) reviewed habitat criteria or indicators required by sage-grouse for specific seasonal needs (breeding, summer, and wintering). While general criteria were recommended, Connelly et al. (2000) recognized that ecological site potential should be considered at the site scale. Hagen et al. 2007 provided a meta-analysis of existing research on nesting and brood-rearing habitats. Generalized seasonal habitats are characterized as (1) breeding habitat—habitat for prelaying hens, leks, nesting, and early brood-rearing; (2) summer/late brood-rearing habitat; (3) fall habitat; and (4) winter habitat. Connelly et al. (2000) provided extensive treatment of each of these seasonal ranges. Tables 4 through 7 summarize seasonal habitat indicators at the fourth order.

Figure 11. Site-scale (fourth-order) habitat selection.

The various site-scale seasonal habitat criteria or indicators referenced above have been further interpreted in the HAF to provide a range of habitat categories that facilitate sage-grouse habitat evaluations and conservation planning. Suitable habitats provide the appropriate protective cover (sagebrush and herbaceous plants), food (forbs, insects, and sagebrush), and security (few or no trees or tall structures for predators) needs for sage-grouse to survive and reproduce (Connelly et al. 2000; Sather-Blair et al. 2000). Marginal habitats include habitat components to support sage-grouse, but habitat conditions are lower in quality compared to suitable habitats and does not provide shelter from predators and weather. Survival and reproduction rates are assumed lower in marginal habitats compared to suitable habitats (Cooperrider et al. 1986; Morrison et al. 1998). Unsuitable habitats are currently missing one or more of the basic life requisites of food or shelter, though they may have the potential to provide these life requisites in the future. In all cases, professional judgment and experience are needed to describe suitability in the appropriate context.

Table 4. Site-scale (fourth-order) breeding habitat indicators and suitability characteristics for lek sites (Connelly et al. 2000).

Habitat Indicators	Metric Description	Habitat Suitability Characteristics
1. Availability of Sagebrush Cover	Lek has adjacent sagebrush cover in close proximity.	Adjacent sagebrush cover within 100 meters.
2. Proximity of Detrimental Land Uses	The distance to land uses that have detrimental effects on lek use. Sonic and physical disturbances such as highways, railroads, and industrial parks are examples.	Detrimental land uses are not within line of sight of lek and absent to uncommon within 3 km of lek.
3. Proximity of Trees or Other Tall Structures	The presence of trees or other tall structures within line of sight of leks.	Trees or other tall structures are not within line of sight of lek and absent or uncommon within 3 km of the lek.

Table 5. Site-scale (fourth-order) breeding habitat indicators and suitability characteristics for nesting and early brood-rearing sites.

Habitat Indicators	Metric Description	Habitat Suitability Characteristics	
		Arid Sites[1]	Mesic Sites[1]
1. Sagebrush Cover	Average percent cover for land cover type.	15–25%	15–25%
2. Sagebrush Height	Average sagebrush height for land cover type.	30–80 cm (12–30 inches)	40–80 cm (15–30 inches)
3. Predominant Sagebrush Shape[2]	Number of sagebrush plants by shape and most common sagebrush shape for land cover type.	Spreading	Spreading
4. Perennial Grass and Forb Heights	Average maximum heights in land cover type.	≥18 cm (≥7 inches)	≥18 cm (≥7 inches)
5. Perennial Grass Cover	Average percent cover for land cover type.	≥10%	≥15%
6. Perennial Forb Cover	Average percent cover for land cover type.	≥5%	≥10%
7. Preferred Forb Availability	Number of preferred forbs in land cover type.	Good abundance and availability relative to ecological site potential	

[1] Mesic and arid sites should be defined on a local basis; annual precipitation, herbaceous understory, and soils should be considered (Connelly et al. 2000).

[2] Sagebrush plants that are more tree- or columnar-shaped, with no or few lower branches, provide less protective cover near the ground than sagebrush plants with a spreading shape. Basin big sagebrush (*Artemisia tridentata* spp. *tridentata*) plants often have this columnar shape, as do other sagebrush species or subspecies that have been heavily browsed or rubbed. Sagebrush communities in which the columnar shrub shape is predominant are assumed likely to require more herbaceous cover to compensate to provide adequate protection for nesting sage-grouse and young broods. Conversely, in suitable habitat, the spreading shape should be predominant; however, there may be a small proportion of columnar plants present.

Table 6. Site-scale (fourth-order) habitat indicators and suitability characteristics for summer/late brood-rearing habitat (Connelly et al. 2000).

Habitat Indicators	Metric Description	Habitat Suitability Characteristics	
		Upland Sagebrush Communities[1]	Riparian and Wet Meadow Communities
1. Sagebrush Cover	Average percent cover for land cover type.	10–25%	
2. Sagebrush Height	Average sagebrush height for land cover type.	40–80 cm (15–30 inches)	
3. Availability of Sagebrush Cover	Food site has sagebrush cover in close proximity.		Sagebrush cover is within 100 m of riparian or wet meadow foraging area.
4. Perennial Grass and Forb Cover	Average percent cover for land cover type.	≥15%	
5. Riparian Stability	Functioning condition.		The majority of riparian areas are in proper functioning condition.
6. Preferred Forb Availability	Number and density of preferred forbs in land cover type.	Good abundance, diversity, and availability relative to ecological site potential.	

[1] In areas where agricultural fields provide the food resources, the habitat indicators for protective cover apply.

Table 7. Site-scale (fourth-order) habitat indicators and suitability characteristics for winter habitat (Connelly et al. 2000).

Habitat Indicators	Metric Description	Habitat Suitability Characteristics
1. Sagebrush Cover	Average percent cover exposed above snow in wintering area.	≥10–30% exposed above snow.
2. Sagebrush Height	Average height above snow in wintering area.	≥25–35 cm (10–14 inches) exposed above snow.

To ensure consistency in reporting and communicating field data, seasonal habitat suitability matrices should NOT be revised unless warranted by scientific evidence. Guidelines for managing sage-grouse habitats have been published by Connelly et al. (2000) and evaluated by Hagen et al. (2007). These guidelines describe characteristics of productive sage-grouse habitats based on a large number (n=24) of studies conducted throughout the species' range. These guidelines are often included in various management plans and planning documents. However, this information should not be viewed as providing standards by which to judge the overall quality of sagebrush habitats. Instead, these sage-grouse habitat characteristics should be used as tools for assessing habitats and guiding management actions.

Connelly et al. (2000) stated that there may be a need to develop adjustments to height and cover requirements and emphasized that any such adjustments should be reasonable and ecologically defensible. To foster consistency, making adjustments to site suitability indicator values at the local scale should be avoided unless there is strong, scientific justification for doing so. Regional adjustments must be supported by regional plant productivity and habitat data and in floristic provinces and sage-grouse management zones as reported by Connelly et al. (2004) and Stiver et al. (2006). If adjustments are made to the site-scale indicators, they must be made using nesting and brood-rearing data collected from sage-grouse studies found in the relevant area and peer reviewed by the appropriate wildlife management agency(ies) and researchers.

Similarly, regional research may suggest the need to adjust habitat management guidelines or quantitative indicator values in the HAF's site-scale suitability matrices. However, these matrices are designed to organize field data into a useful format for consistency and communication, so changes in criteria should only be made after considerable coordination and only if scientific evidence warrants their adjustment. There is a tendency to review each indicator and its suitability category independently, but site suitability is determined by the relationship among the several indicator values in each matrix. The suitability classes for these matrices are based on rangewide plant productivity and structural data and expert opinion relative to sage-grouse use. Finally, it is important to recognize that the term "suitable" is not synonymous with "optimum."

In some parts of the range, the indicators will need to be interpreted with a regional perspective. For example, the sagebrush cover may be naturally high in some portions of the sage-grouse range, but herbaceous cover capability, based on site potential, may be below the height identified in the guidelines; thus, adequate cover for sage-grouse may still be present. In other portions of the range, sagebrush cover may be below those found in the guidelines, but herbaceous cover may be high and providing adequate cover for nesting.

Invasive plants, especially invasive annual grasses, that occur in many sagebrush habitats can have deleterious effects on sage-grouse habitat and therefore should be documented. While sage-grouse habitat may be directly affected by invasive plants through competitive exclusion of native plants that provide cover and forage (Rowland et al. 2010; Mooney and Cleland 2001), the most significant impacts of invasive plants on sage-grouse habitat are indirect through alteration of fire regimes. Invasive annual grasses generally provide for continuous ground cover that facilitates greater frequency and intensity of fires creating annual grass dominated habitats compared to native perennial habitats that are dominated by sparse, discontinuous fuels (Balch et al. 2013; Antonio and Vitousek 1992). The resulting increased frequency and intensity of fires result in changes in life form classes from shrubs to grasses, and species composition becomes dominated by annuals, providing little value for food and cover for sage-grouse (Connelly et al. 2004; Davies et al. 2011; Miller et al. 2011).

While sage-grouse may occupy habitats where shorter statured Sandberg bluegrass (*Poa secunda*) is dominant in the understory, this is not sufficient reason to assume that the suitability indicator value for grass height should be reduced, especially if the ecological site potential is for larger bunchgrasses. Rather, this condition may indeed reflect reduced habitat suitability and likely indicates a rangeland health issue that should be addressed via appropriate restoration activities or management changes. These examples illustrate that individual indicator values do not define site suitability and that overall site suitability descriptions require an interpretation of the relationships between the indicators and other factors. Professional expertise and judgment are required.

Habitat Description Steps

Habitat description steps are identified for each scale. Descriptions for the first and second order are brief. Descriptions and evaluations of habitat at these scales have been completed or are in the process of being completed through ecosystemwide assessments. These assessments have been tasked by agencies including the BLM, U.S. Forest Service, and U.S. Geological Survey and nongovernmental organizations, including The Nature Conservancy. Policy-level officials, scientists, spatial analysts, and resource managers need to access these evaluation efforts to reach decision points for each scale.

Broad Scale (First Order) and Mid Scale (Second Order)

Considerable broad-scale and mid-scale information is available for Greater Sage-Grouse range (Schroeder et al. 2004) and populations (Connelly et al. 2004) as well as for Gunnison Sage-Grouse (Gunnison Sage-Grouse Rangewide Steering Committee 2005). Stiver et al. (2006) identified seven sage-grouse management zones that conform to seven clusters of habitat and populations described in Connelly et al. 2004 from Kuchler (1970), West (1983), and Miller and Eddleman (2001) (figure 7). The management zones provide a first- and second-order context for management purposes. There are also several regional assessments describing shrub steppe habitat (table 8). These assessments provide critical information necessary for finer scale habitat descriptions as they provide scale context to habitats and populations (Connelly et al. 2004; Wisdom et al. 2005; Aldridge et al. 2008). In addition, these assessments describe and evaluate disturbances to landscapes and resulting habitat patterns operating at the population and species range scales. Large landscape features and disturbances influence the distribution and abundance of sage-grouse on the landscape. The BLM has also conducted six rapid ecoregional assessments over the range of Greater and Gunnison Sage-Grouse that examine ecological values, conditions, and trends within ecoregions. Greater and Gunnison Sage-Grouse populations/subpopulations as described by the U.S. Fish and Wildlife Service (2013) and Gunnison Sage-Grouse Rangewide Steering Committee (2005) are shown in figure 3 (see chapter 1).

From a practical standpoint, the management of sagebrush/sage-grouse habitats at the first order of habitat selection requires policy at the management zone that contributes to policy for

Table 8. Rangewide and regional assessments containing information on sage-grouse or their habitat.

Species	Assessment Area	Citations
Greater Sage-Grouse	Rangewide (OR, WA, CA, NV, ID, UT, MT, WY, CO, NM, AB, SK)	Connelly et al. 2000; Miller and Eddleman 2001; Connelly et al. 2004; Aldridge et al. 2008; Knick and Connelly 2011
Greater Sage-Grouse	Upper Columbia River Basin (OR, WA)	Hann et al. 1997; Wisdom et al. 2000
Greater Sage-Grouse	Great Basin (ID, NV, UT, CA)	Wisdom et al. 2005
Greater Sage-Grouse	Wyoming Basin (WY, CO, MT, UT, ID)	Rowland et al. 2006a
Gunnison Sage-Grouse	Rangewide (CO, UT)	Gunnison Sage-Grouse Rangewide Steering Committee 2005

the range of sage-grouse. Each management zone, evaluated by the various regional assessments, provides policymakers with parameters to match policy to realistic outcomes.

Management and management direction for second-order scales require the use of existing broad-scale data and the application of GIS tools for analysis. These evaluations should document existing conditions (see form M-1 in appendix B), assess potential for habitat manipulation, and consider landscape constraints. Landscape scientists and spatial analysts may provide decisionmakers with a vision of the future landscape matrix.

Fine Scale (Third Order)

Ecological processes of interest at the third order of habitat selection are those that may affect sage-grouse movements between seasonal habitats within a home range (table 9). Habitat needs and the indicators that describe life requisite needs vary by season. Third-order habitat assessments take into account seasonal use areas or home ranges of sage-grouse associated with a lek or group of leks. Seasonal habitat availability, connectivity, and anthropogenic disturbances should be described at this scale. Third-order habitat mapping uses the information gathered at the mid-scale and refines it to show seasonal habitat patterns for a home range of interest.

At this scale, identifying seasonal habitat use areas to the extent possible is important. Habitat and wildlife resource specialists, along with people with local knowledge, should jointly evaluate sage-grouse seasonal distribution evidence to determine the presence or absence of sage-grouse. In the absence of telemetry data or other seasonal use data or models, wildlife biologists who understand sage-grouse habitat selection and needs can effectively predict how sage-grouse make seasonal use of their habitats. In many cases, mapping seasonal habitats will occur incrementally over time and in higher priority landscapes first due to limited staffing and funding resources.

Table 9. Summary of fine-scale (third-order) ecological processes (Johnson 1980), mapping features, and management levels for sage-grouse habitat descriptions.

Ecological Processes	
Ecological Time Period	5–20 years in the future
Climatic Processes	Local weather patterns: localized drought, rain shadow areas
Landscape Processes	Local-scale processes that have long- and short-term consequences on home range use, seasonally and year-round: conversion of sagebrush habitat between seasonal ranges to nonhabitat or unsuitable habitat, anthropogenic features that act as filters or barriers to seasonal movements
Population Processes - Habitat Dynamics	Connectivity of sagebrush habitat and other adjacent habitats provide for effective use of seasonal habitats within a home range, seasonal migration corridors are maintained, collective fitness of birds within the home range is sufficient for long-term persistence
Mapping Features	
Extent	Seasonal habitats within a home range
Grain	Fine grain (30-meter pixel size)
Vegetation Cover Types	Associations or groups thereof
Geographic Extent Equivalents	Subbasins or group of watersheds
Cartographic Scale Range	e.g., 1:24,000–1:100,000
Management Levels	
Administrative Hierarchical Level	Local county governments, BLM field offices or subunits, Forest Service national forests/ranger districts
Planning and Assessment Documents	BLM activity plans (e.g., habitat or allotment management plans), forest plans, watershed assessments, and land use plans

The steps to describe sage-grouse habitat at the fine scale (third order) are as follows:

Step 1. Determine the extent and grain size appropriate for a habitat description of the home range area. Develop a vegetation map using appropriate third-order land cover types.

Identify sage-grouse populations or subpopulations as described by the U.S. Fish and Wildlife Service (2013) and Gunnison Sage-Grouse Rangewide Steering Committee (2005) and shown in figure 3 (see chapter 1). Delineate the home range area of interest and document the grain size for the analyses needed. Generally, a 30-meter pixel size is desired for third-order descriptions. Remote data should be collected at as fine a scale as available and affordable and should be aggregated at the 30-meter pixel resolution. Third-order habitat descriptions require more detailed vegetation information for an area. Identify natural vegetation cover types using information from the National Vegetation Classification System (see http://usnvc.org/ or http://www.natureserve.org/conservation-tools/projects/us-national-vegetation-classification).

Land cover datasets are constantly being refined or improved upon, so use the latest, most appropriate product or version. Distinguishing between sagebrush alliances (Reid et al. 2002) to help identify seasonal habitat availability and connectivity of different sagebrush communities is important (table 10). Distinguishing between certain nonhabitat types, such as salt desert shrub, forest/woodland, and agricultural lands, is also important. Pasture lands or conservation reserve program lands adjacent to sagebrush habitat may provide summer food resources with little risk from pesticides or mowing. Conversely, sage-grouse use of agricultural lands, such as row crops adjacent to sagebrush, may be hazardous to sage-grouse because of risk of mortality from mechanical equipment (e.g., mowing) or chemicals.

Step 2. Map occupied seasonal habitats and identify potential habitat by seasonal use period.

Occupied and potential seasonal habitats should be mapped in cooperation with the state wildlife agency. Historic and current data and knowledge

Table 10. Example of basic sagebrush land cover types needed for mid-scale (second-order) habitat descriptions. Fine-scale (third-order) cover types are generally shrubland alliances as described by Reid et al. (2002). NP = native perennial grass, EP = exotic perennial grass, EA = exotic annual grass.

Mid-Scale Cover Types (overstory/understory)	Fine-Scale Cover Types (overstory/understory)	
Sagebrush/Native Perennial Grass	Wyoming and basin big sagebrush/NP Black sagebrush/NP Low sagebrush/NP Low sagebrush – mountain big sagebrush/NP Low sagebrush – Wyoming big sagebrush/NP Mountain big sagebrush/NP	Rigid sagebrush/NP Silver sagebrush/NP Threetip sagebrush/NP Wyoming big sagebrush – squawapple/NP Gambel Oak – Basin big sagebrush shrubland/NP
Sagebrush/Exotic Perennial Grass	Wyoming and basin big sagebrush/EP Black sagebrush/EP Low sagebrush/EP Low sagebrush – mountain big sagebrush/EP Low sagebrush – Wyoming big sagebrush/EP	Mountain big sagebrush/EP Rigid sagebrush/EP Silver sagebrush/EP Threetip sagebrush/EP Wyoming big sagebrush – squawapple/EP
Sagebrush/Exotic Annual Grass	Wyoming and basin big sagebrush/EA Black sagebrush/EA Low sagebrush – mountain big sagebrush/EA Low sagebrush – Wyoming big sagebrush/EA Mountain big sagebrush/EA	Rigid sagebrush/EA Silver sagebrush/EA Threetip sagebrush/EA Wyoming big sagebrush – squawapple/EA

from local sage-grouse experts should be used to help identify seasonal use areas and to determine the migratory status of the population. In some areas, seasonal habitats will overlap (e.g., breeding and winter or late brood-rearing/summer). In other areas, seasonal habitat may be separated by many miles. Three main sage-grouse seasonal habitats (breeding, which is composed of lekking, prelaying, nesting, and early brood-rearing; summer/late brood-rearing; and winter) should be identified (table 11). If seasonal use patterns are unknown, mapping the vegetation and elevations will help identify them. State wildlife agencies, federal agencies, or university researchers may have telemetry data or other information that can be used as well. In addition, predictive modeling as described by Yost et al. (2008) can be used to help identify seasonal habitats.

Breeding Habitat: The breeding period typically occurs from March 1 through late June and includes the period when sage-grouse attend leks to breed, prepare nutritionally for nesting, nest, and raise young chicks (Connelly et al. 2000). Breeding habitat includes all sagebrush types that may be used during this timeframe. Sage-grouse require a mixture of sagebrush, grasses, and forbs for adequate breeding habitat. Sagebrush cover types within 18 km (11 miles) of a lek for migratory populations and 5 km (3.1 miles) for nonmigratory populations are considered breeding habitat and are mapped as such unless this distance includes sagebrush communities that sage-grouse would not use for nesting (e.g., deep canyon areas, sagebrush areas typically covered by deep snow, or sagebrush areas compromised by anthropogenic disturbances). Mapping sagebrush habitats at this scale, with the exclusion of canyon areas and other areas not used for nesting, can be readily accomplished using routine GIS techniques and available land cover and digital elevation data. The accuracy of some thematic vegetation data can be problematic, so users need to understand the limitations of the data. In addition, there may be some sagebrush cover types that do not provide suitable breeding habitat due to plant structure characteristics or because of edaphic conditions, steep slopes, aspect, or other factors that are important locally. Map known nesting and early brood-rearing areas if telemetry data or other observational data are available.

Summer/Late Brood-Rearing Habitat: Summer is generally described as that period between July 1 and September 30 (Connelly et al. 2000). During summer, sage-grouse are found in areas with succulent forbs adjacent to or intermixed with sagebrush. Hens generally move their chicks to more mesic conditions, such as higher elevation sagebrush communities, mountain shrub communities, wet meadow complexes, agricultural fields, perennial lakes, streams, ponds, or lakebeds adjacent to sagebrush, during the summer months. Riparian areas associated with steep drainages or canyons typically are not used by sage-grouse and should not be mapped as summer habitat. Several information sources are available to help identify summer habitats within the home range area:

1. Observations by local residents and agency field personnel.

Table 11. General seasonal habitat descriptions modified from Connelly et al. (2000).

Habitats	General Use Period[1]	General Description[2]
Breeding Habitat	March 1–June 30	Includes leks, prenesting, nesting, and early brood-rearing habitats. A variety of sagebrush plant communities in close proximity to leks and big sagebrush communities.
Summer/Late Brood-Rearing Habitat	July 1–September 30	Variety of mesic or moist habitats in close proximity to sagebrush communities.
Winter Habitat	December 1–February 28 or 29	Variety of sagebrush communities that have sagebrush above the snow.

[1] Use periods may vary based on elevation and annual weather conditions.

[2] General descriptions for some areas; primary vegetation communities may vary based on local conditions and availability.

2. Historic observations in BLM or other agency files.
3. Telemetry data.
4. National Wetlands Inventory (NWI) maps.
5. National Hydrography Dataset (NHD) maps.
6. Riparian proper functioning condition (PFC) assessments and maps.
7. Remote sensing data (NAIP, GAP, Landfire, etc.).
8. Digital elevation models.
9. Current and historic brood survey routes/ area surveys conducted by wildlife agencies.

Mesic sagebrush communities adjacent to breeding habitats should be considered summer habitat and may occur beyond the 18 km distance from leks, particularly in higher elevation areas. In addition, within breeding and summer sagebrush habitat, all riparian, wetland, and other forb-rich habitat should be considered summer habitat. Ground–truthing of historic brood routes should be conducted to determine continued presence of sage-grouse.

Fall Habitat: Fall is the period when sage-grouse transition from feeding on forbs, insects, and sagebrush to primarily sagebrush. Use of fall habitats may occur from September to December due to yearly variability in temperature and precipitation as plants desiccate or die from frost (Connelly et al. 2011). Fall habitats are generally not believed to be a limiting life history component for most populations and therefore are not discussed further.

Winter Habitat: Sage-grouse are entirely dependent on sagebrush for food and cover during winter. Sage-grouse use sagebrush that is exposed above the snow or on windswept ridges. Sagebrush that is covered by deep snow, such as at some higher elevations, is not available to sage-grouse. Sage-grouse typically congregate in large groups during winter and use traditional wintering areas (Berry and Eng 1985; Schroeder and Robb 2003). Wintering areas are likely the most difficult habitats to map for sage-grouse. Wintering areas may be inaccessible, may vary based upon annual weather/snow conditions, or may be found long distances from other known habitats. Mapping known traditional winter use areas, particularly those that are used by large numbers of birds, is important. Due to access constraints during winter, potentially important areas may be identified any time during the year based on topography, sagebrush type, and evidence of roost (pellet group) sites. Areas should eventually be verified for winter use, if possible, by documenting birds, tracks, and scat observed. Particularly during years of above average snowfall, biologists should attempt to document sage-grouse winter-use areas to identify the critical habitat areas. Additionally, biologists should conduct directed searches of likely areas during the winter based upon topography, slope and aspect, elevation, and vegetation. The state wildlife agency, local landowners, or other field personnel may have information regarding winter use. Information sources that may be useful include:

1. Observations by local residents, local working groups, or agency personnel.
2. Telemetry data.
3. Historic observations from land management and wildlife agency files.
4. Aerial flights during winter.
5. Graduate theses, dissertations, and published literature.

Step 3. Describe seasonal habitat availability.

Using the information from steps 1 and 2, describe occupied and potential seasonal habitats in the home range area. Breeding, summer, and winter habitats are important to describe. Calculate:

1. The estimated amounts of occupied breeding, summer, and winter habitats.
2. The estimated amounts of potential breeding, summer, and winter habitats.

Documenting the amount of existing sage-grouse seasonal habitat relative to potential habitat is important because it provides critical information for restoration planning.

Step 4. Describe and map anthropogenic features within and between seasonal habitats.

Overlay spatial data for anthropogenic features that was gathered at the second order (mid scale; indicator 6). For the home range area, document the following information:

1. The location and density of highways, major roads (km/km^2), railroads, transmission lines, oil/gas pipelines, and other large linear features.
2. The location, number, and density (sites/km^2) of communication sites, energy pads, mineral sites, wind turbines, meteorological towers, geothermal sites, landfills, gravel pits, and other anthropogenic features.
3. If planning a habitat trend analysis, the estimated decade or year (the latter if within the last 10 years) when the anthropogenic feature occurred within the home range.
4. The cumulative suitability of the home range based on anthropogenic features.

Step 5. Describe vegetation connectivity characteristics between seasonal use areas.

Home ranges with contiguous sagebrush cover between seasonal use areas are more suitable as habitat than those with discontinuous land cover. For home ranges with separated seasonal use areas, habitat suitability improves as the amount of shrub cover between seasonal use areas increases and tree or annual grass cover decreases. Shrub cover connectivity is particularly important for movements between breeding and summer habitat when chicks are incapable of making long-distance flights. Describe the vegetation between each seasonal use area: breeding to summer, summer to winter, and winter to breeding. Also describe the natural barriers (canyons, mountains) and anthropogenic barriers (reservoirs, canals, major highways, intensive agriculture) between each seasonal use area that may hinder the birds' ability to move between the areas.

Step 6. Summarize the information from steps 3-5 to describe existing third-order habitat suitability of the home range area of interest.

Organize and summarize the information for each third-order indicator on the "Fine-Scale (Third-Order) Sage-Grouse Habitat Description" (form F-1 in appendix B). An example of a completed form for a hypothetical site is shown in figure 12. Baseline third-order habitat data can be used in the future for trend analyses, so documenting the data sources and software, computer programs, and process steps used to describe third-order habitat conditions is important. Identifying where the data for the assessment are stored and can be retrieved in the future is also important. Good documentation of the data, including metadata, and analyses will help future biologists assess changes, causes, and effects.

Once the habitat indicator descriptions have been completed, the suitability of the seasonal-use area can be determined using the descriptive criteria on form F-1.

The habitat suitability of the home range area should be depicted spatially on the map created in steps 1 and 2.

Step 7 (optional). Repeat steps 1-6 and identify a reference period to assess habitat trends.

At the third order, comparing existing habitat suitability data for all or selected indicators to some previous reference period is useful for identifying habitat trends. Land cover type data for the fine-scale indicators of interest as well as sage-grouse lek or other historical data should be available for the reference period. Identify the habitat indicators of interest, measure them with appropriate computer and GIS tools, and describe them in terms of positive, neutral, or negative trends. A summary of this description should be included on form F-1 for each seasonal habitat time period.

Form F-1: Fine-Scale (Third-Order) Sage-Grouse Habitat Description		
Description Year: 2008	**Counties:** Humboldt	**State:** NV
Evaluator(s): Stiver	**Agency:** NDOW	
Home Range Name: Lone Willow	**Population:** Western Great Basin	
Lek Group Name:	**General Location:** Lone Willow	

Data Sources	
Land Cover Type Data Sources: GAP	
Anthropogenic Features Data Sources: Nevada Heritage	
Population Data Sources: NDOW	
Data Storage Location: ftp://ftp.ndow.org/sagegrouse/habitat/HU	
Software and Version: ArcView 10.2	
Mapping Grain: 30 meter pixel	**Home Range Area Extent (km^2):** 240

Habitat Indicator Descriptions	
1. Seasonal Habitat Availability	a. Area of occupied breeding habitat (km^2) = 80
	a. Area of occupied summer habitat (km^2) = 120
	a. Area of occupied winter habitat (km^2) = 140
	b. Area of potential breeding habitat (km^2) = 100
	b. Area of potential summer habitat (km^2) = 150
	b. Area of potential winter habitat (km^2) = 200
	c. Area of nonhabitat (km^2) (optional) =
	Discussion:
2. Seasonal Use Area Connectivity	Breeding to summer (km edge/km^2 of habitat) = 3.2
	Summer to winter (km edge/km^2 of habitat) = 2.5
	Winter to breeding (km edge/km^2 of habitat) = 3.8
3. Anthropogenic Disturbances	a. Densities of linear features (km/km^2) = .75
	b. Densities of point features (sites/km^2) = 1.45
	c. Area of nonhabitat or unsuitable habitat inclusions (km^2) =
	Discussion:

Fine-Scale (Third-Order) Suitability Summary	
✔	**Check the one description below that best describes the home range:**
✔	**Suitable:** Home ranges have connected seasonal use areas. Anthropogenic features that can disrupt seasonal movements or cause mortality are generally absent or at least not widespread.
	Marginal: Home ranges have poorly connected or disjunct seasonal use areas. Anthropogenic features that can disrupt seasonal movements or cause mortality may occur within the home range.
	Unsuitable: Home ranges have seasonal use areas with predominantly grassland, woodland, or incompatible land uses (anthropogenic features) not conducive to sage-grouse seasonal movements or habitat use. Most leks have been abandoned or have few remaining birds.
Discussion:	Large intact habitat. Priorities are to protect winter range on the east side of the range and restore winter range south of the main mountain.

Figure 12. An example of a completed fine-scale (third-order) habitat description form.

Site Scale (Fourth Order)

Ecological processes that may affect individual sage-grouse selection of leks, nest sites, feeding locations, and winter-use areas are important at the fourth order (table 12). Ecological processes of interest take into account seasonal habitat needs related to the life requisites of shelter and food for birds associated with a lek or lek group. Habitat needs and the indicators that describe life requisite needs vary by season. Seasonal habitat availability, connectivity, and anthropogenic disturbances were described at the mid and fine scales. At the fourth order, availability of protective vegetation cover and food resources within seasonal habitats are described.

The basic seasonal habitat suitability matrices developed for the HAF (forms S-2 through S-6 in appendix B) were based largely on Connelly et al. (2000) as a starting point because they used data collected across the species range. However, while Connelly et al. (2000) describe characteristics of *productive* seasonal habitats, generally equivalent to the HAF's "suitable" class, the HAF also describes marginal and unsuitable habitats in an effort to reflect a range of conditions that a land manager may be faced with in performing a habitat assessment.

For the purpose of standardizing habitat descriptions and improving communication, discrete ranges of numeric values or other measurements (e.g., visual shape guide) are used to describe seasonal habitat indicators as suitable, marginal, or unsuitable (Sather-Blair et al. 2000). The numeric values described for productive habitat by Connelly et al. (2000) are guidelines and are not intended to be used as strict prescriptions. To a sage-grouse there may not be much difference between a sagebrush community with 14 percent sagebrush canopy cover and one with 15 percent canopy cover; however, discrete ranges are needed to organize the field information for interpretation.

Table 12. Summary of site-scale (fourth-order) ecological processes (Johnson 1980), mapping features, and management levels for sage-grouse habitat descriptions.

Ecological Processes	
Ecological Time Period	Current to 5 years; average lifespan of sage-grouse
Climatic Processes	Seasonal weather patterns that can affect individual fitness (e.g., excessive spring rains during nesting or early brood-rearing)
Landscape Processes	Fourth-order processes that have short-term consequences on seasonal habitat selection and suitability: natural variation in potential of ecological sites to provide suitable seasonal habitats; herbivory effects on food and shelter habitat needs; human disturbance of birds during critical periods (breeding and wintering); anthropogenic features that increase predation potential during critical periods
Population Processes Habitat Dynamics	Habitat provides for food and shelter needs of the birds for effective daily use within seasonal use areas; individual fitness is sufficient
Mapping Features	
Extent	Seasonal use areas
Grain	Sampling plots (transects)
Vegetation Cover Types	Associations and ecological sites
Geographic Extent Equivalents	Cover type within an ecological site
Cartographic Scale Range	e.g., <1:24,000
Management Levels	
Administrative Hierarchical Level	Grazing allotments, pastures, state wildlife management areas, etc.
Planning and Assessment Documents	Site evaluations; project-specific assessments and plans

Individual indicator values cannot be used independently to describe habitat suitability; rather, site suitability is described using all of the appropriate indicators. For example, the predominant shape of sagebrush plants in an area affects the herbaceous cover needs during the breeding season. A columnar-shaped (tree-shaped) sagebrush plant does not provide the shelter that a spreading-shaped plant provides (figure 13), and an area dominated by this type of sagebrush shape may be of marginal suitability if the accompanying understory has little grass or forb cover. However, in another area of predominantly columnar-shaped sagebrush plants, the presence of abundant grass, forb, or other shrub species cover may make the site suitable as nesting habitat. At another site, shrub and grass cover may be suitable, but the absence of forbs would reduce overall site suitability. These examples illustrate that individual indicator values do not define site suitability in and of themselves and that overall site suitability descriptions require an interpretation of the relationships between all of the indicators and other factors. Professional expertise and judgment are required for these steps.

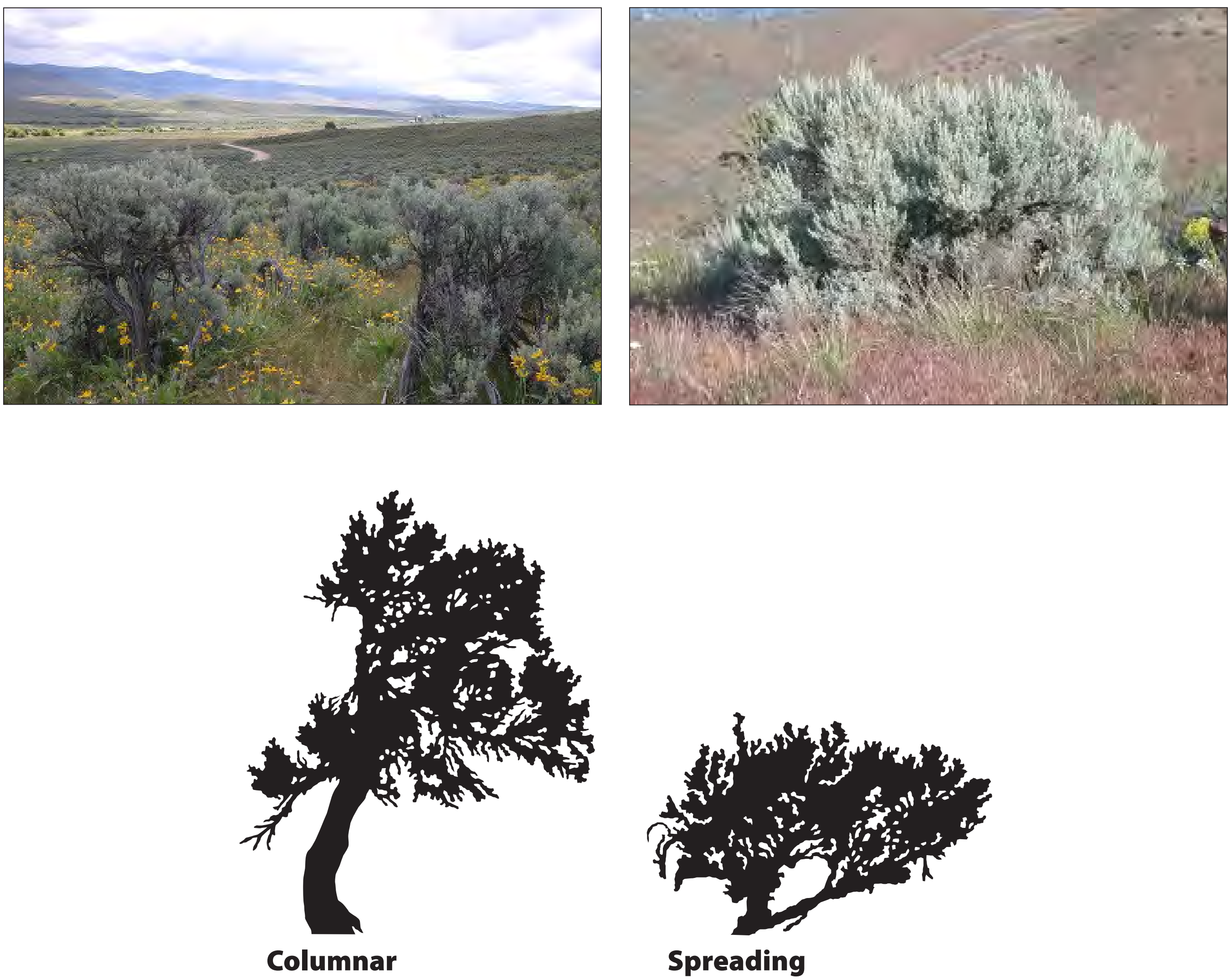


Figure 13. Sagebrush shape is an important habitat cover indicator. Sagebrush communities with more columnar-shaped plants need more herbaceous cover for shelter needs than communities with more spreading-shaped plants.

The steps to describe sage-grouse habitat at the site scale (fourth order) are as follows:

Step 1. Identify seasonal use areas and associated third-order cover types of interest for third-order descriptions. Determine the extent of these land cover types within the seasonal use area.

Refining fine-scale cover type maps of a home range area may be helpful for site-scale descriptions. For a home range area, describing all (e.g., for a small, mountain valley subpopulation) or some (e.g., for a larger, basin subpopulation) of the seasonal use areas may be important. Depending on the scope and purpose of the habitat description, not all land cover types within a seasonal use area may need to be sampled at the project level. For long-term monitoring, only one or two sagebrush cover types for breeding habitat descriptions or certain known wet meadow complexes for brood-rearing habitat descriptions may be needed.

Grasslands or other currently unsuitable cover types that have the potential to become habitat in the future should also be measured because the information collected may be useful for conservation planning. Fourth-order information for these cover types can provide important information on shrub and forb recruitment, linkage area suitability, conifer encroachment, or other aspects of habitat condition.

Step 2. Overlay soil or ecological site maps on land cover type maps to determine ecological site potential.

Ecological site potential, the potential vegetation community, and the production of plant material of a site is based on soil, topography, and climate. For sagebrush communities, site potential (in terms of shrub, grass, and forb composition) is mostly determined by precipitation patterns and soil characteristics (Cronquist et al. 1972; Miller and Eddleman 2001). Ecological site descriptions and soil maps can be obtained from local Natural Resources Conservation Service (NRCS) offices or from the Internet (https://esis.sc.egov.usda.gov). Herrick et al. (2005) provided recommendations on types and numbers of samples as well as background information on ecological sites and site potential. This information is needed for interpreting habitat data for the suitability matrices (e.g., forb abundance related to site potential) and for predicting potential natural habitat changes (i.e., composition and rates of change in community composition relative to natural disturbances and succession) and alternative habitat changes (i.e., composition and rates of change to plant communities not anticipated for a site and from which it is more difficult to recover the natural community). Site potential data would be particularly valuable for predicting future conditions of sagebrush shrubland areas that are now grasslands (native perennial versus exotic annual) due to fire or anthropogenic disturbances.

Soils are mapped in units (e.g., soil mapping units) that can and often do include a mixture of soils correlated to a mixture of ecological sites. For example, a soil map unit may include two soils with two different ecological sites. One ecological site may result from small inclusions of soils that support a mountain big sagebrush (*Artemisia tridentata vaseyana*) community, but the vast majority of the soil map unit consists of a soil that supports a different ecological site with a low sagebrush (*Artemisia arbuscula*) community. These intermixed communities are valuable because big sagebrush is used by males and females for protective cover or nesting, while low sagebrush sites provide important forbs for prelaying hens and broods and loafing sites for adult birds.

Soil maps have not been completed for the entire range of sage-grouse. However, NRCS state soils information is available and provides basic information at a coarse resolution. Data are available at http://www.nrcs.usda.gov/wps/portal/nrcs/site/soils/.

Step 3. Obtain ecological reference sheets, if available, for the ecological sites contained within the seasonal habitat area of interest.

Pellant et al. (2005) described reference sheets as the primary reference for an evaluation of rangeland health. The reference sheet describes a range for each indicator based on expected spatial and temporal variability within each ecological site (or equivalent). Reference sheets provide important information about the 17 indicators of rangeland health and how well the ecological processes are functioning. This information, along with other components of the ecological site descriptions can provide context for more detailed studies on sage-grouse habitat suitability. However, ecological site descriptions have not been completed on portions of the sage-grouse range. If ecological reference areas (ERAs) (Pellant et al. 2005) for the important cover types in the seasonal use area are available, then a visit may be valuable when the expected forb species composition for an ecological site is not well described in ecological site descriptions. Collecting fourth-order data at one or more ERAs for reference purposes might be useful.

Step 4. Design the sampling approach.

Prior to sampling habitat at the fourth order, an appropriate design must be determined. Using the information from steps 1-3, develop an appropriate sampling design and collect field data using one of the methods outlined in the next step and explained further in appendix B. Consulting with other biologists, statisticians, soil scientists, arid land ecologists, or rangeland management specialists to develop an appropriate sampling design for seasonal use areas based on available soils and ecological site data may be helpful. See the Craters of the Moon National Monument case study in appendix A for one example of a sampling approach.

For most fourth-order descriptions, stratified, random sampling of the seasonal habitat area based on land cover types and soils (ecological sites) will be appropriate. In some cases, the seasonal use area may be further stratified by sagebrush canopy cover (e.g., recently disturbed versus mature) or anthropogenic disturbance strata (e.g., grazing pastures, density of anthropogenic features) depending on the intent of the assessment and logistical capacity.

In many areas, patches of big sagebrush (or other tall-statured sagebrush) occur in expansive low or dwarf sagebrush areas. These areas should be treated as two separate cover types or strata. However, there are heterogeneous sagebrush communities that are not easily teased apart and may be better sampled as one stratum. There may be other situations where only certain sagebrush areas are of interest due to steepness of slope, aspect, or other reasons. For example, in "basin and range" topography, seasonal sagebrush habitats may be distributed in narrow, linear stringers adjoining ridges or alluvial fans. In such cases, extra effort is needed to map and stratify these areas to ensure adequate representation in the sample. Use of shorter transects may also be warranted in these situations to ensure that they do not extend beyond the boundary of the cover type of interest. In other cases, only the priority breeding habitat cover types may be sampled due to costs. The rationale for decisions concerning sampling design should always be clearly explained and documented.

Multiple samples (i.e., transects) are likely to be needed in each stratum to account for variability of vegetation and to characterize uncertainty in the habitat indicator estimates. At a minimum, three samples should be collected per stratum because calculating a sample variance per stratum with fewer samples is not possible. The desired number of samples required for each cover type depends on the vegetation heterogeneity of the land cover type and desired degree of precision (or amount of change to be detected). Elzinga et al. (1998) and Herrick et al. (2005) provided guidance on sampling design, and there are many sample size estimation tools available online, including:

http://www.landscapetoolbox.org/mssret/MSSRET.html https://www.dssresearch.com/KnowledgeCenter/toolkitcalculators/samplesizecalculators.aspx

Specialists may also want to seek assistance in sample design from a statistician. Ultimately, decisions about the degree of precision and sample sizes should be tempered by what is practical given the budget and time available.

Ideally, sample size requirements should be determined using previously collected habitat data from the study area or from a pilot study. If this is not possible, sample sizes can be estimated by using data collected from nearby, similar areas. When calculating sample sizes, pay attention to specifying realistic degrees of precision, depending on the purpose of the assessment. Some sample size calculators specify precision in terms of percent variation or change from the mean, which can be confusing when specifying precision for proportion or percent cover indicators (e.g., a difference of 10 percentage points for sagebrush cover that is at 20 percent is actually a difference of 50 percent). Variability in a stratum can also vary by indicator. Ideally, sample sizes should be estimated individually for several important indicators such as sagebrush cover, grass height and forb cover, and a sample size that provides sufficient precision for all three should be selected. However, this practice may not be practical in many instances due to logistical realities.

Regardless of the technique used to determine sample size prior to sampling, an evaluation of sampling sufficiency should be conducted at the end of each data collection effort to determine if the data collected meet the stated precision requirements. The same equations and tools used to estimate sample sizes can also be used to assess sample sufficiency. If sample sufficiency is determined to be too low, additional samples may be warranted.

The timing of sampling fourth-order habitat characteristics depends on what is being measured (table 13). Nesting habitat vegetation should be measured toward the end of the nesting period, generally between May 1 and June 30 to assess forb and grass presence, and annual variation in precipitation should be evaluated to determine when samples should be measured. Late brood-rearing habitat should be measured between July 1 and August 30 depending on latitude and elevations. Fall is a transitional time when the birds are moving from summer to winter habitat. During September, birds may still be concentrated on summer use areas where succulent forbs and

Table 13. Seasonal timing of vegetation data collection associated with habitat indicators for site-scale descriptions.

Seasonal Habitat	Window for Vegetation Data Collection	Comments
Breeding (leks)	Anytime	Vegetation data can be collected at any time of year.
Breeding (nesting and early brood-rearing)	April–June	Data should be collected as soon as hens are off the nest (generally May 1–June 30). Timing within this window will vary based on latitude and elevation.
Summer/Late Brood-Rearing	July–August	Data should be collected based on timing of seasonal movements. Data collection for higher elevation late brood-rearing habitat areas should occur later than for areas of lower elevation.
Fall	September–November	See comments under summer season for early fall use areas. As fall progresses, seasonal movements begin and diets shift.
Winter	November–March	Data can be collected at any time in this window. Snow levels may dictate when data should be collected for wintering areas. Consider mapping all sagebrush habitats as a starting point until more use can be verified. Historical and extreme snow depths should be assessed.

insects can be found. As temperatures cool and their diet changes to sagebrush, sage-grouse begin moving from forb-rich areas to winter range. Winter habitat can be evaluated throughout the year as related to sagebrush species and subspecies diversity and general sagebrush distribution on the landscape; however, the availability of sagebrush to sage-grouse in winter (i.e., above the level of snow cover) is contingent on local snow depths. In some cases, therefore, winter site visits are recommended.

Step 5. Collect field data.

Measuring vegetation at the fourth order generally involves collecting field data on composition and structure of habitat within a seasonal use area (table 14). There are additional measurements (e.g., lek proximity to sagebrush) for some seasonal habitats as well. Connelly et al. (2003) described methods that have previously been used to measure sage-grouse habitat at the fourth order. Line intercept and ocular (using a Daubenmire frame) (LIDF) and line-point intercept (LPI) methods can produce different though comparable cover results (Floyd and Anderson 1987; Symstad et al. 2008; Thacker 2010; Santini 2012). True cover parameters are seldom known in natural ecosystems (Bonham et al. 2004). Advantages and disadvantages of each technique are discussed in Elzinga et al. (1998), Connelly et al. (2003), and Bonham (2013). For the HAF, a key objective is that cover averages fall within the appropriate suitability class. Since transect data are averaged and suitability classes are relatively broad, the differences between techniques used to arrive

Table 14. List of seasonal habitat measurements and associated data collection methods. LPI = line point intercept, LIDF = line intercept—Daubenmire frame, PFC = proper functioning condition.

Seasonal Habitat	Habitat Indicator	Life Requisite(s)	Measurement Technique
Lek	Availability of Sagebrush Cover	Cover	Field or remote sensing measurement
	Proximity of Detrimental Land Uses	Security	Field or remote sensing measurement*
	Proximity of Trees or Other Tall Structures	Security	Field or remote sensing measurement*
Breeding	Sagebrush Cover	Cover, Food	LPI/LIDF
	Sagebrush Height	Cover	LPI/LIDF
	Predominant Sagebrush Shape	Cover	LPI/LIDF
	Perennial Grass and Forb Height	Cover	LPI/LIDF
	Perennial Grass Cover	Cover	LPI/LIDF
	Perennial Forb Cover	Cover	LPI/LIDF
	Preferred Forb Availability	Food	Forb diversity transect/plot species inventory
Summer/Late Brood-Rearing – Riparian	Riparian Stability	Cover, Food	PFC data, if available
	Preferred Forb Availability	Food	Forb diversity transect/plot species inventory
	Availability of Sagebrush Cover	Cover	Field or remote sensing measurement
Summer/Late Brood-Rearing – Upland	Sagebrush Cover	Cover, Food	LPI/LIDF
	Sagebrush Height	Cover	LPI/LIDF
	Perennial Grass and Forb Cover	Cover	LPI/LIDF
	Preferred Forb Availability	Food	Forb diversity transect/plot species inventory
Winter	Sagebrush Cover	Cover, Food	LPI/LI (part of LIDF)
	Sagebrush Height (above snow)	Cover	LPI/vegetation height (part of LIDF)

* Proximity of trees, other tall structures, and anthropogenic disturbances to be noted in comment field of data collection forms for all seasonal habitats.

at those estimates should have minimal impact on the end result. Once a technique or multiple techniques are selected, the technique(s) should be used consistently throughout the assessment or monitoring period for future comparability.

For the BLM, the HAF can be implemented in conjunction with the core indicators and methods that were developed as part of the assessment, inventory, and monitoring (AIM) strategy to improve the efficiency and effectiveness of BLM's assessment and monitoring activities (Toevs et al. 2011). The purpose of the core indicators and methods is to provide consistent, quantitative, land cover and vegetation data using standardized measurements that will allow data to be integrated across the entire range of sage-grouse as well as used for other assessment and monitoring purposes (MacKinnon et al. 2011). The core methods were designed to be a minimal set of methods that should be supplemented with additional methods to meet specific resource needs such as sage-grouse habitat assessments or monitoring.

Procedures for the LIDF and LPI data collection methods, including illustrations and data forms, are provided in appendix B. These methods have been used for sage-grouse habitat descriptions across the range of the species.

This chapter and appendix B provide instructions and illustrations to aid in the technical aspects of these habitat measurements (e.g., determining sagebrush shape, measuring grass and sagebrush height). Additional fourth-order notes and measurements, including local drought conditions, presence of anthropogenic noise disturbance, other shrub canopy cover (besides sagebrush), annual grass canopy cover, and noxious weed abundance, are addressed for some seasonal habitats to aid in interpreting overall site suitability. For example, sagebrush cover is a crucial habitat indicator for fourth-order descriptions. However, in some locations the composition and percent cover of other shrubs can affect site suitability. For instance, sagebrush may only provide 10 percent canopy cover for a particular cover type, but antelope bitterbrush (*Purshia tridentata*) is also present with a canopy cover of 5 percent. The density of bitterbrush may positively affect the overall site suitability.

Once field data are collected, summarize the data for the seasonal habitats of interest on the "Sage-Grouse Site-Scale Seasonal Habitat Data Summary" (form S-1, appendix B). An example of a completed form for a hypothetical site is shown in figure 14.

Step 6. Transfer field data for land cover types of interest into suitability matrix categories associated with the seasonal habitat. Determine fourth-order suitability.

Once the field data have been summarized for land cover types of interest on form S-1, they can be transferred to the suitability worksheets (forms S-2 through S-7) for the appropriate seasonal use periods. Seasonal habitat suitability worksheets with detailed instructions are provided in appendix B. One worksheet should be completed for each cover type stratum sampled in the seasonal use area and administrative unit (e.g., pasture). Where otherwise similar vegetation cover type strata differ substantially due to slope, aspect, or other factors, summarizing those areas separately may be prudent, depending on local conditions and expertise. The mean, mode, or other appropriate summary statistics for each indicator are recorded on the worksheet, and the corresponding suitability category is checked (✔). Describing overall site suitability requires some level of professional judgment because rarely will all indicators fall in the same suitability range. The rationale for suitability criteria must be explained, particularly if it is not obvious on the worksheet. Examples illustrating suitability interpretations are shown in figures 15 through 18.

Leks (form S-2): Suitability should be described for each lek regardless of status (occupied, unoccupied, or undetermined). Site suitability for leks is relatively easy to describe because there are only two indicators: (1) sagebrush cover (presence and amount of sagebrush in close proximity to

Form S-1: Sage-Grouse Site-Scale Seasonal Habitat Data Summary

Date: 06/23/12	County: Blaine	State: ID	Evaluator(s): Janet Hill
Population: Snake, Salmon, and Beaverhead			Home Range Name: Big Hill
Seasonal Habitat: Breeding			Associated Leks: RB05; RB02

Land Cover Type	Ecological Site	Area (ha/ac) or Length (km/mi)	Transects (#)	Indicator Values from Data Forms (mean in most cases)								
				$\bar{x}$ Sage Cover (%)	$\bar{x}$ Sage Ht. (cm)	$\bar{x}$ Predominant Sage Shape (# of S and C)	$\bar{x}$ PG Ht. (cm)	$\bar{x}$ PF Ht. (cm)	$\bar{x}$ PG Cover (%)	$\bar{x}$ PF Cover (%)	Preferred Forb Species (#)	Lek Hbt. Avg. Distance to Sage Cover (m)
Wyoming big sage-brush/bluebunch wheatgrass	Loamy 8-12 ARTRW8/PSSPS	2300 ha	7	13	56	S=36 C=12	19	6	17	13	10	
Threetip sage-brush/bluebunch wheatgrass	Loamy 8-12 ARTRW8/PSSPS	1400 ha	4	19	45	S=32 C=14	15	8	9	5	3	
Bluebunch wheatgrass	Loamy 8-12 ARTRW8/PSSPS	5600 ha	3	4	19	S=0 C=2	25	7	16	8	13	
Threetip sage-brush/ crested wheat-grass	Loamy 8-12 ARTRW8/PSSPS	2100 ha	3	16	64	S=15 C=23	17	8	8	7	6	
Crested wheatgrass	Loamy 8-12 ARTRW8/PSSPS	700 ha	3	3	23	-	26	5	4	1	3	

Figure 14. An example of a seasonal habitat fourth-order data summary form completed with data from field measurements for the cover types of interest.

Form S-2: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Breeding Habitat (Leks)

Date: 4/3/12	County: Owyhee	State: ID	Evaluator(s): Janet Hill
Population: Northern Great Basin			Home Range Name: Triangle
Land Cover Type: ARTRW8/ARTRV/PSSPS/JUOC			Lek ID#: 20702
GPS file #: Xxxxxxxxx			Lek Status (circle one): Occupied Unoccupied Undetermined
UTM: NAD83, Zone 11, 542335E 4912479N			

Habitat Indicator Suitability Range

Habitat Indicator	Suitable	✔	Marginal	✔	Unsuitable	✔
Availability of Sagebrush Cover	*Lek has adjacent protective sagebrush cover (within 100 m)*	✔	*Sagebrush within 100 m provides very little protective cover*		*Adjacent sagebrush cover is >100 m*	
Proximity of Detrimental Land Uses	*Detrimental land uses are not within line of sight of lek and absent to uncommon within 3 km of lek*	✔	*Detrimental land uses are within line of sight of lek and uncommon or few within 3 km of lek*		*Detrimental land uses are within the vicinity of the lek site*	
Proximity of Trees or Other Tall Structures	*Trees or other tall structures are not within line of sight of lek and none to uncommon within 3 km of lek*		*Trees or other tall structures are within line of sight of lek and uncommon or scattered within 3 km of lek*		*Trees or other tall structures are within the vicinity of the lek site*	✔
Site-Scale Suitability	**Suitable**		**Marginal**	✔	**Unsuitable**	

Anthropogenic Noise Description:

N/A. Isolated from human presence. Some livestock can be heard in the lower valley.

Rationale for Overall Suitability Rating:

Site is generally a good lek site. It is a natural opening in a patch of Wyoming and Mountain big sage, relatively short grasses, forbs, and rocks. However, juniper has encroached to within 50 meters of the lek, creating perch sites for raptors. Removal of all juniper within 100 meters of the lek would greatly improve the site. Also, surrounding habitat may be used for nesting if trees are removed. Mostly big sage/bluebunch wheatgrass community with balsamroot, phlox, buckwheat, and goatsbeard in understory.

Figure 15. An example of a completed lek suitability worksheet.

Form S-3: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Breeding Habitat (Nesting/Early Brood-Rearing)

Date: 5/15/12	**County:** Blaine	**State:** ID	**Evaluator(s):** Janet Hill
Population: Snake, Salmon, and Beaverhead			**Home Range Name:** Big Hill
Land Cover Type: ARTRW8/PSSPS			**Ecological Site:** Loamy 8-12 ARTRW8/PSSPS
Associated Leks: RB05, RB02			**Number of Transects:** 7
Area Sampled (ha/ac): 2300 ha			**Site Info. (circle one):** Arid Site (circled) Mesic Site

List UTM Coordinates (coordinates, zone, datum) of All Transects: NAD83, Zone 11, 542335E 4912479N; 542416E 4912520N; 542599E 4912520N; 542721E 4912540N; 542680E 4912357N; 542253E 4912296N; 541867E 4912235N

Habitat Indicator Suitability Range

Habitat Indicator	$\bar{x}$	Suitable	✔	Marginal	✔	Unsuitable	✔
Sagebrush Canopy Cover (mean)	13	15 to 25%		5 to <15% or >25%	✔	<5%	
Sagebrush Height Mesic Site (mean) Arid Site (mean)	56	 40 to 80 cm 30 to 80 cm	✔	 20 to <40 cm or >80 20 to <30 cm or >80		 <20 cm <20 cm	
Predominant Sagebrush Shape (mode) Spreading (*n*) Columnar (*n*)	 36 12	Spreading	✔	Mix of spreading and columnar		Columnar	
Perennial Grass Height (mean)	19	≥18 cm	✔	10 to <18 cm		<10 cm	
Perennial Forb Height (mean)	6	≥18 cm		10 to <18 cm		<10 cm	✔
Perennial Grass Cover Mesic Site (mean) Arid Site (mean)	17	 ≥15% ≥10%	✔	 5 to <15% 5 to <10%		 <5% <5%	
Perennial Forb Cover Mesic Site (mean) Arid Site (mean)	13	 ≥10% ≥5%	✔	 5 to <10% 3 to <5%		 <5% <3%	
Preferred Forb Availability (relative to site potential)		Preferred forbs are common with several species present	✔	Preferred forbs are common but only a few species are present		Preferred forbs are rare	
Number of Preferred Forb Species (*n*)	10						

Site-Scale Suitability	Suitable	✔	Marginal		Unsuitable	

Does ecological site potential limit suitability potential? (circle one) Yes **No** (circled) Unknown

Drought Condition (circle one): Extreme Drought, Severe Drought, **Moderate Drought** (circled), Mid-Range, Moderately Moist, Very Moist, Extremely Moist

Rationale for Overall Suitability Rating:

Site is in suitable condition. Sagebrush cover is not quite in the suitable range, but all of the other indicators are in the suitable range. Sagebrush plants are healthy and there are signs of recruitment. Herbaceous cover heights are barely suitable but similar to ecological reference area. Poor winter and spring moisture may account for herbaceous heights.

Figure 16. An example of a Wyoming big sagebrush/bluebunch wheatgrass (*Pseudoroegneria spicata*) cover type with suitable breeding habitat conditions.

Form S-3: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Breeding Habitat (Nesting/Early Brood-Rearing)

Date: 5/27/12	**County:** Blaine	**State:** ID	**Evaluator(s):** Janet Hill
Population: Snake, Salmon, and Beaverhead			**Home Range Name:** Big Hill
Land Cover Type: Threetip sagebrush/bluebunch wheatgrass			**Ecological Site:** Loamy 8-12 ARTRW8/PSSPS
Associated Leks: RB05, RB02			**Number of Transects:** 4
Area Sampled (ha/ac): 1400 ha			**Site Info. (circle one):** Arid Site Mesic Site

List UTM Coordinates (coordinates, zone, datum) of All Transects: NAD83, Zone 11, 542335E 4912479N; 542416E 4912418N; 542599E 4912520N; 542721E 4912540N; 542680E 4912357N

Habitat Indicator Suitability Range

Habitat Indicator	$\bar{x}$	Suitable	✔	Marginal	✔	Unsuitable	✔
Sagebrush Canopy Cover (mean)	19	15 to 25%	✔	5 to <15% or >25%		<5%	
Sagebrush Height Mesic Site (mean) Arid Site (mean)	45	40 to 80 cm 30 to 80 cm	✔	20 to <40 cm or >80 20 to <30 cm or >80		<20 cm <20 cm	
Predominant Sagebrush Shape (mode) Spreading (*n*) Columnar (*n*)	 32 14	Spreading	✔	Mix of spreading and columnar		Columnar	
Perennial Grass Height (mean)	15	≥18 cm		10 to <18 cm	✔	<10 cm	
Perennial Forb Height (mean)	8	≥18 cm		10 to <18 cm		<10 cm	✔
Perennial Grass Cover Mesic Site (mean) Arid Site (mean)	9	≥15% ≥10%		5 to <15% 5 to <10%	✔	<5% <5%	
Perennial Forb Cover Mesic Site (mean) Arid Site (mean)	5	≥10% ≥5%	✔	5 to <10% 3 to <5%		<5% <3%	
Preferred Forb Availability (relative to site potential)		Preferred forbs are common with several species present		Preferred forbs are common but only a few species are present	✔	Preferred forbs are rare	
Number of Preferred Forb Species (*n*)	3						
Site-Scale Suitability		**Suitable**		**Marginal**	✔	**Unsuitable**	

Does ecological site potential limit suitability potential? (circle one) Yes No Unknown

Drought Condition (circle one): Extreme Drought Severe Drought Moderate Drought Mid-Range Moderately Moist Very Moist Extremely Moist

Rationale for Overall Suitability Rating:

Understory conditions are only marginal with forb cover barely suitable. The predominance of columnar-shaped sagebrush plants, marginal herbaceous cover conditions, and lack of preferred forbs makes this site marginal as breeding habitat.

Figure 17. An example of a threetip sagebrush/bluebunch wheatgrass cover type with marginal breeding habitat conditions.

Form S-3: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Breeding Habitat (Nesting/Early Brood-Rearing)

Date: 6/23/12	County: Blaine	State: ID	Evaluator(s): Janet Hill
Population: Snake, Salmon, and Beaverhead			Home Range Name: Big Hill
Land Cover Type: Bluebunch wheatgrass			Ecological Site: Loamy 8-12 ARTRW8/PSSPS
Associated Leks: RB05, RB02			Number of Transects: 3
Area Sampled (ha/ac): 5600 ha			Site Info. (circle one): Arid Site Mesic Site
List UTM Coordinates (coordinates, zone, datum) of All Transects: NAD83, Zone 11, 542335E 4912479N; 542416E 4912418N; 542599E 4912520N; 542721E 4912540N			

Habitat Indicator Suitability Range

Habitat Indicator	$\bar{x}$	Suitable	✔	Marginal	✔	Unsuitable	✔
Sagebrush Canopy Cover (mean)	4	15 to 25%		5 to <15% or >25%		<5%	✔
Sagebrush Height Mesic Site (mean) Arid Site (mean)	19	40 to 80 cm 30 to 80 cm		20 to <40 cm or >80 20 to <30 cm or >80		<20 cm <20 cm	✔
Predominant Sagebrush Shape (mode) Spreading (*n*) Columnar (*n*)	 0 2	Spreading		Mix of spreading and columnar		Columnar	N/A
Perennial Grass Height (mean)	25	≥18 cm	✔	10 to <18 cm		<10 cm	
Perennial Forb Height (mean)	7	≥18 cm		10 to <18 cm		<10 cm	✔
Perennial Grass Cover Mesic Site (mean) Arid Site (mean)	16	≥15% ≥10%	✔	5 to <15% 5 to <10%		<5% <5%	
Perennial Forb Cover Mesic Site (mean) Arid Site (mean)	8	≥10% ≥5%	✔	5 to <10% 3 to <5%		<5% <3%	
Preferred Forb Availability (relative to site potential)		Preferred forbs are common with several species present	✔	Preferred forbs are common but only a few species are present		Preferred forbs are rare	
Number of Preferred Forb Species (*n*)	13						

Site-Scale Suitability	Suitable		Marginal		Unsuitable	✔

Does ecological site potential limit suitability potential? (circle one) Yes **No** Unknown

Drought Condition (circle one): Extreme Drought Severe Drought **Moderate Drought** Mid-Range Moderately Moist Very Moist Extremely Moist

Rationale for Overall Suitability Rating:

Site is currently unsuitable due to the lack of sagebrush cover. All habitat components (sagebrush, grasses, and forbs) are present, therefore site has potential to become suitable habitat in the future.

Figure 18. An example of a bluebunch wheatgrass cover type with unsuitable breeding habitat conditions. Data indicate that cover type may provide suitable habitat in the future.

the lek); (2) proximity of detrimental land uses; and (3) sage-grouse security (proximity of tall structures such as trees and power poles) (table 15). Describing anthropogenic noise levels (from highways, oil and gas wells, and wind turbines) may also be valuable. Habitat descriptions are intended to help with identifying conservation actions, such as opportunities that might improve the status of a lek. In the example shown in figure 15, removal of avian predator perching structures (e.g., trees, fenceposts) near the lek would likely increase security. In addition, the influence of anthropogenic disturbances on lek use and lekking behavior may be better understood by reviewing how sage-grouse may be using adjacent seasonal habitats (e.g., winter or breeding and nesting).

Breeding Habitat (form S-3): The breeding habitat suitability matrix is the most complicated of the suitability worksheets (table 16). This matrix reflects the importance of breeding habitat, its complexity, and the amount of scientific data available on fourth-order habitat needs. There are different suitability ranges for some indicators depending on whether the breeding area is associated with mesic or arid sagebrush sites. For much of the Greater Sage-Grouse range, arid sites will be those closely associated with Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) and mesic sites will be associated with mountain big sagebrush. Determine whether the land cover type of interest is mesic or arid as defined locally (Connelly et al. 2000) before completing the suitability worksheet.

Where sagebrush cover types are highly interspersed (e.g., small patches of mountain big sagebrush inclusions occurring within a matrix of low sagebrush), sampling patches separately may not be possible or efficient. In such cases, sampling the area as a unit (i.e., one or more transects crossing the mosaic of various cover types) and acknowledging the inherent variability may be the best course of action. The big sagebrush inclusions may provide suitable cover for nesting while the low sagebrush communities may provide a greater diversity of forbs for prelaying hens and broods. Individually, these cover types may lack a life requisite need, but together they provide suitable habitat. The site field data for these intermixed cover types can be combined on one suitability worksheet.

Three examples of completed breeding habitat suitability worksheets using field data for a hypothetical breeding area are shown in figures 16 through 18. In the first example (figure 16), all indicators are in the suitable range except

Table 15. Breeding (lek) habitat life requisites, indicators, and suitability categories for site-scale habitat descriptions.

Life Requisite	Habitat Indicator	Suitability Categories		
		Suitable	Marginal	Unsuitable
Cover	Availability of Sagebrush Cover	*Lek has adjacent sagebrush cover (within 100 m)*	*Sagebrush provides very little protective cover adjacent to the perimeter of the lek*	*Adjacent nesting habitat unavailable*
Security	Proximity of Detrimental Land Uses	*Detrimental land uses are not within line of sight of lek and absent to uncommon within 3 km of lek*	*Detrimental land uses are within line of sight of lek and uncommon or few within 3 km of lek*	*Detrimental land uses are within the vicinity of the lek site*
	Proximity of Trees or Other Tall Structures	*Trees or other tall structures are not within line of sight of lek and absent to uncommon within 3 km of lek*	*Trees or other tall structures are within line of sight of lek though uncommon or scattered within 3 km of lek*	*Trees or other tall structures are within the vicinity of the lek site*

Table 16. Breeding (prelaying, nesting, and early brood-rearing) habitat life requisites, indicators, and suitability categories for site-scale habitat descriptions (adapted from Connelly et al. 2000; Sather-Blair et al. 2000; Hagen et al. 2007).

Life Requisite	Habitat Indicator	Suitability Categories		
		Suitable	Marginal	Unsuitable
Cover	**Sagebrush Cover** (%)	*15 to 25*	*5 to <15 or >25*	*<5*
	Sagebrush Height (cm)			
	Mesic Site[1]	*40 to 80*	*20 to <40 or >80*	*<20*
	Arid Site	*30 to 80*	*20 to <30 or >80*	*<20*
	Predominant Sagebrush Shape	*Spreading*	*Mix of spreading and columnar*	*Columnar*
	Perennial Grass and Forb Height (cm)	*≥18*	*10 to <18*	*<10*
	Perennial Grass Cover (%)			
	Mesic[1]	*≥15*	*5 to <15*	*<5*
	Arid	*≥10*	*5 to <10*	*<5*
Cover and Food	**Perennial Forb Cover** (%)			
	Mesic[1]	*≥10*	*5 to <10*	*<5*
	Arid	*≥5*	*3 to <5*	*<3*
Food	**Preferred Forb Availability**[2]	*Preferred forbs are common with several species present*	*Preferred forbs are common but only a few preferred species are present*	*Preferred forbs are rare*

[1] Mesic and arid sites should be defined on a local basis; annual precipitation, herbaceous understory, and soils should be considered (Connelly et al. 2000).

[2] Relative to ecological site potential.

for sagebrush cover, which is barely marginal. Overall, the habitat is rated as suitable. In the second example, indicator measurements are in the marginal range for three out of the eight indicators (figure 17). Sagebrush cover is adequate, but understory cover conditions and food resources provide only marginal fourth-order suitability. The last example, which is native perennial grassland, is clearly unsuitable due to lack of sagebrush cover (figure 18). However, native perennial grassland in the breeding habitat area has the ecological potential and the habitat components (i.e., forb and sagebrush recruitment) to become suitable in the future.

Summer Sites (form S-4, upland, and form S-5, riparian): Suitability is described differently for summer/late brood-rearing seasonal habitats depending on whether they are associated with upland sagebrush communities or riparian/wet meadow communities (tables 17 and 18) in close proximity to sagebrush communities. The indicators for upland summer habitats are similar to those for breeding habitat, but the ranges for the suitability categories differ. For riparian areas and wetlands, their functioning condition, as defined by Prichard et al. (1998, 2003), is used to describe site stability, which impacts the likelihood that cover and food resources are provided annually (fourth-order temporal scale). Functioning conditions, though they differ slightly between lentic and lotic areas, are generally defined as follows:

- **Proper functioning condition (PFC):** An area is considered to be in PFC when adequate vegetation or other structure components are present to:
  - Dissipate energy, reduce erosion, and improve water quality.
  - Filter sediment and aid in floodplain development.

Table 17. Summer/late brood-rearing habitat life requisites, indicators, and suitability categories for upland sagebrush site-scale habitat descriptions (adapted from Connelly et al. 2000; Sather-Blair et al. 2000; Hagen et al. 2007).

Life Requisite Feature	Habitat Indicator	Suitability Categories		
		Suitable	Marginal	Unsuitable
Cover	Sagebrush Cover (%)	*10 to 25*	*5 to <10 or >25*	*<5*
	Sagebrush Height (cm)	*40 to 80*	*20 to <40 or >80*	*<20*
Cover and Food	Perennial Grass and Forb Cover (%)	*≥15*	*5 to <15*	*<5*
Food	Preferred Forb Availability[1]	*Preferred forbs are common with appropriate numbers of species present*	*Preferred forbs are common but only a few preferred species are present*	*Preferred forbs are rare*

[1] Good abundance, diversity, and availability relative to ecological site potential.

Table 18. Summer/late brood-rearing habitat life requisites, indicators, and suitability categories for riparian or wet meadow site-scale habitat descriptions (adapted from Connelly et al. 2000; Sather-Blair et al. 2000; Hagen et al. 2007).

Life Requisite	Habitat Indicator	Suitability Categories		
		Suitable	Marginal	Unsuitable
Cover and Food	Riparian and Wet Meadow Stability	*Majority of areas are in PFC*	*Majority of areas are FAR*	*Majority of areas are NF*
Food	Preferred Forb Availability[1]	*Preferred forbs are common with appropriate numbers of species present*	*Preferred forbs are common but only a few preferred species are present*	*Preferred forbs are rare*
Cover	Availability of Sagebrush Cover	*Sagebrush cover is adjacent to brood-rearing areas (<100 m)*	*Sagebrush cover is in close proximity to brood-rearing areas (100-275 m)*	*Sagebrush cover is unavailable (>275 m)*

[1] Good abundance, diversity, and availability relative to ecological site potential.

- Improve flood-water retention and ground-water recharge.
- Stabilize streambanks and shorelines.
- Develop diverse ponding and channel characteristic for fish and wildlife habitat and other uses.
- Support greater biodiversity.

- **Functional–at risk (FAR):** An area is considered to be FAR when it possesses some or most of the elements for PFC but has at least one component/process that gives it a high probability of degradation.

- **Nonfunctioning (NF):** An area is considered NF when it clearly lacks the elements listed for PFC.

PFC data are available for most perennial streams and some wet meadows located on federal public lands. There are training opportunities and detailed procedures available for assessing PFC (Prichard et al. 1998, 2003). PFC data should be used whenever possible to help describe sage-grouse habitat. If PFC data cannot be obtained from other sources or collected directly, then the other two indicators should be used to assess habitat suitability.

Forb diversity should be described for brood-rearing areas associated with sagebrush uplands, including those adjacent to agricultural lands (e.g., alfalfa fields). With respect to the latter, descriptions should address whether sage-grouse are exposing themselves to unnecessary risks associated with agricultural fields when forbs are present in the uplands or are taking advantage of the only forbs available. Not all agricultural lands provide good brood-rearing habitat. Certain agricultural practices (e.g., herbicide and pesticide spraying, mowing, use of domestic animals considered to be sage-grouse predators) create risks to sage-grouse survival. Potential risks associated with agricultural fields should be noted (e.g., pesticides (Blus et al. 1989), direct mortality by mower, West Nile virus, etc.).

Proximity to taller sagebrush communities may be an important habitat indicator in some situations. For instance, some brood-rearing habitat occurs in forb-rich, low sagebrush communities adjacent to big sagebrush. In other cases, the available forbs such as arrowleaf balsamroot (*Balsamorhiza sagittata*) may be providing adequate cover, especially for very young broods (≤21 days old).

Winter Habitat (form S-6): There are only two closely related indicators of concern for winter habitat (table 19). Identifying all existing potential or likely winter areas is generally more important than describing individual areas. However, evaluating wintering areas during years of above average snowfall can be helpful in identifying critical winter habitats that need protection.

Step 7. Describe fourth-order habitat suitability for the seasonal habitats of interest.

Summarize the seasonal suitability descriptions for the home range area on the “Sage-Grouse Site-Scale Seasonal Habitat Site Suitability Summary” (form S-7, appendix B). Be sure to summarize only those seasonal habitats for which data have been collected during the appropriate season. Further, summarize habitat potential for each area based on the presence of habitat components (e.g., sagebrush and forb recruitment) and ecological site potential. An example for a hypothetical home range area is presented in figure 19 based in part on the field data for the land cover types previously discussed. This summary, with the associated field data, represents a fourth-order habitat description for the home range area. Depict the habitat suitability of the seasonal use areas spatially within the home range on the map created in steps 1 and 2. Copies of completed fourth-order summary descriptions should be provided to the sage-grouse data coordinator for each state.

Table 19. Winter habitat life requisites, indicators, and suitability categories for site-scale habitat descriptions (adapted from Connelly et al. 2000; Sather-Blair et al. 2000).

Life Requisite	Habitat Indicator	Suitability Categories		
		Suitable	Marginal	Unsuitable
Cover and Food	Sagebrush Cover (%)	*≥10*	*5 to <10*	*<5*
	Sagebrush Height (above snow) (cm)	*≥25*	*>10 to <25*	*≤10*

Form S-7: Sage-Grouse Site-Scale Seasonal Habitat Site Suitability Summary			
Date: 6/23/12	**County:** Blaine	**State:** ID	**Evaluator(s):** Janet Hill
Population: Snake, Salmon, Beaverhead		**Home Range Name:** Big Hill	
Associated Leks: RB05; RB02			

Seasonal Habitat Information						Suitability		
						Current	Future	
Seasonal Habitat	Land Cover Type	Ecological Site	Area (ha/ac) (*upland*)	Length (km/mi) (*riparian*)	Number of Sites (#) (*leks, wet meadows, springs, etc.*)	**S**uitable, **M**arginal, **U**nsuitable	Site potential limiting?	Habitat components present?
Lek	Wyoming big sagebrush/ bluebunch wheatgrass				4	S		
Lek	Wyoming and mountain big sagebrush/bluebunch/ wheatgrass/western juniper				2	M	No	Yes
Breeding	Wyoming and big sagebrush/bluebunch/ wheatgrass	Loam 8-12 ARTRW8/ PSSPS	2300 ha			S		
Breeding	Threetip sagebrush/ bluebunch wheatgrass	Loam 8-12 ARTRW8/ PSSPS	1400 ha			M	No	Yes
Breeding	Bluebunch wheatgrass	Loam 8-12 ARTRW8/ PSSPS	5600 ha			U	No	Yes
Breeding	Threetip sagebrush/ crested wheatgrass	Loam 8-12 ARTRW8/ PSSPS	2100 ha			M	No	Yes
Breeding	Crested wheatgrass	Loam 8-12 ARTRW8/ PSSPS	700 ha			U	No	No
Summer	Riparian			10		S		
Summer	Riparian			2		M	No	Yes
Summer	Wet Meadow				4	S		
Summer	Wet Meadow				2	U	No	No
Winter	Not Measured							

Figure 19. An example of a completed seasonal habitat fourth-order suitability summary that includes information from the previous seasonal habitat worksheet examples.

Glossary

Abundance: The total number of organisms in an area (Wisdom et al. 2003; Braun 2005).

Adaptive Management: An approach to natural resource management that involves identifying areas of scientific uncertainty, devising field management activities as real-world experiments to test that uncertainty, learning from the outcome of such experiments, and revising management guidelines on the basis of the knowledge gained (Morrison et al. 1998).

Adult (sage-grouse): A sage-grouse that is greater than 15 months of age and has entered or is about to enter its second breeding season (Connelly et al. 2003).

Alliance (plant): A physiognomically uniform group of plant associations sharing one or more dominant or diagnostic species, which as a rule are found in the uppermost stratum of the vegetation. Dominant species are often emphasized in the absence of detailed floristic information (such as quantitative data), whereas diagnostic species (including characteristic species, dominant differential, and other species groupings based on constancy) are used where detailed floristic data are available (Reid et al. 2002).

Annual (plant): A plant that completes its life cycle and dies in 1 year or less (Pellant et al. 2005).

Anthropogenic Disturbance: The direct loss or fragmentation of habitat due to human development and increased human activity causing the displacement of individuals through avoidance behavior (Holloran 2005).

Anthropogenic Feature: Any human-caused disturbance on the landscape that results in the direct loss or fragmentation of habitat.

Assessment: The process of estimating or judging the functional status of ecosystem structures, functions, or processes within a specified geographic area at a specific time (United States Department of the Interior 2001).

Association (plant): A plant community of definite floristic composition, uniform habitat conditions, and uniform physiognomy. The association level is differentiated from the alliance level by additional plant species, found in any stratum, which indicate finer scale environmental patterns and disturbance regimes (Reid et al. 2002).

Breeding Habitat: Leks and the sagebrush habitat surrounding leks that are collectively used for prelaying, breeding, nesting, and early brood-rearing activities from approximately March through June (Connelly et al. 2000; Connelly et al. 2003).

Brood (sage-grouse): A hen or group of hens with at least one chick.

Canopy Cover: The percentage of the ground (1) included in a vertical projection of imaginary polygons drawn about the total natural spread of foliage of the individuals of a species (usually used for herbaceous plants), or (2) covered by a projection of the crown, stems, and leaves of the plant onto the ground surface (usually used for shrubs).

Chick (sage-grouse): A sage-grouse up to 10 weeks of age (Connelly et al. 2003).

Community: A set of two or more interacting species, such as members of a trophic web, that live in a particular habitat (Meffe and Carroll 1997).

Condition (vegetation): The ability of a community or ecosystem to function naturally (Wisdom et al. 2005).

Connectivity: The degree to which habitats for a species are continuous or interrupted across a spatial extent. Habitats defined as continuous are within a prescribed distance over which a species can successfully conduct key activities (e.g., effective dispersal distances of seeds or juveniles; mean distances moved for foraging, nesting, and brood-rearing). Habitats defined as interrupted are outside the prescribed distance (Wisdom et al. 2003).

Cover: An indication of the relative amount of shelter or protection provided by all vegetation at a given point; it is normally used to assess nesting habitat (Connelly et al. 2003).

Cover Type: A vegetation classification depicting genera, species, group of species, or life forms of trees, shrubs, grasses, or sedges or a dominant physical feature (e.g., water or rock) or land use (e.g., urban or road) of an area. When a genus or species name is given to the cover type at a broad-scale, it is typically representative of a complex of species or genera with similar characteristics (Wisdom et al. 2000).

Daubenmire Frame: A rectangular frame, 20 x 50 cm, used to estimate canopy cover. The frame has a painted pattern that provides visual reference areas equal to 5, 25, 50, 75, and 95 percent of the plot area (Daubenmire 1959).

Dispersal: Movement of individuals to new living areas, including initial movements from place of birth to first attempted breeding area (natal dispersal) and subsequent movements from one breeding location to another (adult dispersal) (Elphick et al. 2001).

Distribution: The spread or scatter of an organism within its range (Morrison and Hall 2001).

Disturbance: Any relatively discrete event in time that disrupts ecosystem, community, or population structure, and changes resources, substrate availability, or the physical environment (White and Pickett 1985). *See also Anthropogenic Disturbance.*

Droop Height: The height of a grass or forb measured from the ground to the point where the plant naturally bends (maximum natural height). There may be no droop to some plants with relatively short stature (Connelly et al. 2003).

Early Brood-Rearing Habitat: Upland sagebrush sites relatively close to nest sites, typically characterized by high species richness with an abundance of forbs and insects, where sage-grouse hens raise young chicks (<21 days old) (Connelly et al. 2000).

Ecological Reference Area (ERA): Land in which ecological processes are functioning within a normal range of variability and the plant community has adequate resistance to and resilience against most disturbances. This area best represents the potential of a site in both physical function and biological health (Herrick et al. 2005).

Ecological Site: An area of land with a specific potential plant community and specific physical site characteristics, differing from other areas of land in its ability to produce vegetation and to respond to management (United States Department of the Interior 1996).

Ecological Site Description: A description of the soils, uses, and potential of a kind of land with specific physical characteristics to produce distinctive kinds and amounts of vegetation (Pellant et al. 2005).

Ecological Site Potential: The plant community that can be supported in an area given its edaphic and climatic potential (Habich 2001).

Ecosystem: The totality of components of all kinds that make up a particular environment; the complex of a biotic community and its abiotic, physical environment (Wisdom et al. 2005).

Edge: The intersection of two vegetation types (Morrison et al. 1998).

Edge Effect: The influence of a habitat edge on interior conditions of a habitat or on species that use interior habitat (Meffe and Carroll 1997).

Encroachment: Advancement beyond the usual or proper limits; often used to describe the advancement of pinyon pine or juniper woodlands into sagebrush communities (Wisdom et al. 2005).

Erosion: Detachment and movement of soil or rock fragments by water, wind, ice, or gravity (Habich 2001).

Exotic: Not native; an organism or species that has been introduced into an area and is thus outside of its native range (Wisdom et al. 2005).

Extent: (1) [*general*] The area over which observations are made (e.g., study area, species range); (2) [*spatial*] The geographic limits of a geographic dataset specified by the minimum bounding area (Wisdom et al. 2005).

Extirpation: The loss or removal of a species from one or more specific areas but not from all areas (Wisdom et al. 2005).

Fall Habitat: The matrix of sagebrush habitat areas that sage-grouse slowly move through from September through November, transitioning from summer habitat to winter habitat and shifting their diet from large amounts of forbs to exclusively sagebrush (Connelly et al. 2000).

Foliar Cover: The percentage of ground covered by the vertical projection of the aerial portion of plants. Small openings in the canopy and intraspecific overlap are excluded.

Forb: An herbaceous plant other than a grass, sedge, or rush, that has little or no woody material (United States Department of the Interior 1996).

Fragmentation: The process by which a species habitat is reduced and fragmented into pieces separated by areas of unsuitable habitat or nonhabitat. Habitat fragmentation <u>has not</u> occurred when habitat has been separated by unsuitable habitat but occupancy, reproduction, or survival of the species <u>has not</u> been affected (Franklin et al. 2002).

Geographic Information System (GIS): A collection of computer hardware, software, and geographic data for capturing, managing, analyzing, and displaying all forms of geographically referenced information (ESRI 2006).

Grain: (1) [*general*] The smallest resolvable unit of study (e.g., 1- x 1-m quadrant), which generally determines the lower limit of what can be studied (Morrison and Hall 2001); (2) [*spatial*] The mapping resolution at which spatial patterns are measured (Wisdom et al. 2000).

Grass: Any plant of the family Poaceae (United States Department of the Interior 1996).

Grassland: Vegetation dominated by grasses and grasslike plants, including sedges and rushes (Reid et al. 2002).

Habitat: An area with a combination of resources (such as space, food, cover, and water) and environmental conditions (such as temperature, precipitation, presence or absence of predators and competitors) that promotes occupancy by individuals of a given species and allows those individuals to survive and reproduce (Morrison et al. 1998).

Habitat Indicator: A component or attribute of habitat that can be observed and or measured to characterize suitability for space, food, cover, and water.

Habitat Patch: A species habitat unit, appropriate for the scale of interest, surrounded by unsuitable habitat (adapted from Franklin et al. 2002).

Habitat Quality: A measure of two components: (1) habitat use (selection) by animals, and (2) fitness consequences associated with that habitat (Van Horne 1983; Aldridge 2005; Aldridge and Boyce 2007).

Habitat Selection: The process by which an animal chooses its habitat or habitat components (Johnson 1980). The orders of selection are as follows:

First-Order Selection: Selection of the physical or geographic range of a species.

Second-Order Selection: Selection of the physical or geographic home range for a subpopulation (e.g., for a sage-grouse lek or lek group).

Third-Order Selection: Selection of seasonal habitats (cover types) within a home range (e.g., sage-grouse seasonal habitat areas).

Fourth-Order Selection: Selection of habitat components (food items and shelter provisions for feeding, nesting, and roosting areas) within a seasonal use area.

Habitat Suitability: The relative appropriateness of a certain ecological area for meeting the life requirements of an organism (i.e., space, food, cover, and water). Categories of habitat suitability include:

Suitable Habitat: An area that provides environmental conditions necessary for successful survival and reproduction to sustain stable populations (Cooperrider et al. 1986; Morrison et al. 1998).

Marginal Habitat: An area that supports the species but has generally lower survival rates and reproductive success by comparison and may or may not have the potential to become suitable in the future (Cooperrider et al. 1986).

Potential Habitat: An area that is currently unoccupied but has the potential for occupancy in the foreseeable future (<100 years) through succession or restoration.

Unsuitable Habitat: An area that does not currently provide one or more of the life requisites and therefore does not provide habitat, but it may provide habitat sometime in the foreseeable future (<100 years) through succession or restoration.

Nonhabitat: An area within the historical distribution of sage-grouse that is unoccupied, does not currently provide habitat, and does not have the potential to provide habitat in the foreseeable future (<100 years).

Herbaceous (vegetation): Plants that die back to the ground each year, normally with soft, nonwoody stems (Connelly et al. 2003).

Home Range: The area traversed by an animal during its activities during a specified period of time (Morrison and Hall 2001).

Indicator: *See Habitat Indicator*.

Invasive (plant): A plant species that is not part of, or is a minor component of, a predisturbance plant community and that has the potential to become a dominant or codominant species on the site if its future establishment and growth is not actively controlled by management interventions (Pellant et al. 2005).

Inventory: A point-in-time measurement of a resource to determine its location or condition (Elzinga et al. 1998).

Land Cover Type: A classification of the observed biophysical cover on the surface of the earth (Wisdom et al. 2005).

Landscape: A mosaic of landforms, vegetation, and land uses; a heterogeneous land area that is often hierarchically structured and varies in extent with the organism(s) being studied and the purpose for defining a landscape (Urban et al. 1987; Liu and Taylor 2002).

Landscape Matrix: A broad-scale pattern of varied vegetation classes and land uses throughout a region (Urban et al. 1987; Crow 2002).

Late Brood-Rearing Habitat: A variety of habitats used by sage-grouse from July through September, including, but not limited to, wet meadows, farmland, riparian areas, dry lakebeds, and sagebrush areas (Connelly et al. 2000).

Lek: Open area surrounded by sagebrush, without trees or other tall structures in close proximity, where males traditionally display and breeding occurs (Connelly et al. 2000). Categories of leks are as follows:

> **Occupied lek:** (1) [*Greater Sage-Grouse*] A lek that has been active during at least one breeding season within the prior 5 years; (2) [*Gunnison Sage-Grouse*] A lek that has been attended by males in the previous 5 years. Note: The specific terms and definitions for lek status may vary by state. Use the terminology appropriate for your area.
>
> **Unoccupied lek:** (1) [*Greater Sage-Grouse*] A lek that has not been active during a period of 5 consecutive years; (2) [*Gunnison Sage-Grouse*] A lek that has been inactive for 5 years. Note: The specific terms and definitions for lek status may vary by state. Use the terminology appropriate for your area.
>
> **Undetermined lek:** Any lek that has not been documented as active in the last 5 years, but for which survey information is insufficient to designate the lek as unoccupied. Note: The specific terms and definitions for lek status may vary by state. Use the terminology appropriate for your area.

Lek Group: A group of leks with 5-km overlapping or contiguous buffers (Moynahan et al. 2007).

Life Form (plant): Characteristic form or appearance of a species at maturity, such as a grass, forb, tree, or shrub (Habich 2001).

Life Requisite: An item an animal needs to survive, including food, shelter or cover, water (Morrison et al. 1998), and space.

Line Intercept—Daubenmire Frame (LIDF): Two techniques for measuring canopy cover that involves placing a measuring tape between two points and measuring the amount of plant (crown, stems, leaves) that intersects a vertical projection of this line (Canfield 1941). The line intercept technique is used for measuring shrub cover and the Daubenmire frame technique is used for measuring herbaceous cover. *See Daubenmire Frame*.

Line Point Intercept (LPI): A rapid, accurate method for quantifying soil cover, including vegetation, litter, rocks, and biotic crusts (Herrick et al. 2005). The methodology uses a measuring tape, two pins for anchoring the tape, and a straight, small-diameter rod to determine plant cover and composition.

Linkage Area: A land cover type, other than occupied sagebrush shrubland, that sage-grouse frequently use and may move through to another habitat patch. If made into suitable habitat, this area will increase movement between populations and decrease the probability of extinction of the species by stabilizing population dynamics (Gunnison Sage-Grouse Rangewide Steering Committee 2005).

Marginal Habitat: *See Habitat Suitability*.

Monitoring: The collection and analysis of repeated observations or measurements to evaluate changes in condition and progress toward meeting a management objective (Elzinga et al. 1998).

Native (plant): Indigenous to a given place (Wisdom et al. 2005).

Nesting Habitat: Area with protective grass and high lateral shrub cover where hens nest, typically under sagebrush shrubs (Connelly et al. 2000).

Nonhabitat: *See Habitat Suitability*.

Noxious Weed: An unwanted plant specified by federal or state laws as being especially undesirable, troublesome, and difficult to control. It grows and spreads in places where it interferes with the growth and production of desired species (Habich 2001).

Occupied Habitat (sage-grouse): All sagebrush and associated plant communities known to be used by sage-grouse within the last 10 years. Sagebrush areas that are contiguous with areas of known use and that do not have effective barriers to sage-grouse movement from those areas are considered occupied unless specific information exists that documents the lack of sage-grouse use.

Overstory: The upper canopy or canopies of plants, usually referring to trees, shrubs, and vines (United States Department of the Interior 1996).

Patch: *See Habitat Patch*.

Perennial (plant): A plant that has a lifespan of 3 or more years (Pellant et al. 2005).

Population: A collection of organisms of the same species that freely share genetic material (i.e., breed) (Morrison et al. 1998; Braun 2005). *See also Subpopulation.*

Potential Habitat: *See Habitat Suitability*.

Precision: The closeness of repeated measurements of the same quantity (Elzinga et al. 1998; Braun 2005).

Proper Functioning Condition (PFC) Assessment: A consistent approach for considering hydrology, vegetation, and erosion/deposition (soils) attributes and processes to assess the condition of riparian-wetland areas (Prichard et al. 2003). Function ratings are as follows:

> **Proper Functioning Condition (PFC):** A riparian-wetland area in which adequate vegetation or other structure components are present to dissipate energy, reduce erosion and improve water quality, filter sediment and aid in floodplain development, improve flood-water retention and ground-water recharge, stabilize streambanks and shorelines, develop diverse ponding and channel characteristics for fish and wildlife habitat among other things, and support greater biodiversity.
>
> **Functional—At Risk (FAR):** A riparian-wetland area that is in functional condition but has at least one attribute or process that makes it susceptible to degradation.
>
> **Nonfunctioning (NF):** A riparian-wetland area that clearly does not provide adequate vegetation, landform, or large woody debris to dissipate energies associated with high flow and thus does not reduce erosion, improve water quality, etc. (Prichard et al. 2003).

Quantitative: Data derived from measurements, such as counts, dimensions, weights, etc., and recorded numerically. Qualitative numerical estimates, such as ocular cover and production estimates, are often referred to as "semiquantitative" (Pellant et al. 2005).

Range: The limits within which an organism lives or can be found (Morrison and Hall 2001).

Range Site: *See Ecological Site*.

Recruitment: The addition of new individuals (typically only breeding individuals) to a population through reproduction (Dinsmore and Johnson 2005).

Reference Period: A period of time during which data were collected at an area that can be chosen to provide a basis or standard for evaluation or comparison of trend over time. *See also Ecological Reference Area*.

Restoration: The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. An ecosystem is recovered or restored when it contains sufficient biotic and abiotic resources to continue its development without further assistance or subsidy (Society for Ecological Restoration International 2004).

Riparian (habitat): An area that is saturated or inundated at a frequency and duration sufficient to produce vegetation typically adapted for life in saturated soil conditions (Prichard et al. 2003).

Risk: The potential or probability of an adverse event (Wisdom et al. 2005).

Road: A linear route declared a road by the owner, managed for use by low-clearance vehicles having four or more wheels, and maintained for regular and continuous use (United States Department of the Interior 2006).

Sagebrush Ecosystem: Arid and semiarid, sagebrush-dominated lands in the western United States and Canada that encompass the approximate boundaries of the historical range of Greater and Gunnison Sage-Grouse (Wisdom et al. 2005).

Scale: The resolution at which patterns are measured, perceived, or represented. Scale can be broken into several components, including grain and extent (Morrison and Hall 2001). For sage-grouse, scales are as follows:

> **Broad Scale:** Entire species range and populations (first-order habitat selection).
>
> **Mid Scale:** Subpopulations (second-order habitat selection).
>
> **Fine Scale:** Seasonal use areas (third-order habitat selection).
>
> **Site Scale:** Seasonal foraging and shelter habitat (fourth-order habitat selection).

Selection: *See Habitat Selection*.

Shrub: A plant that has persistent woody stems and a relatively low growth habit (less than 5 meters tall) and that generally produces several basal shoots instead of a single bole (Pellant et al. 2005).

Shrubland: Vegetation dominated by shrubs that are generally greater than 0.5 m tall and less than 5 m tall and that generally form greater than 25 percent cover, with trees forming less than 25 percent cover (Reid et al. 2002).

Shrub Steppe: Habitats characterized in western North America by woody, midheight shrubs and perennial bunchgrasses; typically arid, with annual precipitation averaging <36 cm (14 in) over much of the region (Wisdom et al. 2003).

Sink Habitat: Habitat in which local mortality exceeds reproductive success and, therefore, the number of individuals occupying the habitat is declining (Meffe and Carroll 1997).

Site: An area of uniform physical and biological properties and management status (Morrison and Hall 2001).

Site Suitability: The suitability of a specific land cover type or other sampling unit in a seasonal use area based on field data collection.

Source Habitat: Habitat in which local reproductive success exceeds local mortality, thus producing an excess of individuals to emigrate to other areas (Meffe and Carroll 1997).

Species: Groups of populations that can potentially interbreed or are actually interbreeding and can successfully produce viable, fertile offspring (Mayr 1969).

Species Composition (plant): The proportions of various plant species in relation to the total in a given area; it may be expressed in terms of relative cover, density, or weight (Habich 2001).

Subpopulation: A portion of a population in a specific geographic location (Morrison et al. 1998). *See also Population*.

Succession (plant): An orderly and predictable process in which vegetation change represents the life history of a plant community, developing to a distinct climax condition (Morrison et al. 1998).

Succulent: Juicy, watery, or pulpy, as the moist stems of cacti (Habich 2001).

Suitable Habitat: *See Habitat Suitability*.

Summer Habitat: The summer or late brood-rearing period from July through August, when hens and chicks use a variety of moist and mesic habitats where succulent forbs and insects are found in close proximity to sagebrush (Connelly et al. 2000).

Trend: The direction of change in ecological status or resource value rating observed over time (Herrick et al. 2005).

Understory: Plants growing beneath the canopy of other plants; usually refers to grasses, forbs, and low shrubs under a tree or shrub canopy (United States Department of the Interior 1996).

Unsuitable Habitat: *See Habitat Suitability*.

Upland (habitat): An area that is not inundated with water and typically supports vegetation types adapted to life in nonsaturated soil conditions (Prichard et al. 2003).

Watershed: A group of streams that flow into a subbasin (Wisdom et al. 2000).

Wet Meadow: A meadow where the surface remains wet or moist throughout the summer, usually characterized by sedges and rushes (United States Department of the Interior 1996).

Winter Habitat: Sagebrush habitats that provide access to sagebrush above the snow for all food and cover requisite needs (Connelly et al. 2000).

Woodland: Vegetation dominated by open stands of trees with crowns not usually touching (generally forming 25-60 percent cover); canopy tree cover may be less than 25 percent in cases where it exceeds shrub, dwarf-shrub, herb, and nonvascular cover, respectively (Reid et al. 2002).

Literature Cited

Aldridge, C.L. 2005. Identifying habitats for persistence of Greater Sage-Grouse (*Centrocercus urophasianus*) in Alberta, Canada. Ph.D. dissertation, University of Alberta, Edmonton, Alberta, Canada.

Aldridge, C.L. and M.S. Boyce. 2007. Linking occurrence and fitness to persistence: Habitat-based approach for endangered Greater Sage-Grouse. Ecological Applications 17:508-526.

Aldridge, C.L., S.E. Nielsen, H.L. Beyer, M.S. Boyce, J.W. Connelly, S.T. Knick and M.A. Schroeder. 2008. Range-wide patterns of Greater Sage-Grouse persistence. Diversity and Distributions 17:983-994.

Balch, J.K., B.A. Bradley, C.M. D'Antonio, and J. Gomez-Dans. 2013. Introduced annual grass increases regional fire activity across the arid western USA (1980-2009). Global Change Biology 19:173-183.

Baruch-Mordo, S., J.S. Evans, J.P. Severson, D.E. Naugle, J.D. Maestas, J.M. Kiesecker, M.J. Falkowski, C.A. Hagen, and K.P. Reese. 2013. Saving sage-grouse from the trees: A proactive solution to reducing a key threat to a candidate species. Biological Conservation 167:233-241.

Beck, J.L., K.P. Reese, J.W. Connelly, and M.B. Lucia. 2006. Movements and survival of juvenile Greater Sage-Grouse in southeastern Idaho. Wildlife Society Bulletin 34: 1070-1078.

Berry, J.D. and R.L. Eng. 1985. Interseasonal movements and fidelity to seasonal use areas by female sage grouse. Journal of Wildlife Management 49:237-240.

Blus, L.J., C.S. Staley, C.J. Henny, G.W. Pendleton, T.H. Craig, E.H. Craig, and D.K. Halford. 1989. Effects of organophosphorus insecticides on sage grouse in southeastern Idaho. Journal of Wildlife Management 53:1139-1146.

Bonham, C.D. 2013. Measurements for terrestrial vegetation. Second edition. Wiley-Blackwell, Chichester, West Sussex, Hoboken, NJ.

Bonham, C.D., D.E. Mergen, and S. Montoya. 2004. Plant cover estimation: A contiguous Daubenmire frame. Rangelands 26:17-22.

Bottrill, M.C., L.N. Joseph, J. Carwardine, M. Bode, C. Cook, E.T. Game, H. Grantham, S. Kark, S. Linke, E. McDonald-Madden, R.L. Pressey, S. Walker, K.A. Wilson, and H.P. Possingham. 2008. Is conservation triage just smart decision making? Trends in Ecology and Evolution 23:649-654.

Braun, C.E., ed. 2005. Techniques for wildlife investigations and management. Sixth edition. The Wildlife Society, Bethesda, MD.

Canfield, R.H. 1941. Application of the line-intercept method in sampling range vegetation. Journal of Forestry 39:388-394.

Chambers, J.C., B.A. Bradley, C.S. Brown, C. D'Antonio, M.J. Germino, J.B. Grace, S.P. Hardegree, R.F. Miller, and D.A. Pyke. 2013. Resilience to stress and disturbance to *Bromus tectorum* L. invasion in cold desert shrublands of western North America. Ecosystems 17:360-375.

Connelly, J.W., H.W. Browers, and R.J. Gates. 1988. Seasonal movements of sage grouse in southeastern Idaho. Journal of Wildlife Management 52:116-122.

Connelly, J.W., S.T. Knick, M.A. Schroeder, and S.J. Stiver. 2004. Conservation assessment of Greater Sage-Grouse and sagebrush habitats. Unpublished report. Western Association of Fish and Wildlife Agencies, Cheyenne, WY.

Connelly, J.W., K.R. Reese, and M.A. Schroeder. 2003. Monitoring of Greater Sage-Grouse habitats and populations. Station Bulletin 80. College of Natural Resources Experiment Station, University of Idaho, Moscow, ID.

Connelly, J.W., E.T. Rinkes, and C.E. Braun. 2011. Characteristics of Greater Sage-Grouse habitats: A landscape species at micro- and macroscales. *In* Greater Sage-Grouse: Ecology and conservation of a landscape species and its habitats, edited by S.T. Knick and J.W. Connelly, 69–83. Studies in Avian Biology, vol. 38. University of California Press, Berkeley, CA.

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Cooperrider, A.Y., R.J. Boyd, and H.R. Stuart, editors. 1986. Inventory and monitoring of wildlife habitat. U.S. Department of the Interior, Bureau of Land Management, Denver, CO.

Copeland, H.E., A. Pocewicz, D.E. Naugle, T. Griffiths, D. Keinath, J. Evans, and J. Platt. 2013. Measuring the effectiveness of conservation: A novel framework to quantify the benefits of sage-grouse conservation policy and easements in Wyoming. PLoS ONE 8(6):e67261.

Cowardin, L.M., V. Carter, F.C. Golet, and E.T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service. Washington, DC.

Cronquist, A., A.H. Holmgren, N.H. Holmgren, and J.L. Reveal. 1972. Intermountain flora: Vascular plants of the Intermountain West, U.S.A. Volume 1. Hafner Publishing Company, New York, NY.

Crow, T.R. 2002. Putting multiple use and sustained yield into a landscape context. *In* Integrating landscape ecology into natural resource management, edited by J. Liu and W.W. Taylor, 349-365. Cambridge University Press, Cambridge, United Kingdom.

D'Antonio, C.M. and P.M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Review of Ecology and Systematics 23:63-87.

Daubenmire, R.F. 1959. A canopy-coverage method of vegetation analysis. Northwest Science 33:43-64.

Davies, K.W., C.S. Boyd, J.L. Beck, J.D. Bates, T.J. Svejcar, and M.A. Gregg. 2011. Saving the sagebrush sea: An ecosystem conservation plan for big sagebrush plant communities. Biological Conservation 144:2573-2584.

Dinsmore, S.J. and D.H. Johnson. 2005. Population analysis in wildlife biology. *In* Techniques for wildlife investigations and management, edited by C.E. Braun, 154-184. Sixth edition. The Wildlife Society, Bethesda, MD.

Doherty, K.E., J.D. Tack, J.S. Evans, and D.E. Naugle. 2010. Mapping breeding densities of Greater Sage-Grouse: A tool for range-wide conservation planning. Completion Report, Interagency Agreement #L10PG00911. Bureau of Land Management, Washington, DC.

Elphick, C., J.B. Dunning, Jr., and D.A. Sibley, eds. 2001. The Sibley guide to bird life and behavior. The National Audubon Society. Alfred A. Knopf, New York, NY.

Elzinga, C.L., D.W. Salzer, and J.W. Willoughby. 1998. Measuring and monitoring plant populations. Technical Reference 1730-1. U.S. Department of the Interior, Bureau of Land Management, Denver, CO.

Esri. What is GIS? www.esri.com/what-is-gis/. Accessed May 9, 2006.

Fedy, B.C., C.L. Aldridge, K.E. Doherty, M. O'Donnell, J.L. Beck, B. Bedrosian, M.J. Holloran, J.D. Johnson, N.W. Kaczor, C.P. Kirol, C.A. Mandich, D. Marshall, G. McKee, C. Olson, C.C. Swanson, and B.L. Walker. 2012. Interseasonal movement of Greater Sage-Grouse, migratory behavior, and an assessment of the core regions concept in Wyoming. Journal of Wildlife Management 76:1062-1071.

Floyd, D.A. and J.E. Anderson. 1987. A comparison of three methods for estimating plant cover. Journal of Ecology 75:221-228.

Franklin, A.B., B.R. Noon, and T.L. George. 2002. What is habitat fragmentation? Studies in Avian Biology 25:20-29.

Gilbert, F.F. and D.G. Dodds. 1987. The philosophy and practice of wildlife management. Robert E. Krieger Publishing, Malabar, FL.

Gunnison Sage-Grouse Rangewide Steering Committee. 2005. Gunnison Sage-Grouse rangewide conservation plan. Colorado Division of Wildlife, Denver, CO.

Habich, E.F. 2001. Ecological site inventory. Technical Reference 1734-7. U.S. Department of the Interior, Bureau of Land Management, Denver, CO.

Hagen, C.A., J.W. Connelly, and M.A. Schroeder. 2007. A meta-analysis of Greater Sage-Grouse *Centrocercus urophasianus* nesting and brood-rearing habitats. Wildlife Biology 13 (Suppl. 1):42-50.

Hann, W.J., J.L. Jones, M.G. Karl, P.F. Hessburg, R.E. Keane, D.G. Long, J.P. Menakis, C.H. McNicoll, S.G. Leonard, R.A. Gravenmier, and B.G. Smith. 1997. Landscape dynamics of the Basin. In An assessment of ecosystem components in the interior Columbia Basin and portions of the Klamath and Great Basins, edited by T.M. Quigley and S.J. Arbelbide, 337-1055. Volume 2. U.S. Forest Service, Pacific Northwest Research Station, Portland, OR.

Hanser, S.E. and S.T. Knick. 2011. Greater Sage-Grouse as an umbrella species for shrubland passerine birds: a multiscale assessment. *In* Greater Sage-Grouse: Ecology and conservation of a landscape species and its habitats, edited by S.T. Knick and J.W. Connelly, 475-488. Studies in Avian Biology, vol. 38. University of California Press, Berkeley, CA.

Herrick, J.E., J.W. Van Zee, K.M. Havstad, L.M. Burkett, and W.G. Whitford. 2005. Monitoring manual for grassland, shrubland and savanna ecosystems. Volumes I and II. United States Department of Agriculture, Agricultural Research Service, Jornada Experimental Range, Las Cruces, NM.

Holloran, M.J. 2005. Greater Sage-Grouse (*Centrocercus urophasianus*) population response to natural gas field development in western Wyoming. Ph.D. dissertation, University of Wyoming, Laramie, WY.

Johnson, D.H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology 61:65-71.

Johnson, D.H., M.J. Holloran, J.W. Connelly, S.E. Hanser, C.L. Amundson, and S.T. Knick. 2011. Influences of environmental and anthropogenic features on Greater Sage-Grouse populations, 1997–2007. *In* Greater Sage-Grouse: Ecology and conservation of a landscape species and its habitats, edited by S.T. Knick and J.W. Connelly, 407-450.

Studies in Avian Biology, vol. 38. University of California Press, Berkeley, CA.

Knick, S.R and J.W. Connelly, editors. 2011. Greater Sage-Grouse: Ecology and conservation of a landscape species and its habitats. Cooper Ornithological Society Studies in Avian Biology, vol. 38. University of California Press, Berkeley, CA.

Knick, S.T. and S.E. Hanser. 2011. Connecting pattern and process in Greater Sage-Grouse populations and sagebrush landscapes. *In* Greater Sage-Grouse: Ecology and conservation of a landscape species and its habitats, edited by S.T. Knick and J.W. Connelly, 383-406. Studies in Avian Biology, vol. 38. University of California Press, Berkeley, CA.

Knick, S.T., S.E. Hanser, R.F. Miller, D.A. Pyke, M.J. Wisdom, S.P. Finn, E.T. Rinkes, and C.J. Henny. 2011. Ecological influence and pathways of land use in sagebrush. *In* Greater Sage-Grouse: Ecology and conservation of a landscape species and its habitats, edited by S.T. Knick and J.W. Connelly, 203-251. Studies in Avian Biology, vol. 38. University of California Press, Berkeley, CA.

Knick, S.T., S.E. Hanser, and K.L. Preston. 2013. Modeling ecological minimum requirements for distribution of Greater Sage-Grouse leks: Implications for population connectivity across their western range, U.S.A. Ecology and Evolution 3:1539-1551.

Kuchler, A.W. 1970. Potential natural vegetation. *In* The national atlas of the United States of America, 89-92. U.S. Department of the Interior, Geological Survey, Washington, DC.

Leonard, K.M., K.P. Reese, and J.W. Connelly. 2000. Distribution, movements and habitats of sage grouse *Centrocercus urophasianus* on the Upper Snake River Plain of Idaho: Changes from the 1950s to the 1990s. Wildlife Biology 6:265-270.

Liu, J. and W.W. Taylor, eds. 2002. Integrating landscape ecology into natural resource management. Cambridge University Press, Cambridge, United Kingdom.

Lyon, A.G. 2000. The potential effects of natural gas development on sage grouse near Pinedale, Wyoming. Master's thesis, University of Wyoming, Laramie, WY.

Lyon, A.G. and S.H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin 31:486-491.

MacKinnon, W.C., J.W. Karl, G.R. Toevs, J.J. Taylor, M. Karl, C.S. Spurrier, and J.E. Herrick. 2011. BLM core terrestrial indicators and methods. Technical Note 440. U.S. Department of the Interior, Bureau of Land Management, National Operations Center, Denver, CO.

Mayr, E. 1969. Principles of systematic zoology. McGraw-Hill, New York, NY.

Meffe, G.K. and C.R. Carroll. 1997. Principles of conservation biology. Sinauer Associates, Inc., Sunderland, MA.

Miller, R.F. and L.L. Eddleman. 2001. Spatial and temporal changes of sage grouse habitat in the sagebrush biome. Technical Bulletin 151. Oregon State University, Corvallis, OR.

Miller, R.F., S.T. Knick, D.A. Pyke, C.W. Meinke, S.E. Hanser, M.J. Wisdom, and A.L Hild. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. *In* Greater Sage-Grouse: Ecology and conservation of a landscape species and its habitats, edited by S.T. Knick and J.W. Connelly, 145-184. Studies in Avian Biology, vol. 38. University of California Press, Berkeley, CA.

Mills, L.S. 2013. Focal species to bridge from populations to ecosystems. *In* Conservation of wildlife populations, 244-250. Second edition. Wiley-Blackwell Publishing, Oxford, United Kingdom.

Mooney, H.A. and E.E. Cleland. 2001. The evolutionary impact of invasive species. Proceedings of the National Academy of Sciences 98:5446–5451.

Morrison, M.L. and L.S. Hall. 2001. Standard terminology: Toward a common language to advance ecological understanding and application. *In* Predicting species occurrences: Issues of accuracy and scale, edited by J.M. Scott, P.J. Heglund, M.L. Morrison, J.B. Haufler, M.G. Raphael, W.A. Wall, and F.B. Samson, 43-52. Island Press, Washington, DC.

Morrison, M.L., B.M. Marcot, and R.W. Mannan. 1998. Wildlife—habitat relationships: Concepts and applications. University of Wisconsin Press, Madison, WI.

Moynahan, B.J., M.S. Lindberg, J.J. Rotella, and J.W. Thomas. 2007. Factors affecting nest survival of Greater Sage-Grouse in north central Montana. Journal of Wildlife Management 71:1773-1783.

Nielson, R.P., L.F. Pitelka, A.M. Solomon, R. Nathan, G.F. Midgley, J.M.V. Fragoso, H. Lischke, and K. Thompson. 2005. Forecasting regional to global plant migration in response to climate change. BioScience 55:749-760.

Oyler-McCance, S.J. and T.W. Quinn. 2011. Molecular insights into the biology of Greater Sage-Grouse. *In* Greater Sage-Grouse: Ecology and conservation of a landscape species and its habitats, edited by S.T. Knick and J.W. Connelly, 85-94. Studies in Avian Biology, vol. 38. University of California Press, Berkeley, CA.

Pellant, M., P. Shaver, D.A. Pyke, and J.E. Herrick. 2005. Interpreting indicators of rangeland health. Version 4. Technical Reference 1734-6. U.S. Department of the Interior, Bureau of Land Management, Denver, CO.

Prichard, D., J. Anderson, C. Correll, J. Fogg, K. Gebhardt, R. Krapf, S. Leonard, B. Mitchell, and J. Staats. 1998. Riparian area management: A user guide to assessing proper functioning condition and the supporting science for lotic areas. Technical Reference 1737-15. U.S. Department of the Interior, Bureau of Land Management, National Applied Research Science Center, Denver, CO.

Prichard, D., F. Berg, W. Hagenbuck, R. Krapf, R. Leinard, S. Leonard, M. Manning, C. Noble, and J. Staats. 2003. Riparian area management: A user guide to assessing proper functioning condition and the supporting science for lentic areas. Technical Reference 1737-16. U.S. Department of the Interior, Bureau of Land Management, National Applied Research Science Center, Denver, CO.

Reid, M., P. Comer, H. Barrett, S. Caicco, R. Crawford, C. Jean, G. Jones, J. Kagan, M. Karl, G. Kittel, P. Lyon, M. Manning, E. Peterson, R. Rosentreter, S. Rust, D. Tart, C. Williams, and A. Winward. 2002. International classification of ecological communities: Terrestrial vegetation of the United States. Sagebrush vegetation of the Western United States. Final report for the United States Geological Survey, Forest and Rangeland Ecosystem Science Center, Boise, Idaho. NatureServe, Arlington, VA.

Ries, L.R., R.J. Fletcher, J. Battin, and T.D. Sisk. 2004. Ecological responses to habitat edges: Mechanisms, models, and variability explained. Annual Review of Ecology, Evolution, and Systematics 35:491-522.

Rowland, M.M., M. Leu, S. Hanser, S.P. Finn, C.L. Aldridge, S.T. Knick, L.H. Suring, J.M. Boyd, M.J. Wisdom, and C.W. Meinke. 2006a. Assessment of threats to sagebrush habitats and associated species of concern in the Wyoming Basins. Version 2.0. Unpublished report on file at U.S. Geological Survey, Biological Resources, Snake River Field Station, 970 Lusk St., Boise, ID 83706.

Rowland, M.M., L.H. Suring, and M.J. Wisdom. 2010. Assessment of habitat threats to shrublands in the Great Basin: A case study. *In* Advances in threat assessment and their application to forest and rangeland management, edited by J.M. Pye, H.M. Rauscher, Y. Sands, D.C. Lee, and J.S. Beatty, 673-685. PNW-GTR-802. U.S. Department of Agriculture, Forest Service, Portland, OR.

Rowland, M.M., M.J. Wisdom, L.H. Suring, and C.W. Meinke. 2006b. Greater Sage-Grouse as an umbrella species for sagebrush-associated vertebrates. Biological Conservation 129:323-335.

Sage Grouse Initiative. 2014. Wildfire and cheatgrass: New science helps reduce threats to sage grouse. Science to Solutions Series, Number 5. www.sagegrouseinitiative.com/wp-content/uploads/2014/11/S2S-WildfireCheatgrass-Nov5-2014.pdf.

Sather-Blair, S., P. Makela, T. Carrigan, and L. Anderson. 2000. A framework to assist in making sensitive species-habitat assessment for BLM-administered public lands in Idaho: Sage-grouse. Unpublished report. Bureau of Land Management, Idaho State Office, Branch of Resources and Science.

Santini, K. 2012. Vegetation treatment effects in a Wyoming big sagebrush community and comparison of two methods for estimating shrub canopy cover. Master's thesis, University of Idaho, Moscow, ID.

Schroeder, M.A., C.L. Aldridge, A.D. Apa, J.R. Bohne, C.E. Braun, S.D. Bunnell, J.W. Connelly, P.A. Deibert, S.C. Gardner, M.A. Hilliard, G.D. Kobriger, S.M. McAdam, C.W. McCarthy, J.J. McCarthy, D.L. Mitchell, E.V. Rickerson, and S.J. Stiver. 2004. Distribution of sage-grouse in North America. Condor 106:363-376.

Schroeder, M.A. and L.A. Robb. 2003. Fidelity of Greater Sage-Grouse *Centrocercus urophasianus* to breeding areas in a fragmented landscape. Wildlife Biology 9:291-299.

Schroeder, M.A., J.R. Young, and C.E. Braun. 1999. Sage grouse (*Centrocercus urophasianus*). *In* The birds of North America, edited by A. Poole and F. Gill, Number 425, 1-28. The Birds of North America, Inc., Philadelphia, PA.

Scott, J.M., D.D. Goble, A.M. Haines, J.A. Wiens, and M.C. Neel. 2010. Conservation-reliant species and the future of conservation. Conservation Letters 3:91-97.

Society for Ecological Restoration International. 2004. The SER International Primer on Ecological Restoration. Version 2. Science and Policy Working Group. Tucson, AZ. http://www.ser.org/docs/default-document-library/English.pdf.

Stiver, S.J., A.D. Apa, J.R. Bohne, S.D. Bunnell, P.A. Deibert, S.C. Gardner, M.A. Hilliard, C.W. McCarthy, and M.A. Schroeder. 2006. Greater Sage-Grouse comprehensive conservation strategy. Unpublished report. Western Association of Fish and Wildlife Agencies, Cheyenne, WY.

Symstad, A.J., C.L. Wienk, and A.D. Thorstenson. 2008. Precision, repeatability, and efficiency of two canopy-cover estimate methods in northern Great Plains vegetation. Rangeland Ecology and Management 61:419-429.

Taylor, R.L., J.D. Tack, D.E. Naugle, and L.S. Mills. 2013. Combined effects of energy development and disease on Greater Sage-Grouse. PLoS ONE 8(8):e71256.

Thacker E.T. 2010. Greater Sage-Grouse seasonal ecology and responses to habitat manipulations in northern Utah. Ph.D. dissertation, Utah State University, Logan, UT.

Toevs, G.R., J.J. Taylor, C.S. Spurrier, W.C. MacKinnon, and M.R. Bobo. 2011. Bureau of Land Management assessment, inventory, and monitoring strategy: For integrated renewable resources management. Bureau of Land Management, National Operations Center, Denver, CO.

United States Department of the Interior. 1980. Habitat evaluation procedures (HEP). Ecological Services Manual 102. Fish and Wildlife Service, Washington, DC.

United States Department of the Interior. 1996. Sampling vegetation attributes. Interagency technical reference. Bureau of Land Management, National Applied Resource Sciences Center, Denver, CO.

United States Department of the Interior. 2001. Rangeland health standards. Handbook H-4180-1. Bureau of Land Management, Washington, DC.

United States Department of the Interior. 2006. Roads and trails terminology. Technical Note 422. Bureau of Land Management, Denver, CO. http://www.blm.gov/style/medialib/blm/wo/blm_library/tech_notes.Par.17069.File.dat/TN_422.pdf.

Urban, D.L., R.V. O'Neill, and H.H. Shugart, Jr. 1987. Landscape ecology: A hierarchical perspective can help scientists understand spatial patterns. BioScience 37:119-127.

U.S. Fish and Wildlife Service. 2013. Greater Sage-Grouse (*Centrocercus urophasianus*) conservation objectives: Final report. U.S. Fish and Wildlife Service, Denver, CO.

Van Horn, B. 1983. Density as a misleading indicator of habitat quality. Journal of Wildlife Management 47:893–901.

Walker, B.L. and D.E. Naugle. 2011. West Nile virus ecology in sagebrush habitat and impacts on Greater Sage-Grouse populations. In Greater Sage-Grouse: Ecology and conservation of a landscape species and its habitats, edited by S.T. Knick and J.W. Connelly, 127-142. Studies in Avian Biology, vol. 38. University of California Press, Berkeley, CA.

West, N.E. 1983. Western intermountain sagebrush steppe. *In* Ecosystems of the world 5: Temperate deserts and semi-deserts, edited by N.E. West, 351-397. Elsevier Scientific Publishing Co., New York, NY.

White, P.S. and S.T.A. Pickett. 1985. Natural disturbance and patch dynamics: An introduction. *In* The ecology of natural disturbance and patch dynamics, edited by S.T.A. Pickett and P.S. White, 3-13. Academic Press, New York. NY.

Wisdom, M.J., R.S. Holthausen, B.K. Wales, D.C. Lee, C.D. Hargis, V.A. Saab, W.J. Hann, T.D. Rich, M.M. Rowland, W.J. Murphy, and M.R. Eames. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia Basin: Broad-scale trends and management implications. General Technical Report PNW-GTR-000. U.S. Department of Agriculture, Forest Service, Portland, OR.

Wisdom, M.J., M.M. Rowland, and L.H. Suring. 2005. Habitat threats in the sagebrush ecosystem: Methods of regional assessment and applications in the Great Basin. Alliance Communications Group, Lawrence, KS.

Wisdom, M.J., M.M. Rowland, L.H. Suring, L. Schueck, C.W. Meinke, B.C. Wales, and S.T. Knick. 2003. Procedures for regional assessment of habitats for species of conservation concern in the sagebrush ecosystem. Version 1. U.S. Forest Service, Pacific Northwest Research Station, La Grande, OR.

Yost, A.C., S.L. Petersen, M. Gregg, and R. Miller. 2008. Predictive modeling and mapping sage grouse (*Centrocercus urophasianus*) nesting habitat using maximum entropy and a long-term dataset from southern Oregon. Ecological Informatics 3:375-386.

Appendix A:
Craters of the Moon National Monument Case Study

General Overview

The project area is located on the central Snake River Plain and encompasses nearly 300,000 acres of BLM lands within the Craters of the Moon National Monument (CRMO). Private and state lands are interspersed throughout the area, but do not significantly affect the continuity of the landscape. Shrub cover types are generally Wyoming big sagebrush, antelope bitterbrush, or threetip sagebrush at the lower elevations and mountain big sagebrush at higher elevations. Predominant native perennial understory grasses vary between Thurber's needlegrass, needle-and-thread grass, bluebunch wheatgrass, and Idaho fescue. Roughly two-thirds of the transect sites that were read have been exposed to wildfires in the past 20 years, including 2012 fires that burned 42 sites, and have been treated with a mix of native and nonnative seedings. Roughly one-third of the sites read have greater than 30 percent cover of cheatgrass, although some of those areas are also dominated by sagebrush overstory that could still be important to Greater Sage-Grouse, such as during the winter. The area is habitat for several big game species, raptors, and sagebrush obligates such as Greater Sage-Grouse. The primary land uses are grazing and recreation.

Site Stratification

Prior to the field season, the CRMO interdisciplinary (ID) team developed objectives related to the assessment that would help inform future management decisions. The key questions were "What is the status of Greater Sage-Grouse habitat in the CRMO area?" and "How do we stratify this to answer the questions of habitat suitability compared to current management and site potential?" Objectives were also developed to assist with setting parameters for site stratification. These objectives were to quantify the status of Greater Sage-Grouse in the CRMO by ecological site, pasture, and seasonal habitat designation; determine compatibility between assessment, inventory, and monitoring (AIM) program core indicators and the HAF; and establish locations for long-term monitoring of sage-grouse habitat.

Stratification was completed by the Jornada Experimental Range, New Mexico State University, Las Cruces, using ArcGIS Spatial Analyst. Initially, the current status of spatial and tabular data was determined, and then a boundary for the spatial extent of the study area was defined. Based on the existing data, several parameters were selected for use including existing vegetation, past land treatments, wildfires, and ecological sites. The ecological sites were grouped by similar environmental conditions (e.g., ARTRW8/FEID and ARTRW8/ACTH7) to reduce the number of units needing sampling from 38 to 10. Ecological sites reflect similarities that can be related to the state and transition models, expected potential, and expected vegetation for the site. Allotment and pasture boundaries were used as the analysis unit. A travel management plan had recently been completed, so the official roads and trails layer was used to determine a strategy for getting to sites. The range improvements layer was used to determine potential conflicts prior to field verification to ensure transects avoided structures such as water troughs. The transect locations were reviewed preliminarily and appeared to be well distributed across the study area, and only rarely did they occur in the middle of a reservoir, sheep bed ground, or lava flow. These locations were later omitted. A total of 328 transect locations were identified; of those, 316 were read in 2012.

Field Verification

Although a reliable set of GIS layers was available for the stratification process and GPS was used to navigate to sites, there was still a margin of error as to where the transect sites actually occurred on the ground. Therefore, after sites were selected, field verification that the study sites actually occurred within the correct vegetation and ecological site was necessary. The ID team created a common set of rules for initial site verification. These rules were set prior to field work and were used to determine if the site should be kept, moved, or removed.

- The standard azimuth for transects is 0º (due north).
- If a 0º azimuth causes the transect to cross two or more ecological sites or a nonnatural land cover type (e.g., a road), a random azimuth is then selected for the transect. Sites should not be excluded because they are close to these features, only because the feature itself actually occurs in the transect.
- If the first randomly selected azimuth does not successfully reorient the transect in the target ecological site and vegetation, then the site should be moved 100 meters in a standard direction and selection of random azimuths should not be continued.
- If the site cannot be reoriented or moved due to the shape or size of the target area, then the transect is removed from consideration and a backup transect is selected.
- If accessing the site is dangerous or not possible, then the site is removed and a backup is selected.
- If a site can be moved, move 100 meters into the correct ecological site. If that is not possible, then remove the site and use a backup.

General information regarding how to update the GPS data files used during site verification to reflect any changes to the location, azimuth, ecological site, land cover type (LCT), and general site conditions was also included in the strategy.

The site information worksheet was filled out by the journeyman-level specialist who completed the transect verification. Transects were removed if they landed directly on lava fields, if major anthropogenic disturbances were present (e.g., two power lines and two roads running directly through the site), or if one ecological type was not maintained for the whole transect.

Technicians were able to follow directions laid out during site verification by the specialists and immediately begin data collection. This technique prevented confusion or inconsistency of interpretation by the technicians and removed the burden of determining suitable transect locations. Verifying sites ahead of time also ensured that the specialists were familiar with existing conditions when later reviewing large amounts of data and making habitat suitability decisions from the data. Part of the verification process was to determine if the correct ecological site description (ESD) was represented at the site and to initially confirm the LCT. The LCT was later verified by the line-point intercept (LPI) data. This data is critical for proper grouping of transects when summarizing and assigning a habitat suitability rating.

Protocols for data collection and compilation, naming conventions, and download processes were also established. These protocols ensured that the file structure was widely understood and common threads between field data, processed data, and final data were maintained. Forms were completed in both digital and hard copy format, due to computer program availability issues in the field. No matter what method users select, completed photo cards and photos for each transect provide a simple way to organize hard-copy data, document site completion dates, and verify general information. A GIS specialist created an inclusive data dictionary for the GPS units that were used to collect miscellaneous information, such as range improvements, noxious weed locations, and incidental wildlife and rare plant observations.

Field Data Collection

Initial test sites were read by an ID team to determine the necessary equipment and the methods to implement and to simplify training for technicians. The line intercept—Daubenmire frame method (LIDF) was compared to the line-point intercept (LPI) method for measuring cover by taking 1 week to conduct both techniques at each of the transects completed. The data was subsequently analyzed, and the ID team determined the LPI was the more efficient method, relative to the project area and objectives, because it is the more rapid method for collecting cover data by species and the ground cover data collected is more readily compared to existing range program data. However, if only collecting life form data, the LIDF is the more rapid method. The belt transect, used to document forb species presence, was an adequate method to determine diversity and abundance, but has since been revised. The LIDF method also excels in capturing forb information due to the use of a frame rather than a cover pin.

Seasonal technicians performed most of the data collection. Altogether, there were seven technicians, split into three crews. One technician was designated as the crew lead, and one was responsible for handling data downloads and organization. The technicians had backgrounds in botany, wildlife, and range ecology. With this education and experience, the technicians had an understanding of what was asked and why and had enough interest in what was being collected to ask solid questions that helped improve the process.

Crews were assigned areas to focus efforts into a more logical approach across the analysis area. Two crews were stationed at outlying fire crew guard stations to help reduce drive times, and one crew was based out of the field office. This crew was able to pick up the outlier sites that did not fit in logistically with the other crew locations. Each crew was given a separate set of USGS 1:24k topographic maps that strategically divided the sites to facilitate the most expedient completion of the fieldwork. Habitat type and elevation/precipitation gradients (lowest/driest to highest/most moist) was used to seasonally prioritize the sites. A few nights were spent in the field, and as the season progressed, the terrain became more rugged, increasing the hiking time tremendously. Some of the sites took 2-3 hours to hike into, while others were only a 5-minute walk. However, time spent at each LPI transect was consistently about 1 hour.

Analysis and Reporting

After the field data was collected, it was compiled into the correct format to combine transect data for the appropriate site. From this data, the team derived values for sagebrush shapes, heights, species, perennial grass and forb species height and cover percent, and forb abundance. The ID team made the final determinations of habitat suitability for each site based on the compiled data. The ID team had a good understanding of what to expect from the data, having participated in the earlier field verification, and could identify if anything was missed in the initial collection effort, what should be added to the measurements, which sites to revisit, and which transects should be combined with other transects. Data verification by an ID team is an important step to double check the field data and ensure that no sites were misclassified.

The ID team used telemetry data, field observations, and professional knowledge and judgment to determine the habitat suitability for each site area, in addition to the transect/field data collected. Aspect, slope, elevation, ESDs, past land uses, and disturbance regimes were incorporated into the process. The Excel spreadsheets used to collect field data were imported into an Access database, allowing mass calculations and creating a format that assigned values to each transect and that was compatible with ArcMap for spatial analysis. Joining the tables in ArcMap allows for a completely new level of spatial analysis and display. For example, shrub canopy cover can be displayed for all species/subspecies across the project area. Percent cover, dominant sagebrush species, sagebrush cover only, or percent cover for every shrub except sagebrush are a few examples of data that can be readily displayed. This format also helps display connectivity and distribution of habitat qualities.

Summaries were created using the Access tables joined in ArcMap to ensure the correct spatial attributes such as proper management unit, county, subpopulation, seasonal habitat, and proximity to leks and lek status. These tables were then exported to an Excel workbook and put into a pivot table to simplify data analysis. This process allowed for more rapid and efficient determination of Greater Sage-Grouse habitat suitability of the sites and was compiled using the seasonal habitat data summary (form S-1).

Recommendations and Lessons Learned

As with any process, several items were identified as the assessment progressed that might help simplify and streamline future endeavors:

- If using the LPI instead of the LIDF method, collect the shape of each sagebrush plant encountered along the transect when conducting a separate line intercept for sagebrush cover. This process will expand the number of sagebrush shape samples recorded, especially when there are few plants from which to determine shape.
- Create a general plant species list when completing site verification to prepopulate electronic data forms and facilitate data collection in the field.
- Use of electronic platforms is a more rapid and efficient way to collect information, if only for the ease of processing data later, correcting misspellings, and identifying unknowns. Electronic data management also reduces the amount of paper to file and store. Unfortunately, the requested field tablets were not available until August, so a large amount of the initial data was collected on either GPS units loaded with data dictionaries or on hard-copy forms. The small screen of the GPS units made it difficult to collect forb belt transect and LPI data, but collecting the LPI and LIDF data required about the same amount of time via either hard copy or GPS units. The field tablet screens are roughly the same size as a sheet of paper and accelerate the process.
- Print and store final copies of the transect data. The final version should be clean of errors, easy to read, and organized similarly to the digital formats.
- Plan ahead and create a realistic timeframe and calendar, allowing for training days, prep and closeout time, actual field days, and possible extraneous circumstances (e.g., flat tires, GPS unit malfunctions, wildfires).
- Take time at the beginning of the season to clarify details with resource specialists, both in the field and in the office, prior to field crews starting. This step allowed specialists to have most of our questions answered, so that we could explain the process and needs to the crews.
- Have the specialists spend time in the field with the seasonal crews for training.
- Switch crew members around to make sure each crew maintains similar procedures and perform quality checks at sites.
- The HAF consists of metrics that can be expanded upon to inform more than sage-grouse habitat suitability. The stratification used is consistent with other management objectives and resource needs such as:
  - Documenting invasive annuals and noxious weed presence.
  - Verifying dominant land cover type and plant community with the cover data.
  - Informing land health standards.

Appendix B: Data Forms and Measurement Techniques

This appendix to the sage-grouse HAF contains the data forms for the habitat assessment and specific instructions for completing them. It is organized by scale and is intended to be used in the field or in the lab as appropriate for data collection or summary. Chapter II of the HAF provides the detailed habitat description steps to guide setup and data collection.

Assessments for the broad-scale (first-order) habitat selection require rangewide coverage and policy decisions at either the rangewide scale or the management zone scale. No structured data forms are required for a first-order assessment. Policies establish the management direction for sagebrush habitats and sage-grouse.

The assessment of mid-scale (second-order) habitat selection requires a general delineation of sage-grouse populations, habitat, and habitat patterns such as patch connectivity, linkage, patch edges, and fragmentation. Scientists employing advanced mapping technology will provide decisionmakers with the existing land cover classification (e.g., urban, agriculture, and natural vegetation communities at the alliance level), ecological potential for cover classes, and biotic risk factors across the landscape. Spatial analysts, specializing in anthropogenic features, will add sociological and political layers of constraints on the landscapes. This information will enable managers and decisionmakers working in concert with scientists to describe existing conditions. This assessment can aid in the development of priority conservation focus areas. A single form (form M-1) is required for summarizing the second-order assessment. This form should be applied for each landscape/population assessed at this scale.

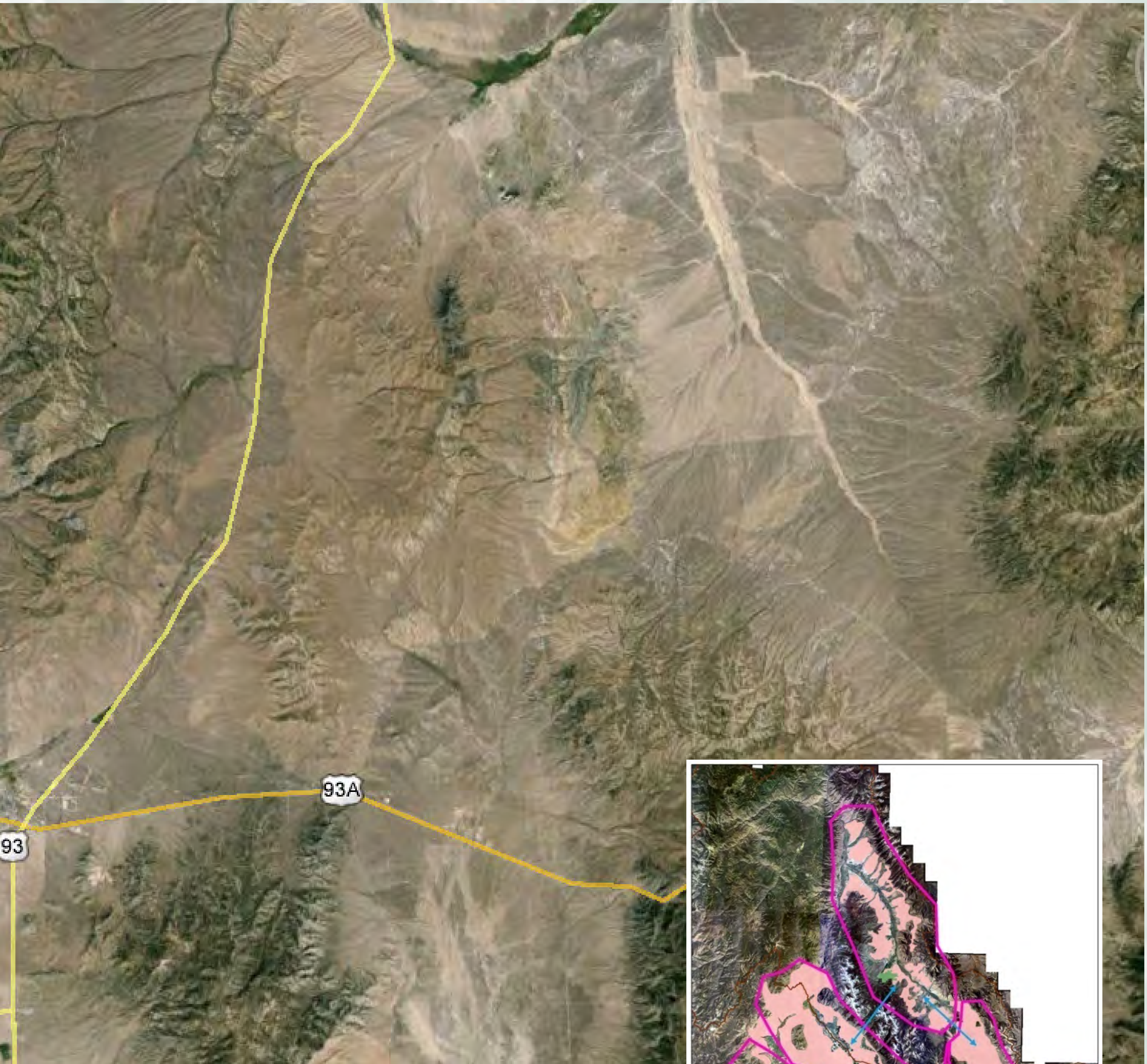
Fine-scale (third-order) habitat selection analysis allows managers to plan and implement conservation actions that promote the objectives of the higher level decisions and policies. Managers can also use fine-scale data collected on a single form (form F-1) to develop project priority lists based on science and available spatial analytical information. Priority conservation focal areas can then be identified and evaluated for potential fourth-order treatments. Following this evaluation, specific projects or other actions can then be proposed.

The remainder of the data forms found in this appendix are site-scale (fourth-order) instruments, adequate to describe vegetation communities to the plant association. The forms include detailed directions and illustrations for measuring vegetation at the site scale. Supplemental information regarding vegetation species and preferred forbs for sage-grouse can be found in table B-1 at the end of this appendix. Managers and resource specialists will find systematic collection and analysis of these data helpful in prescribing appropriate actions or treatments for fourth-order projects.

These forms are available as workbook spreadsheets that can be loaded onto field tablets or ruggedized laptop computers. They can be found on the enclosed flash drive and online at the BLM Library website at www.blm.gov/wo/st/en/info/blm-library/publications/blm_publications/tech_refs.html.


80

Mid-Scale (Second-Order) Data Forms



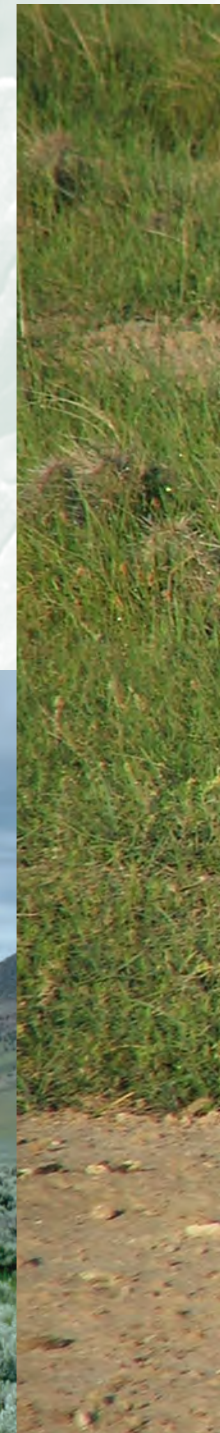
Form M-1: Mid-Scale (Second-Order) Sage-Grouse Habitat Description

Date:	Counties:	State:
Evaluator(s):	Populations:	
General Location:	Map File Name:	
Sage-Grouse Management Zone(s):		
Agencies:		

Data Sources	
Land Cover Type Data Sources:	Date:
Anthropogenic Features Data Sources:	
Population Data Sources:	
Data Storage Location:	
Software and Version:	
Mapping Grain (spatial resolution):	Population Area Extent (km^2):

Habitat Indicator Descriptions	
1. Habitat Availability	a. Area of occupied habitat (km^2) =
	b. Area of potential habitat (km^2) =
	c. Area of nonhabitat (km^2) (optional) =
	Discussion:
2. Patch Size and Number	a. Mean size of occupied habitat patches (km^2) =
	b. # of occupied habitat patches =
	Discussion:
3. Patch Connectivity	Mean distance to nearest occupied habitat patch (km) =
	Discussion:
4. Linkage Area Characteristics	a. % suitable land cover types in linkage areas =
	b. % marginal land cover types in linkage areas =
	c. % unsuitable land cover types in linkage areas =
	Discussion:
5. Landscape Matrix and Edge Effect	a. Mean % positive patch edges =
	b. Mean % negative patch edges =
	Discussion:
6. Anthropogenic Disturbances	a. Densities of linear features (km / km^2) =
	b. Densities of point features (sites / km^2) =
	c. Area of nonhabitat or unsuitable habitat inclusions (km^2) =

Mid-Scale (Second-Order) Suitability Summary	
Landscape Desciption: Check the one description below that best describes the population and subpopulation area:	✔
Suitable: Landscapes have connected mosaics of sagebrush shrublands that allow for bird dispersal and migration movements within the population or subpopulation area. Anthropogenic disturbances that can disrupt dispersal or cause mortality are generally not widespread or are absent.	
Marginal: Landscapes have patchy, fragmented sagebrush shrublands that are not well connected for dispersal and migration in portions of the population or subpopulation area. Anthropogenic disturbances that disrupt dispersal or cause mortality are present throughout all or portions of the landscape. Some lek groups or subpopulations are isolated or nearly isolated.	
Unsuitable: Landscapes were former shrubland habitat now converted to predominantly grassland or woodland cover or other unsuitable land cover or use. Remaining sagebrush patches are predominantly unoccupied or have few remaining birds. Portions of the population or subpopulation area may become occupied in the foreseeable future through succession or restoration.	
Discussion:	




Fine-Scale (Third-Order) Data Forms

Form F-1: Fine-Scale (Third-Order) Sage-Grouse Habitat Description

Description Year:	**Counties:**	**State:**
Evaluator(s):	**Agency:**	
Home Range Name:	**Population:**	
Lek Group Name:	**General Location:**	

Data Sources	
Land Cover Type Data Sources:	
Anthropogenic Features Data Sources:	
Population Data Sources:	
Data Storage Location:	
Software and Version:	
Mapping Grain:	**Home Range Area Extent (km^2):**

Habitat Indicator Descriptions	
1. Seasonal Habitat Availability	a. Area of occupied breeding habitat (km^2) =
	a. Area of occupied summer habitat (km^2) =
	a. Area of occupied winter habitat (km^2) =
	b. Area of potential breeding habitat (km^2) =
	b. Area of potential summer habitat (km^2) =
	b. Area of potential winter habitat (km^2) =
	c. Area of nonhabitat (km^2) (optional) =
	Discussion:
2. Seasonal Use Area Connectivity	Breeding to summer (km edge/km^2 of habitat) =
	Summer to winter (km edge/km^2 of habitat) =
	Winter to breeding (km edge/km^2 of habitat) =
3. Anthropogenic Disturbances	a. Densities of linear features (km/km^2) =
	b. Densities of point features (sites/km^2) =
	c. Area of nonhabitat or unsuitable habitat inclusions (km^2) =
	Discussion:

Fine-Scale (Third-Order) Suitability Summary	
✔	**Check the one description below that best describes the home range:**
	Suitable: Home ranges have connected seasonal use areas. Anthropogenic features that can disrupt seasonal movements or cause mortality are generally absent or at least not widespread.
	Marginal: Home ranges have poorly connected or disjunct seasonal use areas. Anthropogenic features that can disrupt seasonal movements or cause mortality may occur within the home range.
	Unsuitable: Home ranges have seasonal use areas with predominantly grassland, woodland, or incompatible land uses (anthropogenic features) not conducive to sage-grouse seasonal movements or habitat use. Most leks have been abandoned or have few remaining birds.
Discussion:	

Site-Scale (Fourth-Order) Data Forms

Site-Scale (Fourth-Order) Data Forms

Form S-1: Sage-Grouse Site-Scale Seasonal Habitat Data Summary

Date:	County:	State:	Evaluator(s):
Population:			Home Range Name:
Seasonal Habitat:			Associated Leks:

Land Cover Type	Ecological Site	Area (ha/ac) or Length (km/mi)	Transects (#)	Indicator Values from Data Forms (means in most cases)								
				$\overline{x}$ Sage Cover (%)	$\overline{x}$ Sage Ht. (cm)	$\overline{x}$ Predominant Sage Shape (# of S and C)	$\overline{x}$ PG Ht. (cm)	$\overline{x}$ PF Ht. (cm)	$\overline{x}$ PG Cover (%)	$\overline{x}$ PF Cover (%)	Preferred Forb Species (#)	Lek Hbt. Avg. Distance to Sage Cover (m)

Form S-1: Sage-Grouse Site-Scale Seasonal Habitat Data Summary Directions

1. Use this form to summarize seasonal habitat field transect data collected using methods outlined in this document.

2. Complete all location information at the top of the form. Information should be consistent with information on the field data forms. Most of the information should be self-explanatory except the following:

   Population: Identify the population with which the habitat is associated. This definition also includes small populations. Population names are found in figure 3.

   Home Range Name: Identify the home range area using a major drainage area or other distinguishing land feature (e.g., Little Lost River home range).

   Seasonal Habitat: List the one season (breeding, summer, or winter) to which the data pertain. The same area may provide more than one seasonal habitat need, but data must be collected at the appropriate time of year for descriptions.

   Associated Leks: List the two largest occupied leks to which the breeding habitat is associated. Use identification numbers or names that are used in the statewide database.

3. Complete the data section of the form:

   Land Cover Type: Identify the land cover of the seasonal habitat being summarized.

   Upland communities: Use plant alliances or associations (Reid et al. 2002) for sagebrush or grassland communities; use www.natureserve.org/explorer (International Classification of Ecological Communities) or other sampling strata to describe the habitat (e.g., percent sagebrush categories). Use the species symbol for dominant species in the overstory and understory (table B-1), for example ARTRW8 (alliance level – Wyoming big sagebrush) or ARTRW8/FEID (association level – Wyoming big sagebrush/Idaho fescue).

   Riparian or wetland communities: Use site type (riparian areas, wet meadows, springs) or more detailed classification using Cowardin et al. (1979) or riparian type (regional classification systems) to which the data pertain.

   Ecological Site: Refer to soil maps, range site guides, and ecological site descriptions where available and record the appropriate ecological site. Use the species symbol for dominant species in the overstory and understory.

   Area or Length: Record the polygon area (indicating ha/ac) or linear length for riparian areas (indicating km/mi) of the habitat sampled (e.g., the land cover type).

   Transects: Record the number of 50-m transects or sites measured within the land cover type. If transect length was adjusted due to polygon size or shape, annotate as needed.

   Indicator Values: Record the mean or total numbers as indicated for each measurement (sagebrush cover, sagebrush height, sagebrush shape, perennial grass height, perennial forb height, perennial grass cover, perennial form cover, preferred forb species, and lek habitat distance to sage cover).

   Sagebrush Height: Sagebrush height above ground for most seasons and above snow for winter habitat.

   Predominant Sagebrush Shape: Estimate the number of spreading (S) or columnar (C) plants (see visual shape guide, figure 13).

Form S-2: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Breeding Habitat (Leks)

Date:	**County:**	**State:**	**Evaluator(s):**
Population:			**Home Range Name:**
Land Cover Type:			**Lek ID#:**
GPS file #:			**Lek Status (circle one):** Occupied Unoccupied Undetermined
UTM:			

Habitat Indicator Suitability Range						
Habitat Indicator	**Suitable**	✔	**Marginal**	✔	**Unsuitable**	✔
Availability of Sagebrush Cover	*Lek has adjacent protective sagebrush cover (within 100 m)*		*Sagebrush within 100 m provides very little protective cover*		*Adjacent sagebrush cover is >100 m*	
Proximity of Detrimental Land Uses	*Detrimental land uses are not within line of sight of lek and absent to uncommon within 3 km of lek*		*Detrimental land uses are within line of sight of lek and uncommon or few within 3 km of lek*		*Detrimental land uses are within the vicinity of the lek site*	
Proximity of Trees or Other Tall Structures	*Trees or other tall structures are not within line of sight of lek and none to uncommon within 3 km of lek*		*Trees or other tall structures are within line of sight of lek and uncommon or scattered within 3 km of lek*		*Trees or other tall structures are within the vicinity of the lek site*	

Site-Scale Suitability	**Suitable**		**Marginal**		**Unsuitable**	

Anthropogenic Noise Description:

Rationale for Overall Suitability Rating:

Form S-2: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Breeding Habitat (Leks) Directions

1. Complete one form for each occupied, unoccupied, or undetermined lek in the home range or lek group, as needed.

2. Complete all location information at the top of the form. Most of the information should be self-explanatory except the following:

   Population: Identify the population with which the habitat is associated. This definition also includes small populations. Population names are found in figure 3.

   Home Range Name: Identify the home range area using a major drainage area or other distinguishing land feature (e.g., Little Lost River home range).

   Land Cover Type: Identify the cover type at the lek site. Use plant alliances or associations (Reid et al. 2002) for sagebrush or grassland communities; use www.natureserve.org/explorer (International Classification of Ecological Communities) or other sampling strata to describe the habitat (e.g., percent sagebrush categories). Use the species symbol for dominant species in the overstory and understory (table B-1), for example ARTRW8 (alliance level – Wyoming big sagebrush) or ARTRW8/FEID (association level – Wyoming big sagebrush/Idaho fescue). Note whether the lek is located in a nonhabitat (e.g., agriculture, urban, industrial) area. If the lek is located on a road, in a livestock watering area, or on a similar type of surface within a plant community, indicate this cover type in the following manner: ARTRW8:road; ARTRW8:trough area.

   Lek ID #: Use the identification number or name that is used in the statewide database.

   Lek Status: Determine the status using the following definitions. Note that the specific terms and definitions for lek status may vary by state. Use the terminology appropriate for your area.

   Occupied lek: [*Greater Sage-Grouse*] A lek that has been active during at least one breeding season within the prior 5 years. [*Gunnison Sage-Grouse*] A lek that has been attended by males in the previous 5 years.

   Unoccupied lek: [*Greater Sage-Grouse*] A lek that has not been active during a period of 5 consecutive years. [*Gunnison Sage-Grouse*] A lek that has been inactive for 5 years.

   Undetermined lek: Any lek that has not been documented as active in the last 5 years, but for which survey information is insufficient to designate the lek as unoccupied.

3. Complete indicator measurements:

   Availability of Sagebrush Cover: Adjacent sagebrush distance is measured from the edge of the lekking area to the edge of the nearest stand of mature sagebrush of sufficient extent to provide protective cover.

   Proximity of Detrimental Land Uses: Such land uses include oil/gas wells, roads, agricultural fields, subdivisions, etc.

   Proximity of Trees or Other Tall Structures: Trees and tall structures are considered "within the vicinity" when they provide avian perch sites with a view of birds on the lek.

4. Determine the appropriate suitability category and mark (✔) each indicator as suitable, marginal, or unsuitable.

5. Describe **anthropogenic noise**. Indicate the presence of and describe any anthropogenic noises observed during the lekking period. Identify the noise source (highway vehicles, generator, wind turbines, military overflights, etc.) and describe the occurrence frequency (constant or periodic), volume (loud to soft), and pitch (high to low). Use a decibel meter, if available, to record data when anthropogenic noises are a concern for the lek.

6. Determine **site-scale suitability**. Overall suitability takes into consideration the relationship between the indicators and their relative importance. This evaluation is based on professional judgment using the indicators for guidance. Explain overall site suitability in the rationale section.

7. Attach photographs of the lek site.

8. Provide a copy of this form to the state wildlife agency's sage-grouse coordinator.

Form S-3: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Breeding Habitat (Nesting/Early Brood-Rearing)

Date:	County:	State:	Evaluator(s):
Population:			Home Range Name:
Land Cover Type:			Ecological Site:
Associated Leks:			Number of Transects:
Area Sampled (ha/ac):			Site Info. (circle one): Arid Site Mesic Site
List UTM Coordinates (coordinates, zone, datum) of All Transects:			

Habitat Indicator Suitability Range							
Habitat Indicator	$\bar{x}$	**Suitable**	✔	**Marginal**	✔	**Unsuitable**	✔
Sagebrush Canopy Cover (mean)		15 to 25%		5 to <15% or >25%		<5%	
Sagebrush Height Mesic Site (mean) Arid Site (mean)		 40 to 80 cm 30 to 80 cm		 20 to <40 cm or >80 20 to <30 cm or >80		 <20 cm <20 cm	
Predominant Sagebrush Shape (mode) Spreading (*n*) Columnar (*n*)		Spreading		Mix of spreading and columnar		Columnar	
Perennial Grass Height (mean)		≥18 cm		10 to <18 cm		<10 cm	
Perennial Forb Height (mean)		≥18 cm		10 to <18 cm		<10 cm	
Perennial Grass Cover Mesic Site (mean) Arid Site (mean)		 ≥15% ≥10%		 5 to <15% 5 to <10%		 <5% <5%	
Perennial Forb Cover Mesic Site (mean) Arid Site (mean)		 ≥10% ≥5%		 5 to <10% 3 to <5%		 <5% <3%	
Preferred Forb Availability (relative to site potential) Number of Preferred Forb Species (*n*)		Preferred forbs are common with several species present		Preferred forbs are common but only a few species are present		Preferred forbs are rare	

Site-Scale Suitability **Suitable**		**Marginal**		**Unsuitable**	

Does ecological site potential limit suitability potential? (circle one)	Yes	No	Unknown	
Drought Condition (circle one):	Extreme Drought	Severe Drought	Moderate Drought	Mid-Range
	Moderately Moist	Very Moist	Extremely Moist	
Rationale for Overall Suitability Rating:				

Form S-3: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Breeding Habitat (Nesting/Early Brood-Rearing) Directions

1. Use this worksheet to interpret field data collected using methods (LPI/LIDF and forb diversity) outlined in this appendix and summarized on the "Sage-Grouse Site-Scale Seasonal Habitat Data Summary" (form S-1).

2. Complete all site location information at the top of the form. Be sure to list all UTM coordinates or other identifying feature of all sites being summarized. Most of the information should be self-explanatory except the following:

   Population: Identify the population with which the habitat is associated. This definition also includes small populations. Population names are found in figure 3.

   Home Range Name: Identify the home range area using a major drainage area or other distinguishing land feature (e.g., Little Lost River home range).

   Land Cover Type: Identify the cover type of the data collected. Use plant alliances or associations (Reid et al. 2002) for sagebrush or grassland communities; use www.natureserve.org/explorer (International Classification of Ecological Communities) or other sampling strata to describe the habitat (e.g., percent sagebrush categories). Use the species symbol for dominant species in the overstory and understory (table B-1), for example ARTRW8 (alliance level – Wyoming big sagebrush) or ARTRW8/FEID (association level – Wyoming big sagebrush/Idaho fescue).

   Ecological Site: Refer to soil maps, range site guides, and ecological site descriptions where available and record the appropriate ecological site. Use the species symbol for dominant species in the overstory and understory.

   Associated Leks: List the two largest occupied leks to which the breeding habitat is associated. Use identification numbers or names that are used in the statewide database.

   Number of Transects: Record the number of 50-m transects completed within the land cover type.

   Area Sampled: Record the total area (indicating ha/ac) of the land cover type sampled.

   Site Info.:

      Arid Site: Applies to sagebrush ecological sites generally in the 25-30 cm (10-12 in) precipitation zone. Wyoming big sagebrush is a common big sagebrush subspecies for this type of site.

      Mesic Site: Applies to sagebrush ecological sites generally in a >30 cm (12 in) precipitation zone. Mountain big sagebrush is a common big sagebrush subspecies for this type of site.

3. Transfer data from the "Sage-Grouse Site-Scale Seasonal Habitat Data Summary" (form S-1) to this form. Enter the appropriate mean ($\overline{x}$) and number (n) values for the indicators in the column under $\overline{x}$.

      Predominant Sagebrush Shape: Estimate the number of spreading (S) or columnar (C) plants (see visual shape guide, figure 13).

      Perennial Forb Height (Optional): In many situations, perennial forb heights can be quite variable or provide minimal contribution to herbaceous structure. Therefore, in most cases, use perennial grass heights for the suitability rating.

      Preferred Forb Availability: Check the appropriate suitability category based on data derived using the "Sage-Grouse Forb Diversity Data Form." The suitability evaluation must be relative to ecological site potential.

4. Determine the appropriate suitability category and mark (✔) each indicator as suitable, marginal, or unsuitable.

5. Determine **site-scale suitability**. Overall suitability takes into consideration the relationship between the indicators and their relative importance. This evaluation is based on professional judgment using the indicators for guidance. Explain overall site suitability in the rationale section.

6. Indicate if **site potential** is a factor for a suitability description of marginal or unsuitable. Explain further in the rationale section.

7. Indicate **drought condition** using local weather station data or as reported for the region of concern on the National Weather Service website: www.ncdc.noaa.gov/oa/climate/research/us-drought-monthly.html.

8. Attach field data sheet(s) and photographs used for this site-scale description.

9. Provide a copy of this form to the state wildlife agency's sage-grouse coordinator.

Form S-4: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Upland Summer/Late Brood-Rearing Habitat

Date:	County:	State:	Evaluator(s):
Population:			Home Range Name:
Land Cover Type:			Ecological Site:
Number of Transects:			Area Sampled (ha/ac):
List UTM Coordinates (coordinates, zone, datum) of All Transects:			

Habitat Indicator Suitability Range							
Habitat Indicator	$\overline{x}$	Suitable	✔	Marginal	✔	Unsuitable	✔
Sagebrush Cover (mean)		10 to 25%		5 to <10% or >25%		<5%	
Sagebrush Height (mean)		40 to 80 cm		20 to <40 or >80 cm		<20cm	
Perennial Grass and Forb Cover (mean)		≥15 %		5 to <15%		<5%	
Preferred Forb Availability (relative to site potential)		Preferred forbs are common with appropriate numbers of species present		Forbs are common but only a few preferred species are present		Preferred forbs are rare	
Number of Preferred Forb Species (*n*)							

Site-Scale Suitability	Suitable		Marginal		Unsuitable	

Does site potential limit suitability? (circle one) Yes No Unknown

Drought Condition (circle one): Extreme Drought Severe Drought Moderate Drought Mid-Range Moderately Moist Very Moist Extremely Moist

Rationale for Overall Suitability Rating:

Form S-4: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Upland Summer/Late Brood-Rearing Habitat Directions

1. Use this worksheet to interpret field data summarized on the "Sage-Grouse Site-Scale Seasonal Habitat Data Summary" (form S-1).

2. Complete all location information at the top of the form. Be sure to list all UTM coordinates or other identifying feature of all sites being summarized. Most of the information should be self-explanatory except the following:

   Population: Identify the population with which the habitat is associated. This definition also includes small populations. Population names are found in figure 3.

   Home Range Name: Identify the home range area using a major drainage area or other distinguishing land feature (e.g., Little Lost River home range).

   Land Cover Type: Identify the cover type of the data collected. Use plant alliances or associations (Reid et al. 2002) for sagebrush or grassland communities; use www.natureserve.org/explorer (International Classification of Ecological Communities) or other sampling strata to describe the habitat (e.g., percent sagebrush categories). Use the species symbol for dominant species in the overstory and understory (table B-1), for example ARTRW8 (alliance level – Wyoming big sagebrush) or ARTRW8/FEID (association level – Wyoming big sagebrush/Idaho fescue).

   Ecological Site: Refer to soil maps, range site guides, and ecological site descriptions where available and record the appropriate ecological site. Use the species symbol for dominant species in the overstory and understory.

   Number of Transects: Record the number of 50-m transects completed within the land cover type.

   Area Sampled: Record the total area (indicating ha/ac) of the land cover type sampled.

3. Transfer data from the "Sage-Grouse Site-Scale Seasonal Habitat Data Summary" (form S-1) to this form. Enter the appropriate mean ($\overline{x}$) and number (n) values where appropriate for the indicators in the column under $\overline{x}$.

   Preferred Forb Availability: Check the appropriate suitability category based on data derived using the "Sage-Grouse Preferred Forb Diversity Form." The suitability evaluation must be relative to abundance, diversity, and availability relative to ecological site potential. Write a short narrative in the notes section, based on the species observed and available site information.

4. Determine the appropriate suitability category and mark (✔) each indicator as suitable, marginal, or unsuitable.

5. Determine **site-scale suitability**. Overall suitability takes into consideration the relationship between the indicators and their relative importance. This evaluation is based on professional judgment using the indicators for guidance. Explain overall site suitability in the rationale section.

6. Indicate if **site potential** is a factor for a suitability description of marginal or unsuitable. Explain further in the rationale section.

7. Indicate **drought condition** using local weather station data or as reported for the region of concern on the National Weather Service website: www.ncdc.noaa.gov/oa/climate/research/us-drought-monthly.html.

8. Attach field data sheet(s) and photographs used for this site-scale description.

9. Provide a copy of this form to the state wildlife agency's sage-grouse coordinator.

Form S-5: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Riparian Summer/Late Brood-Rearing Habitat

Date:	County:	State:	Evaluator(s):
Population:			Home Range Name:
Land Cover Type:			

Site Type (circle one): Riparian Areas Wetland/Wet Meadows Springs Lakebeds All Other

Number of Transects:	Area (ha/ac) or Distance (km/mi) Sampled:

List UTM Coordinates (coordinates, zone, datum) of All Transects:

Habitat Indicator Suitability Range							
Habitat Indicator	$\bar{x}$ or n	**Suitable**	✔	**Marginal**	✔	**Unsuitable**	✔
Riparian Stability PFC (*n*) FAR (*n*) NF (*n*)		Majority of areas are in PFC		Majority of areas are FAR		Majority of areas are NF	
Preferred Forb Availability (relative to site potential) Number of Preferred Forb Species (*n*)		Preferred forbs are common with appropriate numbers of species present		Preferred forbs are common but only a few species are present		Preferred forbs are rare	
Availability of Sagebrush Cover (mean)		Sagebrush cover is adjacent to brood-rearing areas (<100 m)		Sagebrush cover is in close proximity to brood-rearing areas (100 to 275 m)		Sagebrush cover is unavailable (>275 m)	

Site-Scale Suitability	**Suitable**		**Marginal**		**Unsuitable**	

Drought Condition (circle one): Extreme Drought Severe Drought Moderate Drought Mid-Range Moderately Moist Very Moist Extremely Moist

Rationale for Overall Suitability Rating:

Form S-5: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Riparian Summer/Late Brood-Rearing Habitat Directions

1. Use this worksheet to interpret field data collected using the forb diversity method outlined in this appendix and summarized on the "Sage-Grouse Site-Scale Seasonal Habitat Data Summary" (form S-1).

2. Complete all location information at the top of the form. Be sure to list all UTM coordinates or other identifying feature of all sites being summarized. Most of the information should be self-explanatory except the following:

   Population: Identify the population with which the habitat is associated. This definition also includes small populations. Population names are found in figure 3.

   Home Range Name: Identify the home range area using a major drainage area or other distinguishing land feature (e.g., Little Lost River home range).

   Land Cover Type (Optional): Identify the wetland (Cowardin et al. 1979) or riparian type (regional classification systems) of the habitat sampled. This data may be important to record when more detailed descriptions of summer habitats are desired (i.e., with sites stratified by cover type).

   Site Type: Identify the type of habitat sites sampled.

   Number of Transects: Record the number of 50-m transects or sites measured within the land cover type.

   Area or Distance Sampled: Record the total area (indicating ha/ac) or distance for riparian areas (indicating km/mi) of the site type or land cover type sampled.

3. Transfer data from the "Sage-Grouse Site-Scale Seasonal Habitat Data Summary" (form S-1) to this form. Enter the appropriate mean ($\overline{x}$) and number (n) values and PFC data where appropriate for the indicators in the column under $\overline{x}$.

   Riparian Stability: Record the number of sampling sites that were in proper functioning condition (PFC), functional–at risk (FAR), or nonfunctional (NF) (Prichard et al. 1998, 2003). Current PFC data can be used, if available. If PFC data cannot be obtained from other sources or collected directly, then the other two indicators should be used to assess habitat suitability. Include lotic and lentic riparian habitats.

   Preferred Forb Availability: Check the appropriate suitability category based on data derived using the "Sage-Grouse Forb Diversity Data Form." The suitability evaluation must be relative to abundance, diversity, and availability relative to ecological site potential.

   Availability of Sagebrush Cover: Distance is measured from the edge of the riparian area to the edge of the nearest stand of mature sagebrush of sufficient extent to provide protective cover.

4. Determine the appropriate suitability category and mark (✔) each indicator as suitable, marginal, or unsuitable.

5. Determine **site-scale suitability**. Overall suitability takes into consideration the relationship between the indicators and their relative importance. This evaluation is based on professional judgment using the indicators for guidance. Explain overall site suitability in the rationale section.

6. Indicate **drought condition** using local weather station data or as reported for the region of concern on the National Weather Service website: www.ncdc.noaa.gov/oa/climate/research/us-drought-monthly.html.

7. Attach field data sheet(s) and photographs used for this site-scale description.

8. Provide a copy of this form to the state wildlife agency's coordinator for sage-grouse conservation.

Form S-6: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Winter Habitat

Date:	County:	State:	Evaluator(s):
Population:			Home Range Name:
Land Cover Type:			Ecological Site:
Number of Transects:			Area Sampled (ha/ac):
List UTM Coordinates (coordinates, zone, datum) of All Transects:			

Habitat Indicator Suitability Range							
Habitat Indicator	$\bar{x}$	**Suitable**	✔	**Marginal**	✔	**Unsuitable**	✔
Sagebrush Cover (mean)		≥10 %		5 to <10%		<5%	
Sagebrush Height (above snow) (mean)		≥25 cm		>10 to <25 cm		≤10 cm	

Site-Scale Suitability	**Suitable**		**Marginal**		**Unsuitable**	

Rationale for Overall Suitability Rating:

Form S-6: Sage-Grouse Site-Scale Habitat Suitability Worksheet – Winter Habitat Directions

1. Use this worksheet to interpret field data summarized on the "Sage-Grouse Site-Scale Seasonal Habitat Data Summary" (form S-1).

2. Complete all location information at the top of the form. Be sure to list all UTM coordinates or other identifying feature of all sites being summarized. Most of the information should be self-explanatory except the following:

   Population: Identify the population with which the habitat is associated. This definition also includes small populations. Population names are found in figure 3.

   Home Range Name: Identify the home range area using a major drainage area or other distinguishing land feature (e.g., Little Lost River home range).

   Land Cover Type: Identify the cover type of the data collected. Use plant alliances or associations (Reid et al. 2002) for sagebrush or grassland communities; use www.natureserve.org/explorer (International Classification of Ecological Communities) or other sampling strata to describe the habitat (e.g., percent sagebrush categories). Use the species symbol for dominant species in the overstory and understory (table B-1), for example ARTRW8 (alliance level – Wyoming big sagebrush) or ARTRW8/FEID (association level – Wyoming big sagebrush/Idaho fescue).

   Ecological Site: Refer to soil maps, range site guides, and ecological site descriptions where available and record the appropriate ecological site. Use the species symbol for dominant species in the overstory and understory.

   Number of Transects: Record the number of 50-m transects completed within the land cover type.

   Area Sampled: Record the total area (indicating ha/ac) of the land cover type within the administrative area assessed (e.g., pasture, allotment).

3. Transfer data from the "Sage-Grouse Site-Scale Seasonal Habitat Data Summary" (form S-1) to this form. Enter the mean ($\overline{x}$) for the indicators in the column under $\overline{x}$.

4. Determine the appropriate suitability category and mark (✔) each indicator as suitable, marginal, or unsuitable.

5. Determine **site-scale suitability**. Overall suitability takes into consideration the relationship between the indicators and their relative importance. This evaluation is based on professional judgment using the indicators for guidance. Explain overall site suitability in the rationale section.

6. Attach field data sheet(s) and photographs used for this site-scale description.

7. Provide a copy of this form to the state wildlife agency's sage-grouse coordinator.

Form S-7: Sage-Grouse Site-Scale Seasonal Habitat Site Suitability Summary

Date:	County:	State:	Evaluator(s):
Population:			Home Range Name:
Associated Leks:			

Seasonal Habitat Information						Suitability		
						Current	Future	
Seasonal Habitat	Land Cover Type	Ecological Site	Area (ha/ac) (*upland*)	Length (km/mi) (*riparian*)	Number of Sites (*leks, wet meadows, springs, etc.*)	**S**uitable, **M**arginal, **U**nsuitable	Site potential limiting?	Habitat components present?

Form S-7: Sage-Grouse Site-Scale Seasonal Habitat Site Suitability Summary Directions

1. Use this form to summarize site-scale seasonal habitat suitability descriptions (forms S-2 through S-6) for land cover types within a home range area.

2. Complete all location information at the top of the form. Most of the information should be self-explanatory except the following:

   Population: Identify the population with which the habitat is associated. This definition also includes small populations. Population names are found in figure 3.

   Home Range Name: Identify the home range area using a major drainage area or other distinguishing land feature (e.g., Little Lost River home range).

   Associated Leks: List the two largest occupied leks to which the breeding habitat is associated. Use identification numbers or names that are used in the statewide database.

3. Transfer data from the seasonal habitat suitability worksheets (forms S-2 through S-6) to this form.

   Seasonal Habitat: List one of the following: lek, nesting/early brood-rearing, summer/late brood-rearing, or winter, for each seasonal habitat summarized.

   Land Cover Type: Identify the land cover type of the seasonal habitat.

   Upland communities: Use plant alliances or associations (Reid et al. 2002) for sagebrush or grassland communities; use www.natureserve.org/explorer (International Classification of Ecological Communities) or other sampling strata to describe the habitat (e.g., percent sagebrush categories). Use the species symbol for dominant species in the overstory and understory (table B-1), for example ARTRW8 (alliance level – Wyoming big sagebrush) or ARTRW8/FEID (association level – Wyoming big sagebrush/Idaho fescue).

   Riparian or wetland communities: Use site type (riparian areas, wet meadows, springs) or more detailed classification using Cowardin et al. (1979) or riparian type (regional classification systems) to which the data pertain.

   Ecological Site: Refer to soil maps, range site guides, and ecological site descriptions where available and record the appropriate ecological site. Use the species symbol for dominant species in the overstory and understory.

   Area/Length/Number of Sites: Record the area for upland habitat (indicating ha/ac), linear length for riparian habitat (indicating km/mi), or number of sites (leks, wet meadows, springs, etc.) sampled.

   Current Suitability: Record the overall site-scale suitability as suitable (S), marginal (M), or unsuitable (U).

   Future Suitability: Record any site-scale ecological constraints for the cover type to provide habitat in the future. This information applies only to those sites that are currently providing marginal or unsuitable site-scale conditions.

   Site potential limiting?: If ecological site potential indicates that the site may provide suitable habitat in the future, record "No." If ecological site potential is limiting suitability, record "Yes."

   Habitat components present?: If there is sagebrush recruitment and forbs and perennial grasses are present in suitable amounts, record "Yes." If recruitment of these life forms is lacking, record "No."

Plot Metadata Form

Site:	Ownership:	Establishment Date:
Plot ID:		Visit Date:
Evaluator(s):		

GPS Coordinate System:				Datum :	Zone (if applicable):	Elevation: ☐m ☐ft
Transect	Azimuth	Length ☐m ☐ft		Latitude/Northing	Longitude/Easting	Slope (%)
			Start			
			Start			Aspect (˚)
			Start			

Directions to the Plot:

Population:	Home Range Name:
Land Cover Type:	Ecological Site:
Associated Leks:	Area (ha/ac) or Distance (km/mi) Sampled:
Site Info.: ☐ Arid Site ☐ Mesic Site	Seasonal Habitat:

PFC Status (riparian areas only): ☐ PFC ☐ FAR ☐ NF ☐ Unknown

Comments:

Plot Photos:

Photo	Description

Plot Metadata Directions

1. Complete all location information at the top of the sheet. Be sure to list UTM coordinates and other identifying features of the site. Most of the information should be self-explanatory except the following:

   Population: Identify the population with which the habitat is associated. This definition also includes small populations. Population names are found in figure 3.

   Home Range Name: Identify the home range area using a major drainage area or other distinguishing land feature (e.g., Little Lost River Home Range).

   Land Cover Type: Identify the land cover type of the data. Use plant alliances or associations (Reid et al. 2002) for sagebrush or grassland communities; www.natureserve.org/explorer (International Classification of Ecological Communities) or other sampling strata used to describe the habitat (e.g., % sagebrush categories). Use the species symbol (table B-1) for dominant species in the overstory and understory (Examples: ARTRW8 (alliance level – Wyoming big sagebrush) or ARTRW8/FEID (association level – Wyoming big sagebrush/Idaho fescue).

   Ecological Site: Refer to soil maps and range site guides, and ecological site descriptions where available and record the appropriate ecological site. Use the species symbol for dominant species in the overstory and understory.

   Associated Leks: List the two largest occupied leks to which the breeding habitat is associated. Use identification numbers or names that are used in the statewide database.

   Area or Distance Sampled: Record the total area (indicating ha/ac) or distance for riparian areas (indicating km/mi) of the site type or land cover type sampled.

   Site Info.:

      Arid Site: Applies to sagebrush ecological sites generally in the 25-30 cm (10-12 in) precipitation zone. Wyoming big sagebrush is a common big sagebrush subspecies for this type of site.

      Mesic Site: Applies to sagebrush ecological sites generally in a >30 cm (12 in) precipitation zone. Mountain big sagebrush is a common big sagebrush subspecies for this type of site.

   Seasonal Habitat: List one or more of the following, as appropriate: lek, nesting/early brood-rearing, summer/late brood-rearing, or winter.

2. Take photographs of the study site. At least one photograph must be taken at each transect/evaluation area. Photos will prove invaluable in locating evaluation areas in subsequent years. They will also be of substantial utility in the office when preparing evaluation documents and documenting habitat condition.

   a. Complete a photo card showing, at a minimum, the date, location, allotment, and transect number.
   b. With the photo card near the zero end of the tape, take a general photo of the area, sighting down the tape from eye level, showing landmarks in the background, if possible. A cover board or meter stick should be in the picture for a frame of reference.
   c. In a representative location along or near the tape, place the photo card near the base of a sagebrush plant, and take a tangential closeup photo from near ground level (2-3 ft) toward the shrub/ground interface, to document herbaceous conditions and cover. A cover board or meter stick should be in the picture for a frame of reference.
   d. Optional: Take one or more other closeups or panoramic photos as needed. A photo showing sagebrush canopy cover percent may also be desirable, following completion of the line intercept.

Line-Point Intercept Data Form

Page of	Date:	Plot ID:	Transect:
Evaluator(s):			
Azimuth:	Intercept (Point) Spacing: ☐cm ☐in	Height: ☐cm ☐in	

Pt.	Top Layer	Lower Layers			Soil Surface	Pt.	Top Layer	Lower Layers			Soil Surface
		Code 1	Code 2	Code 3				Code 1	Code 2	Code 3	
1						26					
2						27					
3						28					
4						29					
5						30					
6						31					
7						32					
8						33					
9						34					
10						35					
11						36					
12						37					
13						38					
14						39					
15						40					
16						41					
17						42					
18						43					
19						44					
20						45					
21						46					
22						47					
23						48					
24						49					
25						50					

% foliar cover = _____ top layer pts (1st col) x 2 = _____%

% bare ground = _____ pts (w/NONE over S) x 2 = _____%

Top layer codes: Species code, common name, or NONE (no cover).

Lower layers codes: Species code, common name, L (herbaceous litter), WL (woody litter, >5 mm (~1/4 in) diameter), VL (vagrant lichen).

Unknown Species Codes:

AF# = annual forb
PF# = perennial forb
AG# = annual grass
PG# = perennial grass
SH# = shrub
TR# = tree

Soil Surface (do not use litter):

G = gravel (≤5 mm or ~1/4 in)
R = rock (>5 mm or ~1/4 in)
BR = bedrock
EL = embedded litter
D = duff
M = moss
LC = visible lichen crust on soil
S = soil

Line-Point Intercept Directions

Note: The HAF site-scale protocol for line-point intercept is the same as the BLM's core method. Directions for the method are given below, but readers can refer to Herrick et al. (2005) (or the most current version) for more detail.

Equipment:

Tape, 5 m	Stakes for tape (at least two spikes; old, medium to large screwdrivers work well)
Pin flag or pointer or other point intercept device: straight piece of wire or rod at least 1 m long and less than 2.5 mm in diameter	Meter stick (for measuring shrub and grass/forb heights)
Digital camera (5 megapixel minimum), extra camera battery	Photo cards and markers or small dry-erase board and marker
Topographic map and aerial photographs with project area, general cover types, and pasture boundaries delineated	GPS unit, compass
Forms and/or electronic data entry device with extra battery, pencils	Ecological Site Guides
Calculator	

Protocol:

1. Complete all metadata information at the top of the LPI field form for each transect, making sure that the plot identification information (i.e., plot number) matches that recorded on the overall plot metadata form. If more than 50 points are being recorded on a transect, attach additional forms as needed.

2. Pull out the tape and anchor each end with a steel pin. Keep measuring tape taught and straight. Keep measuring tape as close to the ground as possible (thread under shrubs using a steel pin as a needle), but not so close that it disturbs the soil surface or affects the natural way the vegetation stands below the tape (figures B-1 and B-2).

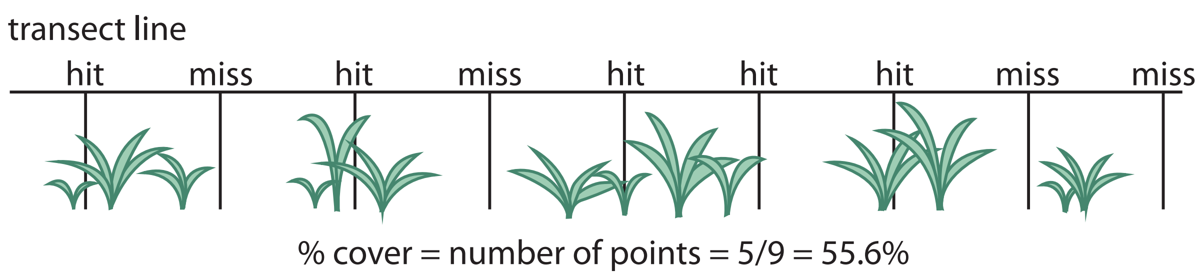


Figure B-1. The line-point intercept method can be used to measure foliar cover and vegetation height of all grass, forb, and shrub species at a site or foliar cover of a single life form (e.g., sagebrush cover for winter habitat areas).

Figure B-2. Measuring plant species using the line-point intercept technique (pin size exaggerated to emphasize method).

3. Begin at the "0" end of the tape.

4. Working from left to right, record cover at each 1 m mark (or ½ m and 1 m mark for 100 points per transect). Begin recording at the first ½ m or 1 m mark depending on the number of points desired. Always stand on the same side of the line. Drop a pin flag to the ground from a standard height next to the tape. Keep the pin vertical. Make a "controlled drop" of the pin from the same height each time. Position the pin so its lower end is several centimeters above the vegetation, release it and allow it to slip through the hand until it hits the ground. A low drop height minimizes "bounces" off of vegetation but increase the possibility for bias. Do not guide the pin all the way to the ground. It is more important for the pin to fall freely to the ground than to fall precisely on the mark.

5. A laser with a bubble level can be used instead of the pin. This tool is useful in savannas where plant layers may be above eye level.

6. Once the pin flag is flush with the ground, record every plant species it intercepts:

    a. Record the species of the uppermost or first stem, leaf, or plant base intercepted in the "Top Layer" column using the USDA PLANTS database species code (http://plants.usda.gov), a four- to six-letter code based on the first two letters of the genus and species, subspecies, or the common name. If no leaf, stem, or plant base is intercepted, record "NONE" in the "Top Layer" column. Woody sagebrush plants should be identified to the subspecies.
    b. Record all additional species intercepted by the pin in the order that they are intercepted from top to bottom.
    c. Record all foliage whether alive or dead, but denote dead vegetation by using the appropriate checkbox in an electronic data collection database or circling the species on the data form. If both alive and dead canopy for a species is hit on the same point, record the live canopy. Sagebrush indicators for sage-grouse habitat are calculated from only live canopy hits and do not include dead stems of shrubs. Residual plant cover can be very important for sage-grouse nesting, so it is also important to denote live versus standing dead herbaceous vegetation on the field form. See Connelly et al. (2003), Monitoring of Greater Sage-grouse Habitats and Populations; http://sagemap.wr.usgs.gov/docs/grouse_habitat_book.pdf; and http://www.cnr.uidaho.edu/range357/notes/cover.pdf) for discussions on cover.
    d. Record each plant species only once, the first time it is intercepted, even if it is intercepted several times.
    e. Record the following codes for lower layers: "L" for herbaceous litter, if present (litter is defined as detached stems, roots, and leaves); "WL" for detached woody litter > 5 mm (~1/4 in) in diameter; or "VL" for vagrant lichen.
    f. If a sagebrush plant is intercepted, record the shape of the sagebrush as "S" for spreading or "C" for columnar (figure 13).

7. Record a species code (if the pin flag intercepts a plant base) or another soil surface code in the "Soil Surface" column.

    a. Use the following abbreviations for soil surface type: G = gravel (≤5 mm diameter or ~1/4 in), R = rock (>5 mm diameter or ~1/4 in), BR = bedrock, EL = embedded litter, D = duff, M = moss, LC = visible lichen crust on soil, and S = soil, without any other soil surface code.
    b. Record plant species (or life form, if species is unknown) when present. For unidentified plants, use the following codes and a sequential number: AF# = annual forb, PF# = perennial forb, AG# = annual grass, PG# = perennial grass, SH# = shrub, and TR# = tree.
    c. An intercept with a plant base is defined as when the end of the pin rests either on or immediately adjacent to and touching living or dead plant material that is rooted in the soil.
    d. Record embedded litter (EL) only where removal of the litter would leave an indentation in the soil surface or would disturb the soil surface, breaking the soil crust. Record duff (D) when there is no clear boundary between litter and mineral soil and litter is not removed during typical storms (occurring annually).
    e. Record lichen (LC) only if it is growing on soil, but not if it is attached to rock substrate. If mosses and lichens are recorded to species, write the species code in the "Soil Surface" column.

Vegetation Height/Sagebrush Shape Data Form

Page of	Date:	Plot ID:	Transect:
Evaluator(s):			
Azimuth:		Intercept (Point) Spacing: ☐cm ☐in	Height: ☐cm ☐in

Point	Species	Woody Height	Sagebrush Shape	Species	Grass Height	Species	Forb Height

Average sagebrush height =

Average grass height =

Average forb height =

Vegetation Height/Sagebrush Shape Directions

Note: The HAF site-scale protocol for vegetation height is similar to the BLM's core method, but there are important differences between the two methods. Data collected using the HAF method can be used to supplement the BLM's core method for assessing the site-scale height indicators of sage-grouse habitat.

Protocol:

1. Record the species of woody and herbaceous plants for which the heights will be recorded.
2. Measure plants heights at the ½ m or 1 m intervals per transect. Do not record the height of the same plant twice.
3. Record the height of plants 0-2 m to the nearest centimeter and plants >2 m to the nearest 30 cm (~12 in).
4. For shrubs, record the maximum height in cm/in of the live portion of the shrub that is touched by the pin, excluding flower or seed stalks.
5. Record the shape of sagebrush only: S = spreading or C = columnar.
6. For perennial grasses and forbs, record the droop height (i.e., the highest point measured with no straightening by the observer or maximum natural height, figure B-3) of the tallest perennial grass or forb plant that is touched by the pin.
7. Woody or herbaceous litter are not measured.

Figure B-3. Grass and forb height measurements. Record natural or "droop" height of grasses and forbs. Note the dashed red reference line.

Line-Point Intercept Data Summary

Page **of**	**Plot:**	**Transect:**
Evaluator(s):		
Shrubs	**Forbs**	**Grasses**
Sagebrush Cover *# Hits* ________ *%* ______	**Perennial Forb Cover** *# Hits* ________ *%* ______	**Perennial Grass Cover** *# Hits* ________ *%* ______
Other Shrub Cover *# Hits* ________ *%* ______	**Annual Forb Cover** *# Hits* ________ *%* ______	**Annual Grass Cover** *# Hits* ________ *%* ______
Sagebrush Shape (n) *S* ________ *C* ________	**Total Forb Cover** *# AF+PF Hits* ________ *%* ______	**Total Grass Cover** *# AG+PF Hits* ________ *%* ______
Avg. Sagebrush Height (cm/in)	**Avg. PF Height (cm/in)**	**Avg. PG Height (cm/in)**
Comments:		

Sage-Grouse Habitat Indicator Calculations – Line-Point Intercept Data Summary Directions

Once the data has been collected, calculate the sage-grouse habitat indicators as described below. If using a tablet, computer, or other electronic data collection device, these indicators may be calculated automatically. If not, summarize the data and write the indicator calculations at the top of your field forms.

Cover of shrubs, forbs, and grasses:

- *Sagebrush Cover: Hits* = # of sagebrush hits, *% cover* = # points where a sagebrush was hit divided by the total number of transect points. Multiply the result by 100.
- *Other Shrub Cover: Hits* = # of total shrub hits, excluding sagebrush, *% cover* = # of points where a shrub was hit divided by the total number of transect points. Multiply the result by 100.
- *Perennial Forb Cover: PF Hits* = # of perennial forb hits, *% cover* = # of hits divided by total number of transect points. Multiply the result by 100.
- *Annual Forb Cover: AF Hits* = # of annual forb hits, *% cover* = # of hits divided by total number of transect points. Multiply the result by 100.
- *Total Forb Cover: PF+AF Hits* = # of perennial and annual forb hits, *% cover* = # total forb hits divided by total number of transect points. Multiply the result by 100.
- *Perennial Grass Cover: PG Hits* = # of perennial grass hits, *% cover* = # of hits divided by total number of transect points. Multiply the result by 100.
- *Annual Grass Cover: AG Hits* = # of annual grass hits, *% cover* = # of hits divided by total number of transect points. Multiply the result by 100.
- *Total Grass Cover: AG+PG Hits* = # of annual and perennial grass hits, *% cover* = # total grass hits divided by total number of transect points. Multiply the result by 100.

Height of shrubs, forbs, and grasses:

- *Avg. Sagebrush Height* = sum of all sagebrush recorded heights divided by total number of sagebrush plants measured.
- *Avg. Perennial Forb (PF) Height* = sum of all perennial forb recorded heights divided by total number of perennial forbs measured.
- *Avg. Perennial Grass (PG) Height* = sum of all perennial grass recorded heights divided by total number of perennial grass plants measured.
- *Note:* Relative to perennial forbs, it is recommended the suitability rating should focus on the cover estimates and preferred forb availability ratings rather than on height due to the variability in heights that can be encountered between forbs and grasses. However, average perennial forb height and/or average perennial forb and grass height (combined) can be calculated, if desired, to provide additional context to the description of the assessment area.
- *Sandberg bluegrass (or similar species):*

    1. Summarize cover and height for perennial grasses, excluding Sandberg bluegrass or similar short-statured perennial grasses.
    2. Summarize cover and height for Sandberg bluegrass.
    3. Summarize cover and height inclusive of all perennial grasses.

    Because shorter-statured perennial grasses such as Sandberg bluegrass may influence cover and height averages especially where abundant, the authors recommend that perennial grass metrics be summarized using all three methods, to provide additional context for the perennial grass suitability rating. For example, if cover, and height for perennial grasses, excluding Sandberg bluegrass (#1), are within the range of the suitable category in the HAF, then consider a ranking of "suitable" for the perennial grass indicator. However, if average cover (regardless of height) of these perennial grasses is not within the suitable category, use the cover and height averages for all perennial grasses, including Sandberg bluegrass (#3). Then, use the cover and height averages for the non-Sandberg perennial grasses (#1), as well as for Sandberg bluegrass itself (#2), to inform the rationale for the rating of the perennial grass indicator. Also, consider the capability of the site to provide species composition, cover, and structure for productive sage-grouse habitat on an annual basis.

Sagebrush shape:

- *Sagebrush Shape* = total # of sagebrush plants of each shape, spreading (S) or columnar (C), divided by total number of sagebrush plants measured.

Line Intercept and Daubenmire Frames Data Form (Electronic Version)

Date:	**County:**	**State:**	**Evaluator(s):**
Population:			**Home Range Name:**
Land Cover Type:			**Associated Leks:**
Ecological Site:			**Seasonal Habitat:**
Transect #:			**Site Info. (circle one):** Arid Site Mesic Site
UTM (coordinates, zone, datum):			

Transect Data Summary (see directions)		
Shrubs	**Forbs**	**Grasses**
Sagebrush Cover (line intercept) % ____________	**Perennial Forb Cover** % ____________	**Perennial Grass Cover** % ____________
Avg. Sagebrush Height (cm)	**Annual Forb Cover** % ____________	**Annual Grass Cover** % ____________
Sagebrush Shape (n) Spreading :________ Columnar:_________		
Other Shrub Cover % ____________	**Total Forb Cover** % ____________	**Total Grass Cover** % ____________
	Avg. PF Height (cm):	**Avg. PG Height (cm):**

Line Intercept Shrub Cover				
Shrub Species	**Intercept Start**	**Intercept End**	**Total Length**	**% Cover by Species**

Line Intercept and Daubenmire Frames Data Form (Paper Version)

Date:	County:	State:	Evaluator(s):
Population:			Home Range Name:
Land Cover Type:			Associated Leks:
Ecological Site:			Seasonal Habitat:
Transect #:			Site Info. (circle one): Arid Site Mesic Site
Area Sampled (ha/ac):			UTM (coordinates, zone, datum):

Transect Data Summary (see directions)		
Shrubs	Forbs	Grasses
Sagebrush Cover (line intercept) % ______	Perennial Forb Cover % ______	Perennial Grass Cover % ______
Avg. Sagebrush Height (cm)	Annual Forb Cover % ______	Annual Grass Cover % ______
Sagebrush Shape (n) Spreading: ______ Columnar: ______		
Other Shrub Cover % ______	Total Forb Cover % ______	Total Grass Cover % ______
	Avg. PF Height (cm):	Avg. PG Height (cm):

Species Name	Shrub Species							Notes
								Totals
Totals								
% Cover								

Daubenmire Cover Class and Vegetation Height Data Form (recorded at 2-m intervals)

Date:	County:	State:	Evaluator(s):
Transect #:		UTM (coordinates, zone, datum):	

Daubenmire Frame Number																										
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	Notes
Ground Cover 1																										
Ground Cover 2																										
Ground Cover 3																										
Ground Cover 4																										
PF Ht																										
PG Ht																										
Other Shrub Ht																										
ARTR Ht																										
ARTR Shape																										

*Cover Class 1=0-5%, 2=>5-25%, 3=>25-50%, 4=>50-75%, 5=>75-95%, 6=>95-100%

Line Intercept and Daubenmire Frame Method Directions

Equipment:

Tape, 50 m	Stakes for tape (at least two spikes; old, medium to large screwdrivers work well)
Daubenmire frame 20 x 50 cm	Meter stick (for measuring shrub and grass/forb heights)
Digital camera, extra camera battery	Photo cards and markers or small dry-erase board and marker
Topographic map with project area, general cover types, and pasture boundaries delineated	Aerial photographs
Ecological Site Guides	GPS unit, compass
Clipboard, data forms and/or data logger with extra battery, pencils	Calculator

Protocol:

- Seasonal habitat has been stratified by land cover types prior to field evaluation (see chapter II for more directions).
- Conduct an appropriate number of transects in each seasonal habitat by each land cover type. Repeat all steps for each transect.

1. Complete all metadata information at the top of the appropriate field forms for each transect, making sure that the plot identification information (i.e., plot number) matches that recorded on the overall plot metadata form. If more than 25 Daubenmire plots are being recorded on a transect, attach additional forms as needed. Most of the information should be self-explanatory except the following:

   Population: Identify the population with which the habitat is associated. This definition also includes small populations. Population names are found in figure 3.

   Home Range Name: Identify the home range area using a major drainage area or other distinguishing land feature (e.g., Little Lost River home range).

   Associated Leks: List the two largest occupied leks to which the breeding habitat is associated. Use identification numbers or names that are used in the statewide database.

   Land Cover Type: Identify the cover type of the data collected. Use plant alliances or associations (Reid et al. 2002) for sagebrush or grassland communities; use www.natureserve.org/explorer (International Classification of Ecological Communities) or other sampling strata to describe the habitat (e.g., percent sagebrush categories). Use the species symbol for dominant species in the overstory and understory (table B-1), for example, ARTRW8 (alliance level – Wyoming big sagebrush) or ARTRW8/FEID (association level – Wyoming big sagebrush/Idaho fescue).

   Ecological Site: Refer to soil maps, range site guides, and ecological site descriptions where available and record the appropriate ecological site. Use the species symbol for dominant species in the overstory and understory.

   Seasonal Habitat: List one of the following: lek, nesting/early brood-rearing, summer/late brood-rearing, or winter.

   Site Info:

   Arid Site: Applies to sagebrush ecological sites generally in the 25-30 cm (~10-12 in) precipitation zone. Wyoming big sagebrush is a common big sagebrush subspecies for this type of site.

   Mesic Site: Applies to sagebrush ecological sites generally in a >30 cm (12 in) precipitation zone. Mountain big sagebrush is a common big sagebrush subspecies for this type of site.

   Transect #: Assign a unique identifier to each transect within the land cover type.

   Area Sampled: Record the total area (indicating ha/ac) or distance for riparian areas (indicating km/mi) of the site type or land cover type sampled.

2. Anchor the tape with a steel pin and pull the tape out 50 meters. Keep the tape as taught and straight as possible. Anchor the tape on the far end. For smaller cover type inclusions or stringers or other unique situations, the transect length may be increased or decreased, as appropriate, to adequately sample the site. This will necessitate modifying the sampling distance for Daubenmire frames along the tape to accommodate 25 frames.

3. Begin at the "0" end of the tape.

4. On the data form, record **shrub cover** by species and subspecies using the line intercept method. Two forms are provided. The electronic version provides an example of data to be collected when using a laptop computer or data logger. The paper version is for collecting data via nonelectronic means.

    a. For the entire length of the line, determine the *intercept length* of any shrub species that touches the line. Only live portions of the shrub canopy are recorded. Intercept length is the portion of the transect length intercepted by the shrub, measured by a perpendicular projection of the shrub foliage over the line (figure B-4).

    b. List all cover increments for each species measured to the nearest 1 cm. Ignore spaces or gaps in the canopy *less than 5 cm* across. Gaps in the live canopy in excess of 5 cm *will not* be included as canopy intercepts (figure B-5). Record only live (leaves, live stems, and shrub trunk) canopy cover.

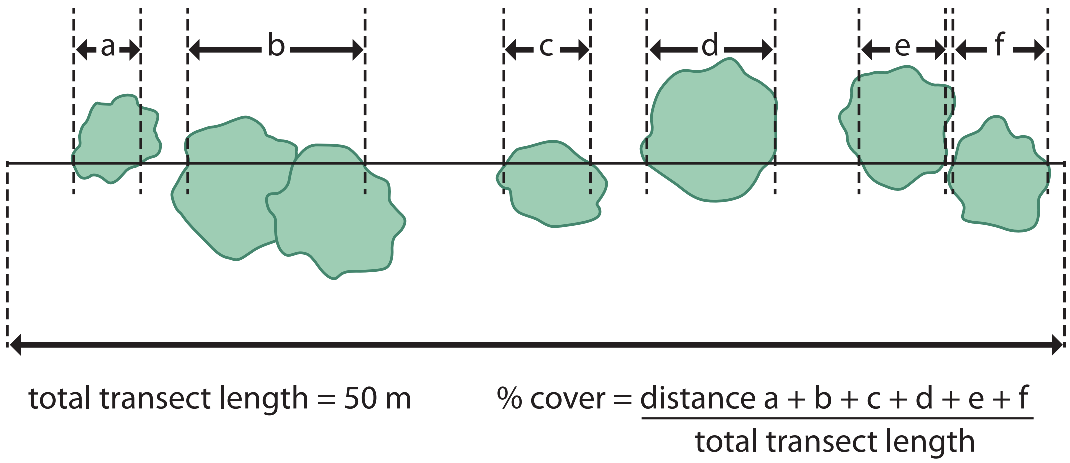


Figure B-4. The line intercept method can be used to measure canopy cover of sagebrush species.

Figure B-5. Measuring shrub canopy cover using the line intercept method. Group sagebrush with gaps smaller than 5 cm. Record sections of sagebrush separated by greater than 5 cm as separate intercepts.

5. Estimate **cover class** and **vegetation height** using the Daubenmire method at each 2-m increment (n = 25 plots per transect) along the tape:

    a. Place a 20 x 50 cm Daubenmire frame (figure B-6) along the tape with the long axis perpendicular to the tape (figure B-7). For each plot, estimate and record cover class for annual forbs, perennial forbs, annual grasses, and perennial grasses by species (based on Connelly et al. 2003):

Cover classes:	1 = 0-5%	midpoint of range 2.5%
	2 = >5-25%	midpoint of range 15%
	3 = >25-50%	midpoint of range 37.5%
	4 = >50-75%	midpoint of range 62.5%
	5 = >75-95%	midpoint of range 85%
	6 = >95-100%	midpoint of range 97.5%

    b. Count plants providing cover over the plot, regardless of if they are rooted in the plot or not.
    c. Record the height in cm of the nearest sagebrush plant (or other shrub species if no sagebrush is present) that is overhanging the Daubenmire frame.
    d. Record the shape of the nearest sagebrush plant that is overhanging the Daubenmire frame: S = spreading or C = columnar (figure 13).
    e. Record the maximum "natural" or "droop height" in cm of the tallest perennial grass and perennial forb overhanging the Daubenmire frame (natural = the highest point of a leaf or seed stalk is measured with no straightening by the observer (figure B-3). This includes seed stalks or inflorescences.

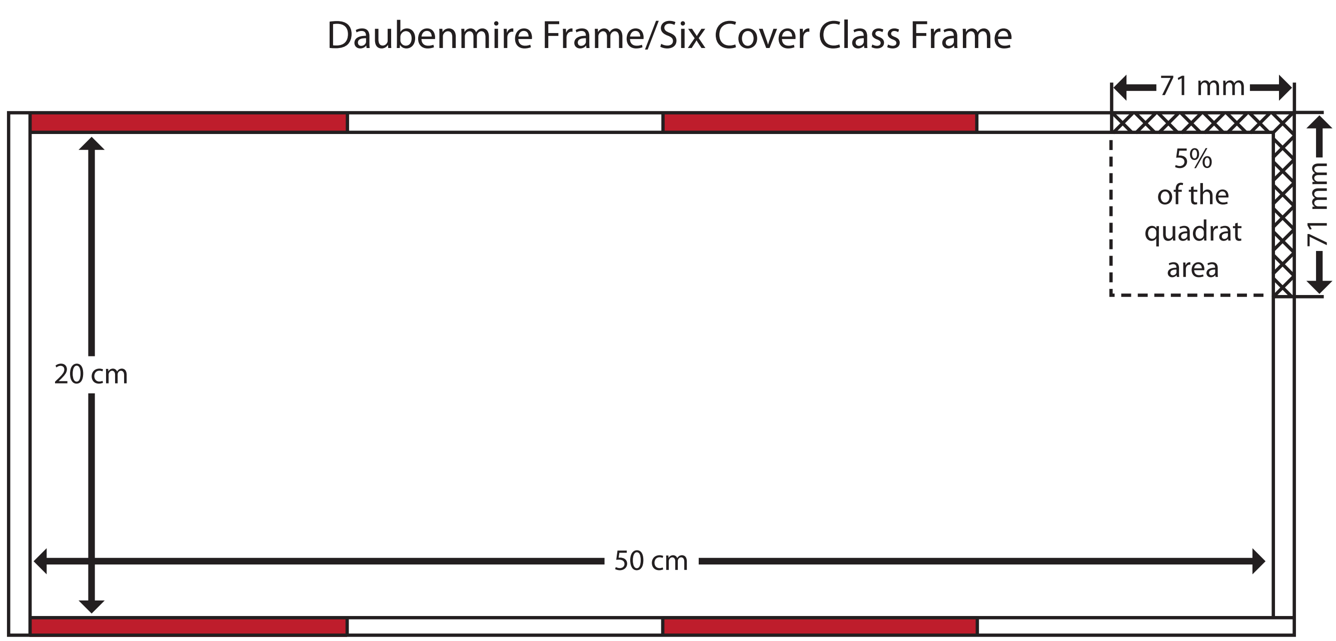


Figure B-6. The Daubenmire frame is used for estimating grass and forb canopy covers. Estimate canopy cover class of species rooted within or overhanging the frame using lines on the frame as guides.

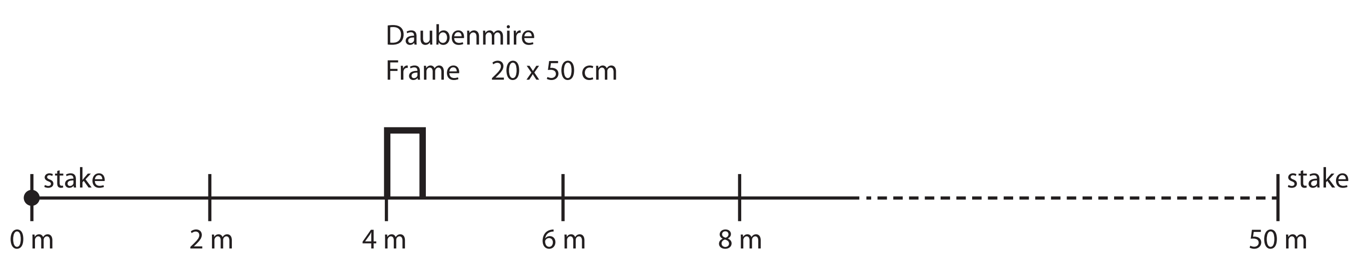


Figure B-7. A line transect with Daubenmire frames positioned every 2 meters.

6. Summarize the data under Line Intercept Shrub Cover:

   a. **Shrub Species:**
      - *Total* = sum of intercept lengths for each shrub species.
      - *% Cover* = total shrub intercept length by species divided by full transect length.

   b. **All Shrubs:**
      - *% Cover* = sum of above % cover calculations by species. The total could exceed 100% if the intercepts of overlapping canopies are recorded.

7. Summarize the data at the top of the form:

   a. **Shrubs:**
      - *Sagebrush Cover: % Cover* = sum of % covers of all sagebrush species listed under Shrub Species in the Cover section.
      - *Avg. Sagebrush Height* = sum of all sagebrush recorded heights divided by total number of sagebrush plants measured in the Vegetation Height section.
      - *Sagebrush Shape* = total # of sagebrush plants of each shape, spreading (S) and columnar (C).
      - *Other Shrub Cover: % Cover* = sum of % covers of all shrub species listed under All Shrubs in the Cover section.

   b. **Forbs:**
      - *Perennial Forb Cover: PF % Cover* = number of plots with perennial forbs in each of the six cover classes, multiplied by the midpoint of each cover class, added together as the sum of products for all cover classes, divided by total number of plots sampled on the transect (e.g., [(15 plots in cover class 1 * 2.5 midpoint) + (10 plots in cover class 2 * 15 midpoint)] / 25 = 7.5% canopy cover).
      - *Annual Forb Cover: AF % Cover* = number of plots with annual forbs in each of the six cover classes, multiplied by the midpoint of each cover class, added together as the sum of products for all cover classes, divided by total number of plots sampled on the transect (e.g., [(15 plots in cover class 1 * 2.5 midpoint) + (10 plots in cover class 2 * 15 midpoint)] / 25 = 7.5% canopy cover).
      - *Total Forb Cover: PF+AF % Cover* = sum of *PF % Cover* and *AF % Cover* (e.g., 7.5 + 7.5 = 15% canopy cover).
      - *Avg. PF Height* = sum of all perennial forb heights recorded divided by the total number of perennial forb plants measured. Relative to perennial forbs, the suitability rating should focus on the cover estimates and preferred forb availability ratings rather than on height due to the variability in heights that can be encountered between forbs and grasses. However, average perennial forb height and/or average perennial forb and grass height (combined) can be calculated, if desired, to provide additional context to the description of the assessment area.

   c. **Grasses:**
      - *Perennial Grass Cover: PG % Cover* = number of plots with perennial grasses in each of the six cover classes, multiplied by the midpoint of each cover class, added together as the sum of products for all cover classes, divided by total number of plots sampled on the transect.
      - *Annual Grass Cover: AG % Cover* = number of plots with annual grasses in each of the six cover classes, multiplied by the midpoint of each cover class, added together as the sum of products for all cover classes, divided by total number of plots sampled on the transect.
      - *Total Grass Cover: PG+AG % Cover* = sum of *PG % cover* and *AG % cover*.
      - *Avg. PG Height* = sum of all perennial grass recorded heights divided by total number of perennial grass plants measured.
      - *Sandberg bluegrass* (*or similar species*):

        1. Summarize cover and height for perennial grasses, excluding Sandberg bluegrass, or similar short-statured perennial grasses.
        2. Summarize cover and height for Sandberg bluegrass.
        3. Summarize cover and height inclusive of all perennial grasses.

        Because shorter-statured perennial grasses such as Sandberg bluegrass may influence cover and height averages especially where abundant, the authors recommend that perennial grass metrics be summarized using all three methods to provide additional context for the perennial grass suitability rating. For example, if cover and height for perennial grasses, excluding Sandberg bluegrass (#1), are within the range of the suitable category in the HAF, then consider a ranking of "suitable" for the perennial grass indicator. However, if average cover (regardless of height) of these perennial grasses is not within the suitable category, use the cover and height averages for all perennial grasses, including Sandberg bluegrass (#3). Then, use the cover and height averages for the non-Sandberg perennial grasses (#1), as well as for Sandberg bluegrass itself (#2), to inform the rationale for the rating of the perennial grass indicator. Also, consider the capability of the site to provide species composition, cover, and structure for productive sage-grouse habitat on an annual basis.

8. OPTIONAL: Complete the "Sage-Grouse Forb Diversity Data Form," or use the forb data collected in the Daubenmire frame to compile forb information for the site. Later, write a short narrative describing forb diversity relative to the site.

9. OPTIONAL: Record ground cover at each of the four outside corners of the Daubenmire frame in the four ground cover cells for each plot. See the codes below:

G = gravel (≤5 mm or ~1/4 in)
R = rock (>5mm or ~1/4 in)
BR = bedrock
D = duff (when there is no clear boundary between litter and mineral soil and litter is not removed during typical storms (occurring annually))
M = moss
LC = visible lichen crust on soil
S = soil
L = herbaceous litter (≤5 mm or ~1/4 in; defined as detached stems, roots, and leaves)
WL = woody litter (>5mm or ~1/4 in)
EL = embedded litter (where removal of the litter would leave an indentation in the soil surface or would disturb the soil surface, breaking the soil crust)
V = live vegetation

Sage-Grouse Forb Diversity Data Form

Date:	County:	State:	Evaluator(s):
Population:			Home Range Name:
Land Cover Type:			Ecological Site:
Associated Leks:			Seasonal Habitat:
Transect #:			Site Info. (circle one): Arid Site Mesic Site

Forb Diversity (see directions)

Type	Species	Forb Plot Number																									
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	T

Type: P= preferred, N=noxious, I= invasive, O=other. **Note:** The forb type can be determined later in the office or via automated approaches.

Sage-Grouse Forb Diversity Summary Form

Date:	County:	State:	Evaluator(s):
Population:			Home Range Name:
Land Cover Type:			Ecological Site:
Associated Leks:			Transect #:
Area (ha/ac) Sampled:			Site Info. (circle one): Arid Site Mesic Site
Seasonal Habitat:			UTM:
PFC Status (riparian areas only, circle one):	PFC	FAR	NF Unknown

Transect Data Summary (see directions)

Preferred Forb Species	Noxious Weeds	Invasive Annual Forbs	Other Forbs
Total Species (#): ___________	Total Species (#): ___________	Total Species (#): ___________	Total Species (#): ___________
List major species:	List major species:	List major species:	List major species:

Comments (describe the diversity, availability, and relative abundance of preferred forbs in relation to site potential):

Sage-Grouse Forb Diversity Data and Summary Form Directions

Equipment:

Tape, 50 m	Stakes for tape (at least two spikes; old, medium to large screwdrivers work well)
Meter stick (for delineating 180-degree arc)	GPS unit
Pencils, clipboard, and plant identification guide; a local plant species list may be helpful	Calculator

Protocol:

- This worksheet should be used to collect forb availability and diversity information at various breeding and summer habitat sites.
- Forb availability should be evaluated as close to the end of nesting as possible (May-June) to allow for easier identification of plant species, as well as more relevant application to the evaluation of breeding habitat. For low elevation areas, this will be May; for higher elevation areas, it will be June.
- Seasonal habitat has been stratified by land cover types prior to field evaluation (see chapter II for additional discussion).
- Conduct an appropriate number of transects in each seasonal habitat by each land cover type, in association with the LPI transects, as appropriate. Repeat all steps for each transect.
- If a more in-depth, quantitative data collection method (e.g., density or other) is desired by the interdisciplinary team, use the Daubenmire method, by species.

1. Fill out all location information at the top of the sheet (transfer information from the LPI or LIDF data form if used on the same transect line). Be sure to list UTM coordinates or other identifying features of the site. Most of the information should be self-explanatory except the following:

   Population: Identify the population with which the habitat is associated. This definition also includes small populations. Population names can be found in figure 3.

   Home Range Name: Identify the home range area using a major drainage area or other distinguishing land feature (e.g., Little Lost River home range).

   Land Cover Type: Identify the cover type of the data collected:

   > **Upland Communities:** Use plant alliances or associations (Reid et al. 2002) for sagebrush or grassland communities; use www.natureserve.org/explorer (International Classification of Ecological Communities) or other sampling strata used to describe the habitat (e.g., percent sagebrush categories). Use the species symbol for dominant species in the overstory and understory (table B-1), for example, ARTRW8 (alliance level – Wyoming big sagebrush) or ARTRW8/FEID (association level – Wyoming big sagebrush/Idaho fescue).
   >
   > **Riparian or Wetland Communities:** Use site type (riparian areas, wet meadows, springs) or more detailed classification using Cowardin et al. (1979), or riparian type (regional classification systems) to which the data pertain.

   Ecological Site: Refer to soil maps, range site guides, and ecological site descriptions where available, and record the appropriate ecological site. Use the species symbol for dominant species in the overstory and understory.

   Associated Leks: List the two largest occupied leks to which the breeding habitat is associated. Use identification numbers or names that are used in the statewide database.

   Seasonal Habitat: List one of the following: lek, nesting/early brood-rearing, summer/late brood-rearing, or winter.

   Transect #: Assign a unique number to each transect within the land cover type (use the same transect number as for the LPI or LIDF data form).

   Site Info:

   > **Arid Site:** Applies to sagebrush ecological sites generally in the 25-30 cm (10-12 in) precipitation zone. Wyoming big sagebrush is a common big sagebrush subspecies for this type of site.
   >
   > **Mesic Site:** Term applies to sagebrush ecological sites generally in a >30 cm (>12 in) precipitation zone. Mountain big sagebrush is a common big sagebrush subspecies for this type of site.

2. At every 2 meters, record the presence of forbs, by species (in the species column on the form), which are rooted within a 1-meter radius, 180-degree arc, centering on the respective 2-meter mark. Place a check in the box on the form for the appropriate plot if the species is present. See figure B-8 for transect layout.

3. In the office later, or via automated means, annotate the type of forbs encountered as to whether they are preferred (by sage-grouse), noxious, invasive, or other. Invasive forbs are considered of low palatability and ecologically undesirable. Noxious weeds are limited to listed state weeds. Other forbs are any forbs that are not considered to be preferred, noxious, or invasive (e.g., ecologically desirable, but unpalatable forbs such as *Lupinus* spp.) Other forbs may not be preferred by sage-grouse as forage, but may still provide substrate for insects important to young sage-grouse. For preferred forbs, see table B-1.

    a. Calculate the total occurrences by species and sum by forb type (preferred, noxious, invasive, and other) on the "Sage-Grouse Forb Diversity Summary Form." In the comments section of the form, describe, relative to site potential, the general availability, diversity (number of species), and relative abundance of preferred forb species, based on the number of species encountered on the transect and number of plots with preferred forbs. Also discuss other, noxious, and invasive forbs as appropriate. Use professional judgment and augment with other forb information that may have been collected from point intercept or Daubenmire transects.

    b. Use this information to help describe preferred forb availability for breeding and summer habitat evaluations.

4. Provide any additional pertinent information that describes the site in the comments section.

5. Attach this form to the other field data sheet(s) (LPI or LIDF) used for this transect.

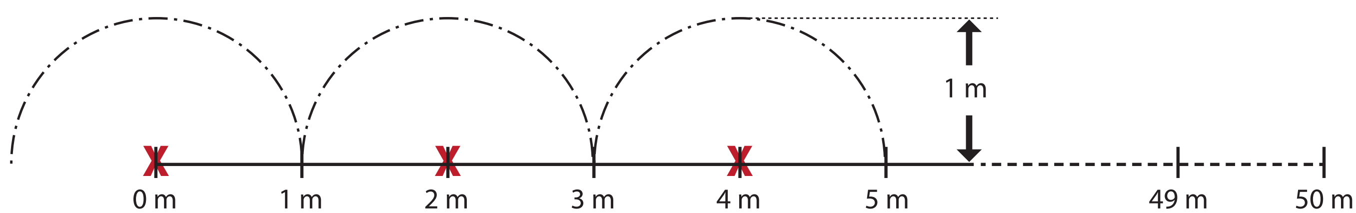


Figure B-8. Forb diversity transect layout. At each 2-m increment, use a 1-m stick to scribe a 180-degree arc. On the "Sage-Grouse Forb Diversity Data Form," record forb species that are rooted within the arc for a total of 25 plots along each transect.

Table B-1. Sagebrush community vegetation species and preferred forbs for sage-grouse. To be used for LPI, LIDF, and forb diversity data collection. Space is provided for the addition of local species. P = preferred forb, W = (noxious) weeds, I = invasive annuals, O = other forbs, N/A = not applicable. Species symbols are current as of 10-01-2013. See the USDA PLANTS database for the most up-to-date species symbols. Other forbs may be palatable at the cotyledon or bud stage.

Scientific Name	Common Name	Symbol	Most Likely Category
SHRUBS			
Dwarf sagebrush			
Artemisia arbuscula	Low sagebrush	ARAR8	N/A
A. arbuscula spp. *longicaulis*	Lahontan sagebrush	ARARL3	N/A
A. arbuscula spp. *longiloba*	Early sagebrush	ARARL	N/A
A. bigelovii	Bigelow sage	ARBI3	N/A
A. nova	Black sagebrush	ARNO4	N/A
A. papposa	Fuzzy sage	ARPA16	N/A
A. pygmaea	Pygmy sagebrush	ARPY2	N/A
A. rigida	Stiff sagebrush	ARRI2	N/A
A. spinescens *Syn = Picrothamnus desertorum*	Bud sagebrush	ARSP5/ PIDE4	N/A
A. tripartita spp. *rupicola*	Wyoming threetip sagebrush	ARTRR2	N/A
Tanacetum nuttallii *Syn = Sphaeromeria argentea*	Silver chickensage	TANU2/ SPAR2	N/A
Tall sagebrush			
A. cana spp. *bolanderi*	Bolander's silver sagebrush	ARCAB3	N/A
A. cana spp. *cana*	Plains silver sagebrush	ARCAC5	N/A
A. cana spp. *viscidula*	Mountain silver sagebrush	ARCAV2	N/A
A. tridentata spp. *spiciformis*	Subalpine big sagebrush	ARTRS2	N/A
A. tridentata spp. *tridentata*	Basin big sagebrush	ARTRT	N/A
A. tridentata spp. *vaseyana*	Mountain big sagebrush	ARTRV	N/A
A. tridentata spp. *wyomingensis*	Wyoming big sagebrush	ARTRW8	N/A
A. tridentata spp. *xericensis*	Xeric big sagebrush	ARTRX	N/A
A. tripartita spp. *tripartita*	Threetip sagebrush	ARTRT2	N/A
Subshrub sagebrush			
A. frigida	Fringed sagewort	ARFR4	N/A
A. pedatifida	Birdfoot sagebrush	ARPE6	N/A
Other shrubs			
Amelanchier alnifolia	Saskatoon serviceberry	AMAL2	N/A
Amelanchier utahensis	Utah serviceberry	AMUT	N/A
Atriplex canescens	Fourwing saltbush	ATCA2	N/A
Atriplex confertifolia	Shadscale saltbush	ATCO	N/A
Ceanothus velutinus	Snowbrush ceanothus	CEVE	N/A
Chrysothamnus nauseosus *Syn = Ericameria nauseosa* spp. *nauseosa* var. *nauseosa*	Rubber rabbitbrush	CHNA2/ ERNAN5	N/A
Chrysothamnus viscidiflorus	Green rabbitbrush	CHVI8	N/A
Grayia spinosa	Spiny hopsage	GRSP	N/A
Gutierrezia sarothrae	Broom snakeweed	GUSA2	N/A

Scientific Name	Common Name	Symbol	Most Likely Category
Juniperus occidentalis	Western juniper	JUOC	N/A
Juniperus osteosperma	Utah juniper	JUOS	N/A
Krascheninnikovia lanata	Winterfat	KRLA2	N/A
Pachystima myrsinites	Oregon boxleaf	PAMY2	N/A
Purshia tridentata	Antelope bitterbrush	PUTR2	N/A
Rosa woodsii	Woods' rose	ROWO	N/A
Sarcobatus vermiculatus	Greasewood	SAVE4	N/A
Symphoricarpos albus	Common snowberry	SYAL	N/A
Symphoricarpos oreophilus	Mountain snowberry	SYOR2	N/A
Tetradymia canescens	Spineless horsebrush	TECA2	N/A
FORBS			
Annuals/Occasionally Biennials			
Alyssum desertorum	Desert alyssum	ALDE	I
Asperugo procumbens	German-madwort	ASPR	I
Camelina microcarpa	Littlepod false flax	CAMI2	I
Carthamus tinctorius	Safflower	CATI	W
Chenopodium spp.	Goosefoot	CHENO	P
Chorispora tenella	Purple mustard	CHTE2	W
Collinsia spp.	Blue eyed Mary	COLLI	P
Collomia spp.	Trumpet	COLLO	P
Cryptantha spp.	Cryptantha	CRYPT	O
Descurainia spp.	Tansymustard	DESCU	I
Epilobium spp.	Willowherb	EPILO	O
Eriastrum sparsiflorum	Great Basin woollystar	ERSP3	P
Eriogonum spp.	Buckwheat	ERIOG	P
Erodium cicutarium	Stork's bill	ERCI6	P
Galium aparine	Stickywilly	GAAP2	I
Halogeton glomeratus	Saltlover	HAGL	I
Helianthus annuus	Common sunflower	HEAN3	O
Kochia scoparia	Kochia	KOSC	W
Lactuca serriola	Prickly lettuce	LASE	P
Lappula texana Syn = *Lappula occidentalis* var. *cupulata*	Flatspine stickseed	LATE3/ LAOCC	I
Lepidium spp.	Pepperweed	LEPID	O
Malacothrix spp.	Desertdandelion	MALAC3	P
Medicago spp.	Alfalfa	MEDIC	P
Melilotus officinalis	Yellow sweetclover	MEOF	P
Microsteris spp.	Microsteris (phlox)	MICRO22	P

Scientific Name	Common Name	Symbol	Most Likely Category
Plantago patagonica	Woolly plantain	PLPA2	P
Plectritis macrocera	Longhorn plectritis	PLMA4	P
Polygonum spp.	Knotweed	POLYG4	P
Ranunculus testiculatus Syn = *Ceratocephala testiculata*	Bur buttercup	RATE/ CETE5	W
Salsola kali	Russian thistle	SAKA	W
Sonchus spp.	Sowthistle	SONCH	P
Stephanomeria spp.	Wirelettuce	STEPH	P
Thlaspi arvense	Field pennycress	THAR5	I
Tragopogon spp.	Goatsbeard	TRAGO	P
Trifolium spp.	Clover	TRIFO	P
Veronica spp.	Speedwell	VERON	I
Biennials			
Cirsium spp.	Thistle	CIRSI	W
Cynoglossum officinale	Hound's tongue	CYOF	W
Gilia aggregata Syn = *Ipomopsis aggregata* spp. *aggregata*	Scarlet gilia	GIAG/ IPAGA3	P
Machaeranthera canescens	Hoary aster	MACA2	O
Perennials/Occasionally Biennials			
Achillea millefolium	Common yarrow	ACMI2	O
Agoseris spp.	Agoseris	AGOSE	P
Allium spp.	Onion	ALLIU	P
Androsace septentrionalis	Pygmyflower rockjasmine	ANSE4	P
Antennaria spp.	Pussytoes	ANTEN	O
Arabis holboellii	Holboell's rockcress	ARHO2	P
Arenaria kingii	King's sandwort	ARKI	P
Artemisia dracunculus	Tarragon	ARDR4	P
Aster chilensis Syn = *Symphyotrichum chilense* var. *chilense*	Pacific aster	ASCH2/ SYCHC	P
Astragalus spp.	Milkvetch	ASTRA	P

Scientific Name	Common Name	Symbol	Most Likely Category
Balsamorhiza hookeri	Hooker's balsamroot	BAHO	P
Balsamorhiza sagittata	Arrowleaf balsamroot	BASA3	P
Berberis repens	Creeping barberry	MARE11	O
Brodiaea spp.	Brodiaea	BRODI	P
Calochortus spp.	Mariposa lily	CALOC	P
Camassia spp.	Camas	CAMAS	P
Castilleja spp.	Indian paintbrush	CASTI2	O
Chaenactis douglasii	Douglas's dustymaiden	CHDO	P
Comandra umbellata	Bastard toadflax	COUM	P
Convolvulus arvensis	Field bindweed	COAR4	W
Crepis spp.	Hawksbeard	CREPI	P
Cymopterus spp.	Springparsley	CYMOP2	P
Camassia spp.	Camas	CAMAS	P
Dalea spp.	Prairie clover	DALEA	P
Delphinium nuttallianum	Twolobe larkspur	DENU2	O
Erigeron spp.	Fleabane	ERIGE2	P
Eriogonum spp.	Buckwheat	ERIOG	O
Erysimum spp.	Wallflower	ERYSI	P
Fritillaria spp.	Fritillary	FRITI	P
Geranium viscosissimum	Sticky purple geranium	GEVI2	P
Geum spp.	Avens	GEUM	P
Grindelia squarrosa	Curlycup gumweed	GRSQ	I
Hackelia spp.	Stickseed	HACKE	O
Haplopappus acaulis	Stemless mock goldenweed	HAAC	O
Hedysarum spp.	Sweetvetch	HEDYS	P
Helianthella spp.	Helianthella	HELIA	P
Hydrophyllum capitatum	Ballhead waterleaf	HYCA4	P
Iva axillaris	Povertyweed	IVAX	P
Lathyrus spp.	Pea	LATHY	P
Leptodactylon pungens *Syn = Linanthus pungens*	Granite prickly phlox	LEPU/ LIPU11	P
Linanthus spp.	Linanthus	LINAN2	P
Linum perenne	Blue flax	LIPE2	P
Lithophragma spp.	Woodland-star	LITHO2	P
Lithospermum ruderale	Western stoneseed	LIRU4	P
Lomatium spp.	Desertparsley	LOMAT	P
Lotus corniculatus	Bird's-foot trefoil	LOCO6	P
Lupinus spp.	Lupine	LUPIN	O
Lygodesmia spp.	Skeletonplant	LYGOD	P
Mentha spp.	Mint	MENTH	I
Mentzelia spp.	Blazingstar	MENTZ	P
Mertensia spp.	Bluebells	MERTE	P

Scientific Name	Common Name	Symbol	Most Likely Category
Microseris spp.	Silverpuffs	MICRO6	P
Oenothera spp.	Evening-primrose	OENOT	O
Opuntia polyacantha	Plains pricklypear	OPPO	N/A
Penstemon spp.	Beardtongue	PENST	P
Perideridia spp.	Yampah	PERID	P
Phacelia spp.	Phacelia	PHACE	P
Phlox spp.	Phlox	PHLOX	O
Ranunculus spp.	Buttercup	RANUN	O
Rumex spp.	Dock	RUMEX	O
Sanguisorba minor	Small burnet	SAMI3	P
Sedum spp.	Stonecrop	SEDUM	P
Senecio spp.	Ragwort	SENEC	O
Smilacina racemosa Syn = *Maianthemum racemosum* spp. *racemosum*	Feathery false lily of the valley	SMRA/ MARAR	P
Solidago spp.	Goldenrod	SOLID	P
Sphaeralcea spp.	Globemallow	SPHAE	P
Taraxacum officinale	Common dandelion	TAOF	P
Townsendia hookeri	Hooker's Townsend daisy	TOHO	P
Vicia spp.	Vetch	VICIA	P
Viola nuttallii	Nuttall's violet	VINU2	O
Viola purpurea	Goosefoot violet	VIPU4	O
Wyethia amplexicaulis	Mule-ears	WYAM	O
Zigadenus spp.	Deathcamas	ZIGAD	O
GRASSES			
Annuals			
Avena fatua	Wild oat	AVFA	N/A
Bromus commutatus Syn = *Bromus racemosus*	Bald brome	BRCO4/ BRRA2	N/A
Bromus japonicus	Japanese brome	BRJA	N/A
Bromus mollis Syn = *Bromus hordeaceus* spp. *hordeaceus*	Soft brome	BRMO2/ BRHOH	N/A
Bromus tectorum	Cheatgrass	BRTE	N/A
Festuca octoflora	Sixweeks fescue	FEOC3	N/A
Triticum aestivum	Common wheat	TRAE	N/A

Scientific Name	Common Name	Symbol	Most Likely Category
Perennials			
Achnatherum thurberianum	Thurber's needlegrass	ACTH7/ STTH2	N/A
Agropyron cristatum	Crested wheatgrass	AGCR	N/A
Agropyron intermedium Syn = *Thinopyrum intermedium*	Intermediate wheatgrass	AGIN2/ THIN	N/A
Agropyron repens Syn = *Elymus repens*	Quackgrass	AGRE2/ ELRE4	N/A
Agropyron smithii Syn = *Pascopyrum smithii*	Western wheatgrass	AGSM/ PASM	N/A
Agropyron spicatum Syn = *Pseudoroegneria spicata* spp. *spicata*	Bluebunch wheatgrass	AGSP/ PSSPS	N/A
Bromus inermis	Smooth brome	BRIN2	N/A
Carex douglasii	Douglas' sedge	CADO2	N/A
Elymus cinereus Syn = *Leymus cinereus*	Basin wildrye	ELCI2/ LECI4	N/A
Elymus junceus Syn = *Psathyrostachys juncea*	Russian wildrye	ELJU/ PSJU3	N/A
Festuca idahoensis	Idaho fescue	FEID	N/A
Koeleria cristata Syn = *Koeleria macrantha*	Prairie junegrass	KOCR/ KOMA	N/A
Melica bulbosa	Oniongrass	MEBU	N/A
Oryzopsis hymenoides Syn = *Achnatherum hymenoides*	Indian ricegrass	ORHY/ ACHY	N/A
Poa bulbosa	Bulbous bluegrass	POBU	N/A
Poa juncifolia Syn = *Poa secunda*	Sandberg bluegrass	POJU/ POSE	N/A
Poa sandbergii Syn = *Poa secunda*	Sandberg bluegrass	POSA12/ POSE	N/A
Poa scabrella Syn = *Poa secunda*	Sandberg bluegrass	POSC/ POSE	N/A
Sitanion hystrix Syn = *Elymus elymoides* spp. *elymoides*	Squirreltail	SIHY/ ELELE	N/A
Stipa comata Syn = *Hesperostipa comata* spp. *comata*	Needle and thread	STCO4/ HECOC8	N/A
Stipa occidentalis Syn = *Achnatherum occidentale* spp. *occidentale*	Western needlegrass	STOC2/ ACOCO	N/A

SEDGES			
Typha spp.	Cattail	TYPHA	N/A

Viability analyses for conservation of sage-grouse populations:

Miles City Field Office, Montana

Completion report

30 June 2010

Prepared for Bureau of Land Management

Miles City Field Office

Miles City, Montana

By

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EXECUTIVE SUMMARY

We conducted population viability analyses of greater sage-grouse for the Bureau of Land Management, to provide decision support for land use planning. We analyzed the effects of different levels of oil and gas development and different levels of agricultural tillage on sage grouse lek counts, for years with West Nile virus (WNv) outbreaks and for years without. We also assessed the potential of grazing as a sage-grouse management tool by quantifying the relationships between grass height, nest success and population growth. Our general conclusions are applicable to Management Zone I, south of U.S. Highway 2. Our quantitative results focus on three focal areas of interest to the Miles City Field Office: they are the Cedar Creek Anticline (CCA), Carter and Haxby sage grouse areas.

The three focal areas contain WNv, but have otherwise divergent profiles with respect to current and potential stressors to sage-grouse populations. Cedar Creek Anticline is already heavily developed for oil and gas extraction, containing an average of 1 well per 364 ac within a 9 mi radius of each lek, and further development is imminent. Tillage in this area is negligible. Haxby currently contains an average of 4% tillage and no wells in the vicinity of leks. The primary threat in this area is a potential increase in tillage. Carter is largely undeveloped, both in terms of tillage and oil and gas wells, but has large potential for both. Notably, Carter is 5-7 times the size of the other areas.

Land use planning must be geared toward preserving sage-grouse in WNv outbreak years, as the disease causes extreme mortality events in this species. Lek counts in relatively intact landscapes such as Haxby and Carter are predicted to drop by approximately 25% when birds are subject to an outbreak. The negative, synergistic effect of WNv and oil and gas development is evident in the 62% decline that is predicted with an outbreak in CCA. Should CCA or Carter be developed to 160 ac spacing throughout—that is, to an average of 160 ac spacing within 9 mi of all leks—we predict the increased level of development combined with a WNv outbreak to precipitate a 95% drop in lek counts. West Nile virus also exacerbates the effects of tillage on sage-grouse. An increase to 10% tillage in Haxby or Carter, combined with a disease outbreak is predicted to result in a 40-50% decrease in lek counts; while an increase to 20% tillage is expected to result in 60-70% declines.

Areas with few human impacts and large numbers of sage-grouse contain a mixture of different sized leks. As human impacts increase and total sage-grouse numbers decrease, the landscape no longer maintains larger leks. In our simulated populations, large leks (> 25 males) were lost when populations had decline by 42-76% from their predicted size without energy development or tillage. Disappearance of leks >10 males occurred when declines reached 77-94%. We consider the presence of large leks to be a leading indicator of population status. As of last count, CCA contained no leks > 25 males.

Planning units for sage-grouse must be large, at least the entirety of a focal area. Small and isolated populations are difficult to conserve, and the large-scale impacts of energy development may exacerbate this problem. We detected the effects of oil and gas development most strongly at the largest scale we tested, a 273 mi^2 area around each lek. Conversely we most strongly detected the effects of tillage at the smallest scale we tested, a 1.2 mi^2 area. The small-scale effect of tillage, and its modest influence on sage-grouse numbers (absent a WNv outbreak) are likely due to the lack of tillage in Management Zone I, at large. The magnitude of tillage impacts, and the scale at which those impacts are felt, will change if the overall landscape changes. Should extensive tillage prevail in Management Zone I, we expect the effects on sage-grouse to be far more devastating than can be predicted by analysis of current data.

The effects of grazing management on sage-grouse have been little studied, but correlations between grass height and nest success suggest that grazing may be one of the few tools available to managers to enhance sage-grouse populations. Our analyses predict that already healthy populations may benefit from moderate changes in grazing practices. For instance, a 2 in increase in grass height could result in a 10% increase in nest success, which translates to an 8% increase in population growth rate. Managing grass height in otherwise intact sagebrush habitat deserves further research; however, grazing management holds little promise in highly impacted landscapes, because research to date suggests that benefits of grass structure at nest sites are secondary to other habitat features.

INTRODUCTION

Energy development and increased food demand will greatly increase the cumulative human impacts on public lands in coming years throughout the West (McDonald et al. 2009). Cumulative impacts are the long-term or permanent negative effects of transmission lines, oil and gas developments, surface mining, tillage, roads and other forms of human infrastructure on the land and its resources. Cumulative impacts present trade-offs in management because public lands lose much of their conservation value following human development (Bottrill et al. 2008, 2009). Given the magnitude of anticipated impacts, identifying and prioritizing lands with low human disturbance is critical for the Bureau of Land Management (BLM) to follow its multiple use mandate (Federal Land Policy and Management Act 1976) by conserving some areas while developing others. The management challenge will be to site future developments away from large, intact landscapes that still maintain biological functions and support other natural resources (Kiesecker et al. 2010).

The sagebrush (*Artemisia* spp.) ecosystem in the West is representative of the struggle to maintain wildlife populations in a landscape that bears the debt of our ever-increasing demands for natural resources (Knick et al. 2003). The greater sage-grouse (*Centrocercus urophasianus*; hereafter 'sage-grouse') is considered a landscape species, in that it requires large, intact expanses of sagebrush habitat to maintain robust populations (Connelly et al. 2010). As a result, the sage-grouse has become a focal species for conservation in the West, and the size of breeding populations is often used as an indicator of the overall health of the sagebrush ecosystem (Hanser and Knick 2010). Loss and degradation of sagebrush habitat has resulted in at least a four decade long sage-grouse decline (Garton et al. 2010, Connelly et al. 2004), and extirpation of the species from 46% of its original range (Schroeder et al. 2004).

Eastern Montana provides habitats that support focal populations of sage-grouse (Doherty et al. 2010). The recent surge in agricultural tillage and energy development has resulted in rapid, large-scale changes in portions of eastern Montana, and a growing recognition of the need to fully understand and monitor potential impacts to wildlife populations. The potential for management to influence populations is large, and options vary from no conservation-oriented action to major land use changes, including curtailing energy development, providing incentives to remove farmlands from cultivation, modifying grazing strategies and manipulating water sources to reduce disease risk.

Reliable knowledge is fundamental to adaptive management because science provides the biological basis for managers to anticipate and plan for desired future conditions. The goal of management-oriented science is to connect the dynamics of focal species, either likelihood of extirpation or potential for recovery, to actions that managers can implement on the ground to maintain or enhance populations. In practice, however, land management actions are oftentimes implemented without any clear connection to how those actions affect the dynamics of the wildlife population of interest. This is particularly true when managers must try to counteract cumulative impacts, because the science on which this management is based often does not capture how population status and habitat availability have changed over space and time. Because current conditions are easiest to use as a baseline for comparison, the cumulative impacts of past decades are often discarded. Furthermore, the disparity between the scale of individual management actions and the scale at which populations respond is a persistent problem in understanding cumulative impacts on population viability (Schultz 2010). Thus, for management-oriented science to be of maximum use, it must be conducted at a spatial scale that captures populations responding to multiple stressors and management actions, and it must capture the accumulation of these impacts over time.

This report links sage-grouse counts and population dynamics with stressors and management actions to evaluate the viability of populations under future land use scenarios. Our objectives were to use lek count data and demographic information from marked birds to 1) evaluate current viability of sage-grouse populations in eastern Montana, 2) formulate potential and realistic future management scenarios for populations and 3) simulate these management scenarios to evaluate future viability of populations to provide decision support to BLM officials at field office, state and national levels.

We used lek count data from throughout Management Zone I to apply to three focal areas that exhibit divergent profiles of exposure to energy development and tillage (Figure 1 and Table 1) interacting with West Nile virus (WNv, Flaviviridae, Flavivirus). To maximize our ability to provide management recommendations, we conducted a second analysis to evaluate the potential effects of grazing management on sage-grouse populations, based on range-wide demographic data from marked birds. With this multi-pronged approach we captured the variability in the response of grouse to stressors and management actions, providing high portability in application to other similar BLM planning units.

Literature Synthesis

Oil and gas development, tillage and disease (WNv) are the primary large-scale factors impacting sage-grouse populations in Montana and across the species' eastern range. Grazing is among the most common land use practices in the West, and may be either detrimental or beneficial, depending on particulars of the grazing management (Beck and

Mitchell 2000). Collectively, these factors represent large-scale stressors that limit populations, and options available to managers to maintain and enhance bird numbers on public lands. Here we synthesize the current scientific literature to provide readers with an understanding of the biological response of sage-grouse populations to each of these factors.

Oil and Gas Development. Oil and gas (energy) development has emerged as a range-wide issue in conservation because areas being developed contain large sage-grouse populations (Connelly et al. 2004) and other sagebrush obligate species (Knick et al. 2003). Breeding sage-grouse populations are severely impacted at oil and gas well densities commonly permitted in Montana and Wyoming (Naugle et al. 2010). Impacts are indiscernible at < 1 well per mi^2, but above this threshold, lek losses are 2-5 times greater inside than outside of development, and abundance at remaining leks declines by 32 to 77% (Doherty et al. 2010). Magnitude of losses vary from one field to another, but impacts are universally negative and typically severe (Harju et al. 2010). High site fidelity, but low survival of adult sage-grouse combined with lek avoidance by younger birds (Holloran et al. 2010) results in time lags of 3-4 years between onset of development activities and local extirpation (Holloran 2005, Walker et al. 2007). Energy development also impacts sage-grouse habitats and vital rates outside the breeding season away from leks. Risk of chick mortality is 1.5 times higher for each additional well site visible within 0.6 mi of brood locations compared to random locations (Aldridge and Boyce 2007), and sage-grouse avoid otherwise suitable winter habitat disturbed by energy development (Doherty et al. 2008, Carpenter et al. 2010).

Tillage. Agricultural tillage is a range-wide stressor to sage-grouse populations (Connelly et al. 2004) that is most pronounced at northern latitudes (Aldridge et al. 2008). Recent changes to the U.S. Food Security Act, coupled with increased commodity prices of grains to meet the demand for biofuels, threatens remaining arable lands as tillage becomes more profitable than ranching (Fargione et al. 2009). Large-scale tillage continues to fragment the once vast tracts of sagebrush dominated grasslands that sage-grouse require for each stage of their life history. Population declines after tilling are known in Montana (Swenson et al. 1987), North and South Dakota (Smith et al. 2005, Tack 2009) and the state of Washington (Schroeder et al. 2000). Disturbance from tillage likely reduces the availability of nesting sites (Holloran et al. 2005), causing females to shift to undisturbed areas, thus decreasing lek size if males recruit to leks outside the disturbance to increase the likelihood of intercepting receptive mates (Holloran et al. 2010). Tillage risk is high in Montana where 58% of active leks (n = 430) are located on private lands (Tack 2009). In Montana, large leks (> 25 males) are 4.5 times less likely to occur than small leks when tillage fragments 21% of land within a 0.6-mi radius of breeding sites (Tack 2009).

West Nile Virus. West Nile virus emerged as a threat to sage-grouse in 2002 and is now an important new source of mortality in low and mid-elevation populations throughout the West (Walker et al. 2010). West Nile virus simultaneously reduces juvenile, yearling, and adult survival, three vital rates important for sage-grouse population growth. Persistent low-level WNv mortality, combined with severe disease outbreaks, results in local and regional population declines (Naugle et al. 2004, 2005). Mortality from this disease reduces growth rate of susceptible populations by an average of 6-9% per year (Walker et al. 2010), and lab experiments show 100% mortality following infection (Clark et al. 2006). Resistance to WNv in the wild is low (Walker et al. 2007) and is expected to increase slowly over time (Walker et al. 2010). Eliminating mosquito breeding habitat from anthropogenic water sources is crucial for reducing impacts (Zou et al. 2006). Better range-wide data are needed on geographic and temporal variation in infection rates, mortality, and seroprevalence. Small, isolated and peripheral populations are most at risk, particularly those at lower elevations, and those experiencing large-scale increases in distribution of surface water (Walker et al. 2010).

Grazing. The future of western rangelands is in developing partnerships that help keep sustainable grazing the prevailing land use on public and private lands (Krausman et al. 2009). Public land managers recognize the importance of top-down strategies that conserve entire landscapes because the scale at which conservation practices are implemented must match the scale of anthropogenic change that threatens populations. Reversing game bird population declines is an example of landscape level conservation that will require regional management of remaining usable space (Williams et al. 2004).

Grazing should not be wrongly classified with other detrimental land use practices that overwhelm management of remaining habitat fragments; rather, grazing is a management tool that depending on its application can be detrimental or beneficial to sage-grouse (Beck and Mitchell 2000). Whether grazing impacts are positive or negative depends on timing and intensity of grazing, and which habitat component is being considered (Beck et al. 2000). For example, light to moderate cattle grazing has been shown to increase forbs eaten by grouse and to induce grouse to use dense, grassy meadows. Conversely, heavy grazing reduces herbaceous cover and promotes invasion by undesirable species, while herbicide application to increase grass production reduces or removes usable sage-grouse habitat (Crawford et al. 2004, Beck and Mitchell 2000). Guidelines describing height and density of herbaceous cover necessary to maintain productive habitats are available for many sage-grouse populations (e.g., Connelly et al. 2000), and decreased herbaceous cover in otherwise suitable habitat is associated with reduced nest success (Holloran et al. 2005, Kaczor 2008), a demographic rate that exerts substantial influence on sage-grouse population growth (Walker and Naugle 2010).

However, little experimental research has been conducted to provide insights into which conservation practices promote the natural heterogeneity of rangelands to benefit sage-grouse. Most contemporary studies lack experimental controls, are too short in duration, or fail to collect pre-treatment data. The best available experimental evidence supports reduced grazing as a conservation practice to recover a declining population of black grouse (*Tetrao tetrix*) in northern England (Calladine et al. 2002). Black grouse numbers averaged 6.3% higher per year and brood survival was 22% higher at sites with reduced grazing than in normally grazed reference sites. This study demonstrated that degradation of habitats for black grouse by intensive grazing is reversible and that manipulation of grazing regimes can contribute to conservation (Calladine et al. 2002). Wildlife managers in Montana readily acknowledge the importance of sustainable grazing to conservation because ranchers that remain profitable are less likely to convert native plant communities to cropland (Licht 1997, Higgins et al. 2002). Measuring sage-grouse response to grazing systems is a priority for all parties interested in maintaining rural ways of life and in conserving healthy sage-grouse populations on working ranches in the West.

METHODS

Focal Areas and Study Region

The three focal areas of our analyses—Cedar Creek Anticline, Haxby and Carter sage-grouse areas—are of particular management interest to the BLM's Miles City Field Office. These focal areas (Figure 1) differ in size, number of sage-grouse remaining during recent lek counts, current stressors (Table 1) and predicted future stressors. Cedar Creek Anticline (CCA, Figure 2), on the Montana-North Dakota border, is the smallest of the three areas, encompassing 780 mi^2 which are already heavily developed for oil and gas extraction. Haxby (Figure 3) is 1.4 times the size of CCA (1090 mi^2), with tillage being the primary stressor to sage-grouse populations. The Carter sage-grouse area (Figure 4) is almost seven times as large as CCA (5200 mi^2), including not only Carter County itself (the southeastern-most county in Montana) but also parts of neighboring counties in Montana and South Dakota. Carter is largely undeveloped, both in terms of tillage and oil and gas wells, yet has large development potential for both. West Nile virus has been documented in sage-grouse in Montana, Wyoming and the Dakotas (Naugle et al. 2004, Naugle et al. 2005, Walker et al. 2004, Walker and Naugle 2010) and has been documented in humans in the three specific areas of interest (Centers for Disease Control 2004).

Because the focal areas vary dramatically in the extent to which the stressors of interest occur, we based our analyses on a larger study region, and were thus able to capture the effects of multiple stressors on sage-grouse populations. For our viability analyses based on location of stressors and nearby counts of breeding males, the study region that provided the strongest foundation was the portion of Management Zone I that lies south of US Hwy 2 (Figure 1, Table 1). We relied on data throughout this region because it encompasses the three focal areas, contains a wide range of stressors and is composed of habitat similar to that found in the focal areas. This habitat is largely dominated by Wyoming big sagebrush (*Artemisia tridentata wyomingensis*), with grass cover typical of the eastern portion of the sage-grouse range. We divided our study region into five supporting areas, based on the Western Association of Fish and Wildlife Agencies (WAFWA) subpopulation designations (Connelly et al. 2004). Our supporting areas (followed by the WAFWA subpopulation name) are as follows: north-central

MT (north-central MT), central MT (central MT), eastern MT (eastern interior MT/northeast tip WY), Dakotas (MT/ND northwest SD) and Wyoming (northeast WY/southeast MT and Fall River SD/eastern edge WY). We combined the latter two because of the small size of the Fall River subpopulation and its proximity to the northeast WY/southeast MT subpopulation.

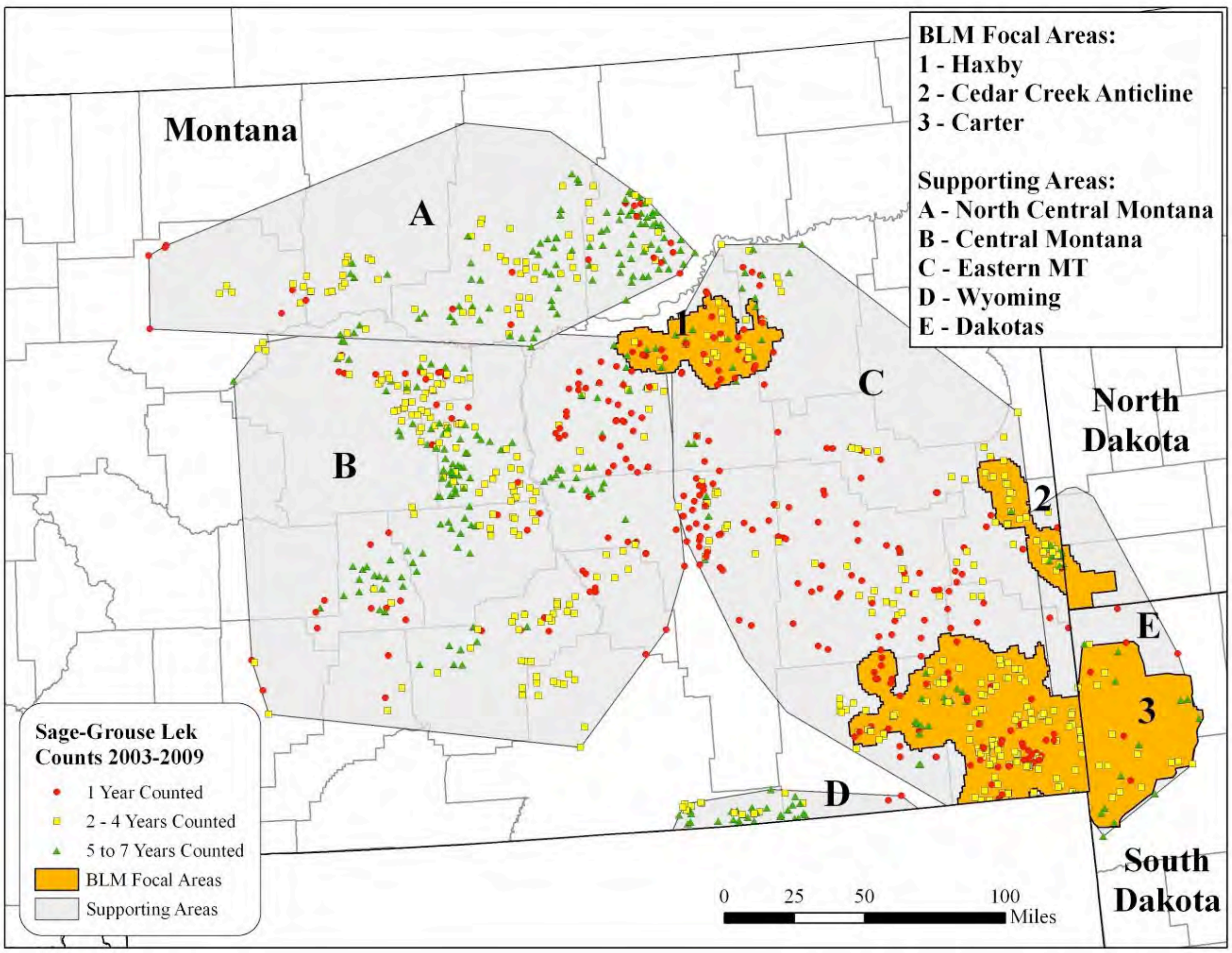

Figure 1. Distribution of lek complex centers with respect to focal areas and supporting areas.

Table 1. Number of lek complex centers used in analysis from a) focal areas and b) supporting areas. Leks are categorized by presence of wells within best fit scale (9.3 mi. radius) and presence of tillage within best fit scale (0.6 mi. radius) [see Results].

a		Area		
Tillage?	Wells?	Carter	CCA	Haxby
No	No	76	0	27
No	Yes	47	28	0
Yes	No	9	0	15
Yes	Yes	2	2	0
Area Total		134	30	42

b		Area				
Tillage?	Wells?	N-cent MT	Central MT	Eastern MT	WY	Dakotas
No	No	74	104	127	197	11
No	Yes	23	57	11	218	22
Yes	No	38	90	18	0	4
Yes	Yes	13	38	3	8	7
Area Total		148	289	159	423	44

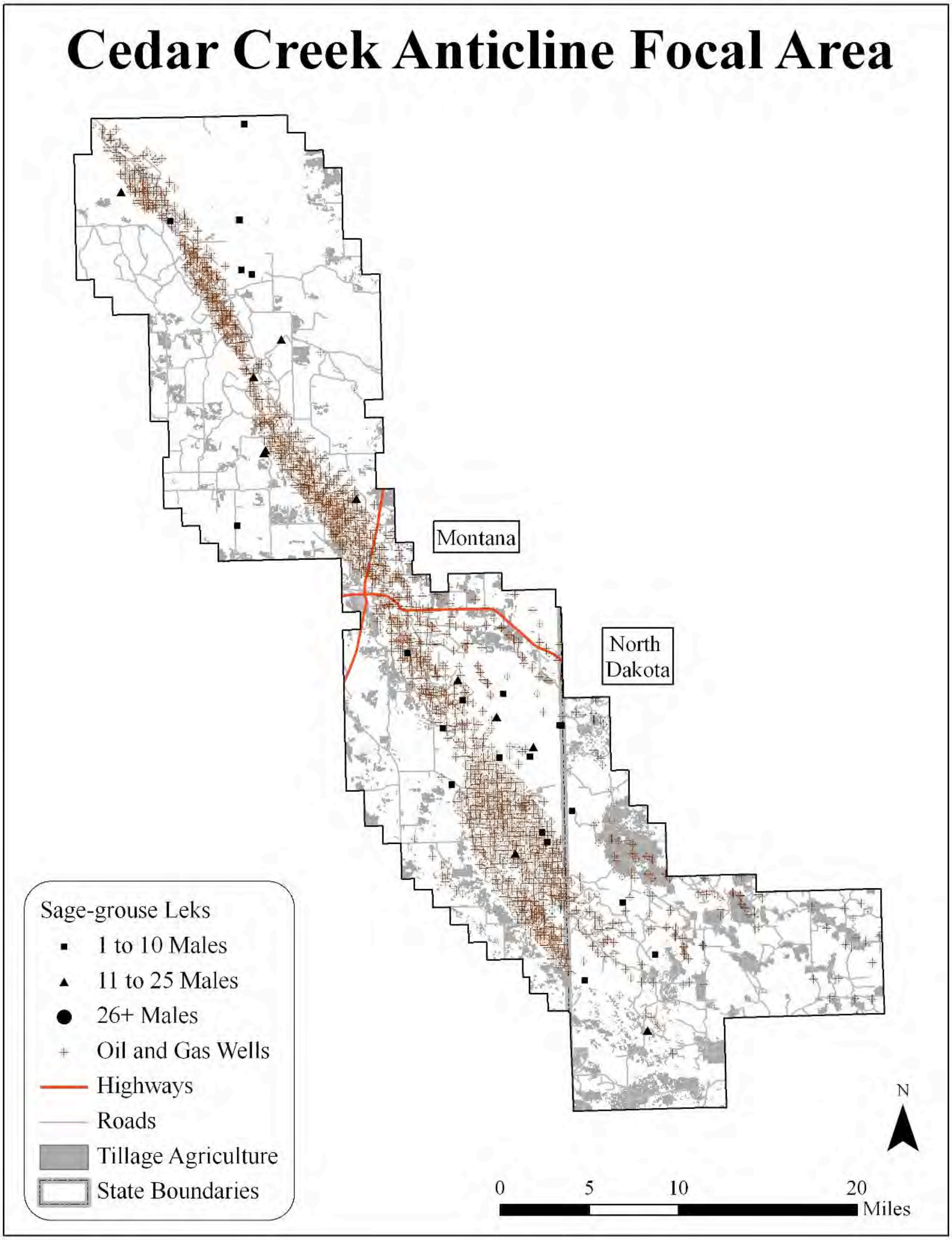

Figure 2. Lek complex centers, oil and gas wells and tillage in Cedar Creek Anticline.

Evaluating efficacy of fence markers in reducing greater sage-grouse collisions with fencing

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Evaluating efficacy of fence markers in reducing greater sage-grouse collisions with fencing

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ABSTRACT

Anthropogenic infrastructure routinely interferes with wildlife movement, habitat use, and survival. Grouse in the family *Phasianidae* may be particularly susceptible to collisions with fences due to their morphology and life history. Because many *Phasianid* species are of conservation concern, managers often deploy markers on fences to reduce collision-associated mortality. However, scarce information on the effectiveness of different marker styles or the effects of local and landscape features on collision risk exists. Our objectives were to (1) determine the effectiveness of different marker styles in reducing collisions, (2) estimate the effects of local and landscape features on collision risk, and (3) evaluate an existing greater sage-grouse (*Centrocercus urophasianus*) collision risk model. We conducted greater sage-grouse collision surveys within Sublette County, Wyoming, USA in March and April of 2014 and 2015. Data were analyzed in a multi-scale occupancy model accounting for incomplete detection of collisions. We found substantial evidence for the ability of all markers to reduce collisions (~57% reduction), with little difference between the tested marker types. We found strong evidence for lower collision probabilities at fences with wood posts and on fences farther from leks. Our results also indicated a negative relationship between collision probabilities and the difference between fence and vegetation heights. We observed little evidence for differences in collision risk between areas defined as "high" or "moderate" risk in a pre-existing collision risk map. We recommend integrating fence marking into conservation practices requiring fencing, and prioritizing fence marking near leks in areas with greater fence exposure.

1. Introduction

Anthropogenic infrastructure such as fences routinely interferes in the movements, habitat use, and survival of a wide variety of wildlife species (Bevanger 1994; Drewitt and Langston 2008; Linnell 2016). Unfortunately, the installation of human infrastructures, including fences, typically witnessed across landscapes of high-income nations is now occurring in low-income countries as well (Bevanger 1994; Drewitt and Langston 2008). The broad-scale erection of fencing has continued due to civil and political unrest throughout the world (Bevanger and Henriksen 1996; Hayward and Kerley 2009; Linnell 2016), the need for maintaining domesticated livestock within an enclosed area (Hayter 1939), the need to exclude undesired animals from certain parcels (Bevanger and Henriksen 1996; Hayter 1939), or to maintain biodiversity (Hayward and Kerley 2009; Linnell et al. 2016).

Wildlife collisions with fencing represent a direct impact on the survival of individuals. Mortality associated with fence collisions has been well documented for numerous avian species, including the *Phasianids* which are thought to be susceptible to collisions with infrastructure due to their high wing loading, lekking behavior, and afoveal retina (Bevanger 1994; Lisney et al. 2012; Sillman 1973). In North America, Wolfe et al. (2007) found that 39.8% of lesser prairie-chicken (*Tympanuchus pallidicinctus*) mortality was caused by collision with fences and, based on a subset of the same data set, Patten et al. (2005) observed elevated mortality rates for female lesser prairie-chickens where habitats were more fragmented by fences, power lines, and roads. Similarly, greater sage-grouse (*Centrocercus urophasianus*; hereafter, sage-grouse) collisions with fencing have been observed in two studies in western North America (Christiansen, 2009, Stevens et al. 2012a). In Europe, collisions with fences and power lines have been observed for the western capercaillie (*Tetrao urogallus*), black grouse (*Tetrao tetrix*), red grouse (*Lagopus lagopus scoticus*), and ptarmigan (*Lagopus spp.*) (Baines and Summers 1997; Bevanger 1995; Catt et al. 1994). Although the impact of this collision-associated mortality on populations is not particularly well understood, there is some evidence indicating infrastructure collisions may contribute substantially

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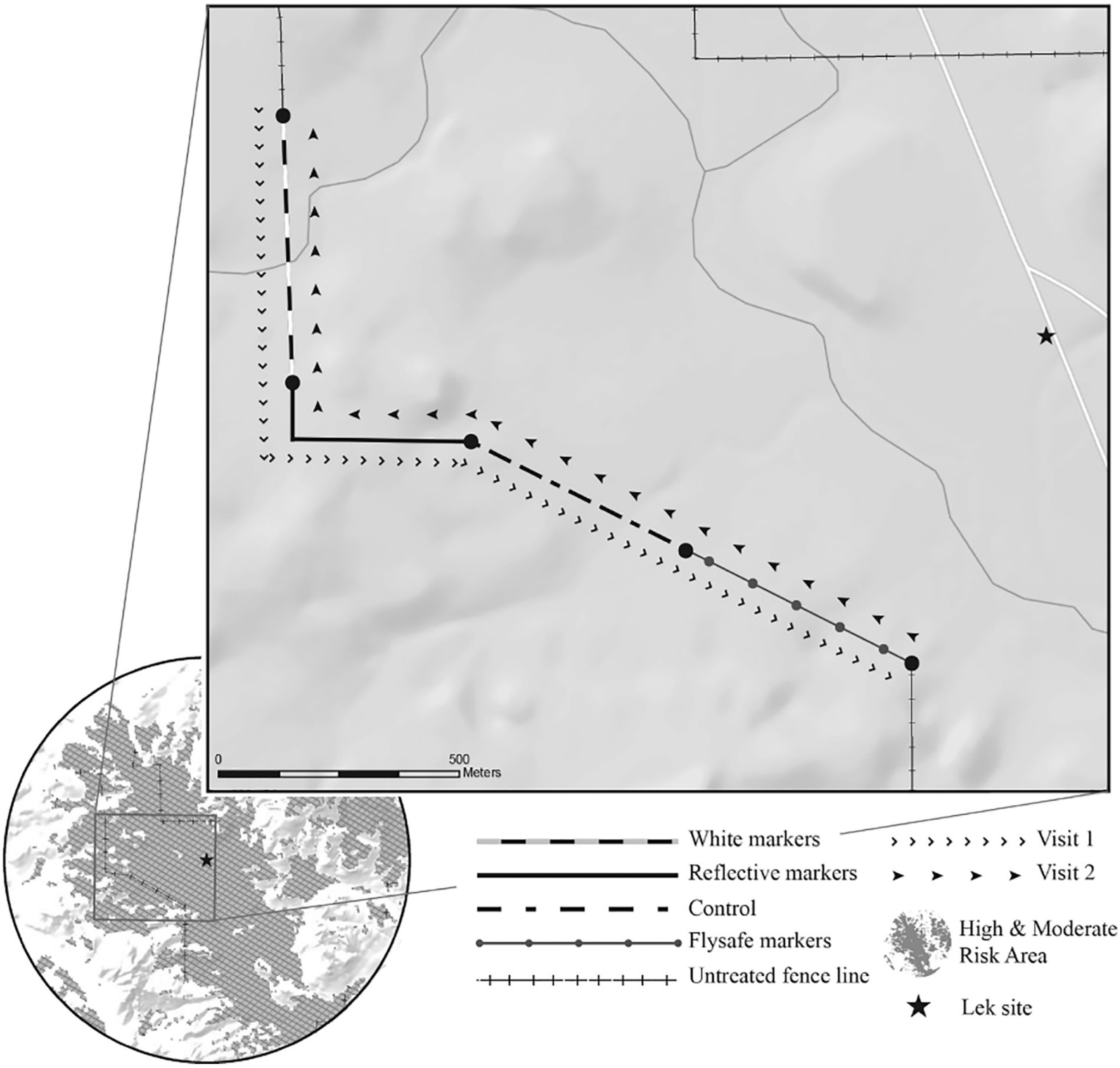

Fig. 1. Illustration of four treated segments of fence-line associated with a focal lek.

to population declines in some species (Baines and Andrew 2003; Bevanger 1995; Moss et al. 2000; Smith and Dwyer 2016).

The risk of wildlife collisions with fencing is likely impacted by a variety of site and landscape-scale factors (Stevens et al. 2012a). Site factors may include the density and height of local vegetation, fence height, type of fence, the type of fence posts, the distance between fence posts, the slope or ruggedness of the nearby landscape, and in the case of lekking species, the distance to surrounding leks and the number of individuals attending adjacent leks (Stevens et al. 2012a). Similarly, landscape-scale factors may include surrounding landcover types (Baines and Summers 1997), the density of individuals throughout the landscape (Baines and Andrew 2003), and movement corridors (including prominent ridges or other vegetative or topographic features that funnel animal movement) (Bevanger 1994; von Schweppenburg 1929).

Marking human infrastructure to increase its visibility is a common practice for reducing collisions for a variety of avian species (Luzenski et al. 2016), including *Phasianids* due to their predisposition for colliding with fences and the level of conservation concern regarding several species within this subfamily (Baines and Andrew 2003; Stevens et al. 2012b). The growing application of fence markers to reduce collisions has prompted government agencies and non-profit organizations to provide significant financial and personnel resources to install them at extensive scales (Natural Resources Conservation Service, 2015). This effort spurred one peer-reviewed study to evaluate the effectiveness of this practice. Stevens et al. (2012b) evaluated the effectiveness of fence markers in reducing greater sage-grouse collisions and found marked fences reduced collisions by 83%. Similarly, marking fences reduced black grouse (91%) and capercaillie (64%) collisions (Baines and Andrew 2003). Although these studies have shown that marking deer and stock fencing can reduce *Phasianid* collisions with fences, to date, no study has compared the efficacy of multiple marker types in reducing collisions, while accounting for imperfect detection, and considering site- and landscape-level factors that may influence collision rates. Durability concerns of marker types in Europe underscore the need for evaluating alternative marker styles (Baines and Andrew 2003). Additionally, few studies have empirically tested site- and landscape-scale factors that may influence the risk of grouse collisions with fencing.

Our research objectives were to 1) determine the effectiveness of different fence marker types, 2) estimate the effects of site and landscape features on collision risk and 3) evaluate an existing greater sage-grouse collision risk model. We evaluated the effectiveness of bright yellow FlySafe markers (FlySafe 2016), white markers with reflective

Fig. 2. Photographs of fence marker types deployed in our study. From left to right the above images represent the Flysafe, reflective, and white marker treatments.

tape and white markers without reflective tape compared to unmarked fence using a dataset collected in western Wyoming where sage-grouse densities are high and leks are abundant. Additionally, we investigated site and landscape features to identify areas with high collision risk and control for potentially confounding variables related to collision risk at multiple spatial scales. We evaluated an existing collision risk map (Stevens et al. 2013) to determine if observed sage-grouse collisions were correlated with areas predicted to have high or moderate collision risk.

2. Methods

2.1. Study area

Our study occurred on both private and public lands within Sublette County, Wyoming, USA. Sublette County contains some of the highest sage-grouse population indices within the occupied range (United States Fish and Wildlife Service, USFWS 2010). It lies within Management Zone II as identified by Stiver et al. (2006). The county covers approximately 3.2 million acres, of which, 80% is publicly owned. Elevations within Sublette County range from 6280 ft to 13,400 ft (Wyoming State Historical Society 2016). Lower elevations are largely characterized as sagebrush steppe habitat with riparian corridors along the Green River and its tributaries. Dominant vegetation within the lower elevation sagebrush steppe largely consists of Wyoming big sagebrush (*Artemesia tridentata* ssp. *wyomingensis*) and basin big sagebrush (*Artemesia tridentate* ssp. *tridentata*). Fencing within our study area largely consisted of three to four metal strands with barbs on all wires. A small amount of fencing within our study area consisted of metal woven wire fencing in which the bottom half of the fence consisted of both vertical and horizontal metal strands without barbs and forming rectangles 9 cm by 12 cm. Above the woven wires were typically one or two single horizontal metal wire strands with barbs.

2.2. Sampling design

We developed the sampling frame for Sublette County, Wyoming, using the 3 km-radius collision risk polygons (Stevens et al. 2013) for sage-grouse leks represented in the Wyoming Game and Fish Department lek database (Christiansen 2012). We reclassified the high and moderate risk zones into a single collision risk category and omitted the low risk zone for each of the 308 lek polygons in Sublette County (Fig. 1) using a Geographic Information System (GIS; ArcGIS Version 10.0, ESRI 2011). Next, we intersected the combined high and moderate risk zones for the lek polygons with the Bureau of Land Management (BLM) fence database (Bureau of Land Management - Pinedale Field Office, GIS Staff 2013). The sampling frame consisted of 77 lek polygons containing a minimum of 2 km of fence within the combined high and moderate risk zone of the lek polygons. We defined the sampling unit as the lek, which was represented by the 3 km-radius collision risk polygon (Stevens et al. 2013).

We selected a spatially balanced sample of 26 lek polygons (hereafter, we refer to randomly selected leks as "focal leks") using Generalized Random Tessellation Stratification (GRTS; Stevens and Olsen 2004). We determined land ownership from the Sublette County Assessor's Office and requested permission to access the sampling units in the rank order of the GRTS sample selection. When landowners denied permission, we selected the next highest rank order of the GRTS sample selection. A useful feature of the GRTS design is the spatially balanced property of the sample was maintained when private landowners denied permission to access the sampling units (Stevens and Olsen 2004).

2.3. Treatments

Each of the four treatments was randomly applied to 500 m stretches of fencing within the selected sample units. Treatments were defined as control (no marker), white (approximately 7.5 × 5 cm piece of white undersill vinyl siding), reflective (white markers with a 7.5 × 1.8 cm strip of lime-yellow Identi-Tape V97 high intensity reflective tape applied to each side), and Fly Safe markers (approximately 12 × 9 cm yellow plastic markers) (FlySafe 2016) (Fig. 2). We selected the marker treatments because they are representative of the gamut of treatments being implemented within the western U.S. to reduce sage-grouse and lesser prairie-chicken collisions with fencing. For the 500 m stretches receiving the white, reflective, or Fly Safe treatments, markers were spaced approximately 1 m from fence-posts and other markers on the top wire of the fencing to be consistent with fence marking recommendations (United States Department of Agriculture, USDA 2016). The design with all three treatments and the control employed at each sampling unit corresponds to a repeated measures design with random order of the treatments levels (Morrison et al. 2008).

2.4. Sampling methods

A total of four observers trained in sage-grouse feather identification and possessing extensive biological survey experience conducted field work throughout the two year study. Observers were intensively trained to ensure they possessed a complete understanding of field protocols, a sufficient ability to identify collision events, and could positively identify sage-grouse remains.

Surveys were conducted approximately biweekly in March and April of 2014 and 2015. A survey of a site entailed either two or four visits. The first visit consisted of an observer walking along the site's fence while scanning for evidence of animal collisions. The observer then crossed the fence and conducted the second visit by doubling back and walking to the starting point of the first visit (Fig. 1). A survey consisted of four visits when a second observer, surveying separately from the first observer, visited the same site on the same day. Observers did not discuss findings during the course of the surveys in order to avoid influencing detection rates.

Observers maintained a distance of 1-2 m from the fence during each visit. While surveying, observers primarily searched the wires of the fence for signs of a collision. Additionally, observers scanned the

Table 1
Covariates included in analyses of fence collisions by Greater Sage-Grouse in Wyoming, 2014–2015, and their expected effect on the parameter of interest (positive effect, +; negative effect, −). Parameters include large-scale occupancy (ψ), small-scale occupancy (θ), and detection probability (*p*). Means and ranges are shown for continuous covariates and levels and frequencies for the categorical covariates.

Covariate	Description	Parameter	Means (ranges) and levels (frequencies)	Expected effect
Occ Lek	Number of occupied leks within 3 km of the focal lek	ψ	1.51 (0–3)	+
Lek Ct	Sum of lek counts for leks within 3 km of focal lek	ψ	72.88 (0–265)	+
Year	Year in which survey was conducted	ψ,θ	2014 (26), 2015 (25)	N/A
Trt	Fence marker type	θ	Control (50), FlySafe (51), White (51), Reflective (50)	Risk of control > white > reflective > FlySafe
Mark	Fence marked or not	θ	Control (50), Marked (152)	Lower for marked
Angle	Angle (°) created by the triangle between the lek and end of fence segment	θ	16.34° (1°–120°)	+
Distance	Distance (km) between the midpoint of the fence segment and the nearest lek	θ	1.85 km (0.15 km–4.60 km)	−
Near Ct	Mean max male lek count for the nearest lek from 2014 to 2015	θ	54.63 (1–265)	+
Fence Exp	Mean difference (cm) between the top strand of a fence and the top of the surrounding vegetation	θ	67.69 cm (26.67 cm–96.10 cm)	+
Risk	Percentage of the fence segment in high risk areas based on Stevens et al. (2013)	θ	45.8% (0.0%–100.0%)	+
Post	Type of posts used in a fence segment	θ	Wood (138), T-post (4), both (62)	Risk of t-post > both > wood
Surv	Biweekly survey (primary) period in which survey was conducted	θ, *p*	1 (200), 2 (202), 3 (189), 4 (189), 5 (188), 6 (190), 7(186)	None
Visit	Visit (secondary period) in which survey took place	*p*	1 (1019), 2 (1014), 3 (114), 4(112)	None
Obs	Observer conducting the survey	*p*	A (432), B (226), C (525), D (1076)	None
Trap	"Trap effects" for the 2nd and 4th visits to account for potential lack of independence between visits by the same observer	*p*	1st/3rd (1133), 2nd/4th (1126)	Higher for 2nd/4th visits
Trap2	"Trap effects" accounting for whether a collision was detected or not on the 1st visit	*p*	Non-detection (1135), detection (1080)	Higher if previously detected
Cloud	Cloud cover (%)	*p*	46.1% (0.0%–100.0%)	−
Snow	Snow cover (%)	*p*	33.8% (0.0%–100.0%)	+

bushes and ground approximately 10 m out from either side of the fence for feathers or carcasses. Observers recorded ocular estimates of average snow and cloud cover (0–100%) during the course of each survey.

We considered a collision to have occurred when sage-grouse feathers were observed in the wires or barbs of a fence. We believe this represents a more accurate count of collisions as other experts have determined carcass recovery can be low due to scavenging (Stevens et al. 2011) and we believe wounded grouse may travel significant distances after striking fences before they expire. Collisions were recorded on each visit during which they were observed. In the event that feathers were found on the fence at multiple locations between two fence posts (the fencing between two fence-posts hereafter is referred to as a "panel"), the evidence was considered a single collision unless the largest gap between feathers on the wire exceeded the average wingspan of a sage-grouse (Sibley 2000). Analyses did not include any evidence in a fence that may have resulted from perching, prey plucking, or preening events, which were generally characterized by a small amount of feathers loosely affixed to the barbs of the fence and primarily distributed near a wooden post.

Observers thoroughly documented all collisions found via photographs and written notes. Observers recorded collision locations with a hand-held Global Positioning System (GPS) unit. Additionally, observers recorded the following information pertaining to the collision evidence: the distance from the evidence on the fence to the nearest fence-post, the distance from the evidence on the fence to the nearest marker, the distance from the ground (or top of the snow layer, when applicable) to the highest evidence on the fence, and the strand of wire containing the collision evidence. Finally, the observers collected the following data to describe the collision site: the distance between the two fence-posts for the panel containing the evidence, the mean height of the vegetation along the fence panel containing the collision evidence, and the number of strands of wire on the panel of fencing containing the evidence. Photographs of feathers were sent to local experts if the field observers could not be sure of identification. Collision events were only included in analyses when species identification was possible (i.e., diagnostic feathers found).

2.5. Covariate data collection

We measured fence exposure by estimating the average height of woody vegetation and the height of the top strand of fencing in centimeters for each panel. We then subtracted the height of the woody vegetation from the height of the top wire of fencing to obtain a value of "fence exposure" in centimeters for the panel. If vegetation was taller than the fence, fence exposure had a negative value. We measured these values for six panels within each 500 m stretch. Values were calculated at the two panels representing the endpoints and systematically at four additional locations at 100 m intervals along each fence segment. The fence exposure values for each of the six panels per stretch were then averaged to derive a single mean fence exposure value for the 500 m stretch. With assistance from BLM personnel, we also noted whether posts within a fence segment were wood posts, metal t-posts, or a combination of the two.

Using ArcGIS 10.0 (ESRI) we calculated several covariates including: 1) the number of occupied sage-grouse leks within 3 km of the focal lek, 2) the sum of mean maximum male lek counts in 2014 and 2015 for all leks within 3 km of the fence segment midpoint, 3) the distance from the midpoint of each fence stretch to the nearest occupied sage-grouse lek and the mean maximum male count for that lek from 2014 to 2015, 4) the proportion of each fence stretch that fell within the high risk category of the collision risk map (Stevens et al. 2013), and 5) the angle of exposure for each stretch of fence (i.e., the angle created by the triangle between the ends of the fence segment and the associated lek).

Lastly, observers estimated cloud cover during each survey and percent of the ground covered by snow to the nearest 10%. In 2014 observers recorded a single value for the average snow cover values surrounding each of the four fence segments during a survey. In 2015 observers recorded a separate value for average percentage of snow cover along each fence segment. For analyses, we calculated the mean of the 2015 values for each survey to produce a single snow cover value consistent with the 2014 data. Table 1 summarizes all covariates included in our models.

2.6. Model justification and hypotheses

We used the method of working hypotheses (Chamberlin 1965) to evaluate alternate a priori hypotheses to understand how different marker types, site- and landscape-features and mapped collision zones affect sage-grouse fence collisions. We used the covariates in Table 1 to represent hypotheses for the objectives and translated the hypotheses into predictive models. We then used the predictive models to evaluate relative strength of evidence for the alternate hypotheses in a model selection framework (Burnham and Anderson 2002). We predicted detection of sage-grouse collisions at the fence segments would be incomplete, potentially biasing the measurement of effect sizes for the fence markers. Therefore, we evaluated several hypotheses for how observers and time occasions may influence the detectability of fence collisions. We predicted the detection of collisions would vary by observer (*Obs*), time of the biweekly surveys (*Surv*), and repeated visits (*Visits*, Table 1). We accounted for potential non-independence of detections when observers visited the fence segment twice on the same day using the *Trap2* covariate (Table 1). In addition, we hypothesized that snow cover (*Snow*) and cloud (*Cloud*) cover may interfere with the ability to detect the signs of collision (Table 1).

When evaluating the effectiveness of fence markers (objective 1), we predicted that collision risk would be lower on fence segments with markers than fence segments without markers (*Mark*, Table 1) since fence marking has been shown to reduce collision risk for grouse species (Stevens et al. 2013). In addition, we hypothesized that collision risk would be lowest on fence segments with yellow Fly Safe markers, intermediate on segments with white markers with reflective tape, and greatest on segments with white markers without reflective tape (*Trt*, Table 1). Because *Phasianid* species are known to see carotenoid-based colors (Mougeot et al. 2007), we predicted the bright yellow Fly Safe markers would be more effective than white markers with reflective tape. We predicted white markers with reflective tape would be more effective than white markers without reflective tape because reflective tape is thought to provide greater visibility for low light and snow background conditions (Stevens et al. 2013). In addition, we hypothesized that fence segments with wood posts would be more effective in reducing collisions than fence segments with iron t-posts and fence segments with both types (*Post*, Table 1) because wooden posts may be more conspicuous than iron t-posts (Stevens et al. 2012a) and sage-grouse are known to avoid areas with vertical woody structure (Stiver et al. 2006).

We evaluated site- and landscape features to identify areas with greater collision risk (objective 2) at multiple scales and to control for potentially confounding variables when evaluating the effectiveness of different marker types (Morrison et al. 2008). At the local scale, we hypothesized that collision risk would be higher on fence segments near active leks (*Distance*) and near leks with greater lek attendance (*Near Ct*, Table 1) as has been shown in previous research (Stevens et al. 2012b). In addition, we predicted that collision risk would be greater on fence segments with greater fence exposure above vegetation and on fence segments (*Fence Exp*) with a larger "exposure angle" in relation to the focal lek (*Angle*, Table 1). Stevens et al. (2012a) considered a variable for the height difference between the fence and the nearest lateral shrub, but did not find strong evidence for this variable. Nevertheless, we felt sage-grouse were more likely to fly above the vegetation than between it and greater fence exposure would therefore lead to greater collision risk. Given the positive association of collisions with lek counts and small lek distances, we hypothesized that birds needing to cross fencing to attend or leave a lek would have a higher risk of collision and used the *Angle* covariate to test this hypothesis. At the landscape scale, we hypothesized that collision risk would be greater in lek polygons with high numbers of occupied leks (*Occ Lek*) and with high lek counts (*Lek Ct*, Table 1). Stevens et al. (2012a, 2012b) measured the distance between fence segments and leks to show that distribution and abundance of leks was related to collision risk at the site-scale. We measured lek density and sage-grouse abundance within the 3-km^2 radius lek buffers (28 km^2) to evaluate the extent that lek distribution and abundance influenced collision risk of lek polygons at the landscape scale. Because sage-grouse are known to move between leks on the landscape (Emmons and Braun 1984), we predicted that lek polygons containing a greater number of leks and greater numbers of birds would also have greater collision risk. If landscape measures of lek distribution and abundance prove important, these covariates can be used to account for the dependence of the treatments within 3-km^2 radius lek polygons using the repeated measures design.

To evaluate an existing collision risk map by Stevens et al. (2013) (objective 3), we predicted that collision risk would be greater along fence segments in areas characterized by high risk than on fence characterized by moderate risk (*Risk*, Table 1). Because the collision risk map was based on terrain ruggedness and distance to nearest lek (Stevens et al. 2013), this hypothesis evaluates collision risk in response to moving farther from a lek with increasing topographic relief.

2.7. Statistical analyses

We developed a multi-scale occupancy model (Nichols et al. 2008) to estimate occupancy probabilities of collision evidence, and the factors influencing them at site- and fence-segment levels. The model allowed estimation of three parameters that corresponded to each level in the nested sampling design. We used repeat visits nested within each survey to estimate detection, repeat surveys of fence segments nested within a site (i.e., lek) to estimate small-scale occupancy (the probability of a collision occurring within a 500 m fence segment), and replicate leks nested within the study area to estimate large-scale occupancy (the probability of a collision occurring within any of the four fence segments associated with the focal lek). All analyses were conducted using Program MARK (version 8.0; White and Burnham 1999) via RMARK (version 2.1.14; Laake 2013). We defined our three general parameters as: (1) the probability that evidence of ≥ 1 new sage-grouse collision was present on ≥ 1 fence segment at site i during any of the surveys, ψ_i, (2) the probability that evidence of ≥ 1 new collision was present at a fence segment during survey j, θ_{ij}, and (3) the probability that a new collision was detected on visit k, given the fence segment was occupied during survey j and visit k, p_{ijk}. The multi-scale occupancy model is well suited for the repeated measures design by allowing the investigation of covariates influencing occupancy at the large-scale (i.e., collisions at any fence segment associated with a focal lek) as well as treatments effects on conditional occupancy at the small-scale (i.e., collisions at individual fence) while accounting for non-independence of fence segments within a lek. This is analogous to how variance is estimated in a mixed model with a random effect on the focal lek (Pavlacky et al. 2012). We assumed fence segments were closed to changes in occupancy within each survey and that new collisions were accurately identified and recorded. The fence segments were allowed to be open between surveys. This model also assumes that detections are independent; however, observers conducted the second visit on the opposite side of the fence immediately after the first visit. We attempted to account for this potential lack of independence by estimating separate detection probabilities for the first and second visits by the same observer during a survey period along with whether a collision was detected during the first visit.

2.8. Model set

To investigate our hypotheses regarding the factors influencing large- and small-scale occupancy and detection, the models in our

Table 2
Model set for models explaining variation in detection probabilities (p) of Greater Sage-Grouse fence collisions in Wyoming, 2014–2015. We fit models using the most general small- (θ) and large-scale (ψ) occupancy probability model structures. Because two covariates on each occupancy probability were different measures of similar hypotheses, we included both model structures on each of those parameters. Covariates included to explain variation in detection probabilities included: fixed visit effects (Visit), fixed survey effects (Surv), fixed observer effects (Obs), "trap effects" for the 2nd and 4th visits (Trap), "trap effects" accounting for whether a collision was detected or not on the 1st visit (Trap.2), cloud cover (Cloud), and snow cover (Snow). Model structure on small-scale occupancy included: fence exposure (Fence Exp), proportion of fence segment in high risk areas (Risk), angle of fence in relation to lek (Angle), Year, biweekly (primary) period (Surv), an interaction between post type and marker type (Post × Trt), and an interaction between distance to nearest lek and the count at that lek (Distance × Near Ct). Model structures on large-scale occupancy included: Year and either the sum of lek counts at nearby leks (Lek Ct) or the number of nearby occupied leks (Occ Lek; indicated in ψ column). The number of parameters (npar), Akaike's Information Criterion adjusted for small sample size (AIC_c), difference between a model's AIC_c value and the minimum AIC_c value (ΔAIC_c), and AIC_c weights are also shown for models with $\Delta AIC_c \leq 10$.

ψ	p	npar	AIC_c	ΔAIC_c	Weight
Occ Lek	Null	25	415.082	0.000	0.582
Lek Ct	Null	25	416.051	0.969	0.358
Occ Lek	Snow	26	423.116	8.034	0.010
Occ Lek	Surv	26	423.388	8.306	0.009
Occ Lek	Cloud	26	423.572	8.490	0.008
Occ Lek	Trap.2	26	423.582	8.500	0.008
Lek Ct	snow	26	424.084	9.002	0.006
Lek Ct	surv	26	424.358	9.275	0.006
Lek Ct	cloud	26	424.541	9.459	0.005
Lek Ct	trap.2	26	424.551	9.469	0.005

model set consisted of various combinations of covariates on each parameter. We included 3 covariates on large-scale occupancy (ψ), 10 on small-scale occupancy (θ), and 7 on detection (p; Table 1). We also included interactions between post type and marker, as well as minimum distance to the nearest lek and maximum male count for that lek on θ. Because the model set was very large when considering all possible combinations of covariates, we used a sequential approach to model selection (Lebreton et al. 1992). We fit models that included all possible additive combinations of covariates on detection, while including additive effects for all covariates for large- (ψ) and small-scale (θ) occupancy. There were two covariates on large-scale occupancy that were different measures of the same hypothesis: (1) the number of occupied leks within 3 km of the focal lek (Occ Lek, Table 1) and (2) the sum of the lek counts for leks within 3 km of the focal lek (Lek Ct). We did not include both covariates in the same model. Therefore, we fit a global model containing all other additive combinations of covariates with Occ Lek and Lek Ct. separately, resulting in two global models. Then, using the most parsimonious detection structure(s), we evaluated hypotheses related to large-scale occupancy. Retaining the best large-scale occupancy model structure(s), we fit models that included all possible combinations of covariates thought to influence small-scale occupancy, including the two interaction terms.

We used an information-theoretic approach for model selection and used Akaike's Information Criterion (AIC) adjusted for sample size (AIC_c) for model comparison (Burnham and Anderson 2002). We used Akaike weights, w_i, as a measure of the relative amount of evidence for each model. Our model set for small-scale occupancy was not balanced because of the interaction terms and mutually exclusive covariates (i.e., Mark and Trt), so we used a modified version of cumulative weights based on the frequency of the covariate in the model set [$w_+(j)$] (Doherty et al. 2012) to determine the relative importance of our covariates,

$$w_+(j) = \left[\frac{w}{1-w}\right] \Big/ \left[\frac{f}{1-f}\right],$$

where w is the cumulative Akaike weight (sum of Akaike weights for models containing the covariate) and f is the frequency of models containing the covariate in the model set. Weights ≫ 1 indicate support for the importance of that variable, weights near 1 are inconclusive, and weights ≪1 indicate little support for importance. We used the odds ratio to express the effect sizes (β) in terms of the percentage increase in the odds of collision.

3. Results

We found evidence of 64 confirmed fence collisions by sage-grouse during the study, with 15 detected in 2014 and 49 detected in 2015. Additionally, we observed 96 instances of possible or likely collisions which were not included in analyses. Over 60% of sites (16 of 26) and 26% of fence segments (27 of 104) contained evidence of ≥1 confirmed collision. Only two fence segments were constructed using t-posts exclusively, and no collisions were detected at those segments; therefore, we fixed small-scale occupancy (θ) of those segments to zero to assist with numerical convergence.

Our global models used in the sequential model selection, included year and either the number of nearby occupied leks or the sum of the lek counts at those leks effects on large-scale occupancy, ψ (Year + Occ Lek) or ψ (Year + Lek Ct); year, survey, treatment × post type, distance to nearest lek × count for nearest lek, fence angle to lek, proportion in high risk areas, and fence exposure effects on small-scale occupancy, θ (Year + Surv + Distance + Angle + Risk + Fence Exp + Post × Trt + Distance × Near Ct); and observer, cloud cover, snow cover, and visit effects on detection, p (Obs + Cloud + Snow + Visit).

3.1. Detection probabilities

Using these two global models, we explored 40 other detection structures, representing simplifications of our general detection structure (Tables 2 and A1). The most parsimonious model included a constant detection probability ($w = 0.59$), as did the 2nd best model, cumulatively accounting for 95.4% of the weight; thus, we retained this detection structure, p (.), in our subsequent models. We estimated the probability of detecting ≥1 collision at 0.935 (SE = 0.026).

3.2. Large-scale occupancy

Large-scale occupancy of collisions increased as the sum of nearby lek counts increased and was higher in 2015. However, the 95% confidence intervals for both of these effects included zero. Because of this uncertainty, the most parsimonious model for ψ was the constant model, which accounted for a majority of the AIC_c weight ($w = 0.85$) (Table 3). On average, large-scale occupancy was estimated to be 0.717

Table 3
Model set for models explaining variation in large-scale occupancy probabilities (ψ) of Greater Sage-Grouse fence collisions in Wyoming, 2014–2015. We fit models using the most parsimonious model on detection probabilities (i.e., null) and the global model structure on small-scale occupancy probabilities (θ). Model structures on large-scale occupancy included: Year and either the sum of counts at leks with 3 km (Lek Ct) or the number of occupied leks within 3 km (Occ Lek; indicated in ψ column). Model structure on small-scale occupancy included: fence exposure (Fence Exp), proportion of fence segment in high risk areas (Risk), angle of fence in relation to lek (Angle), Year, biweekly (primary) period (Surv), an interaction between post type and marker type (Post × Trt), and an interaction between distance to nearest lek and the count at that lek (Distance × Near Ct). We also include the number of parameters (npar), Akaike's Information Criterion adjusted for small sample size (AIC_c), difference between a model's AICc value and the minimum AICc value (ΔAIC_c), and AIC_c weights.

ψ	npar	AIC_c	ΔAIC_c	Weight
Null	23	402.913	0.000	0.852
Lek Ct	24	408.447	5.534	0.054
Year	24	408.498	5.585	0.052
Occ Lek	24	409.084	6.171	0.039
Year + Occ Lek	25	415.082	12.170	0.002
Year + Lek Ct	25	416.051	13.139	0.001

Table 4
Cumulative AIC_c model weights for variables thought to influence small-scale occupancy (θ) of greater sage-grouse fence collisions in Wyoming, 2014–2015. Cumulative weights were adjusted based on the frequency of the covariate in the model set (Doherty et al. 2012). Variables included in the model set are: fence exposure (Fence Exp), proportion of fence segment in high risk areas (Risk), angle of fence in relation to lek (Angle), Year, biweekly (primary) period (Surv), wood post or wood and t-post (Post), marker type (Trt), whether a fence was marked or unmarked (regardless of marker type; Mark), the distance to the nearest occupied lek (Distance), the count at the nearest lek (Near Ct), an interaction between post type and marker type (Post × Trt), an interaction between post type and whether a fence was marked (Post × Mark), and an interaction between distance to nearest lek and the count at that lek (Distance × Near Ct). Modified cumulative model weights ≫ 1 suggest strong support for that variable, weights near 1 are ambiguous, and weights ≪ 1 suggest little support for that variable.

Variable	Cumulative weight
Post	12.797
Mark	4.188
Distance	3.349
Fence Exp	1.699
Year	1.261
Risk	1.246
Near Ct	1.078
Post × Mark	0.908
Surv	0.790
Distance × Near Ct	0.658
Angle	0.476
Trt	0.065
Post × Trt	0.001

(SE = 0.127).

3.3. Small-scale occupancy

We found strong evidence for effects of post type [w_+(Post) = 12.80], whether a fence was marked or not [irrespective or marker type, w_+(Mark) = 4.19], and distance to the nearest lek [w_+(Distance) = 3.35] on small-scale occupancy (Tables 4, 5, and A2). There was some support for the effects of fence exposure [w_+(Fence Exp) = 1.70], year [w_+(Year) = 1.26], the amount of fence segment within the high risk areas based on Stevens et al. (2013) [w_+(Risk) = 1.25], and the count at the nearest lek [w_+(Near Ct) = 1.08; Tables 4 and A2]. Consistent with our hypotheses, wood posts, fence marking, and increasing distance to nearest lek resulted in lower collision occupancy probabilities (Tables 6, A3, and A4 and Fig. 3). The amount of fence exposure and the proportion of fence in high risk areas increased the probability of a collision, as we predicted. Occupancy probabilities were higher in 2015 and as the count at the nearest lek increased, though these coefficients were not significant (Table 6). All marker types performed similarly [β = −0.843,(95% CI = −1.545, −0.141); odds ratio: 0.430, (0.128, 0.732)], with reflective [β = −1.018,(95% CI = −1.967, −0.068); odds ratio: 0.361, (0.018, 0.705)] and white markers [β = −0.808, (−1.703, 0.087); odds ratio: 0.446, (0.047, 0.857)] reducing occupancy probabilities slightly more than Fly Safe markers [β = −0.725, (−1.634, 0.184); odds ratio: 0.484, (0.044, 0.924)] based on the model including treatment and all other covariates with cumulative weights > 1.

4. Discussion

We adapted the multi-scale occupancy framework to investigate landscape- and local-scale features influencing the probability of fence collision, and our results support the anecdotal and limited empirical evidence for the threat of fences to sage-grouse (Christiansen 2009; Flake et al. 2010; Scott 1942; Stevens et al. 2012a, 2012b). Our study also provided insight into the factors influencing fence collisions at two spatial scales by using a multi-scale occupancy model. In addition to accounting for imperfect detection of collisions, this approach allowed us to account for the lack of independence between fence segments associated with a particular lek (Nichols et al. 2008; Pavlacky et al. 2012).

Table 5
Model set for models explaining variation in small-scale occupancy probabilities (θ) of Greater Sage-Grouse fence collisions in Wyoming, 2014–2015. We fit models using the most parsimonious model on detection probabilities (i.e., null) and large-scale occupancy probabilities (i.e., null). Model structures on small-scale occupancy included: distance to nearest lek (Distance), the count at the nearest lek (Near Ct), fence exposure (Fence Exp), wood post or t-post (Post), proportion of fence segment in high risk areas (Risk), angle of fence in relation to lek (Angle), marker type (Trt), marked or unmarked fence (regardless of marker type; Mark), Year, biweekly (primary) period (Surv), an interaction between Distance and Near Ct, and an interaction between Post and Mark or Trt. The number of parameters (npar), Akaike's Information Criterion adjusted for small sample size (AIC_c), difference between a model's AICc value and the minimum AICc value (ΔAIC_c), and AIC_c weights are also shown for the top 10 models.

θ	npar	AIC_c	ΔAIC_c	Weight
Fence Exp + Mark + Distance + Post + Risk + Near Ct	9	364.644	0.000	0.030
Fence Exp + Mark + Distance + Post + Risk + Year	9	364.756	0.111	0.028
Fence Exp + Mark + Distance + Post + Risk + Year + Near Ct	10	364.903	0.259	0.026
Fence Exp + Mark + Post + Risk + Distance × Near Ct	10	365.270	0.626	0.022
Surv + Fence Exp + Mark + Distance + Post + Risk + Year	15	365.647	1.003	0.018
Fence Exp + Mark + Distance + Post + Near Ct	8	365.762	1.118	0.017
Fence Exp + Mark + Post + Risk + Year + Distance × Near Ct	11	365.794	1.150	0.017
Surv + Fence Exp + Mark + Distance + Post + Year	14	365.810	1.166	0.017
Fence Exp + Mark + Distance + Post + Year + Near Ct	9	365.998	1.354	0.015
Fence Exp + Mark + Distance + Post + Risk	8	366.015	1.371	0.015

Table 6
Coefficient estimates, standard errors (SE), and 95% confidence intervals (CI) for all variables explaining variation in small-scale occupancy (θ) probabilities of Greater Sage-Grouse fence collisioins in Wyoming, 2014–2015. Variables include fence exposure, whether a fence was marked (regardless of marker type; Mark), the distance to nearest lek (Distance), fences with wood and t-posts (wood and t-post), proportion of fence segment in high risk areas (Risk), year (2015), and the count at the nearest lek (Near Ct). The intercept represents an unmarked fence with wood posts in 2014 with all continuous variable values set to 0. Variables included had modified cumulative AIC_c weights > 1. Estimates from the third best model are reported because it is the best model including all variables with cumulative weights > 1. All significant coefficients (i.e., 95% CIs do no overlap 0) are indicated by an asterisk.

Parameter	Mean	SE	95% CI
Intercept*	−5.544	1.123	(−7.745, −3.342)
Fence Exp*	0.031	0.013	(0.005, 0.058)
Mark*	−0.843	0.358	(−1.545, −0.141)
Distance*	−0.586	0.192	(−0.962, −0.210)
Wood and T-post*	1.774	0.382	(1.025, 2.523)
Risk*	1.150	0.565	(0.042, 2.258)
2015	0.821	0.473	(−0.105, 1.747)
Near Ct	0.004	0.002	(−0.001, 0.009)

Studies regarding potential risk of collision with human-associated infrastructure have noted that risks to lekking species may be higher in close proximity to lek locations (Baines and Summers 1997; Bevanger 1994; Stevens et al. 2012a, 2012b). Therefore, we tested four hypotheses relating to the risk of collision in association to the number of leks, the number of individuals observed at nearby leks, the position of fencing (angle) in relation to a nearby lek, and the distance to the nearest lek. Unlike Stevens et al. (2012a), we found little evidence for an effect of the number of birds using nearby leks on collision probabilities and therefore failed to confirm our hypothesis. Similarly, there

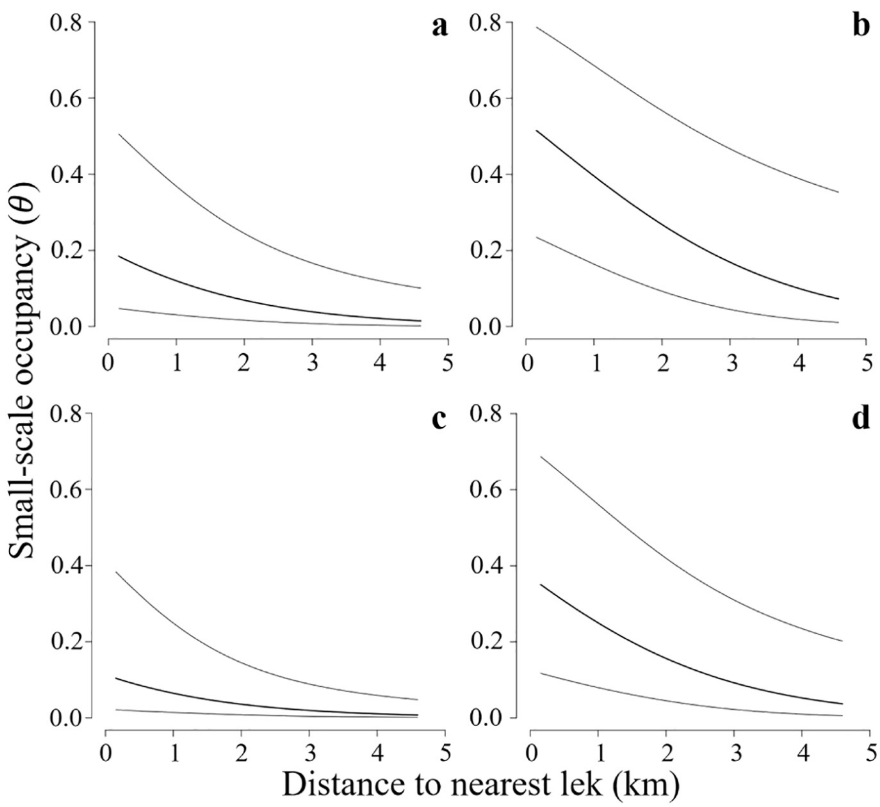

Fig. 3. Small-scale occupancy probability (θ, heavy lines) and associated 95% confidence intervals (light lines) as a function of distance to nearest lek for a) unmarked, wood post, b) unmarked, wood and t-post, c) marked, wood post, and d) marked, wood and t-post fence segments.

was no evidence to support an increased risk of collision near fencelines that are near multiple leks. Baines and Andrew (2003) similarly found no effect of lek indices on collision risk indicating that other factors may be more predicitive. Our findings may be partially due to using presence-absence data to detect differences among leks of various sizes, such that the probability of ≥ 1 collision is high for a fence near even a single smaller lek. Addtionally, lek counts have been criticized for their inability to accurately reflect abundance of sage-grouse (Beck and Braun 1980; Johnson and Rowland 2007; Walsh et al. 2004) but have been shown to be a reasonable index of the population of breeding males when standard survey protocols are followed (Jenni and Hartzler 1978; Emmons and Braun 1984; Walsh et al. 2004; Johnson and Rowland 2007). However, lek counts may not accurately represent the number of birds in the area surrounding a lek, and therefore, may be a poor indicator of the likelihood of a collision. We therefore recommend that future efforts to estimate or account for collision risk use estimated densities when possible.

Although there is an abundance of peer-reviewed work indicating that flight paths may greatly increase the risk of bird collisions with human infrastructure (Bevanger 1994; Bevanger 1998; Everaert and Stienen 2007; Henderson et al. 1996; Scott et al. 1972), we found no evidence for increased collision risk with an increased angle of fence exposure in relation to the lek which failed to confirm our hypothesis. It is possible this covariate was confounded with the distance to the nearest lek (closer distances having a larger angle) which we tested and describe in the following text. Nevertheless, we maintain that flight paths may be important in determining collision risk for some systems and species and encourage researchers to consider other potential vegetative, topographical, biological, and environmental factors that may influence or create flight paths in future studies.

We found the proximity of a fence segment to a lek influenced the probability of a collision (Distance); the average occupancy probability decreased by approximately 39% between distances of 153 m (i.e., smallest distance observed) and 1 km. This is consistent with the findings of Stevens et al. (2012a, 2012b) and confirmed our hypothesis. This relationship is likely due to increased encounters between birds and fences when a fence is closer to an area where birds congregate. We therefore recommend that marking efforts preferentially mark fence close to leks in the future. Additionally, we encourage future studies investigating risks of collisions with human-related infrastructure to consider accounting for water and/or food sources, geophagy sites, or other features that may lure large numbers of individuals into a localized area.

As in Stevens et al. (2012a), our results suggest that fence post type has the largest effect on the occupancy probability of sage-grouse collisions, with the lowest occupancy probabilities for fence segments with wooden posts, which confirmed our hypothesis. Only two fence segments in our study had t-posts exclusively and neither of those segments had evidence of a collision on them; therefore, we were unable to estimate occupancy probabilities for segments with only t-posts. Unmarked fence segments with wooden posts had lower occupancy probabilities than segments with both wooden and t-posts and any of the fence markers; yet, collision rates for fence segments with wooden posts were reduced further by the use of fence markers. These results are consistent with those found by Summers and Dugan (2001), in which, they found full length paling (which resemble wooden posts) to be the most visible fence marker. As such, we recommend future marking efforts consider testing the effectiveness of wooden stays woven into the fencing. Additionally, preferentially marking fencing with t-posts or a mixture of wood and t-posts could maximize the reduction in potential *Phasianid* collisions with fencing as our results indicated fences without wooden posts may have high rates of collisions.

We found a small effect of the amount of exposed fencing on collision risk. As vegetation height near a fence decreased, the probability of a collision increased which supported our hypothesis. *Phasianids* are generally classified as “poor flyers” (Bevanger 1994; Rayner 1988) which characteristically engage in short flights (Viscor and Fuster 1987). These morphological constraints likely result in *Phasianids* engaging in proportionately more of their flight at low altitudes, often near the top of exposed vegetation, than many birds with lower wing loading. As the top of vegetation approaches or exceeds the top of human infrastructure there is thought to be less risk of collisions (Bevanger 1994). Although we observed a weak relationship between the amount of exposed fence and collision risk, we maintain areas with short vegetation may benefit more from the use of markers by making the fence more visible. Similarly, we suggest that taller "elk fences" in the western U.S. and "deer fences" in Europe may increase collision risk beyond that of stock fencing due to the potential for additional fence projection above the vegetation as well as a general increase in total fence area. This idea was not explicitly tested in our study and represents an area for future research.

Our study design was largely based on the collision risk map developed by Stevens et al. (2013) which predicted high risk of collisions in areas close to leks and with little topography. The authors acknowledged their range-wide model was created using data collected within a relatively small geographic area in Idaho. As such, they recommended additional validation efforts be conducted. Our findings suggested a slightly increased collision probability in high risk areas, but this effect was weak. Because we attempted to select fence-line segments within the high and moderate risk areas of this map, much of the fence-line included in our study fell within these areas. Therefore, low risk areas were not well represented in our study, precluding an evaluation of the low risk portions of the risk map. We recommend further investigation of the efficacy of the collision risk map in predicting collision risk, particularly to determine if greater slopes associated with topography do impact collision risk range-wide and to determine if low risk areas on the collision risk map have a lower number of associated fence collisions. Until the collision risk map can be evaluated further, we recommend that managers seeking to reduce sage-grouse collisions focus their fence-marking efforts on fence-lines in both the high and moderate risk zones which are both close to leks and possess local site characteristics which have been shown to increase collision risk in our study and/or in previous studies.

We estimated a detection rate of 0.94, suggesting a false absence rate of 6% in the raw collision data. Our detection rate was similar to the collision detection rate calculated by Baines and Andrew (2003) when they simulated collision events with grouse carcasses. This indicates that detection of collision events is likely quite high when conducting walking surveys, provided that evidence of the collision still persists on the landscape. Stevens et al. (2011) calculated much lower detection rates when conducting walking surveys within 15 m of bird carcasses which were placed in the field; however, their estimates accounted for both detectability and scavenging bias. We suspect the scavenging bias was the driving factor in the reduced detection rates; however, they also placed carcasses beyond the search window of both our study and that of Baines and Andrews (both, of which had an effective search strip width of approximately 5 m). Furthermore, Stevens et al. placed piles of feathers and the carcasses within the habitat whereas in the Baines and Andrews study the carcasses were "vigorously thrown at the fence to simulate flight collisions". Given that we regularly witnessed feathers widely strewn across areas of 30 m or more in our study, we feel the methods used by Stevens et al. (2011) may not have accurately created conditions similar to that of an actual collision event, ultimately underestimating detection probabilities of *Phasianid* collision evidence.

Our results suggest that all three types of fence markers employed in our research were effective at reducing collision probabilities and confirmed our hypothesis, with stretches of marked fence having a 57% (27% - 87%) lower probability of containing ≥ 1 collision. These results align with previous studies by Stevens et al. (2012b) and Baines and Andrew (2003) which found marking fences reduced *Phasianid* collisions with fencing. Our results provided weak evidence that reflective markers were the most effective marker type in our study, with a 64% (30%–98%) reduction in collision probability. Stevens et al. (2012b) saw an 83% reduction in sage-grouse collisions using reflective markers. The smaller effect observed in our study may be due in part to less resolution to detect covariate effects when using occupancy models compared to abundance measures because counts are summarized to presence or absence. In addition, the smaller effect observed in our study may be partially related to accounting for incomplete detection of sage-grouse collisions, despite detection being quite high. The collision reduction estimated in our study aligns well with the estimated 64% reduction for capercaillie, 91% reduction for black grouse, and 49% reduction for red grouse estimated by Baines and Andrew (2003).

Overall, we found little difference in the effectiveness of the three marker types, as models with a marker effect (for any marker type) had substantially more cumulative AIC_c weight than models with effects for all marker types individually. However, contrary to our hypothesis, Fly Safe markers were slightly less effective than both white and reflective markers. We estimated average per marker costs for white markers at \$0.14, reflective markers at \$0.71, and Fly Safe markers at \$0.40 (USD). Therefore, using the plain white markers without reflective tape, may represent the most cost-effective sage-grouse marking strategy of those we tested. In Europe, the only study to our knowledge, which investigated marker utility in preventing *Phasianid* collisions employed two strips of orange plastic netting on the fence (Baines and Andrew 2003). The authors acknowledged that, although effective in reducing collisions within woodlands, this marker style was not suitable for deployment in areas exposed to weather (i.e., open moorland), where red grouse densities may be high. We witnessed very little damage to the three types of markers we deployed and therefore recommend trials using these marker types in open habitats of Europe.

The effectiveness of the fence markers in reducing *Phasianid* collisions highlights the importance of integrating fence marking into ongoing conservation efforts. Prescribed grazing is often recommended to improve nesting and wintering habitat conditions for lekking-species of conservation concern such as the greater-sage-grouse (Monroe et al. in review) and lesser prairie-chicken (Hagen et al. 2016). Because the implementation of rotational grazing systems involves additional fencing to subdivide an area into several pastures (United States Fish and Wildlife Service, USFWS 2010), we recommend marking exposed fence near leks even in areas thought to have only moderate collision risk due to topography. We suggest fence marking may reduce the potential for ecological traps (Battin 2004) associated with conservation practices that require the creation of additional fencing.

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Appendix A

Table A1
Model set for models explaining variation in detection probabilities (p) of Greater Sage-Grouse fence collisions in Wyoming, 2014–2015. We fit models using the most general small- (θ) and large-scale (ψ) occupancy probability model structures. Because two covariates on each occupancy probability were different measures of similar hypotheses, we included both model structures on each of those parameters. Covariates included to explain variation in detection probabilities included: fixed visit effects (Visit), fixed survey effects (Surv), fixed observer effects (Obs), "trap effects" for the 2nd and 4th visits (Trap), "trap effects" accounting for whether a collision was detected or not on the 1st visit (Trap.2), cloud cover (Cloud), and snow cover (Snow). Model structure on small-scale occupancy included: fence exposure (Fence Exp), proportion of fence segment in high risk areas (Risk), angle of fence in relation to lek (Angle), Year, biweekly (primary) period (Surv), an interaction between post type and marker type (Post × Trt), and an interaction between distance to nearest lek and the count at that lek (Distance × Near Ct). Model structures on large-scale occupancy included: Year and either the sum of lek counts at nearby leks (Lek Ct) or the number of nearby occupied leks (Occ Lek; indicated in ψ column). The number of parameters (npar), Akaike's Information Criterion adjusted for small sample size (AIC_c), difference between a model's AIC_c value and the minimum AIC_c value (ΔAIC_c), and AIC_c weights are included.

ψ	p	npar	AIC_c	ΔAIC_c	Weight
Occ Lek	Null	25	415.082	0.000	0.582
Lek Ct	Null	25	416.051	0.969	0.358
Occ Lek	Snow	26	423.116	8.034	0.010
Occ Lek	Surv	26	423.388	8.306	0.009
Occ Lek	Cloud	26	423.572	8.490	0.008
Occ Lek	Trap2	26	423.582	8.500	0.008
Lek Ct	Snow	26	424.084	9.002	0.006
Lek Ct	Surv	26	424.358	9.275	0.006
Lek Ct	Cloud	26	424.541	9.459	0.005
Lek Ct	Trap2	26	424.551	9.469	0.005
Occ Lek	Surv + Snow	27	432.197	17.115	< 0.001
Occ Lek	Cloud + Snow	27	432.347	17.265	< 0.001
Occ Lek	Snow + Trap2	27	432.355	17.273	< 0.001
Occ Lek	Surv + Cloud	27	432.568	17.486	< 0.001
Occ Lek	Surv + Trap2	27	432.627	17.545	< 0.001
Occ Lek	Trap	27	432.720	17.637	< 0.001
Occ Lek	Cloud + Trap2	27	432.811	17.729	< 0.001
Lek Ct	Surv + Snow	27	433.166	18.084	< 0.001
Lek Ct	Cloud + Snow	27	433.315	18.233	< 0.001
Lek Ct	Snow + Trap2	27	433.323	18.241	< 0.001
Lek Ct	Surv + Cloud	27	433.537	18.455	< 0.001
Lek Ct	Surv + Trap2	27	433.597	18.514	< 0.001
Lek Ct	Trap	27	433.688	18.606	< 0.001
Lek Ct	Cloud + Trap2	27	433.780	18.698	< 0.001
Occ Lek	Obs	28	439.208	24.126	< 0.001
Lek Ct	Obs	28	440.177	25.095	< 0.001
Occ Lek	Visit	28	440.748	25.665	< 0.001
Lek Ct	Visit	28	441.716	26.633	< 0.001
Occ Lek	Surv + Cloud + Snow	28	442.205	27.123	< 0.001
Occ Lek	Surv + Snow + Trap2	28	442.276	27.194	< 0.001
Occ Lek	Snow + Trap	28	442.373	27.290	< 0.001
Occ Lek	Cloud + Snow + Trap2	28	442.426	27.344	< 0.001
Occ Lek	Surv + Trap	28	442.621	27.538	< 0.001
Occ Lek	Surv + Cloud + Trap2	28	442.647	27.565	< 0.001
Occ Lek	Cloud + Trap	28	442.789	27.707	< 0.001
Lek Ct	Surv + Cloud + Snow	28	443.173	28.091	< 0.001
Lek Ct	Surv + Snow + Trap2	28	443.245	28.163	< 0.001
Lek Ct	Snow + Trap	28	443.342	28.260	< 0.001
Lek Ct	Cloud + Snow + Trap2	28	443.394	28.312	< 0.001
Lek Ct	Surv + Trap	28	443.589	28.507	< 0.001
Lek Ct	Surv + Cloud + Trap2	28	443.616	28.534	< 0.001
Lek Ct	Cloud + Trap	28	443.758	28.676	< 0.001
Occ Lek	Snow + Obs	29	449.910	34.828	< 0.001
Occ Lek	Cloud + Obs	29	450.240	35.158	< 0.001
Occ Lek	Surv + Obs	29	450.246	35.164	< 0.001
Lek Ct	Snow + Obs	29	450.877	35.795	< 0.001
Lek Ct	Cloud + Obs	29	451.208	36.126	< 0.001
Lek Ct	Surv + Obs	29	451.215	36.133	< 0.001
Occ Lek	Visit + Snow	29	451.315	36.233	< 0.001
Occ Lek	Visit + Surv	29	451.656	36.573	< 0.001
Occ Lek	Visit + Cloud	29	451.786	36.704	< 0.001
Occ Lek	Visit + Trap2	29	451.786	36.704	< 0.001
Lek Ct	Visit + Snow	29	452.283	37.200	< 0.001

Lek Ct	Visit + Surv	29	452.624	37.542	< 0.001
Lek Ct	Visit + Cloud	29	452.754	37.672	< 0.001
Lek Ct	Visit + Trap2	29	452.755	37.672	< 0.001
Occ Lek	Surv + Cloud + Snow + Trap2	29	453.244	38.162	< 0.001
Occ Lek	Surv + Snow + Trap	29	453.256	38.173	< 0.001
Occ Lek	Cloud + Snow + Trap	29	453.403	38.320	< 0.001
Occ Lek	Surv + Cloud + Trap	29	453.607	38.525	< 0.001
Lek Ct	Surv + Cloud + Snow + Trap2	29	454.212	39.130	< 0.001
Lek Ct	Surv + Snow + Trap	29	454.225	39.143	< 0.001
Lek Ct	Cloud + Snow + Trap	29	454.372	39.290	< 0.001
Lek Ct	Surv + Cloud + Trap	29	454.576	39.494	< 0.001
Occ Lek	Surv + Snow + Obs	30	462.022	46.940	< 0.001
Occ Lek	Cloud + Snow + Obs	30	462.034	46.951	< 0.001
Occ Lek	Surv + Cloud + Obs	30	462.383	47.300	< 0.001
Lek Ct	Surv + Snow + Obs	30	462.989	47.907	< 0.001
Lek Ct	Cloud + Snow + Obs	30	463.000	47.917	< 0.001
Lek Ct	Surv + Cloud + Obs	30	463.351	48.269	< 0.001
Occ Lek	Visit + Surv + Snow	30	463.354	48.271	< 0.001
Occ Lek	Visit + Cloud + Snow	30	463.458	48.376	< 0.001
Occ Lek	Visit + Snow + Trap2	30	463.458	48.376	< 0.001
Occ Lek	Visit + Trap	30	463.600	48.517	< 0.001
Occ Lek	Visit + Surv + Cloud	30	463.780	48.698	< 0.001
Occ Lek	Visit + Surv + Trap2	30	463.799	48.716	< 0.001
Occ Lek	Visit + Cloud + Trap2	30	463.929	48.847	< 0.001
Lek Ct	Visit + Surv + Snow	30	464.321	49.239	< 0.001
Lek Ct	Visit + Cloud + Snow	30	464.425	49.343	< 0.001
Lek Ct	Visit + Snow + Trap2	30	464.425	49.343	< 0.001
Lek Ct	Visit + Trap	30	464.567	49.485	< 0.001
Lek Ct	Visit + Surv + Cloud	30	464.748	49.666	< 0.001
Lek Ct	Visit + Surv + Trap2	30	464.767	49.685	< 0.001
Lek Ct	Visit + Cloud + Trap2	30	464.897	49.815	< 0.001
Occ Lek	Surv + Cloud + Snow + Trap	30	465.335	50.252	< 0.001
Lek Ct	Surv + Cloud + Snow + Trap	30	466.304	51.222	< 0.001
Occ Lek	Visit + Obs	31	474.083	59.000	< 0.001
Lek Ct	Visit + Obs	31	475.051	59.969	< 0.001
Occ Lek	Surv + Cloud + Snow + Obs	31	475.438	60.355	< 0.001
Lek Ct	Surv + Cloud + Snow + Obs	31	476.404	61.322	< 0.001
Occ Lek	Visit + Snow + Trap	31	476.629	61.547	< 0.001
Occ Lek	Visit + Surv + Cloud + Snow	31	476.755	61.673	< 0.001
Occ Lek	Visit + Surv + Snow + Trap2	31	476.775	61.692	< 0.001
Occ Lek	Visit + Cloud + Snow + Trap2	31	476.879	61.797	< 0.001
Occ Lek	Visit + Surv + Trap	31	476.984	61.902	< 0.001
Occ Lek	Visit + Cloud + Trap	31	477.020	61.938	< 0.001
Occ Lek	Visit + Surv + Cloud + Trap2	31	477.201	62.119	< 0.001
Lek Ct	Visit + Snow + Trap	31	477.597	62.515	< 0.001
Lek Ct	Visit + Surv + Cloud + Snow	31	477.723	62.641	< 0.001
Lek Ct	Visit + Surv + Snow + Trap2	31	477.742	62.660	< 0.001
Lek Ct	Visit + Cloud + Snow + Trap2	31	477.846	62.764	< 0.001
Lek Ct	Visit + Surv + Trap	31	477.952	62.870	< 0.001
Lek Ct	Visit + Cloud + Trap	31	477.988	62.906	< 0.001
Lek Ct	Visit + Surv + Cloud + Trap2	31	478.169	63.087	< 0.001
Occ Lek	Visit + Snow + Obs	32	488.636	73.554	< 0.001
Occ Lek	Visit + Cloud + Obs	32	488.987	73.905	< 0.001
Occ Lek	Visit + Surv + Obs	32	488.994	73.912	< 0.001
Lek Ct	Visit + Snow + Obs	32	489.603	74.521	< 0.001
Lek Ct	Visit + Cloud + Obs	32	489.955	74.873	< 0.001
Lek Ct	Visit + Surv + Obs	32	489.962	74.880	< 0.001
Occ Lek	Visit + Surv + Snow + Trap	32	491.496	76.413	< 0.001
Occ Lek	Visit + Cloud + Snow + Trap	32	491.542	76.459	< 0.001
Occ Lek	Visit + Surv + Cloud + Trap	32	491.893	76.811	< 0.001
Lek Ct	Visit + Surv + Snow + Trap	32	492.464	77.382	< 0.001
Lek Ct	Visit + Cloud + Snow + Trap	32	492.510	77.427	< 0.001
Lek Ct	Visit + Surv + Cloud + Trap	32	492.861	77.778	< 0.001
Occ Lek	Visit + Surv + Snow + Obs	33	505.266	90.184	< 0.001
Occ Lek	Visit + Cloud + Snow + Obs	33	505.279	90.197	< 0.001
Occ Lek	Visit + Surv + Cloud + Obs	33	505.653	90.571	< 0.001

Lek Ct	Visit + Surv + Snow + Obs	33	506.233	91.150	< 0.001
Lek Ct	Visit + Cloud + Snow + Obs	33	506.245	91.163	< 0.001
Lek Ct	Visit + Surv + Cloud + Obs	33	506.621	91.539	< 0.001

Table A2
Model set for models explaining variation in small-scale occupancy probabilities (θ) of Greater Sage-Grouse fence collisions in Wyoming, 2014–2015. We fit models using the most parsimonious model on detection probabilities (i.e., null) and large-scale occupancy probabilities (i.e., null). Model structures on small-scale occupancy included: distance to nearest lek (Distance), the count at the nearest lek (Near Ct), fence exposure (Fence Exp), wood post or t-post (Post), proportion of fence segment in high risk areas (Risk), angle of fence in relation to lek (Angle), marker type (Trt), marked or unmarked fence (regardless of marker type; Mark), Year, biweekly (primary) period (Surv), an interaction between Distance and Near Ct, and an interaction between Post and Mark or Trt. The number of parameters (npar), Akaike's Information Criterion adjusted for small sample size (AIC_c), difference between a model's AICc value and the minimum AICc value (ΔAIC_c), and AIC_c weights are included for models with $\Delta AIC_c < 4$.

θ	npar	AIC_c	ΔAIC_c	weight
Fence Exp + Mark + Distance + Post + Risk + Near Ct	9	364.644	0.000	0.030
Fence Exp + Mark + Distance + Post + Risk + Year	9	364.756	0.111	0.028
Fence Exp + Mark + Distance + Post + Risk + Year + Near Ct	10	364.903	0.259	0.026
Fence Exp + Mark + Post + Risk + Distance × Near Ct	10	365.270	0.626	0.022
Surv + Fence Exp + Mark + Distance + Post + Risk + Year	15	365.647	1.003	0.018
Fence Exp + Mark + Distance + Post + Near Ct	8	365.762	1.118	0.017
Fence Exp + Mark + Post + Risk + Year + Distance × Near Ct	11	365.794	1.150	0.017
Surv + Fence Exp + Mark + Distance + Post + Year	14	365.810	1.166	0.017
Fence Exp + Mark + Distance + Post + Year + Near Ct	9	365.998	1.354	0.015
Fence Exp + Mark + Distance + Post + Risk	8	366.015	1.371	0.015
Fence Exp + Mark + Distance + Post + Year	8	366.230	1.586	0.014
Surv + Mark + Distance + Post + Year	13	366.584	1.940	0.011
Surv + Fence Exp + Distance + Post + Risk + Year	14	366.689	2.045	0.011
Surv + Fence Exp + Mark + Distance + Post + Risk	14	366.791	2.147	0.010
Surv + Fence Exp + Mark + Distance + Post + Near Ct	14	366.803	2.159	0.010
Surv + Fence Exp + Mark + Distance + Post + Risk + Near Ct	15	366.871	2.227	0.010
Surv + Fence Exp + Distance + Post + Year	13	366.883	2.239	0.010
Surv + Mark + Distance + Post + Risk + Year	14	366.897	2.253	0.010
Fence Exp + Distance + Post + Risk + Year	8	366.926	2.282	0.010
Surv + Distance + Post + Risk + Year	13	366.997	2.353	0.009
Surv + Distance + Post + Year	12	367.005	2.361	0.009
Angle + Surv + Post + Year	12	367.072	2.428	0.009
Surv + Fence Exp + Mark + Distance + Post	13	367.177	2.533	0.008
Fence Exp + Distance + Post + Risk + Year + Near Ct	9	367.183	2.538	0.008
Angle + Surv + Distance + Post + Year	13	367.336	2.692	0.008
Surv + Fence Exp + Mark + Distance + Post + Year + Near Ct	15	367.365	2.721	0.008
Angle + Surv + Mark + Post + Year	13	367.420	2.776	0.007
Fence Exp + Distance + Risk + Near Ct + Post × Mark	10	367.457	2.813	0.007
Mark + Distance + Post + Risk + Year + Near Ct	9	367.459	2.815	0.007
Fence Exp + Distance + Risk + Year + Post × Mark	10	367.587	2.942	0.007
Fence Exp + Mark + Distance + Post	7	367.590	2.946	0.007
Surv + Fence Exp + Mark + Distance + Post + Risk + Year + Near Ct	16	367.591	2.946	0.007
Fence Exp + Distance + Risk + Year + Near Ct + Post × Mark	11	367.717	3.073	0.006
Angle + Fence Exp + Mark + Distance + Post + Risk + Near Ct	10	367.748	3.104	0.006
Angle + Fence Exp + Mark + Distance + Post + Risk + Year	10	367.821	3.177	0.006
Mark + Distance + Post + Risk + Year	8	367.882	3.238	0.006
Angle + Surv + Mark + Post	12	367.902	3.258	0.006
Mark + Distance + Post + Risk + Near Ct	8	367.992	3.348	0.006
Angle + Surv + Mark + Post + Near Ct	13	368.029	3.385	0.006
Fence Exp + Distance + Post + Risk + Near Ct	8	368.075	3.431	0.005
Surv + Mark + Distance + Post + Year + Near Ct	14	368.076	3.432	0.005
Angle + Surv + Post	11	368.076	3.432	0.005
Fence Exp + Distance + Post + Year + Near Ct	8	368.107	3.463	0.005
Angle + Fence Exp + Mark + Distance + Post + Risk + Year + Near Ct	11	368.160	3.516	0.005
Fence Exp + Distance + Post + Year	7	368.210	3.566	0.005
Mark + Distance + Post + Year + Near Ct	8	368.239	3.595	0.005
Angle + Surv + Mark + Distance + Post + Year	14	368.255	3.611	0.005
Surv + Distance + Post + Year + Near Ct	13	368.264	3.620	0.005
Fence Exp + Mark + Post + Distance × Near Ct	9	368.276	3.632	0.005
Fence Exp + Risk + Post × Mark + Distance × Near Ct	11	368.284	3.640	0.005
Surv + Mark + Distance + Post + Near Ct	13	368.308	3.664	0.005
Surv + Distance + Post + Risk + Year + Near Ct	14	368.328	3.684	0.005
Angle + Fence Exp + Mark + Post + Risk + Distance × Near Ct	11	368.379	3.735	0.005
Angle + Surv + Post + Near Ct	12	368.397	3.753	0.005

Surv + Mark + Distance + Post + Risk + Year + Near Ct	15	368.414	3.770	0.005
Surv + Fence Exp + Distance + Post + Year + Near Ct	14	368.431	3.787	0.005
Distance + Post + Risk + Year + Near Ct	8	368.445	3.801	0.004
Angle + Fence Exp + Mark + Distance + Post + Near Ct	9	368.449	3.805	0.004
Angle + Fence Exp + Mark + Distance + Post + Year	9	368.468	3.824	0.004
Fence Exp + Post + Risk + Year + Distance × Near Ct	10	368.499	3.855	0.004
Fence Exp + Distance + Near Ct + Post × Mark	9	368.531	3.886	0.004
Mark + Distance + Post + Year	7	368.550	3.906	0.004
Surv + Fence Exp + Distance + Post + Risk + Year + Near Ct	15	368.591	3.947	0.004
Mark + Distance + Post + Near Ct	7	368.623	3.979	0.004

Table A3
Coefficient estimates, standard errors (SE), and 95% confidence intervals (CI) for all variables from the best model explaining variation in small-scale occupancy (θ) probabilities of Greater Sage-Grouse fence collisions in Wyoming, 2014–2015. Variables include fence exposure (Fence Exp), whether a fence was marked (regardless of marker type; Mark), the distance to nearest lek (Distance), fences with wood and t-posts (wood and t-post), proportion of fence segment in high risk areas (Risk), and the count at the nearest lek (Near Ct). The intercept represents an unmarked fence with wood posts with all continuous variable values set to 0. All significant coefficients (i.e., 95% CIs do no overlap 0) are indicated by an asterisk.

Parameter	Mean	SE	95% CI
Intercept*	− 5.104	1.068	(− 7.197, − 3.012)
Fence Exp*	0.033	0.013	(0.007, 0.059)
Mark*	− 0.922	0.359	(− 1.623, − 0.217)
Distance*	− 0.500	0.197	(− 0.886, − 0.113)
Wood and T-post*	1.783	0.387	(1.025, 2.541)
Risk*	1.128	0.565	(0.020, 2.235)
Near Ct	0.005	0.002	(0.000, 0.010)

Table A4
Coefficient estimates, standard errors (SE), and 95% confidence intervals (CI) for all variables from the second best model explaining variation in small-scale occupancy (θ) probabilities of Greater Sage-Grouse fence collisioins in Wyoming, 2014–2015. Variables include fence exposure (Fence Exp), whether a fence was marked (regardless of marker type; Mark), the distance to nearest lek (Distance), fences with wood and t-posts (wood and t-post), proportion of fence segment in high risk areas (Risk), and the count at the nearest lek (Near Ct). The intercept represents an unmarked fence with wood posts with all continuous variable values set to 0. All significant coefficients (i.e., 95% CIs do no overlap 0) are indicated by an asterisk.

Parameter	Mean	SE	95% CI
Intercept*	− 5.181	1.090	(− 7.317, − 3.046)
Fence Exp*	0.032	0.013	(0.006, 0.058)
Mark*	− 0.818	0.356	(− 1.515, − 0.121)
Distance*	− 0.650	0.186	(− 1.015, − 0.285)
Wood and T-post*	1.685	0.374	(0.952, 2.418)
Risk*	1.161	0.557	(0.069, 2.253)
2015*	0.875	0.431	(0.030, 1.720)

References

Baines, D., Andrew, M., 2003. Marking of deer fences to reduce frequency of collisions by woodland grouse. Biol. Conserv. 110 (2), 169–176.
Baines, D., Summers, R.W., 1997. Assessment of bird collisions with deer fences in Scottish forests. J. Appl. Ecol. 941–948.
Battin, J., 2004. When good animals love bad habitats: ecological traps and the conservation of animal populations. Conserv. Biol. 18, 1482–1491.
Beck, T.D.I., Braun, C.E., 1980. The strutting ground count: variation, traditionalism, management needs. In: Proceedings of the Western Association of Fish and Wildlife Agencies. 60. pp. 558–566.
Bevanger, K., 1994. Bird interactions with utility structures: collision and electrocution, causes and mitigating measures. Ibis 136 (4), 412–425.
Bevanger, K., 1995. Estimates and population consequences of tetraonid mortality caused by collisions with high tension power lines in Norway. J. Appl. Ecol. 32, 745–753.
Bevanger, K., 1998. Biological and conservation aspects of bird mortality caused by electricity power lines: a review. Biol. Conserv. 86, 67–76.
Bevanger, K., Henriksen, G., 1996. Faunistiske effekterav gjerder og andre menneskeskapte barrierer. (In Norwegian with English Summary: Faunistic Consequences of Fences and Other Man-Made Barriers.) Norwegian Institute for Nature Research, Oppdragsmelding. 393. pp. 1–26.
Bureau of Land Management - Pinedale Field Office, GIS Staff. 2013. Unpublished Material. vector digital data.\\ilmwypd6na1\gis\gisdata\Range\fences.shp. Location of Fences within the Bureau of Land Management Pinedale Field Office in Southwest Wyoming, 1:100,000. University of Wyoming Spatial Data and Visualization Center, Laramie, Wyoming.
Burnham, K.P., Anderson, D.R., 2002. Model Selection and Multimodel Inference: A Practical Information-theoretic Approach. Springer-Verlag, New York, New York, USA.
Catt, D.C., Dugan, D., Green, R.E., Moncrieff, R., Moss, R., Picozzi, N., Summers, R.W., Tyler, G.A., 1994. Collisions against fences by woodland grouse in Scotland. Forestry 67, 105–118.
Chamberlin, T.C., 1965. The method of multiple working hypotheses. Science 148, 754–759.
Christiansen, T., 2009. Fence marking to reduce greater sage-grouse (*Centrocercus urophasianus*) collisions and mortality near Farson. In: Wyoming – Summary of Interim Results. Wyoming Game and Fish Department.
Christiansen, T., 2012. Chapter 12: sage grouse (*Centrocercus urophasianus*). In: Tessmann, S.A., Bohne, J.R. (Eds.), Handbook of Biological Techniques, Third edition. Wyoming Game and Fish Department, Cheyenne, Wyoming, USA, pp. 1–55.
Doherty, P.F., White, G.C., Burnham, K.P., 2012. Comparison of model building and selection strategies. J. Ornithol. 152, 317–323.
Drewitt, A.L., Langston, R.H., 2008. Collision effects of wind-power generators and other obstacles on birds. Ann. N. Y. Acad. Sci. 1134 (1), 233–266.
Emmons, S.R., Braun, C.E., 1984. Lek attendance of male sage grouse. J. Wildl. Manag. 48 (3), 1023–1028.
ESRI, 2011. ArcGIS Desktop: Release 10. Environmental Systems Research Institute, Redlands, CA.
Everaert, J., Stienen, E.W.M., 2007. Impact of wind turbines on birds in Zeebrugge (Belgium): significant effect on breeding tern colony due to collisions. Biodivers. Conserv. 16 (12), 3345–3359.
Flake, L.D., Connelly, J.W., Kirschenmann, T.R., Lindbloom, A.J., 2010. Grouse of Plains and Mountains: The South Dakota Story. South Dakota Department of Game, Fish, and Parks, Pierre, South Dakota, USA.
FlySafe, 2016. Fly Safe, LLC. http://www.flysafellc.com (accessed 12.16).
Hagen, C.A., Pavlacky Jr., D.C., Adachi, K., Hornsby, F.E., Rintz, T.J., McDonald, L.L., 2016. Multiscale occupancy modeling provides insights into range-wide conservation needs of lesser prairie-chicken (*Tympanuchus pallidicinctus*). Condor 118, 597–612.
Hayter, Earl W., 1939. Barbed wire fencing: a prairie invention: its rise and influence in the Western States. Agric. Hist. 13 (4), 189–207.
Hayward, M.W., Kerley, I.H., 2009. Fencing for conservation: restriction of evolutionary potential or a riposte to threatening processes? Biol. Conserv. 142, 1–13.
Henderson, I.G., Langston, R.H.W., Clark, N.A., 1996. The response of common terns

Sterna hirundo to power lines: an assessment of risk in relation to breeding commitment, age and wind speed. Biol. Conserv. 77, 185–192.
Jenni, D.A., Hartzler, J.E., 1978. Attendance at a sage grouse lek: implications for spring censuses. J. Wildl. Manag. 46–52.
Johnson, D.H., Rowland, M.M., 2007. The Utility of Lek Counts for Monitoring Greater Sage-grouse. Monitoring Populations of Sage-grouse. 88. College of Natural Resources Experiment Station Bulletinpp. 15–23.
Laake, J., 2013. RMark: An R Interface for Analysis of Capture-Recapture Data with MARK. US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center, Seattle, Washington, USA.
Lebreton, J.D., Burnham, K.P., Clobert, J., Anderson, D.R., 1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case-studies. Ecol. Monogr. 62, 67–118.
Linnell, J.D., 2016. Border controls: refugee fences fragment wildlife. Nature 529 (7585), 156.
Linnell, J.D., Trouwborst, A., Boitani, L., Kaczensky, P., Huber, D., Reljic, S., Kusak, J., Majic, A., Skrbinsek, T., Potocnik, H., Hayward, M.W., Milner-Gulland, E.J., Buuveibaatar, B., Olson, K.A., Badamjav, L., Bischof, R., Zuther, S., Breitenmoser, U., 2016. Border security fencing and wildlife: the end of the transboundary paradigm in Eurasia? Plos Boil 14 (6), e1002483.
Lisney, T.J., Iwaniuk, A.N., Kolominsky, J., Bandet, M.V., Corfield, J., 2012. Interspecific variation in eye shape and retinal topography in seven species of galliform bird (*Aves*: *Galliformes*:*Phasianidae*). J. Comp. Physiol. A. 198, 717–731.
Luzenski, J., Rocca, C.E., Harness, R.E., Cummings, J.L., Austin, D.D., Landon, M.A., Dwyer, J.F., 2016. Collision avoidance by migrating raptors encountering a new electric power transmission line. Condor 118, 402–410.
Monroe, A.P., C.L. Aldridge, T.J. Assal, K.E. Veblen, D.A. Pyke, and M.L. Casazza. In review. Patterns in greater sage-grouse population dynamics correspond with public grazing records at broad scales. Ecol. Appl.
Morrison, M.L., Block, W.M., Strickland, M.D., Collier, B.A., Peterson, M.J., 2008. Wildlife Study Design. Springer, New York, New York, USA.
Moss, R., Picozzi, N., Summers, R.W., Baines, D., 2000. Capercaillie *Tetrao urogallus* in Scotland-demography of a declining population. Ibis 142 (2), 259–267.
Mougeot, F., Martinez-Padilla, J., Perez-Rodriguez, L., Bortolotti, G.R., 2007. Carotenoid-based colouration and ultraviolet reflectance of the sexual ornaments of grouse. Behav. Ecol. Sociobiol. 61, 741–751.
Natural Resources Conservation Service, 2015. Sage Grouse Initiative 2014 Progress Report. http://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/programs/initiatives/?cid=steldevb1027671 (accessed 02.16).
Nichols, J.D., Bailey, L.L., O'Connell Jr., A.F., Talancy, N.W., Grant, E.H.C., Gilbert, A.T., Annand, E.M., Husband, T.P., Hines, J.E., 2008. Multi-scale occupancy estimation and modelling using multiple detection methods. J. Appl. Ecol. 45, 1321–1329.
Patten, M.A., Wolfe, D.H., Shochat, E., Sherrod, S.K., 2005. Effects of microhabitat and microclimate selection on adult survivorship of the lesser prairie-chicken. J. Wildl. Manag. 69 (3), 1270–1278.
Pavlacky Jr., D.C., Blakesley, J.A., White, G.C., Hanni, D.J., Lukacs, P.M., 2012. Hierarchical multi-scale occupancy estimation for monitoring wildlife populations. J. Wildl. Manag. 76, 154–162.
Rayner, J.M.V., 1988. Form and function in avian flight. In: Johnston, R.F. (Ed.), Current Ornithology. vol. 5. Plenum, New York, pp. 1–66.
von Schweppenburg, H.F., 1929. "Zugstraßen" – Leitlinien. J. Ornithol. 77 (4), 17–32.
Scott, J.W., 1942. Mating behavior of the sage-grouse. Auk 59, 477–498.
Scott, R.E., Roberts, L.J., Cadbury, C.J., 1972. Bird deaths from power lines at Dungeness. British Birds 65, 273–286.
Sibley, D.A., 2000. In: Alfred, A. (Ed.), The Sibley Guide to Birds. Knopf, New York, NY.
Sillman, A.J., 1973. Avian vision. In: Farner, D.S., King, J.R. (Eds.), Avian Biology. vol. III. Academic Press, London, pp. 349–387.
Smith, J.A., Dwyer, J.F., 2016. Avian interactions with renewable energy infrastructure: an update. Condor 118, 411–423.
Stevens Jr., D.L., Olsen, A.R., 2004. Spatially balanced sampling of natural resources. J. Am. Stat. Assoc. 99, 262–278.
Stevens, B.S., Reese, K.P., Connelly, J.W., 2011. Survival and detectability bias of avain fence collision surveys in sagebrush steppe. J. Wildl. Manag. 75 (2), 437–449.
Stevens, B.S., Connelly, J.W., Reese, K.P., 2012a. Multi-scale assessment of greater sage-grouse fence collision as a function of site and broad scale factors. J. Wildl. Manag. 76 (7), 1370–1380.
Stevens, B.S., Reese, K.P., Connelly, J.W., Musil, D.D., 2012b. Greater sage-grouse and fences: does marking reduce collisions? Wildl. Soc. Bull. 36 (2), 297–303.
Stevens, B.S., Naugle, D.E., Connelly, J.W., Griffiths, T., Reese, K.P., 2013. Mapping sage-grouse fence-collision risk: spatially explicit models for targeting conservation implementation. Wildl. Soc. Bull. 37 (2), 409–415.
Stiver, S.J., Apa, A.D., Bohne, J.R., Bunnell, S.D., Deibert, P.A., Gardner, S.C., Hilliard, M.A., McCarthy, C.W., Schroeder, M.A., 2006. Greater Sage-grouse Comprehensive Conservation Strategy. Western Association of Fish and Wildlife Agencies. (Unpublished Report. Cheyenne, Wyoming).
Summers, R.W., Dugan, D., 2001. An assessment of methods used to mark fences to reduce bird collisions in pinewoods. Scott. For. 55 (1), 23–29.
United States Department of Agriculture (USDA), 2016. How To Make and Install Fence Markers For Sage-grouse. http://www.sagegrouseinitiative.com/wp-content/uploads/2013/07/NRCD-SGI_FenceMarker_Instructions.pdf accessed 06.16).
United States Fish and Wildlife Service (USFWS), 2010. Endangered and Threatened Wildlife and Plants; 12-month Findings for Petitions to List the Greater Sage-grouse (*Centrocercus urophasianus*) as Threatened or Endangered. 50 CFR Part 17.
Viscor, G., Fuster, J.F., 1987. Relationships between morphological parameters in birds with different flying habits. Comp. Biochem. Physiol. A Physiol. 87 (2), 231–249.
Walsh, D.P., White, G.C., Remington, T.E., Bowden, D.C., 2004. Evaluation of the lek-count index for greater sage-grouse. Wildl. Soc. Bull. 32 (1), 56–68.
White, G.C., Burnham, K.P., 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46 (Suppl. 1), S120–S139.
Wolfe, D.H., Patten, M.A., Shochat, E., Pruett, C.L., Sherrod, S.K., 2007. Causes and patterns of mortality in lesser prairie-chickens *Tympanuchus pallidicinctus* and implications for management. Wildl. Biol. 13 (1), 95–104.
Wyoming State Historical Society 2016. Accessed 3/8/2016. http://www.wyohistory.org/encyclopedia/sublette-county-wyoming.

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Working to protect and restore Western Watersheds and Wildlife

August 15, 2018

Sage-Grouse Amendment Comment
USDA Forest Service - Intermountain Region
Federal Building
324 25th Street
Ogden UT 84401

Submitted by hard copy with CD of attachments and online: https://cara.ecosystem-management.org/Public/CommentInput?project=52904

Re: Comments on the Supplemental Notice of Intent to Prepare an Environmental Impact Statement; Responding to Updated Information Concerning the Forest Service Greater Sage-grouse Land and Resource Management Plan Amendments

The following comments are being submitted by Western Watersheds Project, Center for Biological Diversity, Prairie Hills Audubon Society, American Bird Conservancy, Sierra Club, WildEarth Guardians and Advocates for the West in response to the June 20, 2018 supplemental notice from the U.S. Department of Agriculture's Forest Service ("USFS") of its intent to prepare an Environmental Impact Statement concerning the FS Greater sage-grouse Land and Resource Management Plan amendments in Idaho, Nevada, Utah, Wyoming, and Wyoming/Colorado. 83 F.R. 28608-10.[1] Our organizations submitted timely comments on the agency's original scoping notice published on November 21, 2017; the following comments incorporate those comments and citations by reference and address only the additional issues that have arisen since the previous scoping period.

To summarize, the USFS proposes the following actions:

- Eliminating sagebrush focal areas (SFAs);
- Editing text for clerical errors and to reduce redundancy;
- Specific requirements will be provided for any restrictions on mineral development;
- Revisions to the exceptions process for mineral developments ("Streamlining");
- Revisions to the mapped habitat management areas;
- Livestock management guidelines will be revised to remove restrictions;
- Increased invasive plant management;

[1] The deadline for scoping comments was subsequently extended to August 15, 2018 pursuant to 83 FR. 28608-10. These

- Alignment with state planning processes;
- Focusing plan protections on priority habitat management areas ("PHMA"); and
- Changes to the compensatory mitigation framework, including removing the no net loss and net conservation gain provisions.

Below, we will address the substantive issues with the proposed action and which must be analyzed and disclosed in the forthcoming National Environmental Policy Act ("NEPA") analysis. Our organizations specifically request that the USFS provide full public review of any draft NEPA documents and allow as much time as possible for public comment during the planning process.

Endangered Species Act listing

It's plain from the proposed action that the USFS is planning to walk back the protections provided by the 2015 Greater Sage-Grouse plan amendments, protections that were supposed to sufficiently protect sage-grouse so as to preclude the need for listing the species under the Endangered Species Act ("ESA"). The current proposed actions weaken the 2015 protections and further imperil this iconic bird.

The current effort to revise the ARMPAs undercuts the Not Warranted finding's key assumption that the 2015 GRSG plans will be in effect for 20-30 years and that they are certain to be implemented and effective during that time. As the USFS proceeds with its proposed revisions, it should be aware of the PECE criteria and the impact of further weakening of the ARMPAs on the eligibility of the species for ESA listing.

Importantly, conservation efforts must satisfy the Policy for Evaluating Conservation Efforts ("PECE Policy") in terms of their effectiveness and certainty of implementation. The original RMPAs did not provide science-based effectiveness or certainty of implementation, the latter primarily through provisions for waiver, modifications, and exceptions to conservation measures. To the extent that this latest agency action to further weaken sage-grouse protections further amplifies conditions under which waiver, modifications, or exceptions can be granted, and/or weakens substantive conservation protections and thereby undermines their effectiveness, this planning process will further undermine the legal justification for the "not warranted" finding for the greater sage-grouse.

The U.S Fish and Wildlife Service must consider the PECE as the yardstick to determine the adequacy of existing regulatory mechanisms when considering whether listing is warranted. Under PECE, implementation must be certain and the proposed plan in question must be known to be effective. According to the PECE policy, "We will make this evaluation based on the certainty of implementing the conservation effort and the certainty that the effort will be effective." 68 F.R. 15113. The requirements to qualify for consideration under the PECE policy are as follows:

<u>The certainty that the conservation effort will be implemented:</u>

1. The conservation effort; the parties to the agreement or plan that will implement the effort; and the staffing, funding level, funding source, and other resources necessary to implement the effort are identified.

2. The legal authority of the parties to the agreement or plan to implement the formalized conservation effort, and the commitment to proceed with the conservation effort are described.

3. The legal procedural requirements necessary to implement the effort are described, and information is provided indicating that fulfillment of these requirements does not preclude commitment to the effort.

4. Authorizations (e.g. permits, landowner permission) necessary to implement the conservation effort are identified, and a high level of certainty is provided that the parties to the agreement or plan that will implement the effort will obtain these authorizations.

5. The type and level of voluntary participation (e.g. by private landowners) necessary to implement the conservation effort is identified, and a high level of certainty is provided that the parties to the agreement or plan that will implement the conservation effort will obtain that level of voluntary participation.

6. Regulatory mechanisms (e.g. laws, regulations, ordinances) necessary to implement the conservation effort are in place.

7. A high level of certainty is provided that the parties to the agreement or plan that will implement the conservation effort will obtain necessary funding.

8. An implementation schedule (including completion dates) for the conservation effort is provided.

9. The conservation agreement or plan that includes the conservation effort is approved by all parties to the agreement or plan.

The certainty of effectiveness

1. The nature and extent of threats being addressed by the conservation effort are described, and how the conservation effort reduces the threats is described.

2. Explicit incremental objectives for the conservation effort and dates for achieving them are stated.

3. The steps necessary to implement the conservation effort are identified in detail.

4. Quantifiable, scientifically valid parameters that will demonstrate achievement of objectives, and standards for these parameters by which progress will be measured, are identified.

5. Provisions for monitoring and reporting progress on implementation (based on compliance with the implementation schedule) and effectiveness (based on evaluation of quantifiable parameters) of the conservation effort are provided.

6. Principles of adaptive management are incorporated.

68 Fed. Reg. 15115.

Here, where the USFS is chipping away at the protection afforded by the 2015 ARMPAs and substituting in even more vague and discretionary actions, it is also undermining the certainty of implementation and effectiveness. It is also ensuring that Greater sage-grouse populations on these public lands will continue to decline and their habitats will continue to be diminished, failing the public trust and federal law.

Cumulative impacts

The USFS must analyze any proposed changes with the full consideration that the Bureau of Land Management ("BLM") is similarly reducing protections on BLM lands, thus adding to the cumulative adverse impacts of any of the agency's proposals. Additionally, the USFS must look at what is happening on adjacent private and state lands, on nearby and connected Forest Service lands, and at the interconnectivity of sage-grouse habitats across the West before making an effects determination for its proposals here.

Any and all changes to the livestock grazing provisions of the ARMPAs must also be evaluated in light of the USFS not to implement the ARMPAs in the timeframe originally proposed, but instead to modify term grazing permits "as soon as practicable." *See* https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd568845.pdf. The proposed protections of the ARMPAs haven't been implemented yet, and the 2015 NEPA analysis is no longer accurate. The forthcoming NEPA analysis must admit that even the standards and guidelines it is not proposing to change with these amendments have been undermined by the removal of a definite timeline for implementation.

According to the National Interagency Fire Center, as of August 10, 2018, 1.7 million acres total have burned across the U.S. already this year. The percentage of this within sage-grouse habitat is unknown, but the impacts of this on sage-grouse are certainly significant. As USFS moves to weaken protections for the species on FS lands, it must consider the rangewide impacts of fire to the bird and its habitat.

Eliminating sagebrush focal areas

As our January 2018 comments identified, the current plans are already insufficiently protective of sage-grouse habitat, and fail to protect all of the key sage-grouse habitat areas, winter habitats, and all populations and subpopulations of sage-grouse on public lands. The highest protections of the ARMPAs were reserved for only a subset of all sage-grouse habitats, the Sagebrush Focal Areas ("SFAs"), and our organizations suggested that the USFS consider expanding SFA protections to all Priority Habitat Management Areas (PHMAs) to be more consistent with the scientific recommendations. Instead, the USFS simply proposes now to cut SFAs entirely, removing the special protections that are so important to conserving sage-grouse in the most significant (albeit narrowly defined) habitats.

In Colorado, this means, for example, eliminating the standard to not issue new discretionary written authorizations for anthropogenic disturbances unless the total PHMA and SFA surface disturbance is less than 3 percent of the total sage-grouse habitat.[2] What this means, however, is that the 3 percent is a much larger area and can instead be entirely overlapping with the previously-designated SFA, thus wiping out the protections previously provided for the SFA that only allowed minimal disturbance. The full impact of this change should be identified and assessed in the forthcoming EIS and an assessment of how key habitats could be affected by the revision must be disclosed.

In Idaho, the proposed revision similarly cuts away at the 3 percent cap on anthropogenic disturbance by removing the SFA layer in which the total disturbance will be measured.[3] Additionally, Idaho's proposed revision deletes the No Surface Occupancy and no waivers, exceptions, or modifications for fluid leasing in SFA, opening all of the previously-designated SFA acres of the impacts of this anthropogenic use.

The 3 percent disturbance cap is also removed for SFA in the Nevada, Utah, and Wyoming proposed actions.[4] This changes the disturbance calculation significantly and means that the best habitats for sage-grouse will have far fewer protections under the proposed revisions.

In Utah, the removal of SFA is accompanied also by redefining areas with PHMA and GHMA as "non-habitat" to which the anthropogenic disturbance management would not apply. The current plan limited disturbance in SFA to valid existing rights and authorized uses in all areas of the SFA.[5] The proposed revision only applies the habitat management direction of GRSG-GEN-DC-002-Desired Condition to lands with habitat. The forthcoming EIS must analyze the affected acreage and the edge effects of such changes, and the true acreage of protected vs. unprotected areas should be disclosed.

In sum, the designation of SFAs in the 2015 plans were supposed to set aside these most important habitats for the highest level of protection. The 2018 proposed revisions remove those protections and the forthcoming EIS must assess the impacts in terms of acreage, habitat conditions, local populations, and anthropogenic uses.

Redefining – and reducing – protected areas

The USFS proposes to "evaluate the Habitat Management Area map and Biologically Significant Unit Map when a demonstrated need for change exists." Idaho proposed GRSG-GEN-0-XXX-Objective. But nowhere does the USFS describe what will happen following the evaluation, nor does the proposal define "need." The forthcoming NEPA document must disclose and analyze this proposed action in greater detail.

The proposed changes to desired conditions from being managed in "all GRSG habitat," to "At the landscape scale, in all GRSG habitat," (e.g. Idaho's proposed changes, Utah's proposed changes), render the desired condition requirements less enforceable and certainly vague. "At a landscape scale," is not defined, but would seem to entail resource conditions on non-federal lands, i.e. lands where the USFS has no control. It also fails to protect canopy cover conditions throughout sage-grouse habitat,

[2] https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd584183.pdf
[3] https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd584311.pdf
[4] https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd584291.pdf
[5] https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd584189.pdf

instead vastly enlarging the parameter by which 70 percent is derived (e.g. “70% or more of lands capable of producing sagebrush have from 10 to 30% sagebrush canopy cover and less than 10 percent conifer canopy cover,” Idaho GRSG-GEN-DC-003). Rather than requiring all sage-grouse habitat to meet these conditions, the USFS is blearing the metric by using a wholly vague and subjective denominator. This also renders the impact of the proposed management unknowable. This flies in the face of the Forest Service direction to describe desired conditions in terms that are specific enough to allow progress toward their achievement to be determined. 36 CFR 219.7(e)(1)(i), FSH 1909.12, Chapter 20.

The Nevada proposed revision changes the definitions of lek buffers from “the perimeter of a lek” to “an active or pending lek” in proposed actions such as GRSG-LG-GL-044-Guideline. The limited information about this change fails to provide the distinction, but almost certainly means, that “inactive” leks are no longer protected by such restrictions. Unfortunately, the lack of observation of breeding activity doesn’t mean that a lek is inactive or will not become active, and inactive leks should be provided the same protections as active and pending leks unless there is definitive knowledge that a lek site has been permanently abandoned.

While the removal of SFA is of great concern, we note that the USFS is also proposing to remove protective measures that are supposed to apply to IHMA in Idaho. The USFS proposed action eliminates the need to restrict issuance of special use authorizations for infrastructure, such as high-voltage lines, major pipelines, distribution lines and communication tower sites in IHMA. This reduces the protection the ARMPAs are supposed to protect and, in terms of the Idaho analysis, where the ARMPA says PHMA and IHMA encompass 90 percent of the breeding males in Idaho (e.g. Idaho ARMPA at 181), the new numbers of what percentage of breeding males will be unprotected by the proposed action must be disclosed. The proposed Idaho standard allows new authorizations in IHMA if they are offset by compensatory mitigation – a policy that is itself a target of the Trump Administration and which the USFS has never defined. The Department of Interior gutted its compensatory mitigation policy in a memo released on July 24, 2018, signaling the unwillingness of this administration to require anything meaningful in exchange for the degradation of our public lands. The USFS forthcoming NEPA analysis must be honest about what “compensatory mitigation” means to the agency and how it will impact sage-grouse habitat.

The Utah proposed changes remove protections for GHMA and Anthro Mountain, representing 80,500 acres and 41,200 acres respectively of USFS lands in Utah or nearly 1/6 of the total sage-grouse habitat that was earmarked for protections by the ARMPAs. ROD at 19. The USFS designated Anthro Mountain lands were not specifically designated as PHMA, but “they include similar management allocations and actions as those applicable to PHMA.” Id. Thus, the BLM is actually proposing to remove protections from what is effectively 41,200 acres of PHMA-level lands as well. Anthro Mountain should not be removed from the GRSG-LR-SUA-ST-013-Standard unless the Anthro Mountain Sage-Grouse Management Area is being upgraded to PHMA status. Anthro Mountain is a struggling sage-grouse population, so much so that in 2009 and 2010 resource managers augmented it with sage-grouse translocated from the Emma Park population.

The forthcoming NEPA analysis must explain why it is changing the status of this habitat and removing requirement to require protective stipulations on authorizations for new infrastructure, removal of overhead lines on terminated special user permits, etc.

Wyoming's proposed changes also remove some of the protections that the ARMPAs afforded to GHMA. For example, GRSG-R-ST-065-Standard currently protects PHMA, GHMA, and SFA by prohibiting the authorization of temporary recreation uses that would result in the loss of habitat or would have long-term impacts on habitat. The proposed change would apply this protection to only PHMA. The proposed change to GRSG-M-CM-GL-095-Guideline would completely remove the requirement to adjust coal leases to reduce threats to conserve, enhance and restore sage-grouse and its habitat in PHMA, GHMA, and SFA.

The proposed changes also tie the agency's hands when it comes to allowing land withdrawal as a tool to protect PHMA and GHMA. For example, the Nevada GRSG-LR-LW-GL-0024-Guideline denies the USFS authority to allow land withdrawal and basically waives its authority to protect special areas like the Ruby Mountains. The agency shouldn't be undermining its own autonomy and signing away its authority for blanket permissions for extractive industries; the Forest Service should retain the current language of the 2015 ARMPAs.

In Idaho, the proposed action deletes GRSG-GEN-ST-005-Standard, claiming it is redundant with 13. It is not. There is a significant difference between 005 and 13, namely that the current 005 includes General Habitat Management Areas (GHMAs). 13 does not and thus deleting 005 removes important sage-grouse protections when Forest Service is authorizing new land uses. This is especially important because there are significant knowledge gaps in regard to some Idaho greater sage-grouse populations (e.g., east-central Idaho). As a result, Forest Service cannot reasonably predict the impacts of removing protections from GHMAs by deleting 005 and should not do it unless 13 is revised to include GHMAs. These changes are significant and should be analyzed and disclosed in the forthcoming EIS.

Habitat Objectives

The Idaho proposal removes the requirement for the agency to manage grazing to achieve GRSG habitat guidelines identified in the plan and instead encourages the GRSG habitat assessments to be a part of a determination of habitat condition relative to "desired condition." Idaho GRSG-GL-035-Guideline. This means that the agency doesn't have to use the Ecological Site Descriptions of potential community condition or site potential, but rather to only require the achievement of "Suitable Condition." "Suitability" is not defined.

The GRSG Idaho/SW Montana Plan Amendment required to manage for upland perennial grass height of 7 inches during breeding and nesting season, and for 4 inches upland perennial grass height during post breeding and nesting season, and an average of 4 inches of stubble height for herbaceous riparian/mesic meadow vegetation in all greater sage-grouse habitat. *See* Final LUPA at 82. However, the proposed changes revise the guidelines to no longer require conformance with Table 3. *See* GRSG-LG-GL-035-Guideline. Instead, the new proposed action it to assess habitat conditions relative to the desire conditions, or Table 1 of the scoping doc for Idaho. But Table 1 simply adopts a grass height requirement of "Provide overhead and lateral concealment from predators." This is further defined in a footnote (15) to say, "Projects will be designed to provide overhead and lateral concealment of nests on a site-specific basis." There are no forb height requirements. These proposed changes are completely inadequate to protect sage-grouse seasonal habitat and disregards the direct relationship between hiding cover, forb availability, and nest success.

Stiver et al. (2015) recommended a minimum 18 cm grass height for all breeding and nesting habitats, and explicitly stated that this and other established measures should not be altered unless scientific evidence definitively indicates that the 7-inch threshold is inappropriate. Thus, all available science to date is consistent with standards to maintain at least 7-inches of stubble height rangewide, and more than 10.2 inches in the Dakotas.

Rather than rely on the scientific consensus about the importance of grass height, the agencies seem to want to emphasize the few papers that have questioned the methodology of the grass height studies in order to undermine their conclusions. But sage-grouse sizes haven't changed, and the birds still need sufficient cover for nesting and brood rearing to be successful. Gibson et al. (2016) cast ambiguity on the grass height requirements for nesting sage-grouse by implicating timing of data collection with phenological considerations in ways that seem to undercut the height-at-nesting grass objective. But this study admits that conditions that are present at the time of nest failure are, in fact, important to the concealment strategies of ground-nesting birds, and while taller vegetation may be an artifact of seasonality with successful nests, the inverse relationship of shorter vegetation and failed nests cannot be dismissed. Smith et al. (2018) reanalyzed previous studies and found a limited effect of grass height on nest success, but even this study admits that concealment is important for nest success and that the height of grasses may be more important in context of surrounding vegetation communities. Because none of these studies looked at the impact of livestock-caused reductions in grass height versus abiotic contributions to grass-height, the results are *at most* inconclusive in terms of grazing management. Where the authors attempt to diminish the significance of the grass height parameter in nest success, they also admit that vegetation structure might relate to other parameters of fitness such as insect abundance (*Id.*). Therefore, because grass height remains a determining factor or a proxy indicator of health, it seems premature for the BLM to dismiss the habitat objectives that require maintaining 18 cm in breeding habitat.

The agency is also unduly relying on Smith et al. 2017 to claim that grass height parameters are overemphasized. But the Smith paper really just shows that SGI projects and non-SGI projects are equally bad for sage-grouse. It specifically did not compare grazed areas with idled areas in terms of nesting success. Moreover, the leading cause of nest failure was predation (51.3 percent) and thus the question becomes whether predation is more or less common on grazed lands as a direct or indirect effect of livestock grazing becomes important. Additionally, the idled lands were only idled for 4-12 years; teasing apart differences in these samples would be interesting and looking at longer-term differences in sage-grouse habitat in light of cyclical populations would also be necessary before changing the land use plans in light of these preliminary findings.

Moreover, grass height is not the only important parameter for nesting and brood rearing success; forbs are also an important but little understood component of the sage-grouse diet for both hens and chicks (Curran et al., 2015). Where the ARMPAs had included some management parameters for forbs, the current proposed changes removes the 4 inch stubble height requirement. The proposed plans cite to Stiver et al.. 2015, but the criteria of Idaho's Table 1 for breeding and nesting habitat is specific to *perennial* forb abundance; hens are known to feed on a variety of annuals and perennials (Dumroese et al. 2016). Forbs are a major component of chick diets 2 to 10 weeks after hatching (*Id.*). The responses of forbs to grazing vary by species and grazing management regime (Pennington et al. 2016), but in general, invasive plants have reduced forb availability and grazing facilitates these invasions. The failure of BLM's methods to truly account for this important aspect of grouse habitat is a failure to truly protect the bird. The proposed change to "Preferred forbs are common with several preferred species present," is nonspecific and vague, making it difficult to evaluate and enforce. The forthcoming NEPA should explain the USFS rationale for removing strict requirements for forb structure and cover in important sage-grouse seasonable habitat.

In Utah, the USFS compounds it problematic proposed changes by describing the values in Table 1 ("Seasonal Habitat Desired Conditions for GRSG at the Landscape Scale") as "initial references" that "do not preclude development of local desired conditions or utilizing other indicators/values, based on site selection preferences of the local population and ecological site capability of sagebrush communities." See GRSG-GEN-DC-003-Desired Condition. This is entirely subjective, fails to require a scientific basis for decision making, and doesn't account for the fact that the local population may not be selecting sites that are degraded but it doesn't mean the agency should allow degradation to continue. By neglecting to apply habitat objectives to areas that the local populations aren't actively using, the USFS is managing for status quo, and status quo in Utah has led to severely declining sage-grouse populations.

In Wyoming, the agency proposes to simply delete the word "height" from the requirement for sufficient herbaceous vegetation structure *and height* to provide overhead and lateral concealment and replaces "rich diversity" with "diverse" to describe the desired perennial grass and forb communities in brood rearing habitat, wet meadows and riparian areas. Wyoming GRSG-GRSGH-DC-002-Desired Condition. Both "rich diversity" and "diverse" are highly subjective and unquantifiable, but the latter less so than the former, reducing the requirement for any depth of diversity in the habitat.

Worse, the Wyoming proposed revision makes clear that any habitat attributes are "initial references based on range-wide habitat selection" (and, we note, backed up in the peer-reviewed scientific literature), but that these values "do not preclude collaborative refinement to fit local variables of GRSG habitat use, ecological site capability, and limitations of habitat condition," which weakens the application of the science to management considerably without an objective basis for doing so. It will encourage managers at the local level to maintain status quo rather than recovery of habitats, and to exclude currently poor quality habitat from needing to meet the objectives. We note that the proposed revision also specifically provides for areas in poor condition being exempted from the habitat desired conditions: "Priority, connectivity and general habitat management areas may contain areas of non-habitat within them. Management direction does not apply in the areas of non-habitat if the proposed activity in non-habitat does not preclude effective sage-grouse use of adjacent habitats." Footnote 2 under Wyoming GRSG-GRSGH-DC-002-Desired Condition. Effective use? Adjacent? These qualifiers are vaguely defined, but could be applied to mean that livestock use in "non-habitat," e.g. a highly degraded area with a sagebrush pasture, doesn't preclude "habitat use" by sage-grouse in non-degraded areas of the same pasture, so there is no requirement to management the degraded areas to the same standards.

In Nevada, the proposed amendment to GRSG-LG-ST-042-Standard would chance the current requirement to adhere to seasonal habitat guidelines including vegetation height to instead simply prescribe 50 percent herbaceous utilization in riparian areas and meadows within PHMA and GHMA. This doesn't account for the cover requirements for the species brood-rearing habitat, because 50 percent doesn't leave specific levels of overhead protection from predators. Fifty percent utilization also allows undue impacts to native vegetation.

Reversing adaptive management

In the Idaho proposal, USFS proposes to add, "When habitat or maximum male population count exceeds the 2011 baselines for habitat or population levels within the Conservation Area, IHMA managed as PHMA consistent with [hard triggers being tripped under Idaho's] GRSG-AM-ST-010, will revert to management as IHMA within the Conservation Area." But this fails to reconcile the land management impacts that led to the decline of the habitat or population levels in the first place and instead allows the cycle of degradation to continue. The forthcoming NEPA analysis must fully explore how reverting to IHMA following the achievement of 2011 counts will not again lead to the trigger being tripped. It must also explain why the agency is using 2011 as the benchmark.

In Wyoming, the USFS includes a caveat under proposed revision to GRSG-GRSGH-ST-005-Standard, that it's Adaptive Management Working Group will establish a process to review and reverse adaptive management once the causal factor is resolved and provides, as an example, of returning to previous management once objectives of interim management have been met. But there are no indications of a requirement that the objectives of interim management reflect real recovery of the habitat or population, simply the resolution of the causal factor. The Wyoming proposed changes here cite to Appendix C – Adaptive Management, but that appendix hasn't been released to the public for comment yet. The forthcoming NEPA should provide all the relevant documents to ensure fully informed decision-making.

Due to weaknesses in the management standards, adaptive management, including the use of hard triggers, and mitigation using a net conservation benefit standard are necessary to ensure the plan's effectiveness and that grouse are being conserved. Recent policy changes to eliminate mitigation requirements on projects affecting federal lands in priority sage grouse habitat have eliminated these essential backstops, and increase the risk of endangerment.

Tweaking the language of the plans and removing protections

There are several seemingly small adjustments to language in the proposed revisions that actually have quite large impacts on the landscape.

For example, in Idaho, the USFS proposed to remove the phrase "perch deterrent installation" from the required protection stipulations of Idaho Standard GRSG-LR-SUA-ST-016. The ROD required that perch deterrents would be installed on tall structures within GRSG nesting habitat within 2 years. ROD at 79. There is no scientific rationale for removing this stipulation in Idaho. Taking it off the list of requirements for infrastructure authorizations is arbitrary and capricious. This change is also proposed for Nevada (GRSG-LR-SURA-O-013-Objective); the current language in the ARMPA provides benefits to sage-grouse by removing perching areas for avian predators. Taking this protection away is unjustified and unsupported.

In Wyoming, under GRSG-TDDD-ST-014-Standard, the USFS proposed to remove the words "and disruptive" that currently follow the prohibition on authorizing surface disturbing activities that create noise at the level of 10dB and above. Thus, disruptive activities that make noise would be allowed as long as they aren't disturbing the surface? The point of this standard was to ensure that lekking sage-grouse wouldn't be disturbed overnight, not whether the surface of the ground near the lek is affected. The USFS proposed change renders this prohibition meaningless.

Also in Wyoming, GRSG-LG-DC-036-Desired Condition redefines livestock grazing as "a tool to maintain or move towards desired habitat conditions" rather than desiring that livestock be "managed to maintain or move towards desired habitat conditions." This is a subtle difference but redefines livestock from being something in need of management to management being something in need of livestock. It's an inexplicable shift and reflects an ideological rather than scientific perspective shaping the proposed plan revisions; it's also arbitrary and capricious to make such a sweeping adjustment in the plan's perspective on land use without evidence of need for such a change.

It's curious that the Wyoming proposed plan, as one example, reassigns "Table 1. Seasonal Habitat Desired Conditions for Greater Sage-grouse at the Landscape Scale," to an appendix, "Appendix XX." Is this for the purposes of conforming to new requirements to lower EIS page numbers or is there a difference in the enforceability and implementation of items in the appendices versus those in the text? The Forest Service should make clear that the appendix remains an enforceable part of the plan, and is not merely part of the EIS evaluating the plan amendment.

Additionally in Wyoming, recent research shows that the range of dates in the proposed Wyoming GRSG-TDDD-GL-019-Guideline needs to be extended in order to protect greater sage-grouse. Smith et al. (2016) found that Wyoming sage-grouse used their winter habitats over a longer period than December 1 through March 15. Sage-grouse moved from their fall to winter habitat earlier and moved from their winter to breeding habitat later than current seasonal restrictions. We recommend that Forest Service talk with field researchers and then adjust the proposed dates to ensure that they accurately reflect greater sage-grouse use of winter habitat. In addition, because of the tremendous importance of winter habitat to sage-grouse, this should be a mandatory standard rather than a voluntary guideline.

In Nevada, the proposed addition of the language "at the landscape scale" to GRSG-GEN-DC-003-Desired Condition, would allow larger contiguous areas to be managed without sagebrush canopy cover and without conifer canopy cover because greater acreages could be offset within a larger area. This could be used to justify or permit more or larger vegetation treatments. Therefore, the existing language provides greater protection than this proposed change.

Livestock grazing provisions

The proposed Idaho changes would delete the requirement that new water developments would not be approved unless "beneficial to greater sage-grouse." Idaho GRSG-LG-ST-034.[6] The LUPA/EIS describes the impact of this alternative as benefiting upland and riparian GRSG habitats, reducing direct impacts of GRSG and their seasonal ranges. ID/MT LUPA at 4-52. Because water developments necessarily include a "sacrifice zone" of high impacts at the livestock water site, the proposed removal of this prohibition means that new direct impacts to GRSG will occur. Utah's proposed changes include the same deletion at GRSG-LG-ST-035-Standard. The forthcoming NEPA analysis must explain the agency's about-face on this provision and provide a rationale that isn't arbitrary and

[6] The actual language in the LUPA/EIS is "prohibit construction of water developments unless beneficial to greater sage-grouse habitat." ID/MT LUPA at 218.

capricious. A similar change is proposed for Nevada at GRSG-LG-STl041-Standard, deleting the prohibition on constructing water developments, but retaining the current language provides better protection by preventing further harm to sage-grouse and its habitat.

The proposed change to Idaho GRSG-LG-GL-037-Guideline is not highlighted in the online document, but it reduces by half the required distance for bedding sheep and placing camps in relation to a lek. The original guideline requires 1.2 miles (2 km) between sheep camps and the perimeter of the lek and the proposed alteration halves that to .62 miles (1 km). The forthcoming NEPA documents should provide a rationale for this change given that the original distance was selected in order to reduce disturbance from sheep, human activities, and guard animals to lekking birds. The original distance of 1.2 miles is consistent with the restriction on livestock facilities (also concentration areas) within 1.2 miles of a lek as provided for in Idaho GRSG-LG-GL-040-Guideline. A modification to this distance for sheep bedding and camps should be supported with a robust analysis in the forthcoming NEPA.

In Wyoming, the USFS is walking back the need to manage livestock grazing in accordance with objective, numeric grazing guidelines that were included in the ROD. Instead, the proposed revision to Wyoming GRSG-LG-GL-027-Guideline simply says, "managers *may* use," the Habitat Assessment Framework among many other locally-determined habitat attribute indicators. There is no requirement to use anything specific to GRSG, or to make changes to grazing practices to improve sage-grouse habitat conditions. The proposed changes also deletes GRSG-LG-GL-039 to consider the full-range of options when grazing permits are waived or permits are canceled, thus withdrawing the ROD's provision that such permits may be closed. Effectively, the change is ensuring that grazing – no matter how inappropriate, financially unfeasible, or ecologically destructive – will continue to occur on all the lands in the planning area.

Anti-Science Predator Provisions

The USFS is unnecessarily endorsing the anti-science predator-removal agendas of other agencies. For example, in Nevada's GRSG-P-DC-XX-Desired Condition, the proposed change would include adding language to the ARMPA that says, "Efforts by other agencies to minimize impacts from predators on the greater sage-grouse should be supported and encouraged where needs have been documented." Predators have generally not been shown to be a major impact to sage-grouse and/or predation increases in tandem with other disturbance (i.e. livestock grazing) that the agency should be managing. Therefore, this new provision is unlikely to benefit sage-grouse, and is likely to harm native predators.

The USFS should be aware of recent studies by Coates et al. (2016) which found that the odds of raven occurrences increased by 46 percent in areas where livestock were present, and these authors noted that spatially segregating livestock from sage-grouse breeding areas would likely reduce exposure of predatory ravens to sage-grouse nests and chicks. Though Dinkens et al. (2016) found short-term relief from raven predation to be an interim mitigation measure, they acknowledge the need for long-term solutions such as reducing the availability of dead livestock, and nesting and perching structures. Predator populations rebound quickly and predation rates will return to pre-"control" levels as soon as killing efforts abate (*see, e.g.,* Peebles and Conover 2016).

Weakening Protections from Energy and Mineral Extractive Industries Operating on Public Lands

In Wyoming, the proposed GRSG-TDDD-ST-012-Standard is too weak and does not account for new research that that shows a 0.6 lek buffer is too small to protect greater sage-grouse from oil and gas development. Green *et al.*'s 2017 study of oil and gas development's impacts on Wyoming sage-grouse from 1984 to 2008 found, "When no [oil or gas] wells were present within 6,400m of a lek, attendance for the average [Bureau of Land Management] field office was stable over the study period (Fig. 4). Lek attendance decreased more rapidly as well density increased and reached declines of 17.0%/year at 5.24 wells/km2, the highest observed well densities at a 4-year lag. Declines became significant when well density reached approximately 4 wells/km2 (l¼0.862, 95% CrI: [0.748, 0.999])." Spence *et al.*'s 2017 study of lek collapse in relationship to Wyoming oil and gas development identified an edge effect whereby the "probability of collapse among leks >4.83 km from inside Core Area boundaries was significantly related to well density within 1.61 km (1-mi) and 4.83 km (3-mi) outside of Core Area boundaries." These studies mean that to achieve stable greater sage-grouse populations, lek buffers need to be at least 3.0 miles, not 0.6 miles.

This is of especially high concern for the small and isolated northeastern Wyoming population of sage-grouse, which has been afflicted by high densities of energy development in the Powder River Basin. Gamo and Beck's 2017 study of lek attendance inside and outside of Wyoming Core Areas found that in Management Zone (northeastern Wyoming), trends in peak male attendance at leks were not greater in Core Areas than in non-core areas. Gamo and Beck state, "However, despite implementation of the SGEO [Sage-Grouse Executive Order], we are concerned with the relatively poorer performance of sage-grouse populations in MZ I. Garton et al. (2011) developed a predictive model suggesting continued declines in MZ I potentially leading to extinction in 2107 if projected trends continue." The need to protect this vulnerable population of greater sage-grouse is growing because a dramatic increase in Powder River Basin oil development is reasonably foreseeable, as indicated by recent industry investment and a 10,000 drilling permit backlog at the Wyoming Oil and Gas Conservation Commission.

The proposed Wyoming GRSG-TDDD-ST-014-Standard is too weak and does not reflect research that shows greater sage-grouse need to be protected by a fixed ceiling noise standard, not a relative dB above ambient noise standard. It is also too weak because it would remove existing noise protection from leks outside of PHMA. This change is also present in other states' plans and is problematic wherever it occurs.

The proposed Wyoming GRSG-TDDD-ST-015-Standard is too weak because it would remove existing protections from GHMA and substitutes a no net loss for a net conservation gain standard. In addition, making the standard no net loss at the statewide level would allow habitat for vulnerable sage-grouse populations in northeastern and southwestern Wyoming to be sacrificed. The new proposed GRSG-TDDD-GL-XX-Guideline is a poor substitute because continuing to authorize new development in areas that are no longer pristine will result in the loss of local sage-grouse populations.

The proposed Wyoming GRSG-LR-SUA-ST-025-Standard would weaken existing protection for greater sage-grouse by allowing the authorization temporary lands special-use permits (i.e., facilities or activities) that result in loss of habitat or would have long-term (i.e., greater than 5 years) negative impact on the greater sage-grouse or its habitat in GHMA. This is of concern for two reasons. First, Spence et al. (2017) has demonstrated that disturbance outside of Core Areas is associated with lek collapse inside Core Areas, which are roughly equivalent to PHMA. Second, in northeastern Wyoming, disturbance from intense energy development had already taken place before Core Areas, followed by PHMA, were mapped, so the designation of habitat is not as reliable in that area as in other parts of the state. This is borne out by Gamo and Beck's 2017 study, which in northeastern Wyoming did not find significant differences between male attendance at leks in Core vs. non-core areas. Furthermore, Gamo and Beck have suggested that leks in northeastern Wyoming have difficulty recovering from energy development, which has important implications for a standard about temporary lands authorizations.

The proposed Wyoming GRSG-LR-SUA-ST-029-Standard would weaken existing protection for greater sage-grouse by allowing upgrades to transmission to be sited in GHMA. This is of concern given recent research on power lines and greater sage-grouse. According to a 2018 U.S. Geological Survey greater sage-grouse research review:

> Sage-grouse occurrence increased as distance from a transmission line increased in Washington State; the maximum probability of occurrence was farther than 10 km from the transmission line (Shirk and others, 2015). Additional research reported that leks were greater than 1 km from distribution lines, as well as roads and trees, and home ranges were greater than 6 km away from single 115 kilovolt (kV) transmission lines (Stonehouse and others, 2015). Sage-grouse in this study especially avoided smaller distribution lines (about 12 kV) within their home range, which is consistent with previous research.

The existing Wyoming GRSG-LR-SUA-GL-031-Guideline should not be deleted. It is well established that greater sage-grouse abandon habitat where tall structures are built and that power lines provide perches for raptors that prey on sage-grouse.

The existing Wyoming GRSG-LR-LW-GL-034-Guideline should not be deleted. Land withdrawals are an important tool for protecting greater sage-grouse habitat.

Wyoming's GRSG-M-FMUL-ST-079-Standard should be revised to exclude the offering of new oil and gas leases in PHMA. There is a well-established body of literature that oil and gas development results in loss of greater sage-grouse leks, abandonment of habitat, and loss of sage-grouse population. Allowing new oil and gas leases in the habitat that is most important to greater sage-grouse will move the bird closer to listing under the Endangered Species Act.

Wyoming's GRSG-M-FML-ST-081-Standard should be revised to clarify that Forest Service will authorize Applications for Permits to drill inside PHMA only for valid lease rights that predate this sage-grouse plan.

The existing WY GRSG-M-CM-GL-095-Guideline should not be deleted and in fact should become a mandatory standard rather than a voluntary guideline. Adjustments to a federal coal lease should not occur without additional requirements in the readjusted lease to protect and reduce threats to conserve, enhance, and restore the greater sage-grouse and its habitat for long-term viability.

We support the addition of GHMA to the Wyoming GRSG-M-LM-ST-096-Standard for mining Plans of Operation. However, the standard's mitigation requirement should not be restricted to avoidance and minimization. It should also include rectifying impacts to greater sage-grouse and its habitat.

In Nevada, the changes to GRSG-LR-SUA-ST-014-Standard would change the restriction of special use authorizations, including transmission lines, pipelines, communication towers, to exclusion. However, exceptions may be more easily granted under the new language, for example with mitigation to achieve a net conservation gain. The provision should be clarified, and should not be adopted if it is likely to result in more exceptions to restrictions for special use authorizations.

Additionally, in Nevada, the proposed amendment GRSG-M-FMUL-ST-089 adds geothermal leases to the provision requiring NSO for oil and gas leases, but weakens the process and standard for granting exceptions to NSO for fluid mineral leases from unanimous concurrence by team of sage-grouse experts to concurrence by interagency technical team if net conservation gain.

We are concerned about the potential impacts to sage-grouse of the proposed new GRSG-AM-ST-XXX-Standard in Idaho: "When habitat or maximum male population count exceeds the 2011 baseline for habitat or population levels within the Conservation Area, IHMA managed as PHMA consistent with GRSGAM-ST-010, will revert to management as IHMA within the Conservation Area." This proposed standard could lead to harm to sage-grouse because sage-grouse habitat is not just limited to surface features. Noise is also part of the world that sage-grouse inhabit, and this proposed change does not account for the impact of excessive noise on sage-grouse.

The Idaho GRSG-LR-SUA-ST-014-Standard should not be replaced with GRSG-LR-SUA-GN-014-Guideline. Doing so would weaken protections for sage-grouse in GHMA.

The proposed Idaho GRSG-LR-SUA-ST-017-Standard would weaken existing protection for greater sage-grouse by allowing upgrades to transmission to be sited in GHMA. This is of concern given recent research on power lines and greater sage-grouse. According to a 2018 U.S. Geological Survey greater sage-grouse research review:

> Sage-grouse occurrence increased as distance from a transmission line increased in Washington State; the maximum probability of occurrence was farther than 10 km from the transmission line (Shirk and others, 2015). Additional research reported that leks were greater than 1 km from distribution lines, as well as roads and trees, and home ranges were greater than 6 km away from single 115 kilovolt (kV) transmission lines (Stonehouse and others, 2015). Sage-grouse in this study especially avoided smaller distribution lines (about 12 kV) within their home range, which is consistent with previous research.

The existing Idaho GRSG-LR-LW-GL-034-Guideline should not be deleted. Land withdrawals are an important tool for protecting greater sage-grouse habitat.

The proposed change to Idaho's GRSG-M-FMUL-ST-075-Standard weakens protections for greater sage-grouse from the previous standard by allowing authorized officers to grant exceptions to No Surface Occupancy stipulations for new oil and gas leases in important and priority habitat. The prior version required unanimous concurrence from a team of agency greater sage-grouse experts from the U.S. Fish and Wildlife Service, the Forest Service, and state wildlife agency. It also required a net conservation gain. Controlled surface use and timing limitation stipulations are no substitute for No Surface Occupancy.

The Idaho GRSG-M-FMUL-ST-077-Standard should not be deleted.

Changes to the Compensatory Mitigation Framework

As Doherty et al. (2016) stated, "we suggest that, birds, not acres or dollars spent, would be the best currency in conservation plans…." BLM must document population-level benefits for sage grouse to validate offsetting mitigation efforts. The details of mitigation must not be deferred to subsequent implementation teams because it prevents the EIS from analyzing the impacts of alternatives taking into account "offsetting" mitigation, and fails to analyze the effectiveness of mitigation measures, both of which would violate NEPA. The current political context of mitigation and the general revision to accountability policies must also be addressed.

Climate Change

Palmquist et al. (2016) predict shrinking sagebrush habitat in drier basins, and the potential for expansion of sagebrush in middle and higher elevations, due to climactic changes. Homer et al. (2015) predicted a net loss of 11.6% of current sage-grouse nesting habitat, and 4% of current sage-grouse summer habitats. Balzotti et al. (2016) found that changing climate could result in significant decreases in sagebrush habitat across much of Nevada and Utah, and that in particular, the more xeric sagebrush habitats were at elevated risk for degradation by 2050, according to their model.

The new NEPA analysis must comprehensively analyze how the projected rangewide contraction of sage-grouse habitat will affect species abundance and distribution on a rangewide basis. The new plan amendments should account for the effects of climate change by elevating protections of habitats that may serve as climate refugia.

Consistency

By fragmenting the 2015 planning process into 15 EISs and 4 RODs—and failing to create a Programmatic Environmental Impact Statement to guide the process—the agencies avoided undertaking any comprehensive or rangewide analysis of sage-grouse habitats, populations, threats, or conservation needs. Without this rangewide analysis, the agencies were also unable to properly weigh the effects of climate change, which is expected to drastically reduce the extent of sagebrush habitat on the landscape and facilitate the spread of cheatgrass. The RODs adopted revised or amended land use plans having differing and often inadequate conservation measures, which fail to assure the conservation of sage-grouse in accordance with the best available science.

Now, the USFS perpetuates that fragmentation and, rather than resolve these flaws with a range-wide hard look, instead cuts up the existing plans by state. This tactic is so overtly political rather than based on the species' actual needs, and the proposed changes are largely done to accommodate the state and industry interests in habitat exploitation. The differences between the plans among the states are indicative of the politicization of the process. Sage-grouse either need perch deterrents or they don't, they need 1.2 miles of protection from sheep camps or not; even the minor differences between the plans are evidence that the agency's proposed actions are arbitrary and capricious and not based in the best available science.

In conclusion, the proposed actions of the USFS to revise, degrade, and diminish the protective measures of the already-weak 2015 ARMPAs must be fully analyzed and the potential impacts and cumulative effects of these changes must be disclosed in the forthcoming NEPA. Our organizations incorporate by reference all previous comments, protests, and litigation on the sage-grouse ARMPAs and proposed actions, and to the extent that any of our state-specific comments above apply equally to similar provisions in other state's plans, we herewith ask that they be considered as responsive to those plans as well.

Sincerely,

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References cited and attached via CD to the hard copy sent 8/15/18

Balzotti, C. S., S. G. Kitchen, and C. McCarthy. 2016. Beyond the single species climate envelope: a multifaceted approach to mapping climate change vulnerability. Ecosphere 7(9):e01444. 10.1002/ecs2.1444.

Coates, P.S., Brussee, B.E., Howe, K.B., Gustafson, K.B., Casazza, M.L., and Delehanty, D.J. 2016a. Landscape characteristics and livestock presence influence common ravens— Relevance to greater sage-grouse conservation: Ecosphere, v. 7, no. 2, article e01203, 20 p., https://doi.org/10.1002/ecs2.1203.

Curran, M.F., T.M. Crow, K.M. Hufford, and P.D. Stahl 2015. Forbs and Greater sage-grouse Habitat Restoration Efforts: Suggestions for Improving Commercial Seed Availability and Restoration Practices. Rangelands 37(6):211-216.

Dinkins, J.B, K.T. Smith, K.J. Lawson. J.L. Beck, and A.C. Pratt. 2016. Core Area protections relative to winter concentration areas for greater sage-grouse. Wyoming Landscape Conservation Initiative, abstract only.

Doherty, K. E., J. S. Evans, P. S. Coates, L. M. Juliusson, and B. C. Fedy. 2016. Importance of regional variation in conservation planning: a rangewide example of the Greater Sage-Grouse. Ecosphere 7(10):e01462. 10.1002/ecs2.1462.

Dumroese, R.K., Luna, T., Kilkenny, F.F., Runyon, J.B., and Richardson, B.A., 2015, Conserving and restoring habitat for greater sage-grouse and other sagebrush-obligate wildlife—The crucial link of forbs and sagebrush diversity: Native Plants, v. 16, no. 3, p. 276–299.

Gamo, R.S., and J.L. Beck, J.L. 2017. Effectiveness of Wyoming's sage-grouse core areas—Influences on energy development and male lek attendance. Env. Manage. 59: 189–203.

Garton, E.O., A.G. Wells, J.A. Baumgardt, and J.W. Connelly. 2015. Greater sage-grouse population dynamics and probability of persistence. Final Report to Pew Charitable Trusts, 90 pp.

Gibson, D., Blomberg, E.J., and Sedinger, J.S., 2016, Evaluating vegetation effects on animal demographics—The role of plant phenology and sampling bias: Ecology and Evolution, v. 6, no. 11, p. 3621–3631.

Homer, C.G., G. Xian, C.L. Aldridge, D.K. Meyer, T.R. Loveland, and M.S. O'Donnell. 2015. Forecasting sagebrush ecosystem components and greater sage-grouse habitat for 2050—Learning from past climate patterns and Landsat imagery to predict the future. Ecological Indicators: 131–145.

Palmquist, K. A., D. R. Schlaepfer, J. B. Bradford, and W. K. Lauenroth. 2016. Spatial and ecological variation in dryland ecohydrological responses to climate change: implications for management. Ecosphere 7(11): e01590.10.1002/ecs2.1590.

Peebles, L.W., and M.R. Conover. 2016. Effectiveness of the toxicant DRC-1339 in reducing populations of common ravens in Wyoming. Wildlife Society Bulletin 40: 281–287. https://digitalcommons.usu.edu/wild_facpub/2531/

Pennington, V.E., Palmquist, K.A., Bradford, J.B., and Lauenroth, W.K., 2017, Climate and soil texture influence patterns of forb species richness and composition in big sagebrush plant communities across their spatial extent in the western US: Plant Ecology, v. 218, no. 8, p. 957–970.

Shirk, A.J., M.A. Schroeder, L.A. Robb, and S.A. Cushman. 2015. Empirical validation of landscape resistance models—Insights from the greater sage-grouse (*Centrocercus urophasianus*). Landscape Ecol. 30: 1837–1850.

Smith, K.T., J.L. Beck, and A.C. Pratt. 2016. Does Wyoming's core area policy protect winter habitats for greater sage-grouse? Env. Manage. 58: 585–596.

Smith, J.T., Tack, J.D., Doherty, K.E., Allred, B.W., Maestas, J.D., Berkeley, L.I., Dettenmaier, S.J., Messmer, T.A., and Naugle, D.E., 2018, Phenology largely explains taller grass at successful nests in greater sage-grouse: Ecology and Evolution, v. 8, p. 356–364, https://doi.org/10.1002/ece3.3679.

Spence, E.S., J.L. Beck, and A.J. Gregory. 2017. Probability of lek collapse is lower inside sage-grouse core areas: Effectiveness of conservation policy for a landscape species. PLoS ONE, v. 12, no. 11, article e0185885, 15 pp. https://doi.org/10.1371/journal.pone.0185885.

Stiver, S.J., E.T. Rinkes, D.E. Naugle, P.D. Makela, D.A. Nance, and J.W. Karl, eds. 2015. Sage-Grouse Habitat Assessment Framework: A Multiscale Assessment Tool. Technical Reference 6710-1. Bureau of Land Management and Western Association of Fish and Wildlife Agencies, Denver, Colorado.

WESTERN WATERSHEDS PROJECT * SIERRA CLUB
WILD UTAH PROJECT * AMERICAN BIRD CONSERVANCY * PRAIRIE HILLS AUDUBON SOCIETY * WILDEARTH GUARDIANS *ADVOCATES FOR THE WEST *
CENTER FOR BIOLOGICAL DIVERSITY

January 18, 2018

Sage-grouse Amendment Comments
USDA Forest Service Intermountain Region
Federal Building
324 25th Street,
Ogden, UT 84491

Sent via email to comments-intermtn-regional-office@fs.fed.us

Dear Secretary Purdue,

Advocates for the West, American Bird Conservancy, Center for Biological Diversity, Prairie Hills Audubon Society, Sierra Club, Western Watersheds Project, Wild Utah Project, and WildEarth Guardians submit the following comments on behalf of our members across the United States who care deeply about the persistence and recovery of the iconic greater sage-grouse on the public lands of the west.

These comments are being submitted in response to the Forest Service's "Notice of Intent to prepare an environmental impact statement," as published on November 21, 2017, 82 F.R. 55346: Idaho and Southwestern Montana (Beaverhead-Deerlodge, Boise, Caribou-Targhee, Salmon-Challis, and Sawtooth National Forests and Curlew National Grassland); Nevada (Humboldt-Toiyabe National Forest); Utah (Ashley, Dixie, Fishlake, Manti-La Sal, and Uinta-Wasatch-Cache National Forests); Wyoming (Bridger-Teton National Forest); and Wyoming/Colorado (Medicine Bow-Routt National Forest and Thunder Basin National Grassland); Amendments to Land Management Plans for Greater Sage-Grouse Conservation. The comment period was extended on January 5th in 84 FR 364 until January 19, 2018. These comments are timely.

The NOI identified a number of issues that the FS is considering; our organizations herein provide a brief background, a review of recent science, accounts of plan implementation since 2015, and information on both requested topics as well as additional issues the agency should consider in any revisions.

INTRODUCTION

In 2015, the U.S. Fish & Wildlife Service ("FWS" or "the Service") decided not to list the greater sage-grouse under the Endangered Species Act in light of the protections imposed through the Forest Service Greater Sage-grouse Land-Use Plan amendments ("the Plans") instituted by the National Sage-Grouse Planning Strategy and signed into effect by the Forest Service on September 16, 2015. Now, the Forest Service is soliciting input those Plans and opening a public discussion about the potential need to revise or amend the Plans further. It is our firm belief that if the land-use plans produced through the National Sage Grouse Planning Strategy are revised or amended, they should provide greater, not lesser, protection.

Before revising or amending the plans, the FS must assess the species' status and needs rangewide, including considering how climate change will affect the species, and must adopt the science-based protections outlined by the Service and their own scientists. These protections include refraining from fluid mineral leasing in priority habitats, buffering leks to prevent any impacts from known disturbances, ensuring that all grazing allotments are meeting science-based standards for sagebrush habitat integrity, ceasing vegetation treatments that degrade sagebrush habitat, preserving winter habitats, limiting disturbances to one per section and 3 percent per square mile, and withdrawing sagebrush habitats from anthropogenic disturbance.

We appreciate the opportunity to submit comments and we urge the Secretary to impose science-based sage-grouse protections to preserve our Nation's remarkable natural legacy for future generations.

I. BACKGROUND: GREATER SAGE-GROUSE DECLINE AND EFFORTS AT FEDERAL PROTECTION

The greater sage-grouse is a sagebrush obligate; it depends upon large expanses of contiguous sagebrush to survive. Although greater sage-grouse once numbered in the millions across the United States and Canada, the species has declined with the fragmentation and destruction of its sagebrush habitat. Greater sage-grouse populations have now declined by over 90 percent, with the few remaining birds confined to roughly half of their former range.

"There is an urgent need to 'stop the bleeding' of continued population declines and habitat losses by acting immediately to eliminate or reduce the impacts contributing to population declines and range erosion. There are no populations within the range of sage-grouse that are immune to the threat of habitat loss and fragmentation." U.S. Department of Interior, Greater Sage-Grouse Conservation Objectives: Final Report (COT Report)(COT 2013: 31-32).

A. Endangered Species Act

As early as 1999, groups began petitioning the FWS to list the greater sage-grouse as a threatened or endangered species under the Endangered Species Act ("ESA"). In 2004, the Western Association of Fish and Wildlife Agencies ("WAFWA") documented the declining trends of sagebrush habitats and sage-grouse populations (WAFWA 2004). WAFWA also published its own Conservation Strategy in 2006, with the goal of maintaining and enhancing greater sage-grouse populations and distribution by protecting and improving sagebrush habitats.[1]

Despite the threats to greater sage-grouse persistence identified in WAFWA's Conservation Assessment, in January 2005 FWS responded to several ESA petitions with a finding that the species was "not warranted" for protection under the ESA. 12-Month Finding for Petitions to List the Greater Sage-Grouse as Threatened or Endangered, 70 Fed. Reg. 2244-01 (Jan. 12, 2005). The federal District Court of Idaho reversed and remanded that finding due to improper political interference with the listing process, and the Service's arbitrary treatment of the best available science. *See Western Watersheds Project v. U.S. Fish and Wildlife Serv.*, 535 F. Supp. 2d 1173 (D. Idaho 2007).

The Service then issued a new finding, that the greater sage-grouse was "warranted" for protection under the ESA, but precluded by higher-priority species. *See* 12-Month Findings for Petitions to List the Greater Sage-Grouse As Threatened or Endangered, 75 Fed. Reg. 13910 (March 5, 2010) ("March 2010 Finding"). The Service's conclusion hinged largely on the inadequacy of existing regulatory mechanisms—especially in BLM and U.S. Forest Service land use plans—to protect the greater sage-grouse from threats from energy development (and, in particular, oil and gas development) and other activities that harm sage-grouse. Environmental groups challenged the "precluded" portion of this finding, and a settlement in separate litigation bound the Service to complete a proposed listing rule by the end of fiscal year 2015.

B. Efforts to Avoid Listing By Adopting the Strategy and Amending the Plans

Responding to the Service's "Warranted by Precluded" March 2010 Finding, the BLM and Forest Service initiated an effort to revise or amend some 98 land-use plans across ten western states to include provisions to protect the bird. Both agencies convened teams of sage-grouse scientists to make reports which led to the National Greater Sage-grouse Planning Strategy. *See* Bureau of Land Management and U.S. Forest Service, National Greater Sage-Grouse Planning Strategy (January 2012) available at https://eplanning.blm.gov/epl-front-

[1] *See* Stiver et al., Greater Sage-grouse Comprehensive Conservation Strategy (WAFWA 2006) (unpublished Report).

office/projects/lup/31652/38560/40460/Scoping_Meeting_Conservation-508_Greater_SG_Conservation_Scoping_8.pdf.

1. The NTT Report

The Sage-grouse National Technical Team (including state and federal partners; "NTT") completed a report identifying science-based conservation measures, focusing on threats in each greater sage-grouse Management Zone. The NTT issued a report issued in late December 2011, providing the best available science concerning sage-grouse threats and habitat needs. Sage-grouse National Technical Team, A Report on National Greater Sage-Grouse Conservation Measures, 14 (December 21, 2011) (NTT 2011). The NTT Report addressed key threats as follows:

a. Livestock Grazing

The NTT Report recommended: "Managing livestock grazing to maintain residual cover of herbaceous vegetation so as to reduce predation during nesting may be the most beneficial for sage□grouse populations (Beck and Mitchell 2000, Aldridge and Brigham 2003)…. Treatments used to manipulate vegetation ultimately may have far greater effect on sage□grouse through long□term habitat changes rather than direct impacts of grazing itself (Freilich et al. 2003, Knick et al. 2011)." NTT Report at 14. Accordingly, it identified measures to benefit sage-grouse, including:

- "Within priority sage□grouse habitat, incorporate sage□grouse habitat objectives and management considerations into all BLM grazing allotments through AMPs or permit renewals";
- "Prioritize completion of land health assessments and processing grazing permits within priority sage□grouse habitat areas";
- "Manage riparian areas and wet meadows for proper functioning condition within priority sage□ grouse habitats";
- "Only allow treatments that conserve, enhance or restore sage□grouse habitat (this includes treatments that benefit livestock as part of an AMP/Conservation Plan to improve sage□grouse habitat"; and
- "Maintain retirement of grazing privileges as an option in priority sage□grouse areas…."

Id. at 14-17.

b. Oil and Gas/Fluid Mineral Development

With respect to oil and gas development, the NTT noted:

> There is strong evidence from the literature to support that surface□disturbing energy or mineral development within priority sage□grouse habitats is not consistent with a goal to maintain or increase populations or distribution. None of the published science reports a positive influence of development on sage□grouse populations or habitats. Breeding populations are severely reduced at well pad densities commonly permitted (Holloran 2005, Walker et al. 2007a). Magnitude of losses varies from one field to another, but findings suggest that impacts are universally negative and typically severe.

Id. at 19. It further recognized:

> Impacts as measured by the number of males attending leks are most severe near the lek, remain discernible out to >4 miles (Holloran 2005, Walker et al. 2007, Tack 2009, Johnson et al. 2011), and often result in lek extirpations (Holloran 2005, Walker et al. 2007)…. Past BLM conservation measures have focused on 0.25 mile No Surface Occupancy (NSO) buffers around leks, and timing stipulations applied to 0.6 mile buffers around leks to protect both breeding and nesting activities. Given impacts of large scale disturbances described above that occur across seasons and impact all demographic rates, applying NSO or other buffers around leks at any distance is unlikely to be effective.

Id. at 20. It concluded:

> [T]he conservation strategy most likely to meet the objective of maintaining or increasing sage□grouse distribution and abundance is to exclude energy development and other large scale disturbances from priority habitats, and where valid existing rights exist, minimize those impacts by keeping disturbances to 1 per section with direct surface disturbance impacts held to 3% of the area or less.

Id. at 21.

In general, the NTT Report recommended closing priority sage-grouse habitats to energy development. *Id*. at 22. With respect to *already issued* leases, it recommended imposing certain conservation measures as terms and conditions of the approved RMP. These included (1) prohibiting new surface occupancy on federal leases within priority habitats during any time of the year, within limited exceptions; (2) applying a seasonal restriction on exploratory drilling that prohibits surface-disturbing activities during the nesting and early brood-rearing season in all priority sage-grouse habitat during this period; (3) not applying a Categorical Exclusion (CX) in priority sage-grouse habitats; (4) completing Master Development Plans in lieu of Application for Permit to Drill (APD) by APD processing for all but wildcat wells; (4) when permitting APDs on undeveloped areas, imposing a 3% surface disturbance cap, with limited exceptions.

c. Other Threats from Mineral Extraction.

For similar reasons, the NTT Report also recommended: "[f]ind unsuitable all surface mining of coal under the criteria set forth in 43 CFR 3461.5 [and]…[g]rant no new mining leases unless all surface disturbances (appurtenant facilities) are placed outside of the priority sage□grouse habitat area…." *Id*. at 24. It suggested withdrawing priority sage-grouse habitats from locatable mineral entry, and recommended closing priority habitats to non-energy leasable mineral development and mineral material sales. *Id*. at 25.

d. Wildfire and Vegetation Treatments.

The NTT Report recognized wildfire as a serious threat to sage-grouse, and identified measures to address that threat as well:

> · Do not reduce sagebrush canopy cover to less than 15% (Connelly et al. 2000, Hagen et al. 2007) unless a fuels management objective requires additional reduction in sagebrush cover to meet strategic protection of priority sage□grouse habitat and conserve habitat quality for the species. Closely evaluate the benefits of the fuel break against the additional loss of sagebrush cover in the EA process….
> · Allow no treatments in known winter range unless the treatments are designed to strategically reduce wildfire risk around or in the winter range and will maintain winter range habitat quality.
> · Do not use fire to treat sagebrush in less than 12□inch precipitation zones….
> · Rest treated areas from grazing for two full growing seasons unless vegetation recovery dictates otherwise (WGFD 2011)….

Id. at 26.

<u>2. The FWS COT Report</u>

The Service's Conservation Objectives Team of state and Service representatives made its recommendations in 2013 (the "COT Report"). Its framework relied on the conservation biology concepts of redundancy, representation, and resilience as guiding principles. COT Report, 12.

The COT Report recognized the primary threat to greater sage-grouse as the loss and fragmentation of sagebrush habitat. *Id*. at 10. It identified the causes of these losses as: wildfire and its feedback loop with non-native annual grasses; expansion of pinyon-juniper linked to livestock grazing and climate change; and nonrenewable energy development—in particular, oil and gas. *Id*. at 10. It also mapped "priority areas for conservation," or PACs. *Id*. at 14. These included not individual populations, but areas the states identified as necessary to ensure the redundancy, representation, and resiliency of the species. *Id*. at 13. The COT Report specifically noted that the PACs were not meant to limit the amount of sagebrush habitat to be protected and that "[a]dditional finer scale planning efforts by states may determine that additional areas outside of PACs are also essential." *Id*. Finally, it recommended specific conservation actions to address each threat. *Id*. at 40-52.

The measures the COT Report recommended included the following:

a. Grazing and Invasive Weeds

- Reduce or eliminate disturbances that promote the spread of these invasive species, such as reducing fires to a "normal range" of fire activity for the local ecosystem, *employing grazing management that maintains the perennial native grass and shrub community appropriate to the local site*, reducing impacts from any source that allows for the invasion by these species into undisturbed sagebrush habitats, *and precluding the use of treatments intended to remove sagebrush*. *Id*. at 42-43 (Emphasis added).
- Ensure that [grazing] allotments meet ecological potential and wildlife habitat requirements; and, ensure that the health and diversity of the native perennial grass community is consistent with the ecological site. *Id*. at 45.
- [Range management structures] that are currently contributing to negative impacts to either sage-grouse or their habitats should be removed or modified to remove the threat. *Id*. at 46.

b. Energy Development

- Avoid energy development in [priority habitats] (Doherty *et al*. 2010). Identify areas where leasing is not acceptable, or not acceptable without stipulations for surface occupancy that maintains sage-grouse habitats. *Id*. at 43.
- If avoidance is not possible within PACs due to pre-existing valid rights, adjacent development, or split estate issues, development should only occur in non-habitat areas, including all appurtenant structures, with an adequate buffer that is sufficient to preclude impacts to sage-grouse habitat from noise, and other human activities. *Id*. at 43.
- If development must occur in sage-grouse habitats due to existing rights and lack of reasonable alternative avoidance measures, the development should occur in the least suitable habitat for sage-grouse and be designed to ensure at a minimum that there are no detectable declines in sage-grouse population trends (and seek increases if possible) by implementing the following:

o Reduce and maintain the density of energy structures below which there are not impacts to the function of the sage-grouse habitats (as measured by no declines in sage-grouse use), or do not result in declines in sage-grouse populations within PACs.
o Design development outside PACs to maintain populations within adjacent PACs and allow for connectivity among PACs.
o Consolidate structures and infrastructure associated with energy development.
o Reclamation of disturbance resulting from a proposed project should only be considered as mitigation for those impacts, not portrayed as minimization.
o Design development to minimize tall structures (turbines, powerlines), or other features associated with the development (e.g., noise from drilling or ongoing operations; Blickley et al. 2012). *Id*. at 43.

c. Other Threats From Mineral Extraction

The COT Report also suggested avoiding new mining activities or associated facilities in sage-grouse habitats, and avoiding any new energy infrastructure in sage-grouse habitats.

3. Finalization of the Plans

The majority of BLM and Forest Service sage-grouse land use plan amendments were finalized in four Records of Decision (RODs) in September 2015; the Lander Resource Management Plan's ROD was finalized in June 2014. The two agencies produced 15 Environmental Impact Statements (EISs) associated with the proposed plan amendments. Each EIS considered an "NTT Alternative," which would have adopted the measures recommended by the NTT. However, the Plans generally took an approach that looked more like a compromise between what the science required and what industry and government stakeholders were willing to accept.

The alternatives adopted in the RODs, and crafted through the EIS process, closely tracked state sage-grouse management plans, where available. The Wyoming Plans had few discernable differences from the State of Wyoming's Core Area Strategy, even though sage-grouse populations in Wyoming had been plummeting under the state's strategy due to rampant and unchecked oil and gas development. Likewise, the Idaho and Southwestern Montana Plans applied an approach that closely resembled the State of Idaho's sage-grouse management plan in Idaho and committed to applying protections similar to Wyoming's to habitat in Montana, once the State of Montana adopted a sage-grouse plan like Wyoming's.

While better than no protections, the Plans still failed dismally in numerous respects to do what the science and law require to protect the bird. Environmental groups have filed a pending suit, alleging numerous deficiencies. We discuss some of these deficiencies in Part III below.

II. NOT WARRANTED FINDING

On October 2, 2015, the Service found the greater sage-grouse “not warranted” for listing under the ESA. The Service’s finding relied upon the habitat protections in the new Plans, including restrictions on oil and gas development and mining, disturbance caps, lek buffers, required design features intended to mitigate impacts, and the net conservation benefit mitigation standard:

> The Federal Plans, Wyoming Plan, Montana Plan, and Oregon Plan provide adequate regulatory mechanisms to reduce the threats of human-caused habitat disturbance on the most important sage-grouse habitats (as discussed in detail in the Changes Since the 2010 Finding, above). *** As a result of these measures, the Federal and three State Plans reduce the potential threat of habitat loss caused by human-caused disturbances on approximately 90 percent of breeding habitat across the species' range. These measures were effective immediately upon the implementation of the Federal Plans, the Wyoming Plan, the Montana Plan, and the Oregon Plan and will be in place for the next 20 to 30 years.

Not Warranted Finding, 80 Fed. Reg. 59858, 59934 (Oct. 2, 2015).

Rolling back these vital (yet already inadequate) protections will cause the greater sage-grouse to slide towards rangewide extinction. And, the instant effort to revisit the Plans undercuts the not warranted finding’s key assumption that the Plans will be in effect for 20-30 years. Any change to the Plans must be made with these factors in mind.

III. THE EXISTING PLANS ARE INADEQUATE

A. The Existing Plans Are Insufficiently Protective

As our organizations pointed out during the original planning process in their comments and protests, the current Plans are also seriously flawed. We request that the previous comments and protests from the 2015 decision record be incorporated into this new process and we’ve attached (Attachment A) some of the protests our organizations filed for your convenience. Because the flaws identified in the protests largely persisted into the plans, the protests still accurately identify the issues with the current Plans and the things that must be done to strengthen protections for sage-grouse on FS lands. We are also attaching the lawsuit filed on behalf of some of this letter's signatories in 2016 that identifies the legal deficiencies we've identified in the earlier planning process. Attachment B.

a. Fragmentation of the Planning Process

By fragmenting the planning process into 15 EISs and 4 RODs—and failing to create a Programmatic Environmental Impact Statement to guide the process—the agencies avoided undertaking any comprehensive or rangewide analysis of sage-grouse habitats, populations, threats, or conservation needs. Without this rangewide analysis, the agencies were also unable to properly weigh the effects of climate change, which is expected to drastically reduce the extent of sagebrush habitat on the landscape and facilitate the spread of cheatgrass. The RODs adopted revised or amended land use plans having differing and often inadequate conservation measures, which fail to assure the conservation of sage-grouse in accordance with the best available science.

b. Failure to Adequately Identify and Protect Priority Habitats

The Plans did not adequately identify and protect priority habitats. They identified sage-grouse habitat—in the process, reducing it by millions of acres from the COT Report PACs—then divided it into three or more categories: Sagebrush Focal Areas (SFAs), Priority Habitat Management Areas (PHMAs)[2], and General Habitat Management Areas (GHMAs) are present in most Plans, while the Idaho and Southwestern Montana EIS includes Intermediate Habitat Management Areas (IHMAs), the Nevada and Northeastern California EIS includes Other Habitat Management Areas (OHMAs), and the Wyoming Plans identify "core" and "connectivity" PHMAs. The agencies did not include all key sage-grouse habitats within the priority habitat designations, including all PACs and winter habitats; or encompass all SG populations and subpopulations in priority habitats. Since they did not map or identify winter habitats, they also did not apply the protections the science recommended to these important habitats. In addition, they did not consider or adequately plan for connectivity between priority habitats, providing only downgraded protections to the few habitats (mostly GHMA) supposedly intended to ensure connectivity.

Each category of habitat carries its own management scheme. The only category of habitat that imposes something close to the protections the NTT and COT Reports recommended for priority habitats, including requiring any fluid mineral leasing to occur only subject to No Surface Occupancy and withdrawing the lands from locatable mineral exploration and development, is SFAs. The other two categories rely on lesser protections, some of which are proven *not* to work to protect sage-grouse.

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[2] PHMA as used herein includes SFA. Protections to SFAs should be extended to all PHMA.

c. Failure to Follow the Best Available Science

The planning strategy did not adopt the measures recommended by the best available science.

First, the agencies did not follow the expert scientists' clear directions regarding measures to address the threat from livestock grazing. The Plans and EISs did not adequately analyze livestock grazing impacts and alternatives, or require modifications of livestock grazing necessary to protect and restore sage-grouse habitats. They did not require grazing to be deferred until mid-June as the best available science requires. Instead, the FS included a grass height objective – certainly a good idea – that relies on expensive and frequent monitoring, something the agency has insufficient staff to accomplish and therefore ensure. Worse, since the Plans were signed in September 2015, none of these changes have been implemented, and, as of November 2, 2017, it appears they won't be.[3]

Next, the agencies' treatment of fluid mineral (oil, gas, geothermal) leasing and development also fell short of the agencies' own science. The Plans failed to close priority habitats to future fluid mineral extraction, instead focusing on a "no surface occupancy" (NSO) stipulation that may be modified or excluded in all priority habitats except for SFAs. They claimed oil and gas leasing would be "prioritized" outside of sage-grouse habitats and that use restrictions like timing limitations and controlled surface use would be applied where leasing was allowed, to compensate for this failure. Wyoming's Plans did not even apply the NSO stipulation in priority habitats, and instead focused on the 0.6 mile lek buffer that the NTT Report specifically found inadequate. In addition, with the new plans in place, leases previously deferred to protect sage-grouse may now be offered up for sale. And, like the Wyoming Plans, which created a loophole to allow oil and gas development to continue as usual in fossil-fuel-rich lands in Wyoming, the Nevada/California Plan carved out an exception to the rules to allow geothermal leasing to proceed unfettered on lands with geothermal potential. Priority habitats are not closed to coal leasing, and only a tiny subset of priority sage-grouse habitat, sagebrush focal areas (SFAs), were slated to be withdrawn from locatable mineral development—although, as far as the undersigned know, this never happened even on this minimal subset of lands. The Plans lack a commitment to applying strong protections to existing fluid mineral leases, including the 3% disturbance cap; they rely on inadequate lek buffers; they fail to uniformly apply the 3% disturbance cap; and they fail to make all priority habitats exclusion areas for renewable energy, as the NTT Report recommended.

[3] https://www.fs.usda.gov/detail/ashley/news-events/?cid=FSEPRD565125

The plans create blanket exemptions for several large interstate infrastructure projects slated to cut through sage-grouse habitats: Gateway West, Gateway South, Transwest Express, and Boardman to Hemingway transmission lines.

Finally, rather than following the clear guidance set forth in the NTT Report concerning vegetation treatments, many of the plans allow using prescribed fire in priority/winter habitats, and in precipitation zones with less than 12 inches of annual precipitation. They also permit vegetation treatments in sage-grouse habitat to increase forage for livestock. Not all the plans require closing treated areas to livestock grazing for two full seasons following vegetation treatments. Only one plan included grazing permit retirement as an option in sage-grouse habitats.

In general, the plans failed to require adequate lek buffers for activities that disturb sage-grouse habitat, incorporating instead, as "guidelines", buffers from Manier et al. (2014). The Forest Service claims:

> The Forest Service will assess and address impacts from activities using the lek buffer-distances as identified in the *USGS Report Conservation Buffer Distance Estimates for GRSG – A Review* (Open File Report 2014-1239) http://pubs.usgs.gov/of/2014/1239/pdf/ofr2014-1239.pdf). The lek buffer-distances specified as the lower end of the interpreted range will be applied in the report unless justifiable departures are determined to be appropriate….

GB ROD, 39. These lower-range buffers are the following:

- linear features (roads) within 3.1 miles of leks;
- infrastructure related to energy development within 3.1 miles of leks;
- tall structures (e.g., communication or transmission towers, transmission lines) within 2 miles of leks;
- low structures (e.g., fences, rangeland structures) within 1.2 miles of leks;
- surface disturbance (continuing human activities that alter or remove the natural vegetation) within 3.1 miles of leks; and
- noise and related disruptive activities including those that do not result in habitat loss (e.g., motorized recreational events) at least 0.25 miles from leks.

Generally the buffers are to be applied to "assess and address" impacts. However, USFS may depart from the buffer distances, even in priority habitats, as long as it provides justification for its decision. The question of whether "lower-range" buffers are adequate remains relevant. And, it should be noted that buffers are a poor substitute for closing priority habitats to disruptive activities, which is what the best available science counseled.

d. Failure to Consider Cumulative Impacts.

The failure to take a rangewide perspective also meant the agencies did not consider cumulative impacts from the activities potentially allowed under the plans. The plans adopt a smorgasbord of different "conservation measures" to respond to threats, but their lack of uniformity and complex regulatory web create uncertainty about how they will be applied. There is no analysis of how the exceptions and inconsistencies will affect sage-grouse.

e. FS Must Comply with the National Forests Management Act (NFMA).

The NFMA planning regulations provide, "The responsible official shall use the best available scientific information to inform the planning process required by this subpart." 36 C.F.R § 219.3. Within these comments we have provided a number of referenced to peer-reviewed scientific studies, which we ask the responsible official to consider for the purpose of informing the planning process. Furthermore, NFMA requires a number of specific steps to be taken in the use of the best available science, as follows:

> In doing so, the responsible official shall determine what information is the most accurate, reliable, and relevant to the issues being considered. The responsible official shall document how the best available scientific information was used to inform the assessment, the plan decision, and the monitoring program as required in §§ 219.6(a)(3) and 219.14(a)(4). Such documentation must: Identify what information was determined to be the best available scientific information, explain the basis for that determination, and explain how the information was applied to the issues considered.

36 C.F.R. § 219.3. With this in mind, the Forest Service is bound to render these determinations regarding incorporating the best available science on a point-by-point basis.

B. The Powder River Sage-Grouse Population in in Danger of Extirpation under MVP Analysis

The Powder River Basin sage-grouse population is in deep trouble, and Forest Service management on the Thunder Basin National Grassland, and inadequate sage-grouse protections there., is a major part of the problem. A population persistence study by Garton et al. (2015) incorporates the latest state population data to calculate the probability that various populations will drop below minimum viable population thresholds at the Management Zone and subpopulation levels. *See* Attachment I. According to this study, the prospects for sage grouse populations are even bleaker today than in 2010, when the species was found to be 'warranted, but precluded' for Endangered Species Act listing. According to this study, the Powder River population (all of northeast Wyoming including the Buffalo Field Office, Thunder Basin National Grassland, parts of Casper Field Office, and Newcastle Field Office) has a 98.7% chance of dropping below an effective population size of 50 in 30 years, with a 55% chance of sage grouse populations across the Great Plains (Management Zone I) dropping below 50 in 100 years. These conclusions illustrate that BLM has been failing to uphold its FLPMA obligation to prevent unnecessary or undue degradation to sage grouse habitats, and failing to uphold Sensitive Species requirements, for many years; this plan revision offers BLM the opportunity to reverse this legal failing and the agency is obligated by law to do this.

In its initial designation of Core Areas, the State of Wyoming made some major errors in the Buffalo Field Office that have been implicated in subsequent population declines and threats to long-term viability for sage grouse populations (*see* Taylor et al. 2012). These failures were adopted by the BLM and Forest Service, crippling the ability of the new plan to maintain viable populations of sage grouse in this area. It is important to note that many of the most populous sage grouse leks in the Buffalo Field Office lie outside Core Area boundaries. See Attachments O, P, and Q, Buffalo FEIS 32, 33, 36. The State of Wyoming has developed current lek population density mapping based on 2014 data, which is readily available to BLM. BLM should have included such a population density buffer map with its Buffalo FEIS as part of its NEPA baseline information fulfillment; failure to do so violates NEPA. The majority of identified nesting habitat in the Buffalo Field Office lies outside designated Core and Connectivity Areas. Buffalo RMP FEIS at Map 37.

In particular, the Buffalo Core Area was not designated based on 4-mile or 5.3-mile buffers around the most populous leks, as were most Core Areas designated under the State of Wyoming Executive Orders. As a result, much of the nesting habitat within 5.3 miles of the occupied leks inside the Buffalo Core Area are found outside the Core Area on lands that are slated to receive minimal protections. In Attachment G, a screenshot of a presentation by WGFD, the Buffalo Core Area is delineated at the left side of the screen, with a rectilinear boundary following jagged land ownership. The lek sites, shown with magenta dots with numbers for 2014 maximum male counts, are located inside the core area (infilled with pale green), while a significant amount of the nesting habitat for the most populous leks inside the Core Area (shown as underlying red circles) extend outside the Core Area into unprotected lands. In addition, most of the occupied lek sites themselves lie along the boundary of the Core Area or within 1.9 miles (the distance at which producing well sites have a significant negative impact on lek populations, Holloran 2005), and as a result industrial development has and will continue to drive these leks near the Core Area boundary (FEIS at Map 37) toward extirpation.

BLM also failed to designate a new Core Area along the Powder River (*see* [Buffalo FEIS Attachment 34]) to address the inadequate spatial extent of Core Areas in the Buffalo Field Office. In Attachment G, the proposed new Core Area is outlined in purple at the center of the image, and many of the lek sites in this potential Core Area (shown as magenta dots) have relatively high lek counts (the numbers accompanying the lek symbols), and in addition the underlying red circular buffers indicate the location of nesting habitat that represents part of the densest 25% of the state sage grouse population. This recommendation was submitted to the State of Wyoming (but not recommended for adoption by the state) in 2015. This designation would address the need to designate key sage grouse habitats encompassing some of the most densely populated sage grouse habitats in the Powder River sage grouse population area, which were excluded from Core Area designations in 2008 contrary to the best available science in an act of state obeisance to the coalbed methane industry. Much of these lands was subsequently developed for coalbed methane, but this industry has gone dormant, and the BLM should be managing its lands and minerals estate to provide a level of surface disturbance compatible with maintaining this sage grouse population. In effect, BLM needs to manage this area back to a level of development (3% surface disturbance, maximum one wellpad per square mile, no wellsites or roads within 4 miles of leks) that will optimize sage grouse survival and recovery.

f. The Forest Service Must Comply with the National Environmental Policy Act and Administrative Procedure Act

The National Environmental Policy Act's implementing regulations require that agencies "shall provide full and fair discussion of significant environmental impacts and shall inform decisionmakers and the public of the reasonable alternatives which would avoid or minimize adverse impacts or enhance the quality of the human environment." 40 C.F.R. § 1502.1. The Administrative Procedure Act ("APA"), 5 U.S.C. § 706(2)(A) further mandates that agency decisions, such as Forest Service land use plan revisions or amendments under NFMA, must be consistent with the evidence before the agency and based on reasoned explanation. *See Motor Vehicle Mfrs. Ass'n v. State Farm Mut. Auto Ins. Co.*, 463 U.S. 29, 43 (1983). An agency's decision is arbitrary when it "entirely failed to consider an important aspect of the problem." *Id*. at 43. Although agencies are entitled to change policy positions, such changes are arbitrary and violate the APA if not justified by a reasoned explanation based on the record. *See Organized Vill. of Kake v. U.S. Dep't of Ag.*, 795 F.3d 956, 966-67 (9th Cir. 2015) (en banc) (citing *FCC v. Fox Television Stations, Inc.*, 556 U.S. 502, 515-16 (2009)).

Eliminating or modifying conservation measures which the science requires across part or all of the range of the greater sage grouse would violate both NEPA's requirement for reasoned consideration of significant environmental impacts, and the APA's requirement for reasoned decision making. Applying differing conservation measures within ecologically equivalent "management zones" also fails this test. A geographically inconsistent approach to implementing the science of sage-grouse conservation is arbitrary and capricious under the National Environmental Policy Act, and Administrative Procedure Act.

The science shows that responses of sage-grouse to human-induced habitat alterations are remarkably similar across the species' range. Given that the scientific consensus does not differ significantly across the species' range regarding the impacts of human activities on sage-grouse, does not find different thresholds at which human impacts become significant, and recommends similar (or indeed, identical) conservation measures, different approaches to sage grouse conservation in different geographies indicates a failure to address the conservation needs of the species.

Because science-based thresholds of impact and population and habitat response do not differ from state to state for most types of impacts, the appropriate, science-based sage grouse protections also do not differ.

To the extent that the current plans violate NEPA, FLPMA, NFMA and the APA, those violations will persist in subsequent planning efforts unless the deficiencies we identify here are corrected.

C. The Forest Service Has Not Implemented The Existing Plans

Moreover, USFS has not implemented the protections the Plans promised.

a. Grazing

The 2015 RODs say this about implementing grazing changes in accordance with the FS plans:

> Under NFMA, the Forest Service may conduct implementation "as soon as practicable" after the effective date of the ROD. Our expectation is to implement amended grazing guidance with a phased-in approach within 18-24 months after signing the ROD for the majority of our allotments. However, in some circumstances up to 36 months may be required for permit modification and full implementation. Therefore there will be no immediate change in grazing management or modification of term grazing permits upon signing this ROD and implementation will occur in a phased approach.
>
> The first phase of implementation of the grazing guidance contained in the LMP amendments will be habitat mapping that identifies GRSG habitat and an evaluation of allotments (i.e. specific pastures and riparian/mesic areas). The Habitat Assessment Framework protocol(http://sagemap.wr.usgs.gov/docs/rs/SG%20HABITAT%20ASESSMENT%202010.pdf) will be used to identify habitat condition at the allotment scale. Field visits with permittees may also be conducted to understand the new guidance and expectations,

> evaluate impacts, and explore collaborative solutions to effectively implement this guidance.
>
> In the second phase of implementation, term grazing permits of affected allotments will be modified with new grazing guidance by the 2017 grazing season for most units and no later than 2018 grazing season for all units. In most cases, no additional site-specific NEPA analysis or decision is anticipated. If after a period of time (i.e. 1 to 3 years after modifying permits) of implementation and monitoring, it is determined that existing allotment management plan prevent attainment of standards, guidelines, or desired conditions, then new NEPA may be required to adjust the allotment management plans.

GB ROD at 71. However, on November 27, 2017, the Forest Service issued a public notice and decision[4] that implementation of the grazing objectives would not be forthcoming and that the agency would instead implement grazing changes "as soon as practicable" and stated, "Furthermore, we may resolve these issues so that modification of the permits would not be necessary or may be different than the current plan direction." Hence, the Forest Service has withdrawn assurances that grazing management will be improved to protect sage-grouse under the FS land use plan amendments and the threat of unchecked grazing on sage-grouse (contrary to the scientific evidence and court orders) are continuing on FS lands. This is arbitrary and capricious, violates the current land-use plans, and continues to harm the sage-grouse across the large percentage of its range that is grazed.

b. Oil and gas

As noted above, the Forest Service plan revisions relied on five key components for addressing threats from oil and gas development: (1) outside of Wyoming, any new oil and gas leases within USFS PHMA must require no surface occupancy without waiver, exception, or modification (Rocky Mountain ROD 31-32, Great Basin ROD 33-35) (2) on existing leases, "the Forest Service will limit energy development in PHMAs," with exceptions only where best available science indicates that impacts will be avoided (not merely mitigated) (Rocky Mountain ROD 31, Great Basin ROD 32-33); (3) controlled surface use and timing limitation requirements for GHMA (Rocky Mountain ROD 31, Great Basin ROD 33); (4) limits on Forest Service "discretionary authorizations" to limit density of disturbance to sage-grouse habitat (3% in northwest Colorado, 5% in Wyoming, and 3% in Idaho, Nevada, and Utah) (Rocky Mountain ROD 31, Great Basin ROD 33); (5) a requirement that the Forest Service "encourage" new development in non-habitat areas, and "new development outside of PHMA, where possible." Elimination or weakening of these plan elements, particularly without concrete evidence of how

[4] https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd565108.pdf

they have been applied in practice, will only further exacerbate the inadequacy of the existing plans to conserve sufficient greater sage-grouse habitat to permit recovery.

Expanding the circumstances under which lease stipulations on PHMA and GHMA are subject to exceptions, modifications, and waivers, will further undermine any certainty that USFS activities will maintain functionality of sage-grouse habitat. The USFS revised plans mandate no waiver of no surface occupancy requirements new leases in PHMA, and procedures for limiting waiver on existing leases to circumstances where "impacts will be avoided." Expanding the use of waivers, exceptions, and modifications, which are granted frequently and with little documentation, will further undermine the effectiveness and scientific integrity of the plans. For example,a recent GAO study of BLM field offices found that of the 54 recorded exception decisions, from four offices that could provide data, 49 exception requests were approved and 5 were denied—that is, exception requests were granted 90% of the time. *See* U.S. Gov't Accountability Off., GAO-17-307, Oil and Gas Development: Improved Collection and Use of Data Could Enhance BLM's Ability to Assess and Mitigate Environmental Impacts 16 n. 24 (Apr. 2017). That same study found that BLM's decisions to grant such exceptions, modifications, and waivers often takes place in the dark, without written justification, oversight, documentation of the request or field office's decision, or additional NEPA analysis. *Id*. at 11–21. The report concluded, "BLM may be unable to provide reasonable assurance that it is meeting its environmental responsibilities." *Id*. at Intro. BLM's willingness to grant modifications, waivers, and exceptions—and without transparency or public participation—creates large loopholes that render the lease stipulations ineffective and afford the sage grouse insufficient protection.

As a result of BLM's erroneous interpretation of "prioritization" in oil and gas leasing, both the BLM and USFS the plans have not achieved their goal of guiding development away from identified sage-grouse habitat. In fact, roughly 43% of all parcels offered for oil and gas leasing since 2015 have contained sage-grouse habitat. Mineral leasing on PHMA appears to have actually increased since the issuance of the 2015 Sage-Grouse Plans. A 2017 study of the overlap between sage-grouse habitat and energy development across the West found that only 4% of existing mineral leases contain PHMA. *See* Chad LeBeau and Grant Gardner, Analysis of the Overlap between Priority Habitat Management Areas and Existing and Potential Energy Development across the Western United States at i (June 9, 2017). In contrast, roughly 10% of all newly-offered lease parcels contain PHMA.

Given the extent of sage grouse habitat already encumbered by existing mineral leases and the scientific consensus on regarding the need to protect those habitats, the current Forest Plans' treatment of existing mineral leases is grossly inadequate to protect the species' habitat needs, and falls far short of the agencies' available authority to impose conditions of approval on mineral development.

Although the existing Plans are insufficient, for all of the reasons summarized above, and set forth in more detail in the appended Complaint filed by environmental groups challenging the Plans, they are better than no protections at all. These protections must stay in place pending any plan revisions. And, rather than further weakening the protections sage-grouse require, as the Zinke Report recommended, any new process should strengthen sage-grouse protections to comply with what the best available science explains the birds need.

IV. THE NEW PLANS MUST COMMIT TO STRONG, DURABLE, SCIENCE-BASED PROTECTIONS FOR GREATER SAGE-GROUSE.

A. The Plans must follow the best available science.

a. Vegetation Treatments

In order to conform to the best available science, vegetation projects that reduce or eliminate sagebrush must be prohibited. There is no scientific support for vegetation treatments as a means of improving grouse habitats, and to the contrary, numerous studies highlight negative impacts to sage grouse of this practice.

Baker (2007) and Bukowski and Baker (2013) have shown that natural fire return intervals (without livestock) are far less frequent than current fire return intervals in sagebrush systems (with livestock grazing everywhere), particularly in lowland systems dominated by Wyoming big sagebrush. Hess and Beck (2012) found that neither burned nor mowed areas produced suitable sage grouse habitats. Wamboldt et al. (2002:24) stated:

> Natural or prescribed burning of sagebrush is seldom good for sage grouse. This assessment recommends that fires within sage grouse habitat be avoided in most cases, and should be allowed only after careful study of each local situation. The evidence also indicates that habitat loss due to fire may well be the most serious of all the factors contributing to the decline of sage grouse.

Heath et al. (1997:50) went even farther: “Based on our results, we recommend no reduction or control of sagebrush in areas containing between 18-30% live sagebrush canopy coverage within 4.5 km of leks.” Connelly et al. (2000) recommended against habitat manipulation in sagebrush stands of 10-30% canopy cover heights of at least 25 cm to protect winter habitats. Beck and Mitchell (1997) recommended against sagebrush control projects when canopy cover is less than 20 percent, and recommend against any sagebrush control within 2 miles of leks. The Conservation Objectives Team report (COT 2013: 44) recommended the following: “Avoid sagebrush removal or manipulation in sage grouse breeding or wintering habitats.” In the BLM's

2006 paper titled Review of Livestock Grazing Management of Sage Grouse Habitat the BLM determined from its review of the literature that “No treatment should be considered where sagebrush cover is less than 20 percent or within 2 miles of breeding, nesting, or brood areas.”

Even in areas with less than 3.5% habitat disturbance through vegetation treatments, these vegetation treatments have been found to have a significant negative effect on sage grouse populations (Holloran and Belinda 2009). According to Beck et al. (2012:444), “The preponderance of literature indicates that habitat management programs that emphasize treating Wyoming big sagebrush are not supported with respect to positive responses by sage-grouse habitats or populations.”

Arkle et al. (2014) made a comprehensive study of the effectiveness of restoration activities in burned sagebrush. They found that restoration actions did not increase the probability of burned areas meeting most guideline criteria. Of 313 plots seeded after fire, none met all sagebrush guidelines for breeding habitats. Less than 2% of treated plots met winter habitat guidelines. They concluded that sage-grouse are relatively unlikely to use many burned areas within 20 years of fire, regardless of treatment, and that reestablishing sagebrush cover will require more than 20 years using past restoration methods (Arkle et al. 2014). Their findings reiterate the importance of reducing threats to sage-grouse in their remaining occupied habitats and underline the need to avoid any use of prescribed fire in sage-grouse habitat.

Hess and Beck (2014) also looked at the effectiveness of sage-grouse habitat restoration actions. They found that neither mowing nor prescribed burning promoted statistically significant increases in sage-grouse nesting or early brood-rearing habitat attributes such as cover or nutritional quality of food forbs, or counts of ants, beetles, or grasshoppers compared with reference sites.

Sagebrush is the most critical habitat component for maintaining and recovering sage grouse populations, making up the vast majority of the species’ diet year-round and providing necessary hiding cover and key nesting habitat. The Forest Service must take the legally required ‘hard look’ at impacts that cause surface disturbance, including industrial activities and excessive livestock grazing, as well as disturbances such as fire and sagebrush manipulation projects, by estimating the time it will take for sagebrush to recover to the point where it becomes functioning habitat as food and cover for sage grouse.

Indeed, recovery times following disturbance can be quite long. Past NEPA analysis concedes, “In the absence of cheatgrass, Wyoming big sagebrush sites can take 150 years to recover.” Nevada – Northeast California Greater Sage Grouse RMP Amendment DEIS at 608. When cheatgrass is present, it can take over following disturbance, forming a monoculture characterized by unnaturally frequent fire return intervals that can effectively prevent the recovery of sagebrush and perennial grasses on a long-term if not permanent basis. For Oregon, the FS's own NEPA analysis states, "In Wyoming big sagebrush sites, full recovery to pre-burn sagebrush canopy cover conditions will take over 100 years (Cooper 2007);…." Oregon Greater Sage Grouse RMP Amendment DEIS at 3-70. More generally, the FS states, "Sagebrush recovers slowly from fire; most species do not resprout but must be replenished by wind-dispersed seed from adjacent unburned stands or seeds in the soil. Depending on the species and the size of a burn, sagebrush can reestablish itself within five years, but a return to a full pre-burn community cover can take 50 to over 100 years (Baker 2011)." Oregon Greater Sage Grouse RMP Amendment DEIS at 4-10. All planning must adjust to these parameters.

Federal agencies should prohibit vegetation treatments in Priority Habitats except where they are consistent with maintaining optimal sage grouse habitat (per NTT 2011). In the Wyoming plan amendment, vague language about "ESD or other methods" leave the door open to vegetation management projects that degrade or fragment important sage grouse habitats; only in northeast Wyoming is there a standard that prevents vegetation projects that reduce sagebrush canopy cover below 15%, the bare minimum for quality sage grouse habitat. In these sensitive habitats, vegetation treatments that reduce or eliminate sagebrush should be prohibited entirely, statewide, based on the best available science.

b. Livestock grazing

Extensive scientific literature has confirmed that livestock grazing adversely affects sagebrush ecosystems. Daubenmire (1970) described the lower resilience of sagebrush plant communities to grazing. In addition, Mack and Thompson (1982) discuss the myriad harmful effects of livestock grazing to intermountain and Great Basin sagebrush communities that evolved without large herds of hooved mammals. Fleischner (1994) and Belsky and Gelbard (2000) review the many harmful impacts of livestock grazing to arid western lands, including alteration of plant community composition and structure. Finally, Anderson and Holte (1981) describe significant increases in perennial grass and shrub cover after grazing was removed from sagebrush lands in southeastern Idaho—perennial grass cover increased exponentially and shrub cover was 154 percent greater.

Any plan revisions should focus on improving sage-grouse protections from livestock grazing. Plans must restrict grazing practices known to harm sage-grouse nesting and brood rearing success, particularly during the "well established" timeframes necessary to avoid adversely impacting sage-grouse— June 20 to August 1, and Nov. 15 to March 1 (in other words, avoiding grazing during the spring and fall). *Salazar*, 843 F.Supp.2d at 1115, 1123. The current Forest Service objectives pertaining to vegetation height are insufficient to protect sage-grouse from other threats posed by livestock in their habitat.

In addition to limiting grazing by season, the best science recommends limiting livestock use of herbaceous forage to about 30 percent of annual production. (Braun 2006). The courts have ruled that monitoring is necessary to ensure that grazing utilization stays below this level to improve vegetation and meet the habitat needs of sage-grouse. *W. Watersheds Project v. Dyer*, Nos. CV-04-181-S-BLW, CV-02-521-S-BLW, 2009 WL 484438, *21 (D. Idaho Feb. 26, 2009) (unreported opinion).

Livestock grazing is considered the single most important influence on sagebrush habitats and fire regimes throughout the Intermountain West in the past 140 years (Knick et al 2005). Grazing is the most widespread use of sagebrush steppe and almost all sagebrush habitats are managed for grazing (Connelly et al. 2004, Knick et al. 2003, Knick et al. 2011.). Livestock grazing disturbs the soil, removes native vegetation, and spreads invasive species in sagebrush steppe (Knick et al. 2005). Cattle or sheep grazing in sage-grouse nesting and brood-rearing habitat can negatively affect habitat quality; nutrition for gravid hens; clutch size; nesting success; and/or chick survival (Connelly and Braun 1997, Beck and Mitchell 2000, Barnett and Crawford 1994, Coggins 1998, Aldridge and Brigham 2003). Livestock may directly compete with sage-grouse for grasses, forbs and shrub species; trample vegetation and sage-grouse nests; disturb individual birds and cause nest abandonment (Vallentine 1990, Pederson et al. 2003, Call and Maser 1985, Holloran and Anderson 2005, Coates 2007).

Jankowski et al. (2014) studied stress hormones in greater sage-grouse with a focus “on the effects of cattle grazing because of the potential negative effects on sage-grouse habitats and because cattle grazing practices can be influenced by management decisions.” Jankowski et al., (2014). They found that residence in a cattle-grazed habitat was associated with increased stress hormone levels in a large sample of greater sage-grouse (329 sage-grouse, 160 from grazed sites and 169 from ungrazed sites). They found higher immunoreactive corticosterone metabolites in greater sage-grouse in cattle-grazed versus ungrazed sites and found a positive correlation of immunoreactive corticosterone metabolites in greater sage-grouse with cattle fecal pat count. The maximum rise in immunoreactive corticosterone metabolites associated with the high end fecal pat count approached levels associated with the acute stress from capture. Lower and average fecal pat counts were associated with immunoreactive corticosterone metabolites levels that were comparable or higher than found in male sage-grouse in noise-treated leks. The findings of Jankowski et al. (2014) are thus of considerable concern.

Jankowski et al. (2014) postulate that the increases in the stress hormone may be a physiological response to the direct visual presence of cattle on the landscape, infrastructure associated with cattle grazing, or the use of degraded habitats (e.g., reductions in perennial grasses or trampled riparian areas).

Blickley et al. (2012) found that chronic noise exposure also increased corticosteroid metabolites in lekking male sage-grouse. They note that for sage-grouse, which are highly susceptible to West Nile virus, reduced immune response due to elevated glucocorticoid levels could have a significant effect on survival in areas where sage-grouse are exposed to West Nile virus (Blickley et al. 2012). Because WNV is spread in livestock waters, this is a real threat throughout sage-grouse habitat.

The FWS Finding also explained why the physical presence of livestock poses a risk and threat to sage-grouse during nesting season:

> Other consequences of grazing include several related to livestock trampling of grouse and habitat. Although the effect of trampling at a population level is unknown, outright nest destruction has been documented and the presence of livestock can cause sage-grouse to abandon their nests (Rasmussen and Griner 1938, p. 863; Patterson 1952, p. 111; Call and Maser 1985, p. 17; Holloran and Anderson 2003, p. 309; Coates 2007, p.28). Coates (2007, p. 28) documented nest abandonment following partial nest depredation by a cow. In general all recorded encounters between livestock and grouse nests resulted in hens flushing from nests, which could expose the eggs to predation; there is strong evidence that visual predators like ravens use hen movements to locate sage-grouse nests (Coates 2007, p.33).

75 Fed. Reg. at 13940-41.

Sage-grouse depend almost entirely on sagebrush for food and protection from predators. In the summer, the birds depend on the grasses and plants that grow under the sagebrush to provide nesting material, as well as high protein insects that are critical to the diet of chicks in the first few months of life. In winter, almost 99 percent of their diet is sagebrush leaves and buds. Recent estimates indicate that the sage-grouse populations have declined by approximately 86 percent from historic levels. One of the greatest threats to sage-grouse populations is the destruction and loss of habitat from a variety of management activities including livestock grazing (FWS 2004).

The potential conflict between livestock grazing and sage-grouse intensifies near water sources due to the importance of these areas to sage-grouse, particularly during early brood rearing. Heavy cattle grazing near springs, seeps, and riparian areas can remove grasses used for cover by grouse (Klebenow 1982). "[R]apid removal of forbs by livestock on spring or summer ranges may have a substantial adverse impact on young grouse, especially where forbs are already scarce" (Call and Maser 1985). A recent study on the Hart Mountain National Antelope Refuge in southeastern Oregon demonstrated that the removal of cattle can result in dramatic changes in riparian vegetation, even in semi-arid landscapes (Batchelor et al. 2015).

In presettlement times, the range of the sage-grouse paralleled the range of big sagebrush. Basin big sagebrush provides important cover for sage-grouse (Benson et al. 1991). Populations of sage-grouse have declined primarily because of loss of habitat due to overgrazing, elimination of sagebrush, and land development (Hamerstrom and Hamerstrom 1961). Sage-grouse populations began declining from 1900 to 1915, when livestock utilization of sagebrush rangeland was heavy (Patterson 1952). In the 50's and 60's, land agencies adopted a policy of aggressive sagebrush control in order to convert sagebrush types to grassland. Chaining, frequent fire, and herbicide treatments reduced sagebrush by several million acres and sage-grouse numbers plummeted drastically (Call 1979).

A BLM report (Taylor et al 2010) reveals:

> The effects of grazing management on sage-grouse have been little studied, but correlations between grass height and nest success suggest that grazing may be one of the few tools available to managers to enhance sage-grouse populations….For instance, a 2 inch increase in grass height could result in a 10% increase in nest success, which translates to an 8% increase in population growth rate.

Sage-grouse historically occurred throughout the range of big sagebrush (*A. tridentata*), except on the periphery of big sagebrush distribution or in areas where it has been eliminated (Call and Maser 1985). Sage-grouse prefer mountain big sagebrush (*A. t. ssp. vaseyana*) and Wyoming big sagebrush *(A. t. ssp. wyomingensis)* communities to basin big sagebrush (*A. t. spp. tridentata*) communities. Sage- grouse are totally dependent on sagebrush-dominated habitats (Benson et al. 1991). Sagebrush is a crucial component of their diet year-round, and sage-grouse select sagebrush almost exclusively for cover (Patterson 1952).

When not on the lek, sage-grouse disperse to the surrounding areas (Wallestad 1974). Some females probably travel between leks. Patterson (1952) reported that in Wyoming, 92 percent of sage-grouse nests in Wyoming big sagebrush were in areas where vegetation was 10 to 20 inches (25-51 cm) tall and cover did not exceed 50 percent.

The importance of sagebrush in the diet of adult sage-grouse is impossible to overestimate. Numerous studies have documented its year-round use by sage-grouse (WAWFA 2009, Call 1979, Call and Maser 1985, Patterson 1952, Schneegas 1967, Wallestad 1975). A Montana study, based on 299 crop samples, showed that 62 percent of total food volume of the year was sagebrush. Between December and February it was the only food item found in all crops. Only between June and September did sagebrush constitute less than 60 percent of the sage-grouse diet (Wallestad 1975).

In places, the production of young sage-grouse simply is not enough to sustain a stable population. Sage-grouse have one of the lowest recruitment rates of any upland game bird in North America. Loss of habitat, predation, drought, and poor weather conditions during hatching and brooding periods have all been cited as factors leading to poor recruitment (Mattise 1995). Lack of adequate nesting and brooding cover may account for high juvenile losses in many regions (Kindschy 1986). A decline in preferred prey may also result in increased predation on sage-grouse. Nest losses to predators vary throughout the range of sage-grouse, but predators are more successful in areas of poor-quality nesting habitat.

Manier et al. (2013) provides a fairly comprehensive review of potential impacts of livestock grazing on sage grouse. Manier et al. (2013) point out that a reduction in livestock stocking rates can directly increase residual vegetation substantially, potentially assisting in meeting this target level for grasses.

The paper, "A Blueprint for Sage-grouse Conservation and Recovery (Braun 2006) states "if livestock grazing is permitted on public rangelands, is to not exceed 25-30% utilization of herbaceous forage each year. Grazing should not be allowed until after 20 June and all livestock should be removed by 1 August with a goal of leaving at least 70% of the herbaceous production each year to form residual cover to benefit sage-grouse nesting the following spring."

Sage-grouse experts recommended a minimum 7-inch residual stubble height standard, a level at which vegetation would afford the best chance of nest success (Connelly et al. 2000, Doherty et al. 2011). The same paper recommended disallowing livestock grazing in sagebrush steppe habitats that produce less than 200 lbs/ac of herbaceous vegetation per year "if successful sage-grouse nesting and brood rearing is an objective." According to Gregg et al. (1994: 165), "Land management practices that decrease tall grass and medium height shrub cover at potential nest sites may be detrimental to sage grouse populations because of increased nest predation.... Grazing of tall grasses to <18 cm would decrease their value for nest concealment.... Management activities should allow for maintenance of tall, residual grasses or, where necessary, restoration of grass cover within these stands." Connelly et al. (2000) reviewed the science of that time and recommended an 18-cm residual stubble height standard. Hagen et al. (2007) analyzed all scientific datasets up to that time and concluded that the 7-inch threshold was the threshold below which significant impacts to sage grouse occurred (*see also* Herman-Brunson et al. 2009). Prather (2010) found for Gunnison sage grouse that occupied habitats averaged more than 7 inches of grass stubble height in Utah, while unoccupied habitats averaged less than the 7-inch threshold. According to Taylor et al. (2010: 4),

> "The effects of grazing management on sage-grouse have been little studied, but correlation between grass height and nest success suggest that grazing may be one of the few tools available to managers to enhance sage-grouse populations. Our analyses predict

> that already healthy populations may benefit from moderate changes in grazing practices. For instance, a 2 in increase in grass height could result in a 10% increase in nest success, which translates to an 8% increase in population growth rate."

Heath et al (1997) found that near Farson, Wyoming, nests with taller grass heights were more successful than those with shorter heights. The exception to this 7-inch rule is found in the mixed-grass prairies of the Dakotas, where sparser cover from sagebrush and greater potential for tall grass have led to a recognition that a 26-cm stubble height standard is warranted (Kaczor 2008, Kaczor et al. 2011). Foster et al. (2014) found that livestock grazing could be compatible with maintaining sage grouse populations, but notably stubble heights they observed averaged more than 18 cm during all three years of their study, and averaged more than 10.2 inches in two of the three years of the study. This finding is consistent with our conclusion based on the science that maintaining at least 7 inches of residual stubble is necessary to maintain or recover sage grouse populations. Doherty et al. (2014) found a similar relationship between grass height and nest success in northeast Wyoming and south-central Montana but did prescribe a recommended grass height. Stiver et al. (2015) recommended 18 cm grass height for all breeding and nesting habitats, and explicitly stated that this and other established measures should not be altered unless scientific evidence definitively indicates that the 7-inch threshold is inappropriate. This scientific evidence has never been produced in Wyoming, and therefore the 7-inch threshold should prevail. Thus, all available science to date is consistent with standards that maintain at least 7 inches of stubble height rangewide, and more than 10.2 inches in the Dakotas.

The FWS Finding also articulated the need to ensure sufficient grass cover.

> Sage-grouse need significant grass and shrub cover for protection from predators, particularly during nesting season, and females will preferentially choose nesting sites based on these qualities (Hagen et al. 2007, p. 46). The reduction of grass heights due to livestock grazing in sage-grouse nesting and brood-rearing areas has been shown to negatively affect nesting success when cover is reduced below the 18 cm (7 in.) needed for predator avoidance (Gregg et al. 1994, p. 165).

75 Fed. Reg. at 13939.

Moreover, while Doherty and others (2011) found no support for using Ecological Site Descriptions (ESDs) to predict habitat use by sage-grouse or to base sage-grouse management decisions on, they found average grass height within 15 m of the nest was strongly associated with daily survival rates of nests. And in fact, grass height alone explains much of the observed variation in greater sage-grouse nest survival (Doherty et al., 2014). The concept of "ecological site potential" is therefore less useful for conserving sage-grouse than firm and unwavering criteria that specify minimum grass heights of greater than or equal to 18 cm.

The FWS also explained other aspects of direct competition between livestock and grouse:

> Livestock also may compete directly with sage-grouse for rangeland resources. Cattle are grazers, feeding mostly on grasses, but they will make seasonal use of forbs and shrub species like sagebrush (Vallentine 1990, p. 226) … in general, forb consumption may reduce food availability for sage-grouse. This impact is particularly important for pre-laying hens, as forbs provide essential calcium, phosphorus, and protein (Barnett and Crawford 1994, p. 117). A hen's nutritional condition affects nest initiation rate, clutch size, and subsequent reproductive success (Barnett and Crawford 1994, p.117; Coggins 1998, p. 30).

Id. at 13940.

Additionally, livestock grazing is a well-known vector of invasive, non-native, or noxious species colonization on public lands. Livestock promote the spread and colonization of alien plants, which can increase fire frequencies (Belsky and Gelbard 2000, Billings 1994). Disturbance is a reliable indicator of alien dominance in vegetation composition, and livestock grazing is a significant disturbance to desert ecosystems (Brooks and Berry 2006).

Grazing across many states has led to the invasion of cheatgrass, a highly flammable noxious weed that accelerates the fire cycle to less than five years destroying the sagebrush upon which sage-grouse rely for food and cover. One recent estimate found that approximately 36 percent of the greater sage-grouse range is invaded by cheatgrass (Lebbin et al 2010); that percentage has surely increased with recent fires. Because sagebrush requires at least 15 years (and up to 50) to reoccupy burned sites, restoring invaded areas is a difficult and slow process. Preventing further spread into intact sagebrush should be prioritized, something the Plans fail to consider or manage for.

Biological invasions, especially invasion by exotic annual grasses such as cheatgrass, are consistently cited as among the most important challenges to maintenance of healthy sagebrush communities (Miller et al. 2011, Wisdom et al. 2005). Estimates of the rapid spread of weeds in the West include 2,300 acres per day on BLM lands and 4,600 acres per day on all western public lands (See 65 FR 54544). Clearly, the BLM needs to consider the cause of these infestations and the contribution of domestic livestock grazing to them.

A study published in the Journal of Applied Ecology concludes that livestock grazing contributes to the domination of some western landscapes by cheatgrass, an invasive grass that both destroys sage-grouse habitat and increases the frequency of wildfire (Reisner et al. 2013).

To mitigate the spread of cheatgrass, the study suggests maintaining and restoring bunchgrasses and soil crusts, two ecological features that are quickly degraded under the hooves of livestock. Such mitigation would require the decrease or elimination of livestock grazing in the affected areas.

Anderson and Inouye (2001) found that viable remnant populations of native grasses and forbs are able to take advantage of improved growing conditions when livestock are removed. They found further that despite depauperate and homogeneous conditions of permanent plots in 1950, after 45 years of no livestock grazing, vegetation had been anything but static, clearly refuting claims of long-term stability under shrub dominance. Mean richness per plot of ALL growth forms increased steadily in the absence of domestic livestock grazing. Grasses and forbs increased significantly.

The primary long-term threat is the widespread conversion of mid-stature cool season bunchgrasses that did not evolve with significant herbivory, to short stature, grazing-tolerant species. This livestock-induced conversion has occurred throughout much of the planning area already and is a primary and continuing source of imperilment for sage-grouse.

Additionally, livestock grazing is known to have significant effects on soil and watershed conditions, including directly causing increased soil erosion. The phenomenon has three basic components. Grazing reduces plant cover that binds the soil and, in low desert areas, destroys microbiological soil crusts that stabilize soil surfaces (Beymer and Klopatek 1992, Brotherson and Rushforth 1983). Vegetation that impedes overland flow of rainfall runoff in intact watersheds is lost to grazing (Sharp et al. 1964). Grazing livestock compact the soil, so instead of rainfall soaking down toward the aquifer it flows faster and in greater volume overland (Arnold 1950, Belsky et al.1999, Johnson 1956, Jones 2000). Erosion is far greater on grazing than ungrazed lands (Lusby 1979). Other impacts such as plant community degradation (Yeo 2005) are also well documented.

Eroding soil and manure throughout watersheds end up in streams as increased sediment load, excessive nutrients, and pathogen contamination. Various grazing management strategies have not been found to reduce such watershed degradation (Gifford and Hawkins 1976, Blackburn et al. 1982). A list of impaired waters and the sources of contamination within the watersheds of these public lands would be an appropriate place to begin taking a “hard look” at potential grazing effects from the public lands, but the Plans contain no such analysis.

Livestock related infrastructure also adversely affects sage-grouse. The BLM and the Forest Service have constructed hundreds of thousands of miles of fencing throughout greater sage-grouse habitat to facilitate livestock grazing. Impacts from fences include loss of birds through collisions, fragmentation of habitat, habitat degradation, spread of invasive plants, facilitation of juniper expansion, and increased perching opportunities for predators such as ravens. Mortality associated with fence collisions can be dramatic in sage-grouse habitat. For example, Stevens (2011) found that corrected landscape-scale sage-grouse collision rates ranged from 0.12-0.70 strikes/km in 2009 and 0.18-0.75 strikes/km in 2010 (Stevens 2011:63). Avian fence collision surveys in sagebrush steppe habitats should be conducted with less than 2-week sampling intervals to reduce the impact of survival bias on collision rate estimates and caution should be used when aggregating or comparing uncorrected collision data from areas with differing vegetation, as detection probabilities are likely different between sites (Stevens et al., 2011 p. 447). Marking fences may help reduce collision rates, but collisions still occurred at marked fences <500 m from large leks and moving or removing fences may be necessary (Stevens et al., 2012 p. 297).

Livestock fences also facilitate pinõn-juniper expansion into sage-grouse habitat by providing perch sites for songbirds within sagebrush; rows of juniper seedlings can often be seen along fences where birds perch (Evans, 1988). However, unless the fences are also removed, removal of pinõn-juniper from along those fences may facilitate raven predation on sage-grouse by opening line of sight from fence posts. Howe et al. (2014) found that ravens strongly avoided juniper and showed some selection for non-native vegetation for their nest sites. Sage-grouse select nest sites and brood sites away from avian predators (Dinkins et al. 2013); so, by opening up fences and facilitating raven perching and predation, pinõn-juniper treatments may result in less nesting habitat being available for sage-grouse.

The FWS articulated the threats of infrastructure in the 2010 Finding thusly:

> *Fences*:
> Another indirect negative impact to sage-grouse from livestock grazing occurs due to the placement of thousands of miles of fences for livestock management purposes. Fences cause direct mortality through collision and indirect mortality through the creation of predator perch sites, the potential creation of predator corridors along fences (particularly if a road is maintained next to the fence), incursion of exotic species along the fencing corridor, and habitat fragmentation (Call and Maser 1985, p. 22; Braun 1998, p. 145; Connelly et al. 2000a, p. 974; Beck et al. 2003, p. 211; Knick et al. 2003, p. 612; Connelly et al. 2004, p. 1-2).
>
> 75 Fed. Reg. at 13941.
>
> *Water developments:*
> Water developments for the benefit of livestock and wild ungulates on public lands are common (Connelly et al. 2004, p. 7-35). Development of springs and other water sources to support livestock in upland shrub-steppe habitats can artificially concentrate domestic

and wild ungulates in important sage-grouse habitats, thereby exacerbating grazing impacts in those areas such as heavy grazing and vegetation trampling (Braun 1998, p. 147; Knick et al., in press, p. 42).

Diverting the water sources has the secondary effect of changing the habitat present at the water source before diversion. This impact could result in the loss of either riparian or wet meadow habitat important to sage-grouse as sources of forbs or insects. Water developments for livestock and wild ungulates also could be used as mosquito breeding habitat, and thus have the potential to facilitate the spread of West Nile virus (see discussion under Factor C: Disease and Predation).
Id.

Connelly (2013) raised concerns over the activities of major federal agencies directed at sage-grouse conservation and concludes, “taken as a whole, these efforts appear to be getting sage-grouse conservation nowhere fast, largely because of bureaucratic approaches and continued reliance on rhetoric and dogma.” In his expert view: “Where allotments are not meeting rangeland health standards and livestock grazing is shown to be a major contributing factor, federal agencies could alter grazing systems to improve habitat over a relatively short period of time” (Connelly, 2013: 63).

Bell (2011) studied the nest site characteristics of native and translocated sage-grouse hens at a site in northern California. He recommends that sage grouse habitat management focus on maintaining or enhancing quality nesting habitat by increasing herbaceous cover, promoting moderate levels of grass height, and promoting larger sagebrush diameters (Bell, 2011: iv and 32).

Dinkins et al. (2013) studied predation on greater sage-grouse eggs using cameras deployed at 24 artificial sage-grouse nests. They observed predation at eight of these nests; 4 by badgers, 2 by magpies, and 1 by a domestic cow, with 1 unknown due to camera failure (Dinkins et al. 2013). The Dinkins et al. study confirms that predation by cows on sage-grouse eggs as previously reported in the literature (Coates et al. 2008; USFWS, 2013) is not uncommon. The fact that predation by cattle was observed in two different studies (noting that one was a hen’s egg) designed to probe predation on sage-grouse eggs indicates that this is a recurrent issue and one which must be addressed through improved management.

A number of management efforts have proposed to reduce predation on sage-grouse nests by corvids such as the common raven. Recent studies have documented little or no predation by ravens: both Bell (2011) and Dinkins et al. (2013) found none of their observed predation events to be due to corvids. The presence of livestock also subsidizes ravens, increasing their density and attracting them to grazed areas. Coates et al. (2016) found that odds of raven occurrence increased 45.8% in areas where livestock were present, and that ravens selected for habitats closer to leks. These researchers recommended spatially segregating livestock from sage-grouse breeding areas. Although raven predation is of concern in limited cases where increased raven populations have been facilitated by anthropogenic factors, there is little evidence that predation by ravens is significant at a population level.

Landscapes that are less fragmented provide greater opportunity for species to shift ranges without being blocked (Opdam and Wascher, 2004). Fragmentation of the landscape through vegetation removal or grazing infrastructure such as fencing exacerbates the challenges that species are already dealing with in trying to adapt to a changing climatic regime. According to Beschta et al. (2012) and Beschta et al. (2014), livestock use of public lands in the West remains a major stressor with effects of increasing concern under the overarching stressor of climate change. Its removal or reduction is an ecologically efficient and unambiguous approach for restoring resilience to large areas of these lands. The Beschta et al. (2014) paper includes documentary photographs of sage-grouse habitat on Hart Mountain National Antelope Refuge showing the dramatic recovery following removal of livestock; these results demonstrate that livestock removal is likely to rapidly restore riparian areas, which are important brood-rearing and summer habitats for sage-grouse and which are often heavily degraded by livestock.

Any revisions to the plans must take a hard look at the direct, indirect, and cumulative impacts of livestock grazing on sage-grouse habitats. Sage-grouse depend on large areas of contiguous sagebrush with healthy, native understories. 75 Fed. Reg. 13910, 13916-17 (2010). Allowing ongoing livestock grazing to fragment and degrade what is left of these habitats is unscientific and contrary to the conservation ethic most Americans support.

In addition to the failures to consider all the myriad deleterious impacts of livestock grazing, the plans themselves haven't even been implemented yet. Despite promises in 2015 to prioritize the review and processing of grazing permits and leases in sage-grouse habitat, and a firm commitment to have completed these revisions within 18-24 months, the Forest Service is lagging behind on implementation and, in fact, possibly backing away from its commitments altogether.[5] This is unacceptable; the needs of sage-grouse and the necessary management of livestock grazing in sage-grouse habitat have been well-established.

The Forest Service seems to be overinterpreting the findings of several recent papers in its attempt to weaken the grazing habitat objectives of the plans. For example, the agency seems to be relying heavily on Smith et al. (2017) to cast ambiguity on grass height necessary for nest success. However, that paper really just shows that SGI projects and non-SGI projects are

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[5] "Furthermore, we may resolve these issues so that modification of the permits would not be necessary or may be different than the current plan direction (see the November 21, 2017 notice of intent, referred to above)." https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd565108.pdf

equally bad for sage-grouse. It specifically did not compare grazed areas with idled areas in terms of nesting success. Moreover, the leading cause of nest failure was predation (51.3 percent) and thus the question becomes whether predation is more or less common on grazed lands as a direct or indirect effect of livestock grazing becomes important. Additionally, the idled lands were only idled for 4-12 years; teasing apart differences in these samples would be interesting and looking at longer term differences in sage-grouse habitat in light of cyclical populations would also be necessary before changing the land use plans in light of these preliminary findings.

c. Oil and Gas/Fluid Mineral Leasing and Development

Any new planning process must adopt strong, science-based protections to shield greater sage-grouse from the effects of fluid mineral leasing and development. As noted, the agency's own science, set forth in the NTT and COT Reports suggested excluding these activities within sage-grouse priority habitats. New plans should exclude new fluid mineral leasing and/or surface occupancy within all PHMA.

The science is clear that greater sage-grouse are negatively impacted by activities associated with oil and gas development, including not only the construction and operation of well pads and associated drilling, gathering, processing, and transmission facilities, but also the ongoing human presence, noise and disturbance associated with production. Oil and gas infrastructure and activity can lead not only to direct mortality from collisions, contamination, and poaching, but, more significantly, to the abandonment of necessary habitats, including breeding grounds, winter habitat, and brood-rearing habitat. Sage-grouse ecosystems are slow and difficult to impossible to reclaim once sagebrush is removed.

Recent empirical study confirms the established finding that sage-grouse lek attendance is negatively related to oil and gas density, regardless of sagebrush cover and participation.[4] Green et al. examined greater sage-grouse lek attendance ,oil and gas well, and habitat and precipitation data from Wyoming over the period 1984 to 2008, and, consistent with numerous prior studies, that lek attendance declines are closely associated with the density of oil and gas development:

Oil and gas development correlates well with sage-grouse population declines from 1984 to 2008 in Wyoming, which is supported by other findings (Doherty et al. 2010b, Harju et al. 2010, Hess and Beck 2012, Taylor et al. 2013, Gregory and Beck 2014). Holloran (2005) found that several types of oil and gas infrastructure sited within 1.9 miles of the lek site had a negative impact on populations of breeding males on the lek; these infrastructure feature include both wellpads during the post-drilling, production phase and gravel trunk roads leading to five or more wellpads. It is important to note that a single wellpad or road can cause significant impacts, and these impacts occur even in cases where roads are not visible from the lek site due to intervening terrain (Holloran 2005). Holloran et al. (2007) found that yearling sage grouse avoided otherwise suitable nesting habitat within 930m (almost 0.6 mile) of oil and gas-related infrastructure. This means that individual wellsites, and their access roads and other related facilities, will be surrounded by a 0.6-mile band of habitat that has substantially lost its habitat capability for use by nesting grouse.

As with other studies, we also found support for 4-year lag effects of oil and gas development on lek attendance (Walker et al. 2007, Doherty et al. 2010a, Harju et al. 2010, Gregory and Beck 2014). This result suggests that development likely affects recruitment into the breeding population rather than avoidance of wells by adult males or adult survival. Adult sage-grouse are highly philopatric to lek sites (Dalke et al. 1963, Wallestad and Schladweiler 1974, Emmons and Braun 1984, Dunn and Braun 1985, Connelly et al. 2011a), and males typically recruit to the breeding population in 2–3 years. We would expect a delayed response in lek attendance if development affects recruitment, either by reducing fecundity or avoidance of disturbance by nesting females, as adult males die and are not replaced by young males.

On average, lek attendance was stable when no oil and gas development was present within 6,400m. However, attendance declined as development increased.[5] Importantly, Green et al. confirmed that declines in sage-grouse populations may continue even within Wyoming's "core areas," where density of wells is limited to approximately one pad per square mile.

Federal agencies have not heretofore analyzed the potential impacts of oil shale leasing and development on sage grouse, and has not applied specific protections against this land use, which is far more destructive to sage grouse habitat than even oil and gas leasing and development. Possible impacts of oil shale development include mine sites, retorting plants, and/or massive in-situ wellpad complexes; each for of oil shale extraction requires 100% destruction/industrialization of all lands to be produced for oil shale. To recover oil shale, the land in question must be mined (typically by strip mining), or an in-situ recovery process can be used that requires the entire surface of the deposit to be mined to be converted to a single massive wellpad studded with dozens to hundreds of injection wells, extraction wells, and freeze-wall maintenance wells. Mining has been shown to cause significant sage grouse population declines (*see* Braun 1986, Remington and Braun 1991). In any case, the result is 100% loss of all habitat features on the surface of the deposit to be extracted. No method has ever been developed to "do oil shale right" without massive impacts t the land, and these massive impacts will result in the loss of sage grouse habitats and populations. Oil shale and tar sands leasing in sage grouse Priority Habitats in Utah and Colorado was closed through previous plan amendments, but Wyoming Priority Habitats remain open. Oil Shale – Tar Sands Programmatic EIS Approved Land Use Plan Amendments at 24. Allowing this destructive practice in

Wyoming further jeopardizes sage grouse populations in key parts of the state, and this difference in management across state lines, failing to impose adequate regulatory mechanisms in Wyoming, is arbitrary and capricious and an abuse of discretion. The federal agencies must close all Core Areas, Focal Areas, Winter Concentration Areas, and Connectivity Areas to oil shale leasing and development in Wyoming to protect sage grouse habitats and populations.

To rectify these problems, the Forest Service should impose, as terms of the Land Use Plans, Conditions of Approval on all existing fluid mineral leases consistent with the recommendations of the Sage-Grouse National Technical Team, including no new surface occupancy on existing federal leases (with exceptions for occupancy of no more than 3% outside a 4-mile lek buffer, if the entire leasehold is within such habitat).

d. Roads and Off-Road Vehicles

The Forest Service must provide adequate protection from roads and off-road vehicle impacts. Roads have multiple impacts on sage grouse, including noise and movement from vehicle traffic causing disturbance, habitat fragmentation, and dust pollution that can depress productivity of sagebrush and other plants important to the diet of sage grouse. Sage grouse avoid habitats surrounding roads (Braun 1986, Holloran 2005, Wisdom et al. 2011). According to BLM's:

> Impacts on GRSG accrue over varying distances from origin depending on the type of development:
> …
> ● Interstate highways at 4.7 miles (7.5 kilometers) and paved roads and primary and secondary routes at 1.9 miles (3 kilometers) based on indirect effects measured through road density studies (Connelly et al. 2004; Holloran 2005; Lyon 2000)

Nevada – Northeastern California Greater Sage-grouse RMP Amendment DEIS at 605.

Roads fragment habitats and interfere with natural movements of sensitive species, and with regard to road upgrades, "Any exceptions resulting in road upgrades could further fragment habitat, cause vegetation loss, erosion, and the spread of invasive, non-native plant species." Wyoming Greater Sage-grouse RMP Amendment DEIS at 4-313 and 4-294, respectively.

The National Technical Team (2011: 11) recommended that at minimum, vehicle traffic in Priority Habitats be limited to designated roads and trails, use existing roads for access, limit construction to realignments of existing routes that minimize impacts to sage grouse, prohibit road upgrades that change route category, consider seasonal road closures, and conduct travel planning within 5 years, reclaiming roads and trails not designated for vehicular use.

Road densities are also an issue, because sage grouse avoid habitats adjacent to roads. Holloran (2005) found that road densities greater than 0.7 linear miles per square mile within 2 miles of leks resulted in significant negative impacts to sage grouse populations. This road density should be applied as a maximum density in Priority and General Habitats, and in areas that already exceed this threshold, existing roads should be decommissioned and revegetated to meet this standard on a per-square-mile-section basis.

Limiting road and trail networks and off-road vehicle travel also is critical in limiting the spread of invasive weeds. "Roads and trails are one of the main vectors of invasive weed spread, which leads to increase in FRCC and ecosystems moving away from natural fire regimes (CEC 2012)." Nevada – Northeastern California Greater Sage-grouse RMP Amendment DEIS at 701.

Off-road vehicle travel must be adequately regulated to protect sage grouse under new plans. Off-road vehicles are noisy, and typically exceed the background noise levels by more than 10 dBA. Northwest Colorado Greater Sage-grouse RMP Amendment DEIS at 399. This level of noise exceedance has significant negative consequences for sage grouse, as outlined in the section of this protest addressing noise. Off-road vehicle use also results in habitat degradation and destruction, disturbance of sage grouse, and proliferation of invasive weeds (NTT 2011; *see also* Manier et al. 2011). Limiting motor vehicles to existing roads is problematic because once one motorist illegally ventures off-road, he or she creates an "existing route" that can then legally be followed by every other motorist that follows. BLM characterized this proliferation of motorized routes thusly:

> Each year new trails are being created by a wide range of OHV users including, but not limited to, recreational users. Once a new trail becomes established it is considered by the public to be an existing route.

Wyoming Greater Sage-grouse RMP Amendment DEIS at 3-340.

For sage grouse PHMA, the federal agencies need to require the same "white-arrow" approach as used on many National Forests, in which motorized routes are closed to motorized use unless specifically posted as open. In addition, Special Use Permits need to be limited in Priority Habitats to activities that have neutral or beneficial impacts on sage grouse (NTT 2011).

The Forest Service should adopt the following measures into the plan amendments: New primary, secondary, or high-activity roads should be excluded within 1.9 miles of leks, and all new road construction or location should be excluded within 0.6 miles of leks (with no exceptions, waivers, or modifications); limit new road construction to realignments of existing routes where realignment has minimal impact on sage grouse, and require travel management planning to designate routes within Priority Habitat Management Areas within 5 years of plan amendment adoption.

e. Utility-scale wind and solar projects

Wind power development on a utility scale has the potential for multiple impacts to sage grouse, including habitat fragmentation, behavioral avoidance of tall structures, and disturbance of birds from noise and motion and/or human activity. LeBeau (2012) found that sage grouse had significantly lower nest success and chick survival in habitats in close proximity to wind turbines. The National Technical Team (2011) recommended that Priority Habitats be designated as exclusion areas for wind power development, and that General Habitats be avoidance areas for wind power development. Nevertheless, FS provided that wind energy development will be allowed in priority habitats on National Forests in Wyoming with "special stipulations." GB ROD, 50. This inconsistent treatment of industrial development fails to follow the best available science and will cause losses to affected sage-grouse populations. The importance of these habitats to the overall survival and recovery of sage grouse populations in the planning area require that adequate protections be provided: there is no scientific reason for any inconsistencies.

Guy wires for met towers pose a collision risk for sage grouse, and are unnecessary sources of mortality given the widespread availability of unguyed met tower designs. The plans state that the use of guy wires to be "avoided" in PHMAs (Wyoming RMPA FEIS at 2-30); instead the use of guy wires should be excluded to prevent the unnecessary and undue degradation (pursuant to FLPMA) that results from this unnecessary source of sage grouse mortality. BLM proposes that met towers should be "avoided" within 2 miles of leks in PHMAs (Wyoming RMPA FEIS at 2-30); this also is inadequate to prevent undue degradation to sage grouse habitats. The record establishes that met towers can result in sage grouse population declines (*see Cotterel Mountain data reviewed in* 'Wind Power in Wyoming,' Attachment D), and siting these tall structures in the midst of prime nesting habitat is likely to result in a significant level of habitat abandonment by grouse. The 2-mile buffer for such tall structures is not supported by the science, and instead a 5.3-mile buffer (after Holloran and Anderson 2005) should be applied. In addition, this restriction should not be limited to PHMAs but should also extend to General Habitats, Winter Concentration Areas, Focal Areas, and Connectivity Areas as well.

The Forest Service should designate sage grouse Priority Habitat Management Areas, Connectivity Areas, Focal Areas, and Winter Concentration Areas as exclusion areas for wind energy development, and designate General Habitats as avoidance areas for wind energy development. Met towers in sage grouse habitats should exclude the use of guy wires and siting of these facilities should be excluded within 5.3 miles of active leks in all occupied habitats, and in identified wintering habitats.

f. Transmission lines and renewable energy projects

Wisdom et al. (2011) found that lands within 3.1 miles of transmission lines and highways had an elevated rate of lek abandonment. Nonne et al. (2011) found that raven abundance increased along the Falcon-Gondor powerline corridor in Nevada both during the construction period, and long-term after powerline construction activities had ceased. Braun et al. (2002) reported that 40 leks with a power line within 0.25 mile of the lek site had significantly slower population growth rates than unaffected leks, which was attributed to increased raptor predation. Dinkins (2013) documented sage grouse avoidance of powerlines not just during the nesting period but also during early and late brood-rearing. LeBeau et al. (2014, Attachment E) found that sage grouse avoided habitats within 2.9 miles of transmission lines during the brood-rearing period.

The National Technical Team (NTT 2011) recommended that Priority Habitats be exclusion areas for overhead powerlines, and that General Habitats should be avoidance areas for overheads line. And according to BLM,

> Impacts on GRSG accrue over varying distances from origin depending on the type of development:
>
> ● Tall structures such as power lines, wind turbines, communication towers, agricultural, and urban development based on an avian predator foraging distance of 4.3 miles (6.9 kilometers; Boarman and Heinrich 1999; Leu et al. 2008)

Nevada – Northeastern California Greater Sage-grouse RMP Amendment DEIS at 605.

The National Technical Team (2011) recommended that general habitats be managed as avoidance areas for new rights-of-way, and also recommended that overhead powerlines and other infrastructure that have fallen out of use should be removed, when they occur in Priority Habitats.

The Forest Service cannot rely on perch inhibitors to reduce impacts to sage grouse, as these do not address the behavioral avoidance of sage grouse of tall structures, and don't even completely prevent raptor perching. Prather (2010) provided an empirical test of the effectiveness of perch inhibitors on smaller distribution lines in Utah, and found that they had no significant effect in terms of reducing raptor perching activity. Lammers and Collopy (2007) found similar results for larger transmission lines in Nevada.

The Plans currently manage PHMAs as right-of-way “avoidance areas” instead of exclusion areas (*See, e.g.,* Wyoming RMPA FEIS at 2-25), as recommended by their own experts. This prevents certainty of implementation by allowing new rights-of-way to be granted on a case-by-case basis. “Exclusion” is the appropriate level of management for these habitats based on the best available science, and this level of protection should also apply to Focal Areas and Winter Concentration Areas as well. Only portions of General Habitats would be managed as avoidance areas for rights-of-way based on other resource values (*See, e.g.,* Wyoming RMPA FEIS at 2-26); the importance of protecting sage grouse habitat merits avoidance management for all General Habitats.

The Plans exempt the Gateway West, Gateway South, and TransWest Express transmission line projects from the plan amendments. This loophole renders Forest Service management for large transmission projects essentially meaningless, as these three lines are the only lines of this size likely to be constructed in the planning area over the 20-year time horizon of the plan amendments. The Forest Service must instead subject these transmission lines to protection measures adequate to prevent major impacts to sage grouse habitats and populations.

In Wyoming, new transmission lines larger than 115 kV are allowed only within a two-mile designated corridor through Core Areas, within other designated corridors, or within 0.5 miles of existing large lines. *See, e.g.,* Wyoming RMPA FEIS at 2-26. This is troubling for several reasons. First, a two-mile-wide corridor is unnecessary to accommodate multiple large transmission lines, which no longer must be spaced widely apart, opening the door for unnecessary and undue degradation of sage grouse habitats pursuant to FLPMA. Second, the proposed transmission corridor (referencing a Governor’s Executive Order) shown in Attachment D Map 2 to the Wyoming EO 2011-5 unnecessarily traverses the Greater South Pass, Seedskadee, and Kemmerer Core Areas when the routing could readily have been altered to pass southward near Green River, south of the Seedskadee Core Area, and south of the Kemmerer Core Area to completely avoid PHMAs.

The National Technical Team (2011) reviewed the best available science, noting the sage grouse’s avoidance of tall structures, and recommended that priority habitats be “exclusion areas” for wind energy facilities. LeBeau (2012) found that sage grouse experienced significant declines in nest and brood survival in proximity to wind turbines. The Forest Service proposes to “restrict” wind energy development within Priority Habitats, whatever that means, but this is a discretionary Guideline that is not guaranteed to be enforced. Wyoming RMPA FEIS at 2-71. To remedy this deficiency, federal agencies must provide certainty of implementation by managing these sensitive habitats as “exclusion areas.”

Priority Habitat Management Areas (including SFA), Connectivity Areas, and Winter Concentration Areas should be exclusion areas for new transmission rights-of-way and wind and solar projects, including the TransWest Express, Boardman to Hemingway, Gateway West, and Gateway South projects to prevent unnecessary or undue degradation to sage grouse habitats. General Habitats should be managed as "avoidance areas" for new rights-of-way.

g. Prescribed fire in sage-grouse habitat

Fire is a threat to sage grouse populations, and the USFWS has identified prescribed fire as a threat to sage grouse in this region. Large fires of high frequency can extirpate sage grouse populations (Pedersen et al. 2003). A landscape mosaic of burns may not meet the nesting habitat needs of sage grouse (Nelle et al. 2000), and may also fail to meet grouse habitat requirements during other seasons (Wamboldt et al 2002). Fire was an uncommon occurrence in sagebrush habitats in pre-settlement times, with natural fire return intervals in Wyoming big sagebrush average 100-240 years (Baker 2007). Wyoming big sagebrush recovers slowly after fires, which typically result in 100% sagebrush mortality; recovery to pre-fire canopy cover takes over 100 years (Cooper et al. 2007). Baker (2007) examined the same issue and projected that Wyoming big sagebrush recovery following fire ranges from 50 – 120 years; for mountain big sagebrush, the recovery period was estimated at 35 – 100 years.

But vegetation manipulations to create fuel breaks also can fragment and degrade sage grouse habitat, as discussed elsewhere in this protest. The appropriate management approach will be to minimize the probability of large-scale fire in sage grouse habitat, without resorting to techniques that themselves destroy or degrade sage grouse habitats.

Prescribed fire also has no place in sage grouse habitats. Prescribed fire can result in a loss of sagebrush dominance for 25-45 years, and may also result in increased erosion (Sedgwick 2004). Cooper et al. (2007) projected the full recovery of Wyoming big sagebrush canopy cover would take 625 years based on their observed recovery rates following prescribed fire (a biologically improbable outcome), and no recovery at all was recorded following prescribed fire on 17 of 24 sites. Close proximity to seed sources and moister conditions did not accelerate recovery in this study. These researchers concluded, "Wyoming big sagebrush recovery takes so long that managers considering prescriptive burns need to have a long-term view of the landscape before eliminating a sagebrush habitat that will not return for at least a century" (Cooper et al. 2007:12). Rhodes et al. (2010) found that fires resulted in loss of sagebrush cover and increases in perennial grasses and invasive forbs, while native forbs did not increase in yield or nutritional quality, and ants (a significant part of the diet of sage-grouse chicks) also decreased. Beck et al. (2011) stated, "In particular, prescribed burning leads to pronounced negative response in sagebrush cover that lasts for at least a few decades," and recommended against burning in Wyoming big sagebrush.

The Forest Service should take a renewed look the primacy of cheatgrass invasion in determining patterns of rangeland fire; "The positive feedback loop between fire and invasive plant species may be the greatest impact on fire management and GRSG (Abatzoglou and Kolden 2011)." Nevada – Northeastern California Greater Sage Grouse RMP Amendment DEIS at 701. "In Oregon 19th and early 20th century grazing practices, along with introduction and spread of invasive plant species and the practice of fire suppression in the 20th century, have all contributed to fire suppression and to increasingly destructive wildfires." Oregon Greater Sage Grouse RMP Amendment DEIS at 4-10.

The current plan amendments fail to provide adequate controls on prescribed fire. Currently, there is an almost total absence of reliable protections. According to the best available science, prescribed fire should not be permitted in sage grouse habitats with less than 12" annual precipitation. This should be adopted as an Action if there is any new plan amendment.

h. The widespread creation of firebreaks is harmful to sage grouse and their habitats

The Forest Service should prevent the widespread creation of fire breaks under the sage-grouse plans. Creating firebreaks in sagebrush steppe is a practice unsupported by science. To the extent that the agency considers the use of fuel breaks under these plans, it must provide peer-reviewed, scientific literature that demonstrates that such fuel breaks in sagebrush steppe habitat have been demonstrated to reduce fire. Our review of the literature uncovered only unpublished white papers and "fact sheets" that cited no actual scientific studies to support the assertion that "green strips" slow or halt the spread of fire. If no such evidence can be provided, such "green strips" should be explicitly forbidden in the RMP amendment. It is obvious that "green strips" will only be green in the spring, when precipitation occurs and the risk of fire is negligible. During the dry periods when fire ignitions occur and spread most readily, "green strips" will be brown and represent a concentrated source of fine fuels that will do nothing to slow the advance of a flame front, and may indeed accelerate it. In addition, fuel breaks have no hope of halting (or even slowing) a flame front during a fire unless they are actively defended by firefighting personnel, and it is widely known that neither BLM nor the Forest Service have the combined resources to defend large networks of fuel breaks. Anecdotally, according to Vollmer (2005), fuel breaks that are left untended can become hazards in their own right:

> By the spring of 2003, annual weedy species (cheatgrass, mustards, filaree) dominated [the] fuel break resulting in shrub fuel being replaced by a highly flammable, continues [sic] fuel. Stands or mats of cheatgrass act as a hazardous fuel that can carry very hot fires, quickly. When cheatgrass dominates a fuel break, it acts as a wick, able to bring fire

in to the subdivision or take fire from the subdivision to the wildland. In addition, fire fighter safety is jeopardized due to the fast fire spread and difficulty of getting in front of the fire because blowing embers quickly spread the fire to new areas.

Meanwhile, the negative impacts of "green strips" on sage grouse are proven, as they fragment habitat, create edge environments where increased predation rates occur, and result in direct loss of valuable sagebrush stands that are key to grouse survival in terms of providing food and cover. We are concerned that the widespread implementation of green strips across Priority Habitats will significantly fragment degrade sage grouse habitats, further exacerbating population declines, and in the process will have no net effect on fire frequency or extent.

i. Noise protections need to be strengthened

Noise can mask the breeding vocalizations of sage grouse (Blickley and Patricelli 2012), displaces grouse from leks (Blickley et al. 2012a), and causes stress to the birds that remain (Blickley et al. 2012b). According to Blickley et al. (2010), "The cumulative impacts of noise on individuals can manifest at the population level in various ways that can potentially range from population declines up to regional extinction. If species already threatened or endangered due to habitat loss avoid noisy areas and abandon otherwise suitable habitat because of a particular sensitivity to noise, their status becomes even more critical." Noise must be limited to a maximum of 10 dBA above the ambient natural noise level after the recommendations of Patricelli et al. (2012); the ambient noise level in central Wyoming was found to be 22 dBA (Patricelli et al. 2012) and in western Wyoming it was found to be 15 dBA (Ambrose and Florian 2014, Ambrose et al. 2014, Ambrose 2015). [Wyoming FEIS #117] provides a review of the relevant literature on noise including analysis that indicates sage grouse lek population declines once noise levels exceed the 25 dBA level. With this in mind, ambient noise levels should be defined as 15 dBA and cumulative noise should be limited to 25 dBA in occupied breeding, nesting, brood-rearing, and wintering habitats, which equates to 10 dBA above the scientifically-derived ambient threshold.

It is reasonable to suppose that if noise that mimics oil and gas truck traffic causes elevated levels of stress-related metabolites in grouse on the lek (as demonstrated by Blickley et al. 2012b), that this physiological response would be substantially similar during other parts of this bird's life cycle. Indeed, these researchers stated, "Noise at energy development sites is less seasonal and more widespread and may thus affect birds at all life stages, with a potentially greater impact on stress levels." Patricelli et al. (2012) recognized this explicitly:

> "Second, and much more importantly, if noise levels drop down to stipulated levels at the edge of the lek, then much of the area surrounding the lek will be exposed to higher noise levels (see Figures 3 & 4). This management strategy therefore protects only a fraction of sage-grouse activities during the breeding season — mate assessment and copulation on

> the lek — leaving unprotected other critical activities in areas around the lek, such as foraging, roosting, nesting and brood rearing."

This failing has been incorporated by federal agencies in their ARMPAs by specifying that noise limits will be measured within 0.6 mile of the lek instead of at the periphery of occupied seasonal habitat. In the Wyoming Basins Ecoregional Assessment, the authors pointed out, "Any drilling <6.5 km [approximately 4 miles] from a sage-grouse lek could have indirect (noise disturbance) or direct (mortality) negative effects on sage-grouse populations." WBEA at 131.

Federal agencies adopted a limit of 10 dBA above ambient within 0.6 mile of leks in its Required Design Features, with ambient defined at 20-24 dBA. The ambient level should instead be set at 15 dBA and maximum noise allowed should not exceed 25 dBA to prevent lek declines due to noise. In addition, by setting the noise level within 0.6 mile of the lek, agencies fail to adequately protect nesting habitats, wintering habitats, and brood-rearing habitats from significant noise impacts. Instead, the Forest Service should set a limit of 10 dBA above a defined ambient noise level of 15 dBA within 4 miles of leks and in identified wintering habitats, to be applied across all occupied sage grouse habitats.

j. Mining

We are concerned that future development of coal resources could have a significant impact on remaining sage grouse populations. All priority habitats should be found unsuitable for coal leasing in order to prevent direct destruction of sage grouse habitats through strip mining and indirect impacts from grouse being driven away from otherwise suitable habitats adjacent to mine sites and associated access roads and facilities by increased industrial activity. The agencies should therefore find Priority Habitats unsuitable for surface mining for coal in order to provide regulatory certainty.

In the absence of mineral withdrawals, the current ARMPAs offer essentially no protection from locatable minerals mining, and given the limited latitude that agencies have to regulate projects under the 1872 Mining Law, this is a particularly egregious abdication of the responsibility to protect and restore sage grouse populations. Mining activity is widespread in sage-grouse habitats, so the impacts from mining projects on key sage grouse habitats would be expected to be substantial. All Priority Habitats designated should all be withdrawn from locatable minerals entry, and the federal agencies should accomplish this through the RMP amendment. We lack confidence in federal agencies' abilities to restrict the level of activity and surface disturbance on mining claims filed under the 1872 mining law to accommodate sage grouse habitat needs. Therefore, the appropriate course of action is to avoid allowing claims to issue in these priority habitats. We are particularly concerned about the potential for uranium extraction, be it underground, strip mining, or through *in situ* drilling and extraction methods.

The lack of uranium mining activity thus far is not a reliable measure of future development potential.

In some parts of the sage-grouse range, nonenergy leasable minerals development (especially phosphate) is more widespread and potentially more impactful to sage-grouse than fluid minerals leasing for energy production. The cost of closing all potential Priority Habitats to nonenergy leasable minerals entry would be small, in terms of hindering overall production. As with fluid minerals, the Forest Service should close all Priority Habitats to nonenergy leasable minerals leasing, as this does little to hinder minerals production but much to assure that adequate regulatory mechanisms are in place to address threats to sage grouse persistence.

Salable minerals include gravel pits, limestone quarries, and decorative rock, and sand deposits. Extraction typically entails small-scale operations that nonetheless can have significant direct and indirect impacts on sage-grouse and their habitats. There are abundant opportunities for salable minerals extraction outside sage grouse habitats, and therefore all priority and general habitats should be closed to salable mineral operations in order to foster sage grouse population maintenance and recovery.

B. The Plans Must Adequately Protect Sage-Grouse Winter Habitats.

Priority Habitats were largely designated on the basis of buffers around active lek sites, which encompass the breeding and nesting habitats used by grouse during spring and summer. But protecting wintering habitats is equally important to ensuring the continued existence and ultimate recovery of the species, and these wintering habitats are frequently located outside the protective boundaries of designated Priority Habitats. If sage grouse are unable to survive the winter season due to impacts to their wintering habitats, there will be no sage grouse in Priority Habitats or outside them in the planning area. " The agencies have already conceded that this is necessary: "Doherty et al. (2008) demonstrated that Greater Sage-Grouse in the Powder River Basin avoided otherwise suitable wintering habitats once they have been developed for energy production, even after timing and lek buffer stipulations had been applied." Buffalo RMP Revision DEIS at 367. In addition, Carpenter et al. (2010) found that wintering sage grouse avoided otherwise suitable habitats within a 1.2-mile radius of wellsites. Dzialek et al. (2012: 12, Attachment F) confirmed these relationships for wintering sage grouse in Wyoming, and concluded, "First, we can say with increasing confidence that the winter pattern of occurrence among sage-grouse shows consistency throughout disparate portions of its distribution. Second, avoidance of human activity appears to be a general feature of winter occurrence among sage-grouse." This indicates a broad consistency in sage grouse sensitivity to human development in wintering habitats throughout the species' range.

The Nevada RMPA FEIS provided a literature review of scientific studies on sage grouse winter habitat use, and concludes that distance from development and density of development are key factors. Holloran et al. (2015) determined that increasing wellpad density had a negative impact on sage grouse winter habitat use regardless of whether liquid gathering systems were used to reduce human activity levels or not, and also found a negative impact of distance to wellsites (within 2.8 km or 1.75 miles) and distance to roads. In accordance with this review of the best available science, the Forest Service should apply the following restrictions on development in designated winter habitats: (1) close all lands within 1.75 miles of winter habitats to future oil and gas leasing, coal location, non-energy minerals leasing, mineral materials sales, and seek withdrawal of these lands from locatable mineral entry; (2) for valid existing lease rights, apply a limit of 3% surface disturbance and one energy or mining site per square-mile section.

Yet despite these scientific studies, the Forest Service current plans fall short. This is completely inadequate because industrial facilities constructed in the summer will remain throughout every subsequent winter. Proposed stipulations fail utterly to address the threat of habitat destruction, habitat fragmentation, displacement of and stress to sage grouse resulting from vehicle traffic, noise, and human activity along roads and at industrial sites, displacement of grouse and increased predation resulting from overhead powerlines and tall structures, construction of wind farms, and other human intrusions know to disturb, displace, and causer population declines of sage grouse. Forest Service must do more than merely "consider" limiting over-the-snow vehicle use, with a vague Desired Condition in the context of travel management about grouse experiencing minimal disturbance. For these reasons, winter concentration areas should receive at least the level of protection from permitted industrial activities as recommended by NTT (2011) for priority habitats. As it stands now, unlimited surface disturbance is allowed in all winter concentration areas and winter habitat outside of priority habitats, risking significant winter habitat loss.

Any new revisions or amendments must discuss these impacts resulting from development and sagebrush removal in winter habitat or respond to comments noting these impacts. Moreover, the Forest Service must identify baseline winter habitat and winter concentration areas to create a science-based understanding of any plan amendment's impacts on wintering sage grouse.

Even if it were proper for the Forest Service to postpone the identification of winter habitat, the EIS must analyze any specific plans as to how and when this will occur or the criteria these areas must meet for winter habitat protections to apply. And the planning amendment must provide for interim protections for these areas until mapping is complete. In the absence of interim protections, it is thus entirely possible that sage-grouse wintering areas will be irreparably damaged and sage-grouse populations lost before they can receive minimal protections that apply today under the LUPAs, let alone the full set of protections needed for winter habitat based on the science. At minimum, any leasing or development of parcels that potentially contain winter habitat should be suspended until winter habitat and winter concentration areas are fully mapped and designated appropriate protections. This is extremely critical: Without any restrictions on sagebrush removal in wintering habitats, the habitat loss will be permanent. *See* Minnick 2015 (well sites lacked favorable soil conditions decades after reclamation, preventing sagebrush regrowth); *cf.* FEIS 4-315 (winter concentration areas "could be difficult to restore to original conditions…due to the composition and size of sagebrush in these areas"). Indeed, to the extent the EIS relies on winter habitat restoration as "mitigation" for any habitat loss, this is wishful thinking. Even a short-term loss of winter habitat would likely be detrimental to sage grouse dependent on these areas. As explained in the NTT report:

> Sage grouse exhibit strong site fidelity (loyalty to a particular area *even when the area is no longer of value*) to seasonal habitats, which includes breeding, nesting, brood rearing, and wintering areas. (Connelly et al. 2004, Connelly et al. 2011b). Adult sage grouse rarely switch between these habitats once they have been selected, *limiting their adaptability to changes*.

NTT at 51 (emphases added). Accordingly, loss of critical wintering habitat could lead to extirpation of sage-grouse populations that solely rely on these areas for the winter. *See also* FEIS at 3-5 ("Site fidelity in breeding birds could delay population response to habitat changes, and a clear response may require the death of most site-tenacious individuals.")

The NSO buffers in the plan are likely insufficient to protect wintering sage grouse. While surface disturbance could be prohibited up to 3.1 miles around leks (where winter habitats fall within Priority Habitat Management Areas and those PHMAs are located outside Wyoming with its 0.6-mile lek buffers), sage-grouse will still avoid development within 1.75 miles of wellpads and other development, as discussed above. Thus, development near these buffer zones could still cause sage grouse to avoid otherwise suitable winter areas falling within lek buffer zones. No analysis shows that enough winter habitat will be left undisturbed under existing ARMPAs to support local populations.

Absent clear definitions of "winter habitat" and "winter concentration area" and the distinction between the two, the Forest Service should adopt a plan that provides adequate disturbance and vegetation protection for <u>all</u> identified winter habitats. In the current Plans, it is unclear whether these terms are interchangeable or distinct concepts. The NTT defines "winter concentration areas" as, "Sage-grouse winter habitats which are occupied annually be sage-grouse and provide sufficient sagebrush cover and food to support birds throughout the winter (especially periods with above average snow cover). Many of these areas support several different breeding populations of sage-grouse. Sage-grouse typically show high fidelity for these areas, and loss or fragmentation can result in significant population impacts." NTT 2011, p. 37. Winter habitat, on the other hand, may be areas that have favorable sagebrush conditions for sage grouse throughout the winter, regardless of whether sage grouse annually occupy these areas. Wintering areas not utilized in typical years may become critical in severe winters. Caudill 2013. Thus, all winter habitat should be protected.

Finally, as detailed in previous comments, the Forest Service's winter habitat health objectives must have scientific support. These objectives should require 20-30% crown cover with shrub heights 25-35 cm above the median snow level, or greater than 40 cm in height, whichever is taller. *See* Center for Biological Diversity Nevada RMPA DEIS Comment, p. 22.

The agencies should suspend all development until winter habitat is identified and mapped. Identified winter habitats, whether inside or outside of Priority Habitats, should be closed to future mineral leasing and materials sales and withdrawn from locatable minerals entry. For valid existing rights both agencies should impose a 3% surface disturbance limit and one pad limit, both calculated per square mile section of winter habitat; No Surface Occupancy within 1.75 miles of the edge of wintering habitats; and no high-volume roads within 1.9 miles of wintering habitats. Wintering habitats should be seasonally closed to all vehicular access between November 30 and March 15. These winter concentration areas should receive the same protections as the NTT recommends for priority habitats.

IV. RESPONSES TO FOREST SERVICE REVIEW ISSUES

In addition to our concerns about the existing Plans' inadequacies in both substance and implementation, and our recommendations for further protections required, we provide the following information specific to the issues identified in the scoping notice.

A. SFA Designation: FS should Expand Sage Grouse PHMA Designations to Include All Lands Designated as Priority Areas for Conservation by the USFWS, as Well as Other Key Habitats.

The Forest Service NOI specifically addresses the US District Court of Nevada's finding that the agency erred in failing to provide a public comment period on the SFA designation. 82 FR 223. Our organizations believe that SFA was a last-minute compromise to cut habitat protections away from all PHMA lands and the, if anything, the analysis was undermined by overestimating the amount of protection given to sage-grouse habitat across the west. If there are any changes to SFA in forthcoming amendments, SFA management actions should be expanded to more lands, including all of the PHMA and wintering habitat.

The federal agencies have failed to designate all Priority Areas for Conservation (PACs) as outlined by COT (2013) as PHMAs or Focal Areas in a number of states, including Nevada, California, Utah, and Idaho, instead downgrading many of them to General Habitat Management Areas (GMHAs) or Important Habitat Management Areas ("IHMAs") that receive a lesser (and inadequate) level of protection. *See* NV FEIS #110. In Nevada, millions of acres of PACs were excluded from PHMA designation; the resulting designations are vermiculated with unprotected lands, allowing industrial uses in the midst of priority habitats that will clearly eliminate the habitat capability of the remaining PHMA lands designated. In California, some 70% of PACs were denied PHMA protection despite the fact that California has some of the smallest and most imperiled sage grouse populations remaining in the West. These discrepancies appear to add up to millions of acres, and represent an abject failure to provide an adequate level of habitat protection for lands with the USFWS deemed "key habitats that are essential for sage-grouse conservation" (COT 2013, Dear Reader letter).

In its initial designation of Core Areas, the State of Wyoming made some major errors that have been implicated in subsequent population declines and threats to long-term viability for sage grouse populations (*see* Taylor et al. 2012). These failures are adopted by the agencies in its Wyoming RMP amendment, crippling the ability of the new plan to maintain viable populations of sage grouse in this area. It is important to note that many of the most populous sage grouse leks in northeast Wyoming, the south-central part of the state lie outside Core Areas. *See* WY FEIS #126. The State of Wyoming has developed current lek population density mapping based on 2014 data, updated versions of which are readily available to FS. FS should have included such a population density buffer map with its FEIS as part of its NEPA baseline information fulfillment; failure to do so violates NEPA. Later, areas with high population densities were removed from Core Area status to accommodate industrial projects that are incompatible with maintaining sage grouse on the landscape. *See* WY FEIS 127. At the outset of the State's consensus-based Core Area mapping process, the original boundaries of Core Areas were drawn to exclude sage-grouse habitats that land users were interested in developing, particularly in the Powder River Basin, Atlantic Rim area, and upper Green River Valley. As a result, thousands of acres of undeveloped habitat were denied protection despite their vibrant sage-grouse populations and relatively undeveloped condition. Under the RMP Amendment process, the Forest Service should correct politically-driven changes to Core Area boundaries that exclude lands within 5.3 miles of leks that represent the smallest area 75% of the Wyoming sage grouse population. In addition, the State of Wyoming made several boundary increases in Core Area designations, and these should be incorporated as PHMAs if the plans are amended again. The ARMPAs incorporated these errors and unscientific delineation of Core Area boundaries into the PHMAs in its approved plan amendments in Wyoming, resulting in a failure to protect key habitats that have been wrongfully excluded from Core Areas.

In Idaho, many lands designated as PACs by the USFWS were ultimately designated as IHMAs, with a lesser level of protection. And, in Utah, the agencies failed to designate all lands within 5.3 miles of leks within PHMA as PHMA habitats, and excluded many of the state's important sage grouse breeding and nesting habitats from PHMA designations to accommodate industrial uses.[6]

In sum, designated PHMAs should be expanded to all lands designated as PACs by the US Fish and Wildlife Service in 2013 (COT 2013), and include expansions of Core Areas adopted by the State of Wyoming in 2015. In turn, SFA status and management parameters should be expanded to all lands designated as PHMA if the Forest Service truly wants to protect and conserve sage-grouse throughout its range and the Plans are being used to defer ESA listing.

B. Mitigation Standards

a. Effectiveness of Compensatory Mitigation

If the Forest Service proposes to allow compensatory mitigation in lieu of compliance with disturbance density and other requirements, restrictions must not be waived with the approval of off-setting mitigation. We call upon the Forest Service to reach a determination regarding the effectiveness of each category of compensatory mitigation to result in no net loss of sagebrush populations for the area in question. Please document any and all scientific studies that conclude that compensatory mitigation efforts have yielded an increase in sage grouse populations for the area to which mitigation efforts apply. We are unaware of any cases in which

[6] The cruel irony is that the areas that have been excluded are those that have the most serious threats from energy development; the end result is that the agencies are protecting only those areas that face no imminent threats. In areas such as the Bald Hills population area (Utah RMPA FEIS at Map 1-2), significant amounts of lands designated as PACs in the COT (2013) report were designated not as PHMAs, which the best available science demands, but as GHMAs. FEIS at Map 2-1. In the Panguitch population area (Map 1-2), lands designated as PACs in the COT (2013) report were not designated at all (Utah RMPA FEIS at Map 2-1), leaving remaining designated PHMAs badly fragmented. In the Parker Mountain and Emery population areas (Utah RMPA FEIS at Map 1-2), where the COT report recommends the designation of PACs that provided connectivity along the length of these population areas, the BLM designates instead a few isolated islands of sage grouse habitat (Utah RMPA FEIS at Map 2-1), which will result in the further fragmentation and degradation of sage grouse habitats in these population areas and ultimately result in the extirpation of their grouse populations. In the western Rich population area and the west-central portion of the Carbon population area (Utah RMPA FEIS at Map 1-2), the BLM designates as PHMA significantly less habitat than was designated as PAC by COT (2013), designating some PAC lands as GHMA and some as no sage grouse designation at all. Utah RMPA FEIS at Map 2-1, *and as shown on map*, Attachment H (Utah FEIS #110). This should be evaluated in context of Forest Service management changes in the forthcoming EIS.

a compensatory mitigation program has resulted in a significant increase in sage grouse compared to an untreated landscape. The fact that "compensatory mitigation" funding frequently is used to purchase conservation easements is problematic, because this is a paper transaction with legal ramifications preventing future potential losses, but can never yield population gains to offset the very real and immediate losses of sage grouse habitats and populations incurred as a result of industrial development. The Forest Service must document population-level benefits for sage grouse to validate offsetting mitigation efforts. The details of mitigation must not be deferred to subsequent implementation teams because it prevents the EIS from analyzing the impacts of alternatives taking into account "offsetting" mitigation, and fails to analyze the effectiveness of mitigation measures, both of which would violate NEPA.

C. Lek Buffers In All Habitat Management Area Types

a. A 4-mile No Surface Occupancy Buffer Around Leks is Necessary in PHMAs

Industrial activities directly eliminate and fragment habitat. Equally, or perhaps even more importantly, the resulting facilities are hubs for human and vehicular activity that disturb and displace sage grouse, resulting in lower rates of survival and/or reproduction and leading to population declines; "Human presence and vehicles may force special status species away from desired habitat to lower-quality, less desirable habitat." Wyoming Greater Sage-grouse RMP Amendment DEIS at 4-302. This, in turn, hinders the ability of sage grouse to thrive: "Moving to lower-quality sagebrush could result in lower calorie consumption and reduced health and vigor, making birds more susceptible to disease and predation." Wyoming Greater Sage-grouse RMP Amendment DEIS at 4-298. As a result, facilities and activities deleterious to sage grouse must be kept an adequate distance away from key habitats to prevent significant impacts to grouse.

Holloran (2005) found that several types of oil and gas infrastructure sited within 1.9 miles of the lek site had a negative impact on populations of breeding males on the lek; these infrastructure feature include both wellpads during the post-drilling, production phase and gravel trunk roads leading to five or more wellpads. It is important to note that a single wellpad or road can cause significant impacts, and these impacts occur even in cases where roads are not visible from the lek site due to intervening terrain (Holloran 2005). Drilling activities can have significant impacts when wells are sited within 3 miles of leks (*id*.). Manier et al. (2014) reviewed all available science and found that appropriate lek buffers (the "interpreted range") ranged from 3.1 to 5 miles. Aldridge and Boyce (2007) suggested that even larger buffers (10 km, or 6.2 miles) are warranted.

In addition to significant negative impacts on breeding populations at the lek site, industrial incursions can also have a significant negative impact on nesting females. The lek is the hub of nesting activity, with most females nesting within 4 to 6 miles of a lek site. Holloran et al. (2007) found that yearling sage grouse avoided otherwise suitable nesting habitat within 930m (almost 0.6 mile) of oil and gas-related infrastructure. This means that individual wellsites, and their access roads and other related facilities, will be surrounded by a 0.6-mile band of habitat that has substantially lost its habitat capability for use by nesting grouse. The National Technical Team (2011: 20) observed, “it should be noted that protecting even 75 to >80% of nesting hens would require a 4mile radius buffer (Table 1). Even a 4mile NSO buffer would not be large enough to offset all the impacts reviewed above.” Importantly, a 0.6-mile lek buffer covers by area only 2% of the nesting habitat encompassed by a 4-mile lek buffer, which takes in approximately 80% of nesting grouse according to the best available science.

The NTT experts recommended for existing fluid mineral leases that a 4-mile No Surface Occupancy buffer should be applied to leks, with an exception allowed in cases where the entire lease is within 4 miles of a lek, in which case a single wellsite should be permitted in the part of the lease most distal to the lek (NTT 2011). This recommendation is reinforced by a similar recommendation from western state agency biologists, who also recommended a 4-mile No Surface Occupancy buffer (Apa et al. 2008). According to Taylor et al (2012: 27), “Second, female sage-grouse that visit a lek use an approximately 9-mi (15-km) radius surrounding the lek for nesting; a 2-mi (3.2-km) radius encompasses only 35-50% of nests associated with the lek (Holloran and Anderson 2005, Tack 2009). While a lek provides an important center of breeding activity, and a conspicuous location at which to count birds, its size is merely an index to the population dynamics in the surrounding habitat. Thus attempting to protect a lek, without protecting the surrounding habitat, provides little protection at all.”

In the context of the original Greater Sage-Grouse RMP amendment and revision effort, the agencies’ own Draft EIS analysis supported 4-mile No Surface Occupancy buffers to be applied as Conditions of Approval to existing fluid mineral leases. The Wyoming RMPA DEIS states, “Walker et al. (2007) recommends a buffer distance of at least 4.0 miles containing extensive stands of sagebrush habitat for breeding populations to persist.” Wyoming Greater Sage-grouse RMP Amendment DEIS at 4-291. For the Buffalo RMP revision, BLM’s analysis of the science states,

> “Energy development within two miles of leks is projected to reduce the average probability of lek persistence from 87% to 5% (Walker et al. 2007a). Current research suggests that impacts to leks from energy development are discernible out to a minimum of 4 miles, and that some leks within this radius have been extirpated as a direct result of energy development (Apa et al. 2008). Even with a timing limitation on construction activities, Greater Sage-Grouse avoid nesting in oil and gas fields because of the activities associated with operations and production”

Buffalo RMP Revision DEIS at 367. For Montana, BLM observes, “Impacts from energy development occur at distances between 3 and 4 miles. Impacts to leks caused by energy development would be most severe near the lek.” HiLine RMP Revision DEIS at 4-135. Manier et al. (2014) undertook a comprehensive analysis of the available science on lek buffers, and concluded that the appropriate range for lek buffer protections was 3.1 to 5 miles, which encompasses and buttresses BLM’s earlier NTT (2011) expert recommendations.

State agencies and their wildlife experts have long pointed out the flaws in smaller lek buffers and the need for 4-mile No Surface Occupancy buffers around leks. According to the Nevada Division of Wildlife, “…the current NSO distance is 0.6 miles, which is not based on the best available science (see Coates et al. 2013 which suggests a buffer distance of 5.0 kilometers).” NDOW comments on Nevada – Northeastern California DEIS, January 14, 2014, analysis chart 1. Apa et al. (2008, emphasis added) reviews the best available science by a team of state sage grouse biologists, and states,

> “Yearling female greater sage-grouse avoid nesting in areas within 0.6 miles of wellpads, and brood-rearing females avoid areas within 0.6 miles of producing wells. This suggests <u>a 0.6-mile buffer around all suitable nesting and brood-rearing habitat</u> is required to minimize impacts to females during these seasonal periods.” This report further clarifies, “These suggest that all areas within at least 4-miles of a lek should be considered nesting and brood-rearing habitats in the absence of mapping.”

Thus, state experts in this report in effect recommended a 4.6-mile NSO buffer around active leks.

The U.S. Fish and Wildlife Service also has pointed out the inadequacy of smaller lek buffers. For the Utah RMP effort, the agency states, “There is substantial scientific information that shows that impacts of human disturbance (e.g. oil and gas drilling) to sage-grouse remain discernible out to distances > 4 miles of a lek.” USFWS comments on Utah Conservation Plan 7/12/12, at 3. The agency goes on to conclude, “In summary, we recommend avoiding permanent structures within a 4 mile lek buffer…at all times. Exceptions may be appropriate for the placement of permanent structures on non-habitat areas within the 4 mile lek buffer if it can be determined that the location of these structures will not impact nesting sage-grouse.” USFWS comments Utah Conservation Plan, 5/8/13 at 8. In Nevada, the USFWS states, “We recommend a year-round lek buffer of 4.0 miles.” USFWS Nevada/NE California comments at 26. The Nevada – Northeastern California DEIS, BLM states,

> Impacts on GRSG accrue over varying distances from origin depending on the type of development:
> …

- Energy extraction such as oil and gas, geothermal, and plan of operation mining at 11.8 miles (19 kilometers) based on direct impacts of field development, including associated infrastructure, noise, lighting, and traffic (Johnson et al. 2011; Taylor et al. 2012)

Nevada – Northeastern California Greater Sage-grouse RMP Amendment DEIS at 605. BLM Wyoming Draft EIS analysis arrives at the same conclusion: "Buffer distances from 0.5 to two miles from oil and gas infrastructure have been shown to be inadequate to prevent declines of birds from leks (Walker et al. 2007). Studies have shown that greater distances, anywhere from two to four miles, are required for viable Greater Sage-Grouse populations to persist (Connelly et al. 2000, Holloran and Anderson 2005, Walker et al. 2007)." Wyoming Greater Sage-grouse RMP Amendment DEIS at 4-335.

According to Apa et al. (2008), "Buffer sizes of 0.25 mi., 0.5 mi., 0.6 mi., and 1.0 mi. result in estimated lek persistence of 5%, 11%, 14%, and 30%." BLM concludes, "Studies have shown that greater distances, anywhere from two to four miles, are required for viable Greater Sage-Grouse populations to persist." Wyoming Greater Sage-grouse RMP Amendment DEIS at 4-335. For these reasons, the application of a 0.6-mile lek buffer is arbitrary and capricious, and will contribute to further population declines in Core Areas that will contribute to the need to protect the greater sage grouse under the Endangered Species Act.

Holloran (2005) undertook an empirical test of the adequacy of 0.25-mile No Surface Occupancy buffers and 2-mile Timing Limitation Stipulations, and determined that sage grouse in the Pinedale Anticline and Jonah Fields would be completely extirpated within 19 years of the study as a result of full-field development with this package of protections applied. The BLM's NEPA analysis for a Miles City Field Office oil and gas leasing EA provides a thorough synopsis:

> "Sage grouse are offered species specific protections through a stipulation. Under Alternative B, ¼ mile NSO buffers and 2 mile timing buffers would apply where relevant. Based on research, these stipulations for sage grouse are considered ineffective to ensure that sage grouse can persist within fully developed areas. With regard to existing restrictive stipulations applied by the BLM, (Walker et al. 2007a) research has demonstrated that the 0.4-km (0.25 miles) NSO lease stipulation is insufficient to conserve breeding sage-grouse populations in fully developed gas fields because this buffer distance leaves 98 percent of the landscape within 3.2 km (2 miles) open to full-scale development. Full-field development of 98 percent of the landscape within 3.2 km (2 miles) of leks in a typical landscape in the Powder River Basin reduced the average probability of lek persistence from 87 percent to 5 percent (Walker et al. 2007a).

> Other studies also have assessed the efficacy of existing stipulations for sage grouse. Impacts to leks from energy development are most severe near the lek, and remained discernable out to distances more than 6 km (3.6 miles) (Holloran 2005, Walker et al. 2007a), and have resulted in the extirpation of leks within gas fields (Holloran 2005, Walker et al. 2007a). Holloran (2005) shows that lek counts decreased with distance to the nearest active drilling rig, producing well, or main haul road, and that development influence counts of displaying males to a distance of between 4.7 and 6.2 km (2.9 and 3.9 miles). All well-supported models in Walker et al. (2007a) indicate a strong effect of energy development, estimated as proportion of development within either 0.8 km (0.5 miles) or 3.2 km (2 miles), on lek persistence. Buffer sizes of 0.25 mi., 0.5 mi., 0.6 mi. and 1.0 mi. result in an estimated lek persistence of 5 percent, 11 percent, 14 percent, and 30 percent. Lek persistence in the absence of CBNG development averages approximately 85 percent. Models with development at 6.4 km (4 miles) had considerably less support, but the regression coefficient indicated that impacts were still apparent out to 6.4 km (4 miles) (Walker et al. 2007a). Tack (2009) found impacts of energy development on lek abundances (numbers of males per lek) out to 7.6 miles."

Miles City October 2014 Oil and Gas Leasing EA, Environmental Assessment DOI-BLM-MT-C020-2014-0091-EA, May 19, 2014 at 60.

For most states, federal agencies purported to apply lek buffer distances in accordance with Manier et al. (2014) at the project stage of the NEPA approval process. These typically are set at 3.1 miles for roads and energy infrastructure, 2 miles for tall structures, and 1.2 miles for low structures, and represent the lowest (least protective) end of the protection spectrum described by Manier et al. (2014). We are concerned that these buffer distances (and also the 1.2-mile standard for low structures) are inappropriately small (with the possible exception of the road buffer) because while they be adequate to protect breeding grouse while on the lek based on the best available science, they will allow these disruptive and damaging features to be located in the midst of prime nesting habitat, which extends 5.3 miles from the lek site (Holloran and Anderson 2005). Furthermore, both agencies' Plan amendments allow leeway to relax even these minimum lek buffers, rendering them completely discretionary. Because the nesting period is equally sensitive and equally important to survival of and recruitment to sage grouse populations, larger buffers are necessary.

In addition, in Wyoming and parts of Utah, federal agencies adopted even more inadequate NSO buffers for leks, of 0.6 mile inside Core Areas and within 0.25 mile of leks in General Habitats. *See, e.g.,* Wyoming RMPA FEIS at 2-60. The efficacy of these scientifically inadequate lek buffers is further undermined by the fact that they are discretionary; exceptions, modifications, or waivers could be issued with the concurrence of the Wyoming Game and Fish Department ("WGFD"). *Id*. Males use shrubs <1 km (0.6 mi) from a lek for foraging, loafing, and shelter (Rothenmeier 1979, Autenreith 1981, Emmons and Braun 1984); this does not make 0.6 mile the appropriate NSO buffer for preventing impacts even to breeding bird, much less nesting birds. In Wyoming, State and BLM policies have in the past (and in the ARMPAs) erroneously used this as a basis for a 0.6-mile No Surface Occupancy buffer around leks. However, there is no science to indicate that preventing wells within 0.6 mile of a lek will eliminate or minimize negative population impacts on sage grouse.

For the foregoing reasons, protections applied to existing oil and gas leases both inside Priority Habitats and in General Habitats under the current land use plan amendments are scientifically unsound, and biologically inadequate. Federal agencies' failures to apply adequate lek buffers to conserve sage grouse, both inside and outside of Priority Habitats, in the face of scientific evidence, their own expert opinion, and their own NEPA analysis to the contrary, is and continues to be arbitrary and capricious and an abuse of discretion, and this legal deficiency should be remedied of the ARMPAs are amended.

Should the ARMPAs be further amended, Forest Service must provide 4-mile No Surface Occupancy buffers at minimum for all active leks in PHMAs for existing oil and gas leases, with exceptions available for mineral leases located entirely within this buffer for a wellsite of minimal size and intrusion to be placed at a location most distal from an active lek or leks.

b. Buffer Distances in GHMAs

The agencies' plans to rely on 0.25-mile No Surface Occupancy buffers and 2-mile Timing Limitation Stipulations to govern oil and gas development outside Priority Habitats is radically insufficient to protect this BLM Sensitive Species and is a known recipe for sage grouse extirpation. Holloran (2005) undertook an empirical test of the adequacy of 0.25-mile No Surface Occupancy buffers and 2-mile Timing Limitation Stipulations, and determined that sage grouse in the Pinedale Anticline and Jonah Fields would be completely extirpated within 19 years of the study as a result of full-field development with this package of protections applied.

Should the ARMPAs be further amended, the Forest Service must provide 4-mile No Surface Occupancy buffers at minimum for all active leks in Connectivity Areas and General Habitats for existing oil and gas leases, with exceptions available for mineral leases located entirely within this buffer for a wellsite of minimal size and intrusion to be placed at a location most distal from an active lek or leks.

D. Disturbance Density Caps

A limit of 3% human surface disturbance per square-mile section is the minimum necessary standard for preventing habitat abandonment be sage grouse. Knick et al. (2013) found that 99% of active leks across the western half of the sage grouse's range were surrounded by lands with 3% or less human development. The vast majority were surrounded by much less disturbance. Copeland et al. (2013) found that if all of the State of Wyoming sage grouse policy provisions (which include a 5% disturbance cap calculated using a Disturbance Density Calculation Tool) were implemented fully and to the letter, that a 9 to 15% further decline in greater sage grouse populations would still occur statewide, including a 6 to 9% decline within designated Core Areas (where the 5% disturbance cap would be applied). There is <u>no scientific evidence at all</u> indicating that sage grouse can tolerate a greater percentage of surface disturbance. In particular, the 5% cap on disturbance proposed for the Wyoming RMP amendments and revisions for Core Areas and Connectivity Areas been shown to be effective by <u>no scientific study, ever.</u>

The five percent disturbance threshold, as adopted under the Wyoming ARMPA (including parts of the Utah), Lander, Bighorn Basin, and Buffalo RMP revisions, does not conserve sage-grouse long-term and is only a guess by agencies and others seeking to accommodate development in sage-grouse habitat. Past projects approved prior to implementation of the Wyoming Core Area strategies indicate that sage-grouse are adversely affected at lower levels of disturbance. For example, for the Continental Divide/Wamsutter II Natural Gas Project approved in the year 2000, 3,000 wells were proposed with 22,400 acres of new surface disturbance, representing 2.1 percent of the planning area (with an average well density of 4 wellsites per square mile) (BLM 2000); today, sage grouse are functionally extirpated in this area. In the Atlantic Rim coalbed methane field, 2,000 wells were permitted at a density of eight wells per square mile, far above the threshold known to cause sage grouse declines. Kirol (2012) found for his study area in the Atlantic Rim coalbed methane field of Wyoming's Red Desert that surface disturbance greater than or equal to 4% of the land area had a significant negative impact on greater sage grouse brood rearing habitat. Today, sage grouse are essentially extirpated in developed portions of this field. The projected surface disturbance for this project is 15,800 acres, or 5.85 percent of the project area (BLM 2005). Recent science in the western portion of the sage grouse range found that some 99 percent of active leks were located in areas surrounded by lands with 3% or less surface disturbance from roads, powerlines, pipelines, and other features (Knick et al. 2013). Clearly, a threshold of five percent is too high to sustain sage grouse.

The North Dakota Game and Fish Department also concurred that in order to achieve low to no impact, disturbance percentages needed to be maintained at 3% or below; moderate impact results from disturbance percentages between 3% and 6% (Robinson 2013).

The use of Disturbance Density Calculation Tool ("DDCT"), Project Impact Analysis Area ("PIAA"), or similar area substantially larger than the footprint of proposed development for calculating percentage of lands disturbed also is inappropriate. Knick et al. (2013) used a 3-

mile buffer around leks to calculate their 3% surface disturbance threshold, a land area much smaller than a DDCT area. Wyoming's current 5% disturbance limit using a DDCT would allow much greater surface disturbance density, not less, within the geography (3-mile radius) used by Knick et al. (2013). In addition, Forest Service disturbance caps are discretionary guidelines only. This means that these measures have no certainty of implementation. Densities of oil and gas wellpads indicate that a 3% limit is consistent with one wellpad per square mile (which equates to 2.7% surface disturbance on average), the density of wellpads beyond which significant population declines occur. The NTT Report (2011: 7) was particularly explicit regarding the necessity to implement the 3% disturbance threshold rigorously:

> Manage priority sage□grouse habitats so that discrete anthropogenic disturbances cover less than 3% of the total sage□grouse habitat regardless of ownership. Anthropogenic features include but are not limited to paved highways, graded gravel roads, transmission lines, substations, wind turbines, oil and gas wells, geothermal wells and associated facilities, pipelines, landfills, homes, and mines.[iii]
>
> o In priority habitats where the 3% disturbance threshold is already exceeded from any source, no further anthropogenic disturbances will be permitted by BLM until enough habitat has been restored to maintain the area under this threshold (subject to valid existing rights).
>
> o In this instance, an additional objective will be designated for the priority area to prioritize and reclaim/restore anthropogenic disturbances so that 3% or less of the total priority habitat area is disturbed within 10 years.

The current sage grouse land use plan amendments qualify their imposition of surface disturbance thresholds with the clause "subject to valid existing rights." *See, e.g.*, Wyoming Greater Sage-Grouse Land Use Plan Amendment 2-59; Northwest Colorado Greater Sage-Grouse Proposed LUPA/Final EIS 2-21; Utah Greater Sage-Grouse Proposed LUPA/Final EIS 2-18. The valid existing rights conveyed by a federal mineral lease or mining claim do not include an absolute right to use of the federally-managed surface, but are subject to stipulations, federal law, and conditions of approval or plans of operation pursuant to the reasonable federal regulation of surface use. For the 3% disturbance cap to be effective as a conservation measure, and certain of implementation as a regulatory mechanism under the ESA, it must be applicable to all disturbances, including those on existing leases, claims, or rights-of-way.

If the clause "subject to valid existing rights" means, as certain of the FEISs imply, that existing leaseholders may create surface disturbances that exceed the cap, or where the cap has already been exceeded, then the Forest Service cannot rely on the disturbance cap to mitigate habitat loss on lands already under lease. In order for the disturbance cap mechanism to be effective, it must apply to existing leases or claims, precluding new disturbance that would exceed the cap (or where the cap has already been exceeded), at least until such time as previously-disturbed areas have been restored to adequate habitat function.

E. Changing Habitat Boundaries In Response to "New Information."

It is possible that, if and when sage-grouse begin to recover, they will move into new areas and begin reoccupying restored and improving landscapes. With this in mind, it would be appropriate to provide *increased* habitat protections in these areas. Thus, habitat boundaries might change from GHMA to PHMA as conditions improve and occupation increases. This kind of "new information" (including expansions of Core Areas as proposed in the Wyoming state plan) that results in increased protections would not be objectionable.

However, changing habitat boundaries towards categories with lesser protections incentivizes not following the best management practices within the current category. For example, allowing exemptions and exceptions to lease stipulations and thus degrading the quality of the habitat should not then be used to downgrade protections at a SFA/PHMA/GHMA level for the same lands. It would be too easy to reclassify lands for lesser protections after allowing destruction to diminish the habitat, where the "new information" is that there is now fragmentation, noise, vegetation conversion, higher road densities, etc.

Thus the habitat boundary changes should be confined to changes in one direction: increases in acreages getting greater protections. Until the sage-grouse are no longer in need of protection, there should be no loss in acres of protected habitat.

G. Should Planning Effort Occur Through a regional, state-by-state, or Forest-by-Forest basis?

As described in detail above, the current fragmented planning effort and the political compromises across the plans according to state and region already undermine the effort to conserve greater sage-grouse. Handing the authority to protect the species back to the states is likely to result in more of the same declines that led the species towards federal protection in the first place. Setting state or local population targets for sage-grouse, and waiving habitat protection standards when these are met, is not an acceptable or scientifically valid approach for a number of reasons.

First, sage-grouse populations naturally fluctuate in about a 10-year cycle, rising upward to a peak, then descending to a low point. Thus, using population targets to remove habitat protections when targets are attained at the peak of the cycle risks habitat destruction that will exacerbate cyclical lows and depress future population peak that can be met at the peak of the cycle to remove habitat protections will If habitat protections don't apply at the peak of the cycle, because some arbitrary population target has been met, then habitat destruction will be allowed at levels known to cause population crashes. Once the habitat is lost, the ability of a population to fully rebound is lost too, and future generations of grouse will suffer from the long-term habitat impacts allowed when population were higher. And those habitat losses will be there to depress every population peak, and every population trough, that follows.

Next, sage-grouse population declines take habitat losses two to ten years to show up following habitat losses (Harju et al. 2010) in the form of population declines. This is because adult sage-grouse have incredibly strong ties to the habitats where they live so they continue to occupy degraded habitats even as the juveniles disperse and move on, even after that habitat becomes so decimated that it no longer supports sage grouse. So, like the population of a dying factory town after the mill closes, the young birds leave while the older birds stay and die out, until the population disappears. goes extinct. This phenomenon means that populations can stay above pre-set targets for years before showing signs of distress, while habitat destruction allowed by waiving restrictions obliterates the habitat base that supports the population. Failing to take corrective action in real time as habitats are being destroyed or degraded makes the resulting population losses worse, even if the measurable effects aren't observable until a few years later.

Finally, the outcome-based approach means no sage-grouse habitat – no matter how important – is off-limits to incompatible land uses, or indeed to total destruction. When populations are on the upswing, big chunks of the most important habitat could be opened up for unrestricted conversion to industrial landscapes. It can take up to 120 years for sagebrush habitat to re-grow following elimination (Baker 2006). For some types of habitat conversion, it is gone forever, and so a population-based policy ensures that the Sagebrush Sea continues to disappear, anywhere there is an industrial appetite for habitat destruction. This means sage-grouse populations continue a relentless march toward extinction at the local, state, and worldwide scales.

Moreover, State protections are demonstrably inadequate to stem population and habitat declines as illustrated most strikingly by the example of the State of Wyoming. Recent peer-reviewed scientific publications have reviewed greater sage-grouse population response to oil and gas management measures in Wyoming, and re-confirmed lek attendance by male sage-grouse declines approximately 2.5% per year in response to oil and gas development, and that attendance declines as development increases, even where well pad density is limited.[7] In light of this information, the Forest Service cannot continue to assume, against scientific evidence, that the management measures in the 2015 RMP amendments, much less weakening of those measures as proposed in the Zinke Report, will be sufficient to stem sage-grouse population decline.

V. ADDITIONAL ISSUES

A. Population trends

While some have hailed the Sage-grouse plan amendments as a great success, the reality is that sage-grouse populations are continuing to decline. This is occurring even though major oil and gas projects have not occurred in sage-grouse habitats for the past 10 years due to a bust in both oil and gas commodity prices, and major transmission lines, wind farms, and other significant industrial projects have yet to be built under the new plan amendments. Of course, livestock grazing has continued relatively unchanged by the plan amendments across the range of the greater sage-grouse, and thus its impacts are having an ongoing effect on sage-grouse populations.

We have compiled the maximum lek counts for a number of states, based on state game and fish agency data. Notable in these data is that North Dakota is down to five strutting males, while South Dakota has declined to 228 strutting males. Both of these populations appear to be in immediate danger of extirpation in the near future. For most states, 2016 was the most recent peak in sage-grouse populations, and comparing this peak to the previous (2007) peak year, sage-grouse populations are continuing to decline. While we do not have comparable recent data for the state of Oregon, Oregon Department of Fish and Wildlife data are consistent with this continuing downward trend:

A population persistence study by Garton et al. (2015) incorporates the latest state population data to calculate the probability that various populations will drop below minimum viable population thresholds at the Management Zone and subpopulation levels. See Attachment I. According to this study, the prospects for sage grouse populations were even bleaker in 2015 than in 2010, when the species was found to be ‘warranted, but precluded’ for Endangered Species Act listing. This study characterizes the likelihood of the Northern Great Basin Management Zone falling below an effective population of 50 breeding birds as “very likely” at 72.2% in 100 years. According to this study, “The Western Great Basin population [shared between northeast California, northwestern Nevada, and southeastern Oregon] has declined by 69% over the last 6 years and appears to be experiencing an extinction vortex.” For the Northwest-Interior Nevada population population, “Parametric bootstraps imply that the minimum count of males has a 100% (SE 0%) chance of declining below 20 males in 30 years.” The Southern Great Basin management zone has a more optimistic outlook but still faces a substantial likelihood of functional extinction (25.3%) at the 100-year timeframe.

B. Large Fires Have Occurred

According to the National Interagency Fire Center, over 2 million acres of sage-grouse habitats burned in 2017, with 626,268 acres burning in 2016, and 562,734 acres burning in 2015 (NIFC 2017, Attachment J). This is significant new information that was not considered under the previous RMP Amendment process. The Forest Service must carefully consider the significant losses in sage-grouse habitats that have occurred since the plan amendments were put in place, and factor in the role that these impacts might play, both directly and cumulatively, in sage-grouse population persistence and recovery under all alternatives, while accounting for any changes in sage-grouse habitat protections.

Large fires of high frequency can extirpate sage grouse populations (Pedersen et al. 2003). A landscape mosaic of burns may not meet the nesting habitat needs of sage grouse (Nelle et al. 2000), and may also fail to meet grouse habitat requirements during other seasons (Wamboldt et al 2002). Fire was an uncommon occurrence in sagebrush habitats in presettlement times, with natural fire return intervals in Wyoming big sagebrush average 100-240 years (Baker 2007). Wyoming big sagebrush recovers slowly after fires, which typically result in 100% sagebrush mortality; recovery to pre-fire canopy cover takes over 100 years (Cooper et al. 2007). Baker (2007) examined the same issue and projected that Wyoming big sagebrush recovery following fire ranges from 50 – 120 years; for mountain big sagebrush, the recovery period was estimated at 35 – 100 years.

The Forest Service must take the legally required 'hard look' at effectiveness of proposed mitigation measures and its impact analysis must account for the primacy of cheatgrass invasion in determining patterns of rangeland fire; "The positive feedback loop between fire and invasive plant species may be the greatest impact on fire management and GRSG (Abatzoglou and Kolden 2011)." Nevada – Northeastern California Greater Sage Grouse RMP Amendment DEIS at 701. BLM further elucidates, In Oregon 19th and early 20th century grazing practices, along with introduction and spread of invasive plant species and the practice of fire suppression in the 20th century, have all contributed to fire suppression and to increasingly destructive wildfires. Oregon Greater Sage Grouse RMP Amendment DEIS at 4-10.

In the absence of cheatgrass, Wyoming big sagebrush sites can take 150 years to recover. Nevada – Northeast California Greater Sage Grouse RMP Amendment DEIS at 608. When cheatgrass is present, it can take over following disturbance, forming a monoculture characterized by unnaturally frequent fire return intervals that can effectively prevent the recovery of sagebrush and perennial grasses on a long-term if not permanent basis. In Oregon, "In Wyoming big sagebrush sites, full recovery to pre-burn sagebrush canopy cover conditions will take over 100 years (Cooper 2007);…." Oregon Greater Sage Grouse RMP Amendment DEIS at 3-70. More generally, BLM states, "Sagebrush recovers slowly from fire; most species do not resprout but must be replenished by wind-dispersed seed from adjacent unburned stands or seeds in the soil. Depending on the species and the size of a burn, sagebrush can re-establish itself within five years, but a return to a full pre-burn community cover can take 50 to over 100 years (Baker 2011)." Oregon Greater Sage Grouse RMP Amendment DEIS at 4-10.

For these reasons, the Forest Service must incorporate science-based measures to reduce the spread of cheatgrass, including rest from livestock grazing, into any future sage-grouse plan amendments, and must also rest burned areas for two years or more from livestock grazing, to allow native perennial grasses to recover and to reduce the distribution of weed seeds on newly burned areas.

C. Captive Rearing and Translocations Should Not be a Part of Forest Service Plan Direction.

The Forest Service should **not** incorporate captive breeding/rearing programs for sage grouse, or translocation of sage-grouse, into its conservation plans. These methods are known to fail and provide added stress to wild populations from human-caused disturbance related to raiding nests or brood for eggs or chicks. Of 56 translocation attempts for greater sage-grouse, Reese and Connelly (1997) found only a handful that were even marginally successful, and even these failed to support population increases for the populations receiving them. We are concerned that attempts to gather eggs or chicks from wild grouse will have a negative effect on ness and brood success for affected hens, and for the eggs or chicks taken, survival and success probabilities in the wild are far greater than the prospects for those same eggs and chicks in captivity. Translocated grouse are differentially susceptible to mortality compared to grouse left alone in the wild. For these reasons, capture and translocation and/or captive rearing represents a net loss of sage-grouse, and should not be incorporated into Forest Service plan revisions or amendments.

VI. ANY REVISIONS MUST STRENGTHEN THE EXISTING PLANS

A. Increase acreage of withdrawn lands

The current Greater Sage-Grouse RMP Amendments and Revisions incorporate insufficient Priority Habitat Management Area designations in all states except Oregon, Colorado, and North Dakota. All lands designated as Priority Areas for Conservation ("PACs") by the U.S. Fish and Wildlife Service need to be designated as Priority Habitat Management Areas and given strong, science-based protections in accord with the recommendations of the National Technical Team. In addition, expansions of PHMA are warranted in Wyoming, where reductions in state Core Area designations were made for political, rather than scientific, proposes, and which render this state's Priority Habitat Management Areas scientifically invalid.

In South Dakota, potentially occupied sage grouse habitats in the southwest corner of the state were excluded entirely from the South Dakota RMP provisions for sage-grouse. For this state, the plans should be expanded to designate habitat for the recovery of sage-grouse south of the Black Hills, including the designation of Priority Habitats in occupied portions of this range (*see* Molvar 2015, Attachment M, Figure 7). Given the very small and tenuous state of the South Dakota population (a maximum lek count of 228 males in 2017), the habitat that gets the strongest protections should be maximized in this state.

In Nevada, fully 47% of the PACs designated by the U.S. Fish and Wildlife Service were excluded from PHMA designations under the Nevada-Northeastern California ARMPA (Molvar 2015). This means that 10.5 million acres of land designated as priority habitats by the Service was not protected as such under the plans. This is unacceptable and scientifically unjustifiable. The remaining PHMA designations are tiny, fragmented, and often isolated, in striking contrast to the scientific understanding that sage-grouse area a landscape species that require large, unfragmented tracts of high-quality sagebrush habitats to survive and recover. It is also known that industrial uses cause habitat abandonment and population decreases in an area extending at least 1.9 miles from the edge of an industrial site (*see, e.g.,* Holloran 2005). Nevada's PHMA fragments fall largely within a 1.9-mile zone of influence for industrial projects that would be allowed to be sited along the PHMA boundary under the plans (and indeed, fluid minerals leasing of PHMA lands under No Surface Occupancy stipulations encourages fluid minerals wellsites to be located immediately adjacent to PHMAs to maximize recovery and minimize drilling costs of directional drilling, which is the only allowable method to recover fluid minerals from an NSO lease absent waivers or exceptions to the NSO standard).

In California, fully 70% of PACs designated by the U.S. Fish and Wildlife Service were excluded from PHMA designations under the Nevada-Northeastern California ARMPA (Molvar 2015), with only 0.4 million acres designated as PHMAs, and 1.3 million acres of PACs left undesignated and falling within the much less-protected designation of GHMA. California has a very small and imperiled sage-grouse population, and in light of this tenuous population viability, the Forest Service should be working to maximize, not minimize, the extent of habitat that is designated with the highest level of protection.

In Utah, only 5.5 million acres of PHMA were designated for the entire state, a very small total, leaving 2 million acres of designated PACs unprotected by the Utah ARMPA (Molvar 2015). The Utah PHMA designations tend to be narrow, small, and often isolated, increasing the likelihood of population extirpation. As noted above for Nevada, the designation of small or narrow PHMAs leaves the designated habitats vulnerable to disturbance from industrial activities approved along the boundary of PHMA in less-protected or unprotected habitats. In many cases, Utah sage-grouse habitats consist of valley-bottom sagebrush habitats bordered by hillsides clad in pinyon-juniper woodlands or other coniferous forest that are not habitat for sage-grouse. The Forest Service should nonetheless expand PHMAs in Utah to encompass all lands within 5.3 miles of active sage-grouse leks, regardless of whether the encompassed lands are in fact grouse habitats, because industrial developments in non-habitat woodlands will absolutely have negative impacts that extend for miles into surrounding habitats, including those that are sage-grouse habitats. In order to create a scientifically defensible and legally robust system of PHMAs, the Forest Service must in many cases provide levels of protection that extend beyond the limits of sagebrush itself.

In Wyoming, important sage-grouse habitats that were pristine and should have been designated as Core Areas were omitted from Core Area designation through the collaborative state process in 2008. This was done because the oil industry representatives on the Sage Grouse Implementation Team blackmailed other team members, threatening to block the adoption of any sage grouse plans unless undeveloped sage-grouse habitats with abundant populations were excluded from Core Areas so that future drilling could proceed unimpeded by wildlife habitat conservation measures. Excluded from Core Areas during this process were parts of the Atlantic Rim, Jonah, and Pinedale Anticline oil and gas project areas that remained undeveloped at the time, and significant acreages of important habitats in the Powder River Basin, where a coalbed methane play was in process at the time (*see* Molvar 2015, Figure 4). These excluded lands should be added to PHMA under the federal plans moving forward. Then, in 2010, Core Area boundaries were further gerrymandered to excluded Core Area lands previously designated that were desired for industrial exploitation by the wind industry (notably for the Chokecherry - Sierra Madre Wind Farm, as well as the DKRW coal-to-liquids plant and the Whirlwind LLC White Mountain wind farm, projects never built and subsequently abandoned). All lands eliminated from Core Area designation during the 2010 State of Wyoming boundary revision (*see* Molvar 2015, Figure 5) should be reinstated as PHMA through this federal process. For the Wyoming Basin population, which encompasses the rest of the state and is the most populous sage grouse population remaining worldwide, has a chance of dropping below an effective population of 50 of 4.7% in 30 years and 21% in 100 years (Garton et al. 2015).

In addition, of the Wyoming RMP provisions for sage grouse, the Buffalo Revised RMP stands out as requiring additional increases in PHMA designations above and beyond those listed above. According to Garton et al. (2015, Attachment N), the Powder River population (all of northeast Wyoming including Thunder Basin National Grassland, parts of Casper Field Office, and Newcastle Field Office) has a 98.7% chance of dropping below an effective population size of 50 in 30 years, with a 55% chance of sage grouse populations across the Great Plains (Management Zone I) dropping below 50 in 100 years. An effective population size of 50 is deep in the "extinction vortex." We are particularly concerned that the likely loss of this population through inadequate habitat protections and concomitant industrial development, along with the likely loss of the North and South Dakota populations due to intrinsic small size and vulnerability, will result in the isolation and ultimate extirpation of sage-grouse throughout the

Great Plains ecosystem. In its initial designation of Core Areas, the State of Wyoming made some major errors in the Buffalo Field Office that have been implicated in subsequent population declines and threats to long-term viability for sage grouse populations (*see* Taylor et al. 2012). It is important to note that many of the most populous sage grouse leks in the Buffalo Field Office lie outside Core Area boundaries. *See* Attachments O, P, Q (Buffalo FEIS 32, 33, 36). In particular, the Buffalo Core Area was not designated based on 4-mile or 5.3-mile buffers around the most populous leks, as were most Core Areas designated under the State of Wyoming Executive Orders. As a result, much of the nesting habitat within 5.3 miles of the occupied leks inside the Buffalo Core Area are found outside the Core Area on lands that are slated to receive minimal protections. In Attachment P (Buffalo FEIS 34), a screenshot of a presentation by WGFD, the Buffalo Core Area is delineated at the left side of the screen, with a rectilinear boundary following jagged land ownership. The lek sites, shown with magenta dots with numbers for 2014 maximum male counts, are located inside the core area (infilled with pale green), while a significant amount of the nesting habitat for the most populous leks inside the Core Area (shown as underlying red circles) extend outside the Core Area into unprotected lands. In addition, most of the occupied lek sites themselves lie along the boundary of the Core Area or within 1.9 miles (the distance at which producing well sites have a significant negative impact on lek populations, Holloran 2005), and as a result industrial development has and will continue to drive these leks near the Core Area boundary (FEIS at Map 37) toward extirpation. BLM should also designate a new Core Area along the Powder River (*see* Attachment P) to address the inadequate spatial extent of Core Areas in the Buffalo Field Office. In Attachment P (Buffalo FEIS 34), the proposed new Core Area is outlined in purple at the center of the image, and many of the lek sites in this potential Core Area (shown as magenta dots) have relatively high lek counts (the numbers accompanying the lek symbols), and in addition the underlying red circular buffers indicate the location of nesting habitat that represents part of the densest 25% of the state sage grouse population. This recommendation was submitted to the State of Wyoming (but not recommended for adoption by the state) in 2015. This designation would address the need to designate key sage grouse habitats encompassing some of the most densely populated sage grouse habitats in the Powder River sage grouse population area, which were excluded from Core Area designations in 2008 contrary to the best available science in an act of state obeisance to the coalbed methane industry. To remedy these errors, the Forest Service must designate additional PHMA to include the Core Areas denoted above, and ensure that all lands within 5.3 miles of a Core Area lek also fall entirely within Core Area PHMAs.

In Idaho, only 62% of PACs designated by the Service were given the status of PHMAs under the Idaho – Southwest Montana ARMPA, omitting 3.8 million acres of prime sage-grouse habitats from the level of protection they deserved (Molvar 2015). Some of these excluded lands were designed as Important Habitat Management Areas and granted a weaker level of protection that is inadequate based on the best available science. All PACs in Idaho must be designated as PHMAs and given a level of protection equal to the NTT (2011) recommendations.

In all, the boundaries of PHMAs rangewide should be reset to incorporate all lands within 5.3 miles of the most populous remaining sage-grouse leks mapped by Doherty et al. (2010), as delineated in Attachment L.

B. Strengthen density criteria

Any revisions to the plans should require the regulation of disturbance density and wellpad density on a per-square-mile-section basis. All scientific studies that have tested wellpad density and/or disturbance percentage per square mile have measured these densities per square-mile section or another relatively small area (Knick et al. 2013 used a 3-mile buffer around leks to measure disturbance density). Not one scientific study has ever examined the threshold of significant impacts when densities of wells or surface disturbance are calculated across an area of scores or hundreds of square miles, but presumably, the threshold of significant impacts would be different from the threshold that results from testing densities across a single square mile. Some calculation protocols, such as Wyoming's Disturbance Density Calculation Tool ("DDCT," Wyoming ARMPA at 34), calculate these densities on the basis of land areas much greater than one square mile, often hundreds of square miles. In the case of tightly-packed projects, this results in densities of disturbance and/or wellpads that far exceeds the scientifically determined thresholds of one pad per square mile and 3% per section at which significant negative impacts to sage grouse populations are known to occur.

The amount of cumulative disturbance allowed in sage-grouse core habitat at the project analysis area scale is calculated by an algorithm known as the Density Disturbance Calculation Tool ("DDCT"). The DDCT is used to establish an area for measuring the amount of disturbance that may be allowed under a project proposal. The DDCT essentially buffers a proposed project area by 4 miles, identifies all occupied leks within this area and buffers them by 4 miles, and uses the combined area as the denominator to calculate the total land area from which to derive the total percentage of land that could be disturbed by the project. This results in well densities and percentage of surface disturbance that exceed the threshold of significant impact to sage grouse populations within individual project areas. In cases where the DDCT area/project analysis area is very large, more than one well or mine site is permitted to be developed in a given square mile as long as the surrounding Priority Habitat lands are relatively free from other development disturbance. This can result in a density of wellsites that exceeds science-based thresholds at which significant impacts to sage grouse inhabiting the habitat in question begin to occur.

In other states, similarly large areas are used to dilute the disturbance density calculation. In the Nevada – Northeastern California RMPA, the "Biologically Significant Unit" is the denominator for disturbance density calculations. Nevada – Northeast California RPA at 1-10 *and see* Figure 2-2.

The Lost Creek Uranium In Situ Recovery Project exemplifies how development can exceed disturbance and density limits under the DDCT. The 4,254-acre permit area is located inside a Core Area, and it intersects the 4-mile buffers of 15 sage-grouse leks.[8] The DDCT area for this project is 147,060 acres, almost 230 square miles. If this were a hypothetical oil and gas project with the same 147,060-acre DDCT area, 229 wells would be allowed in the 4,254-acre permit area, for a density of 34.4 wellsites per square mile within the permit area. Within the actual perimeter of development, wellsite density will exceed 50 wells per half-section, or 100 wellsites per square mile. This extreme density would destroy habitat function for sage-grouse locally, even though well density for the DDCT area would still be within the one well per square-mile limit in the Core Area strategies.

In the case of the Lost Creek project, the extra-large DDCT area allowed intense development within the permit area. The project expects to disturb (i.e., bulldoze) 345 acres, which, when combined with preexisting disturbance, amounts to less than one percent for the DDCT area, but when compared to the 4,254-acre permit area, would yield 8.1 percent disturbance, far above the limit in the state and federal Core Area strategies. Note that virtually all development in this project will be along the ore trend (shown in Attachment K), meaning that the actual density within the developed portion of the Permit Area will be much greater than 8.1%. The DDCT area for this project, by contrast, totals 147,060 acres (*see* Attachment K), yielding a percent disturbance of less than 1% when considering the existing and proposed disturbance. The 345-acre development area also violates the strategies' limitation on site density. The DDCT assumes individual development sites (like oil and gas wells) will only each affect 4-5 acres. But for this project, the state wildlife agency classified the entire 4,254-acre development area as a single "site," which, although it meets the one site per square mile requirement, will eliminate half of a square mile section of directly bulldozed land within the 4,254-acre project area where it is located, and certainly have deleterious effects on sage-grouse for miles around. The DDCT area for this project is so large that 229 oil and gas wells could be permitted within the six-square-mile project area without exceeding the putative one wellpad per square mile limit on site density. BLM Resource Management Plan direction must prevent this type of excessive development through scientifically sound calculation methods for site density and disturbance percentage.

Importantly, the NTT (2011) recommended that disturbance density be calculated per square-mile section, based on their review of the best available science. This is supported by subsequent scientific study by Knick et al. (2013), who found a limit of 3% development based on a 3-mile buffer around leks was the threshold beyond which sage grouse populations were rarely able to sustain themselves.

C. Remove grazing in SG habitats

Every Plan should be amended to allow for grazing permit retirement within sage-grouse habitats. Because LUP are the appropriate place to determine various land uses and to weigh the relative values of the lands at issue, consideration for and facilitation of permit retirement following permit relinquishment should be included in every amendment.The lands at issue are federal; the permits are federal. No states should interfere with voluntary relinquishment or permit retirement is that's the preference of the permittee.

CONCLUSION

In conclusion, the undersigned groups believe that any revisions being undertaken to facilitate, encourage, or expedite the development of sage-grouse habitats for extractive industries are contrary to law and the public interest. Whereas the current plans are not perfect, they already reflect serious compromise between the science and the agendas of the western states; continuing to reduce protections for this iconic species will not sustain the bird and will lead to ins continued imperilment. We urge the FS to consider only revisions and amendments that strengthen the plans and comport with the best available science.

Sincerely,

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References cited

Abatzoglou, J.T., and C.A. Kolden. 2011. Climate change and western U.S. deserts: Potential for increased wildfire and invasive grasses. Rangeland Ecol. Manage. 64: 471-478.

Aldridge, C. L. and R. M. Brigham. 2003. Distribution, status and abundance of Greater Sage-grouse, *Centrocercus urophasianus*, in Canada. Canadian Field-Natur. 117: 25-34.

Aldridge, C.L., and M.S. Boyce. 2007. Linking occurrence and fitness to persistence: Habitat-based approach for endangered greater sage-grouse. Ecol. Appl. 17:508-526.

Ambrose, S. 2015. Review of Greens Hollow Sound Study by Tetra Tech (2008), and Summary of Sound Level Measurements at Wildcat Knolls Lek, March 29-31, 2015. Unpublished report, 11 pp.

Ambrose, S., and C. Florian. 2014. Sound levels at greater sage-grouse leks, Pinedale Anticline Project Area, Wyoming, April 2013. Unpublished report prepared for the Wyoming Game and Fish Department, 133 pp. Available online at http://www.wy.blm.gov/jio-papo/papo/wildlife/reports/sage-grouse/2013GSGacoustic-rpt.pdf.

Ambrose, S., C. Florian, and J. MacDonald. 2014. Sound levels at greater sage-grouse leks in the Pinedale Anticline Project Area, WY, April 2013-2014. Unpublished report prepared for the Wyoming Game and Fish Department, 79 pp.

Anderson, Jay E. and Rishard S. Inouye. 2001. Landscape-Scale Changes in Plant Species Abundance and Biodiversity of a Sagebrush Steppe Over 45 Years. Ecological Monograaphs, 71(4), 2001, pp. 531-556.

Apa, T., J. Bohne, T. Christiansen, J. Herbert, B. James, R. Northrup, D. Olsen, A. Robinson, P. Schnurr, T.O. Smith, and B. Walker. 2008. Using the Best Available Science to Coordinate Conservation Actions that Benefit Greater Sage-grouse Across States Affected by Oil & Gas Development in Management

Zones I-II (Colorado, Montana, North Dakota, South Dakota, Utah, and Wyoming). Unpublished multi-state report of game and fish agencies, 10 pp. Online at http://www.ourpubliclands.org/files/upload/ti-State_ScienceGroupDocument_FINAL_01-28-08.pdf.

Arkle, R. S., Pilliod, D. S., Hanser, S. E., Brooks, M. L., Chambers, J. C.,Grace, J. B., Knutson, K. C., Pyke, D. A., Welty, J. L. and Wirth, T. A. 2014. Quantifying restoration effectiveness using multi-scale habitat models: implications for sage-grouse in the Great Basin. Ecosphere 5(3): 31. http://dx.doi.org/10.1890/ES13-00278.1

Arnold, J. F. 1950. Changes in ponderosa pine- bunchgrass in northern Arizona resulting from pine regeneration and grazing. Journal of Forestry 48:118-126;

Autenrieth, R.E. 1981. Sage-grouse management in Idaho. Id. Dept. Fish and Game Wildl. Bull. 9.

Baker, W.L. 2006. Fire and restoration of sagebrush ecosystems. Wildl. Soc. Bull. 34: 177-185.

Balch, J.K., B.A. Bradley, C.M. D'Antonio, and J. Gomez-Dans. 2013. Introduced annual grass increases regional fire activity across the arid western USA (1980-2009) Global Change Biology 19: 173-183

Batchelor, J.L., W.J. Ripple, T.M Wilson, L.E. Painter. 2015. Restoration of riparian areas following the removal of cattle in northwestern Great Basin. Environmental Management. DOI 10.1007/s00267-014-0436-2

Beck, J.L., and D. Mitchell. 1997. Brief guidelines for maintaining and enhancing sage grouse habitat on private lands in Utah. Utah Division of Wildlife Resources, April 16, 1997. http://wildlife.utah.gov/uplandgame/sage-grouse/utguidef.htm, site last visited 5/6/08.

Beck, J. L. and D. L. Mitchell. 2000. Influences of livestock grazing on sage grouse habitat. Wildl. Soc. Bull. 28(4): 993-1002. Barnett, J. F. and J. A. Crawford. 1994. Pre-laying nutrition of sage-grouse hens in Oregon. J. Range Manage. 47: 114-118.

Beck, J.L., J.G. Klein, J. Wright, and K.P. Wolfley. 2011. Potential and pitfalls of prescribed burning big sagebrush habitat to enhance nesting and early brood-rearing habitats for greater sage-grouse. Nat. Res. Envtl. Issues 16:39.

Beck, J.L., J.W. Connelly, and C.L. Wamboldt. 2012. Consequences of treating Wyoming big sagebrush to enhance wildlife habitats. Rangeland Ecol. Manage. 65:444-455.

Bell, C. B. 2011. Nest Site Characteristics and Nest Success of Translocated and Resident Greater Sage Grouse at Clear Lake National Wildlife Refuge. MSc. Thesis. Humboldt State University. 46 pp. including app.

Belsky, J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54:419-431.

Belsky, J., and J.L. Gelbard. 2000. Livestock Grazing and Weed Invasions in the Arid West. Oregon National Desert Association, Bend, OR. 1-31.; Billings, W. D. 1990. *Bromus tectorum*, a biotic cause of ecosystem impoverishment in the Great Basin. Pages 301-322 in G. M. Woodwell, editor. The earth in transition: patterns and processes of biotic impoverishment. Cambridge University Press New York;

Benson, Lee A.; Braun, Clait E.; Leininger, Wayne C. 1991. Sage-grouse response to burning in the big sagebrush type. In: Comer, Robert D.; Davis, Peter R.; Foster, Susan Q.; [and others], eds. Issues and technology in the management of impacted wildlife: Proceedings of a national symposium; 1991 April 8-4. Snowmass Resort, CO. Boulder, CO: Thorne Ecological Institute: 97-104.

Beschta, R. L., DellaSala, D. A., Donahue, D. L., Rhodes, J. J., Karr, J. R. O'Brien, M. H., Fleischner, T. L. and Deacon-Willams, C. 2012. Adapting to climate change on western public lands: addressing the impacts of domestic, wild and feral ungulates. Environmental Management, DOI 10.1007/s00267-012-9964-9

Beschta, R. L., Donahue, D. L., DellaSala, D. A., Rhodes, J. J., Karr, J. R., O'Brien, M. H., Fleischner, T. L. and Williams, C. D. 2014. Reducing Livestock Effects on Public Lands in the Western United States as the Climate Changes: A Reply to Svejcar et al. Environmental Management, 53(6): 1039-1042. Online at: http://www.thewildlifenews.com/wp-content/uploads/2014/07/WWP-Sagegrouse-and-livestock-report.pdf

Beymer, R. J., and J. M. Klopatek. 1992. Effects of Grazing on Biological soil Crusts in Pinyon-Juniper Woodlands in Grand Canyon National Park. American Midland Naturalist 127:139-148;

Billings, W. D. 1994. Ecological impacts of cheatgrass and resultant fire on ecosystems in the western Great Basin. Pp. 170-175 in Monsen, S. B. and S. G. Kitchen (compilers), Proceedings – Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. US Department of Agriculture, Forest Services, Intermountain Research Station, Ogden, UT;

Blackburn, W. H., R. W. Knight, and M. K. Wood. 1982. Impacts of grazing on watersheds: a state of knowledge. Texas Agricultural Experiment Station, Texas A&M University, College Station, Texas

Blickley, J. L. and Patricelli, G. L. 2010. Impacts of Anthropogenic Noise on Wildlife: Research Priorities for the Development of Standards and Mitigation. Journal of International Wildlife Law & Policy, 13: 4, 274 — 292. http://dx.doi.org/10.1080/13880292.2010.524564.

Blickley, J.L., and G.L. Patricelli. 2012. Potential acoustic masking of greater sage-grouse (*Centrocercus urophasianus*) display components by chronic industrial noise. Ornith. Monogr. 74: 23-35.

Blickley, J.L., D. Blackwood, and G.L. Patricelli. 2012a. Experimental Evidence for the Effects of Chronic Anthropogenic Noise on Abundance of Greater Sage-Grouse at Leks. Conserv. Biol. 26:461-471.

Blickley J.L., Word K.R., Krakauer A.H., Phillips J.L., Sells S.N., et al. 2012b. Experimental Chronic Noise Is Related to Elevated Fecal Corticosteroid Metabolites in Lekking Male Greater Sage-Grouse (*Centrocercus urophasianus*). PLoS ONE 7(11): e50462. doi:10.1371/journal.pone.0050462.

(BLM) Bureau of Land Management. 2000. Record of Decision Environmental Impact Statement Continental Divide/Wamsutter II Natural Gas Project, Sweetwater and Carbon Counties, Wyoming. Bureau of Land Management, Rawlins and Rock Springs Field Offices. Rawlins and Rock Springs, WY.

BLM. 2005. Atlantic Rim Natural Gas Field Development Project Draft EIS. Rawlins Field Office.

Braun, C.E. 1986. Changes in sage grouse lek counts with advent of surface coal mining. Proc. Issues and Technology in the Management of Impacted Western Wildlife, Thorne Ecol. Inst. 2:227-231.

Braun, C.E. 2006. A blueprint for sage-grouse conservation and recovery. Tucson, AZ: Grouse, Inc., 20 pp.

Braun, C.E., O.O. Oedekoven, and C.L. Aldridge. 2002. Oil and gas development in western North America: effects on sagebrush steppe avifauna with particular emphasis on sage grouse. In Transactions North American Wildlife and Natural Resources Conference 67:337-349.

Brooks, M.L. and Berry, K.H. 2006. Dominance and environmental correlates of alien annual plants in the Mojave Desert, USA. Journal of Arid Environments 67.

Brotherson, J. D., and S. R. Rushforth. 1983. Influence of Cryptogramic Crusts on Moisture Relationships of Soils in Navajo National Monument, Arizona. Great Basin Naturalist 43:73-78.

Bukowsi, B.E. and W.L. Baker. 2013. Historical fire in sagebrush landscapes of trhe Gunnison sage-grouse range from land-survey records. J. Arid Env. 98:1-9.

Call, M. W. and C. Maser. 1985. Wildlife habitats in managed rangelands – the Great Basin of southeastern Oregon: sage grouse. Gen. Tech. Rep. PNW-187. U.S. Forest Service, Pacific Northwest Forest and Range Exp. Stn. Portland, OR.

Call, Mayo W. 1979. Habitat requirements and management recommendations for sage-grouse. Denver, CO: U.S. Department of the Interior, Bureau of Land Management, Denver Service Center. 37 p.

Carpenter, J., C. Aldridge, and M.S. Boyce. 2010. Sage-grouse habitat selection during winter in Alberta. Journal of Wildlife Management 74:1806-1814.

Coates, P. S. 2007. Greater Sage-grouse (*Centrocercus urophasianus*) nest predation and incubation behavior. Ph.D. Diss. Idaho State Univ. Pocatello, ID.

Coates, P. S., Connelly, J. W. and Delehanty, D. J. 2008. Predators of Greater Sage Grouse nests identified by video monitoring. Journal of Field Ornithology, 79: 421-428.

Coates P. S., B. E. Brussee, K. B. Howe, K. B. Gustafson, M. L. Casazza, and D. J. Delehanty. 2016. Landscape Characteristics and livestock presence influence common ravens: relevance to greater sage-grouse conservation. Ecosphere 7(2): e01203. 10.1002/ecs2.1203

Coggins, K. A. 1998. Relationship between habitat changes and productivity of sage grouse at Hart Mountain National Antelope Refuge, Oregon. M.S. thesis. Oregon State University. Corvallis, OR.

Connelly, J. 2013. Federal agency responses to Greater sage-grouse and the ESA: Getting nowhere fast. Northwest Science 88(1):61-64. http://dx.doi.org/10.3955/046.088.0113

Connelly, J. W. and C. E. Braun. 1997. Long-term changes in sage-grouse Centrocercus urophasianus populations in Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage-grouse populations and their habitats. Wildlife Society Bulletin 28(4): 967-985.

Cooper, S.V., P. Lesica, and G. M. Kudray. 2007. Post-fire recovery of Wyoming big sagebrush shrub-steppe in central and southeast Montana. Report to the United States Department of the Interior, Bureau of Land Management, State Office. Montana Natural Heritage Program, Helena, Montana. 16 pp. plus appendices. Online at http://mtnhp.org/reports/Sage_Succ_Veg.pdf.

Copeland, H.E., A. Pocewicz, D.E. Naugle, T. Griffiths, D. Keinath, J. Evans, and J. Platt. 2013. Measuring the effectiveness of conservation: A novel framework to quantify benefits of sage-grouse conservation policy and easements in Wyoming. PlosONE 8: e67261. 14 pp.

(COT) Conservation Objectives Team, Abele, S., Budd, R., Budeau, D., Connelly, J., Deibert, P.A., Delevan, J., Espinosa, S., Gardner, S.C., Griffin, K., Harja, J., Northrup, R., Robinson, A., Schroeder, M., and Souza, P. 2013. Sage-grouse conservation objectives report: Denver, Colo., U.S. Fish and Wildlife Service, 62 p., appendix, available at *http://www.fws.gov/mountain-prairie/species/birds/sagegrouse/.*

Dinkins, J. B., 2013. Common raven density and greater sage-grouse nesting success in southern Wyoming: Potential conservation and management implications. PhD Dissertation, Utah State Univ. http://digitalcommons.usu.edu/etd/1700.

Dinkins, J. B., Conover, M. R. and Mabray, S. T. 2013. Do artificial nests simulate nest success of greater sage-grouse? Human-Wildlife Interactions, 7(2): 299-312.

Dinkins, J. B., Conover, M. R., Kirol, C. P. and Beck, J. L. 2012. Greater Sage-Grouse (*Centrocercus urophasianus*) Select Nest Sites and Brood Sites Away from Avian Predators. The Auk, 129(4): 600-610. doi: 10.1525/auk.2012.12009

Doherty, K.E., D.E. Naugle, H. Copeland, A. Pocewicz, and J. M. Kiesecker. 2011. Energy development and conservation tradeoffs: Systematic planning for sage-grouse in their eastern range. Pages 505-516 in S.T. Knick and J. W. Connelly, editors, Greater Sage-Grouse-Ecology and Conservation of a Landscape

Species and Its Habitats. Studies in Avian Biology No. 38. Cooper Ornithological Society. University of California Press. Berkeley and Los Angeles, CA.

Doherty, K.E., D.E. Naugle, J.D. Tack, B.L. Walker, J.M. Graham, and J.L Beck. 2014. Linking conservation actions to demography: Grass height explains variation in greater sage-grouse nest survival. Wildlife Biology 20:320-325.

Dusek, R. J., Hagen, C. A., Franson, J. C., Budeau, D. A., and Hofmeister, E. K. 2014. Utilizing hunter harvest effort to survey for wildlife disease: A case study of West Nile virus in greater sage□grouse. Wildlife Society Bulletin. 20 AUG 2014. DOI: 10.1002/wsb.472

Dzialak, M. R., C. V. Olson, S. M. Harju, S. L. Webb, and J. B. Winstead. 2012. Temporal and hierarchical spatial components of animal occurrence: conserving seasonal habitat for greater sage-grouse. Ecosphere 3:30.

Emmons, S. R. and C. E. Braun. 1984. Lek attendance of male sage-grouse. J. Wildl. Manage. 48:1023-1028.

Evans, R. A. 1988. Management of pinyon-juniper woodlands. US Department of Agriculture, Forest Service, Intermountain Research Station. 34 pp.

Foster, M.A., J.T. Ensign, W.N. Davis, and D.C. Tribby. 2014. Greater sage-grouse in the southeast Montana core area. Unpublished report, Montana Fish, Wildlife, and Parks, 108 pp.

Garton, E.O., A.G. Wells, J.A. Baumgardt, and J.W. Connelly. 2015. Greater sage-grouse population dynamics and probability of persistence. Final Report to Pew Charitable Trusts, 90 pp.

Gifford, G. F., and R.H. Hawkins. 1976. Grazing systems and watershed management: a look at the record. Journal of Soil and Water Conservation 31:281-283;

Gregg, M.A., J.A. Crawford, M.S. Drut, and A.K. DeLong. 1994. Vegetational cover and predation of sage grouse nests in Oregon. J. Wildl. Manage. 58:162-166.

Green, Adam et al., Investigating Impacts of Oil and Gas Development on Greater Sage-Grouse, Journal of Wildlife Management, doi: 10.1002/jwmg.21179 (2016).

Hagen, C.A., J.W. Connelly, and M.A. Schroeder. 2007. A meta-analysis of greater sage-grouse *Centrocercus urophasianus* nesting and brood-rearing habitats. Wildlife Biology 13:42–50.

Hamerstrom, Frederick; Hamerstrom, Frances. 1961. Status and problems of North American grouse. Wilson Bulletin. 73(3): 284-294.

Harju, S.M., M.R. Dzialak, R.C. Taylor, L.D. Hayden-Wing, and J.B. Winstead. 2010. Thresholds and time lags in the effects of energy development on greater sage-grouse populations. Journal of Wildlife Management 74: 437–448.

Heath, B.J., R. Straw, S.H. Anderson, and J. Lawson. 1997. Sage grouse productivity, survival, and seasonal habitat use near Farson, Wyoming. Unpublished completion report to the Wyoming Game and Fish Department.

Hess, J.E., and J.L. Beck. 2012. Burning and Mowing Wyoming Big Sagebrush: Do Treated Sites Meet Minimum Guidelines for Greater Sage-Grouse Breeding Habitats? Wildlife Society Bulletin 36: 85-93.

Hess, J. E. and Beck, J. L. 2014. Forb, Insect, and Soil Response to Burning and Mowing Wyoming Big Sagebrush in Greater Sage-Grouse Breeding Habitat. Environmental Management. DOI 10.1007/s00267-014-0246-6.

Holloran, M. J. 2005. Greater sage-grouse (*Centrocercus urophasianus*) population response to natural gas field development in western Wyoming. PhD Dissertation. University of Wyoming. Laramie, Wyoming.

Holloran, M. J. and S. H. Anderson. 2005. Spatial distribution of Greater Sagegrouse nests in relatively contiguous sagebrush habitats. Condor 107(4): 742-752.

Holloran, M., and K. Belinda. 2009. Bighorn Basin greater sage-grouse habitat effectiveness modeling. Unpublished completion report for Biodiversity Conservation Alliance, Laramie, WY, 24 pp.

Holloran, M.J., R. C. Kaiser, and W. A. Hubert. 2007. Population response of yearling greater sage-grouse to the infrastructure of natural gas fields in southwestern Wyoming. Completion report. Wyoming Cooperative Fish and Wildlife Research Unit, Laramie, WY, USA.

Howe, K. B., Coates, P. S. and Delehanty, D. J. 2014. Selection of anthropogenic features and vegetation characteristics by nesting Common Ravens in the sagebrush ecosystem. The Condor, 116(1): 35-49. DOI: http://dx.doi.org/10.1650/CONDOR-13-115-R2.1

Jankowski, M. D., Russell, R. E., Franson, J. C., Dusek, R. J., Hines, M. K., Gregg, M. and Hofmeister, E. K. 2014. Corticosterone Metabolite Concentrations in Greater sage-grouse are Positively Associated with the Presence of Cattle Grazing. Rangeland Ecology and Management. 67(3): 237-246. doi: http://dx.doi.org/10.2111/REM-D-13-00137.1

Johnson, W. M. 1956. The effect of grazing intensity on plant composition, vigor, and growth of pine-bunchgrass ranges in central Colorado. Ecology 37:790-798;

Jones, A. 2000. Effects of cattle grazing on North American arid ecosystems: a quantitative review. Western North American Naturalist 60:155-164.

Kaczor, N. 2008. Nesting and brood-rearing success and resource selection of greater sage-grouse in northwestern South Dakota. M.S. Thesis, South Dakota State Univ., 85 pp.

Kaczor, N. W., K. C. Jensen, R. W. Klaver, M. A. Rumble, K. M. Herman-Brunson, and C. C. Swanson. 2011. Nesting success and resource selection of greater sage-grouse. Pp. 107–118 *in* B. K. Sandercock, K. Martin, and G. Segelbacher (editors). Ecology, conservation, and management of grouse. Studies in Avian Biology (no. 39), University of California Press, Berkeley, CA.

Kindschy, Robert R. 1986. Rangeland vegetative succession—implications to wildlife. Rangelands. 8(4): 157-159.

Kirol, C.P. 2012. Quantifying habitat importance for greater sage-grouse (*Centrocercus urophasianus*) population persistence in an energy development landscape, M.S. Thesis, U. of Wyoming, 203 pp.

Klebenow, D. A. 1982. Livestock grazing interactions with sage grouse. Proc. Wildlife-Livestock Relations Symp. 10: 113-123.

Knick, S. T., D. S. Dobkin, J. T. Rotenberry, M. A. Schroeder, W. M. Vander Haegen, C. van Riper. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. Condor 105(4): 611-634.

Knick, S. T., A. L. Holmes, R. F. Miller. 2005. The role of fire in structuring sagebrush habitats and bird communities. FIRE AND AVIAN ECOLOGY IN NORTH AMERICA. Studies in Avian Biology, no. 30. Page 6. Cooper Ornithological Society. Boise, ID.

Knick, S. T., S. E. Hanser, R. F. Miller, D. A. Pyke, M. J. Wisdom, S. P. Finn, E. T. Rinkes, C. J. Henny. 2011. Ecological influence and pathways of land use in sagebrush. Pages 203-251 in S. T. Knick and J. W. Connelly (eds). GREATER SAGE-GROUSE: ECOLOGY AND CONSERVATION OF A LANDSCAPE SPECIES AND ITS HABITATS. Studies in Avian Biol. Series, vol. 38. Cooper Ornithological Society. Univ. Calif. Press.
Berkeley, CA.

Knick, S.T., S.E. Hanser, and K.L. Preston. 2013. Modeling ecological minimum requirements for distribution of greater sage-grouse leks – Implications for population connectivity across their western range, USA. Ecology and Evolution 3: 1539-1551.

Lammers, W.M., and M.W. Collopy. 2007. Effectiveness of avian predator perch deterrents on electric transmission lines. J. Wildl. Manage. 71: 2752-2758.

Lebbin, Daniel J.; Parr, Michael J.; and Fenwick, George H., The American Bird Conservancy Guide to Bird Conservation. The University of Chicago Press, 2010.

LeBeau, C.W. 2012. Evaluation of greater sage-grouse reproductive habitat and response to wind energy development in south-central Wyoming. M.S. Thesis, Univ. of Wyoming, 120 pp.

LeBeau, C.W., J.L. Beck, G.D. Johnson, and M.J. Holloran. 2014. Short-term impacts of wind energy development on greater sage-grouse fitness. J. Wildl. Manage. 78: 522-530.

Lusby, G. 1979. Effects of Grazing on Runoff and Sediment Yield from Desert Rangeland at Badger Wash in Western Colorado, 1953-73

Manier, D.J., C.L. Aldridge, P.J. Anderson, G. Chong, C.G. Homer, M. O'Donnell and S.J. Schell. 2011. Land use and habitat conditions across the southwestern Wyoming sagebrush steppe: development impacts, management effectiveness and the distribution of invasive plants. *Natural Resources and Environmental Issues* Vol. 17, Article 4. Available at: http://digitalcommons.usu.edu/nrei/vol17/iss1/4.

Manier, D.J., Wood, D.J.A., Bowen, Z.H., Donovan, R.M., Holloran, M.J., Juliusson, L.M., Mayne, K.S., Oyler-McCance, S.J., Quamen, F.R., Saher, D.J., and Titolo, A.J. 2013. Summary of science, activities, programs, and policies that influence the rangewide conservation of Greater Sage-Grouse (*Centrocercus urophasianus*): U.S. Geological Survey Open-File Report 2013–1098, 170 p., http://pubs.usgs.gov/of/2013/1098/.

Manier, D.J., Bowen, Z.H., Brooks, M.L., Casazza, M.L., Coates, P.S., Deibert, P.A., Hanser, S.E., and Johnson, D.H. 2014. Conservation buffer distance estimates for Greater Sage-Grouse—A review: U.S. Geological Survey Open-File Report 2014–1239, 14 p., *http://dx.doi.org/10.3133/ofr20141239*.

Mattise, Samuel N. 1995. Sage-grouse in Idaho: Forum 94'. Technical Bulletin No. 95-15. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office. 10 p.

Miller, R. F., S. T. Knick, D. A. Pyke, C. W. Meinke, S. E. Hanser, M. J. Wisdom, A. L. Hild. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. Pages 145-184 in S. T. Knick and J. W. Connelly (eds). Greater Sage-Grouse: Ecology and Conservation of a Landscape Species and its Habitants. Studies in Avian Biol. Series, vol. 38. Cooper Ornithological Society. Univ. Calif. Press. Berkeley, CA.;

Nelle, P. J., K. P. Reese, and J. W. Connelly. 2000. Long-term effects of fire on sage grouse habitat. Journal of Range Management 53:586–591.

Nonne, D., E. Blomberg, and J. Sedinger. 2011. Dynamics of greater sage-grouse (*Centrocercus urophasianus*) populations in response to transmission lines in central Nevada. Progress Report: Year 9. Unpublished report, 55 pp.

(NTT) Sage-grouse National Technical Team. 2011. A Report on National Greater Sage-grouse Conservation Measures. Available at www.blm.gov/pgdata/etc/medialib/blm/co/programs/wildlife.Par.73607.File.dat/GrSG%20Tech%20Team%20Report.pdf.

Opdam, O. and Wascher, D. 2004. Climate change meets habitat fragmentation: linking landscape and biogeographical scale levels in research and conservation. Biological Conservation, 117: 285-297. Online at: http://research.eeescience.utoledo.edu/lees/Teaching/EEES4760_07/Opdam.PDF

Patricelli, G.L., J.L. Blickley, and S.L. Hooper. 2012. The impacts of noise on greater sage-grouse: A discussion of current management strategies in Wyoming with recommendations for further research and interim protections. Unpubl. Report prepared for: The Bureau of Land Management, Lander Field Office and Wyoming State Office, Cheyenne and Wyoming Game and Fish Department, 25 pp.

Patterson, Robert L. 1952. The sage-grouse in Wyoming. Federal Aid to Wildlife Restoration Project 28-R. Denver, CO: Sage Books, Inc. 341 p.

Pederson, E. K., J. W. Connelly, J. R. Hendrickson, W. E. Grant. 2003. Effect of sheep grazing and fire on sage grouse populations in southeastern Idaho. Ecol. Model. 165(1): 23-47.;

Prather, P.R. 2010. Factors affecting Gunnison sage-grouse (*Centrocercus minimus*) conservation in San Juan County, Utah. PhD Dissertation, Utah State Univ., 134 pp.

Reese, K.P., and J.W. Connelly. 1997, Translocations of sage grouse *Centrocercus urophasianus* in North America. Wildl. Biol. 3: 235-241.

Reisner, M. D., Grace, J. B., Pyke, D. A. and Doescher, P. S. 2013. Conditions favouring Bromus tectorum dominance of endangered sagebrush steppe ecosystems. Journal of Applied Ecology.

Remington, T.E., and C.E. Braun. 1991. How surface coal mining affects sage grouse, North Park, Colorado. Proc. Issues and Technologies in the Management of Impacted Western Wildlife, Thorne Ecol. Inst. 5:128-132.

Rhodes, E.C., J.D. Bates, R.N. Sharp, and K.W. Davies. 2010. Fire effects on cover and dietary resources of sage-grouse habitat. J. Wildl. Manage. 74: 755-764.

Robinson. 2013. Management Plan and Conservation Strategies for Greater Sage-Grouse in North Dakota. North Dakota Game and Fish Department. Bismarck, ND, USA. Online at http://gf.nd.gov/specialty-publications/sage-grouse-management-plan; site last visited 9/26/13.

Rosentreter, R. 1994. Displacement of rare plants by exotic grasses. Pp. 170-175 in Monsen, S. B. and S. G. Kitchen (compilers), Proceedings – Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. US Department of Agriculture, Forest Services,Intermountain Research Station, Ogden, UT.

Rothenmaier, D. 1979. Sage-grouse reproductive ecology: breeding season movements, strutting ground attendance and site characteristics, and nesting. M.S. Thesis, Univ. Wyoming, Laramie.

Sage Grouse Initiative. 2014. Marking high-risk fences saves sage grouse. Science to Solutions Series Number 1. Sage Grouse Initiative. 4pp. http://www. sagegrouseinitiative.com/.

Schneegas, E. R. 1967. Sage-grouse and sagebrush control. Transactions, North American Wildlife Conference. 32: 270- 274.

Sedgwick, J.A. 2004. Habitat Restoration for Gunnison and Greater Sage-Grouse—A Literature Review. Report Prepared for the U.S. Department of Interior. Bureau of Land Management, Gunnison Field Office.

Sharp, A. L., J.J. Bond, J.W. Neuberanger, A.R. Kuhlman, and J.K. Lewis. 1964. Runoff as affected by intensity of grazing on rangeland. Journal of Soil and Water Conservation 19:103-106.

Smith, J.R., J.D. Tack, L.I. Buckley, M. Szczypinski, D.E. Naugle. 2017. Effects of rotational grazing management on nesting Greater sage-grouse. Journal of Wildlife Management, DOI: 10.1002/jwmg.21344

Stevens, B. S. 2011. Impacts of fences on greater Sage grouse in Idaho: collision, mitigation and spatial ecology. Master's Thesis, University of Idaho, Moscow.

Stevens, B.S., D.E. Naugle, B. Dennis, J.W. Connelly, T. Griffiths, and K.P. Reese. 2013. Mapping sage-grouse fence collision risk: Spatially explicit models for targeting conservation implementation. Wildl. Soc. Bull. 37: 409-415.

Stiver, S.J., E.T. Rinkes, D.E. Naugle, P.D. Makela, D.A. Nance, and J.W. Karl, eds. 2015. Sage-Grouse Habitat Assessment Framework: A Multiscale Assessment Tool. Technical Reference 6710-1. Bureau of Land Management and Western Association of Fish and Wildlife Agencies, Denver, Colorado.

Taylor, R. L., D. E. Naugle, L. S. Mills. 2010. Viability analysis for conservation of sage-grouse populations: Miles City Field Office, Montana. BLM Contract 09-3225-0012; Number G09AC00013. Final Report. Prepared for Bureau of Land Management, Miles City Field Office. Miles City, MT.

Taylor, R. L., D. E. Naugle, L. S. Mills. 2012. Viability analysis for conservation of sage-grouse populations: Buffalo Field Office, Wyoming. BLM Contract 09-3225-0012; Number G09AC00013. Final Report. Prepared for Bureau of Land Management, Buffalo Field Office. Buffalo, WY.

Vallentine, J. F. 1990. GRAZING MANAGEMENT. Academic Press. San Diego, CA.

Vollmer, J.L. 2005. New Technology for Fuel Breaks and Green Strips in Urban Interface and Wildland Areas. Modified Poster from Eighth International Wildland Fire Safety Summit, April 26-28, 2005

Missoula, MT, April, 2005. Online at
http://www.gbfiresci.org/storage/docs/additionalinfo/Vollmer_Brte_Plateau_1.pdf.

(WAFWA) Western Association of Fish and Wildlife Agencies. 2004. Conservation assessment of Greater Sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. Cheyenne, WY. (July 22, 2004).

WAFWA 2009. Prescribed Fire as a Management Tool in Xeric Sagebrush Ecosystems: Is it Worth the Risk to Sage-Grouse? June 2009. A White Paper prepared by the Sage and Columbian Sharp-tailed Grouse Technical Committee for the Western Association of Fish and Wildlife Agencies.

Wallestad, Richard; Pyrah, Duane. 1974. Movement and nesting of sage-grouse hens in central Montana. Journal of Wildlife Management. 38(4): 630-633.

Wallestad, R. 1975. Life history and habitat requirements of sage-grouse in central Montana. Helena, MT: Montana Department of Fish and Game. 65 p. In cooperation with: U.S. Department of the Interior, Bureau of Land Management.

Wallestad, R., Peterson, J. G. and Eng, R. L. 1975. Foods of adult sage-grouse in central Montana. Journal of Wildlife Management. 39(3): 628-630.

Wambolt, C. L., A. J. Harp, B. L. Welch, N. Shaw, J. W. Connelly, K. P. Reese, C. E. Braun, D. A. Klebenow, E. D. McArthur, J. G. Thompson, L. A. Torell, and J. A. Tanaka. 2002. Conservation of greater sage-grouse on public lands in the western U.S.: implications of recovery and management policies. Policy Analysis Center for Western Public Lands, Caldwell, Idaho, USA.

Wisdom, M. J., M. M. Rowland, R. J. Tausch. 2005. Effective management strategies for sagegrouse and sagebrush: a question of triage? Trans. N. Wildl. Nat. Res. Conf. 70: 206-227.

Wisdom, M.J., C.W. Meinke, S.T. Knick, and M.A. Schroeder. 2011. Factors associated with extirpation of sage-grouse. Studies in Avian Biology 38:451–472.

Yeo, J. 2005. Effects of grazing exclusions on rangeland vegetation and soils, east central Idaho. Western North American Naturalist 65(1), 2005, pp. 91–102

Western Watersheds Project

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Sage-grouse Amendment Comment
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COMMENTS ON THE DRAFT GREATER SAGE-GROUSE PROPOSED LAND MANAGEMENT PLAN AMENDMENTS AND DRAFT ENVIRONMENTAL IMPACT STATEMENT

The following comments on the draft Greater sage-grouse Proposed Land Management Plan Amendments (LMPAs) and the Draft Environmental Impact Statement (DEIS) are being submitted for consideration in response to the October 5, 2018 announcement of their availability and the commencement of the public comment period. These comments are being submitted on behalf of Western Watersheds Project, Prairie Hill Audubon Society, Sierra Club, Defenders of Wildlife, WildEarth Guardians, American Bird Conservancy and Center for Biological Diversity. We incorporate by reference all previous comments, protests, and litigation filings by Western Watersheds Project pertaining to any of USFS's Greater sage-grouse planning efforts, and the comments and attachments submitted during the two public scoping periods prior to the release of this DEIS, all attached here for your convenience.

In 2015, the U.S. Fish & Wildlife Service (FWS" or "the Service") decided not to list the Greater sage-grouse under the Endangered Species Act in light of the protections imposed through approved land-use plan amendments (ALMPAs) instituted by the National Sage-Grouse Planning Strategy. The ALMPAs were themselves insufficient to protect the grouse, and several of our organizations initiated legal action to persuade the agency to reconsider and strengthen the 2015 decision. Then, in 2017, the USFS began dismantling the ALMPAs. As our organizations insisted in our previous two scoping letters, any additional or revised sage-grouse planning must be strengthened to provide adequate, science-based protections for the birds. Unfortunately, the current proposed

LMPAs weaken, not strengthen, conservation measures for the greater sage-grouse and violate the federal laws that govern the public lands at issue here.

1. The proposed plan reduces the enforceability of the conservation measures.

Numerous provisions in the existing land use plan amendments are weakened in the proposed plans simply by reassigning the plan component titles to the same criteria. Where "Standards" are mandatory constraints, "Guidelines" are discretionary constraints and here, the USFS added the optional content of "Management Approaches," a more vague and visionary direction that is entirely unenforceable as a matter of law. 36 C.F.R. 219.15(d).

Thus, when the agency proposes, as it does dozens of times in the DEIS, to revise standards to guidelines and guidelines to management approaches, it is necessarily removing the mandatory requirements by which future projects must abide, i.e. weakening the enforceability and conformance requirements of all future actions. Therefore, the utility of these plans in protecting sage-grouse habitat is only as strong as the authorized officer in charge of making decisions, and the plans themselves provide less durability, certainty, and security for Greater sage-grouse and its habitat.

Many examples of this enforceability "gerrymandering" are found in the DEIS. One such example is the weakening of adaptive management triggers. In Nevada, the USFS proposes to revise GRSG-AM-ST-011-Standard – requiring immediate action if a population or habitat is in severe declines –to a "Management Approach" that requires only that action "needs to be considered." DEIS/DRMP at 2-84. The level of discretion implicit in this vague language in no way ensures that deviations from conservation objectives will be rectified or that declines in sage-grouse populations in Nevada will be meaningfully and promptly addressed. Similar dilutions are proposed in many other places in the proposed plans, undermining their efficacy in protecting sage-grouse.

Because the DEIS cannot account for the subjectivity inherent in these proposed changes, it also cannot analyze the proposed actions as if they will certainly protect the sage-grouse. The enforceability and meaningfulness of these restrictions is significantly weakened so as to wholly remove the safeguards that the 2015 plans incorporated. As affirmed by the Northern District of California, in *Desert Survivors v. U.S. Department of the Interior* (3:16-cv-01165-JCS; 2018), future conservation efforts must be "sufficiently certain" to be effective.

2. The proposed plan removes the Sagebrush Focal Area (SFA) designation, opening more land to fewer protections.

The 2015 Plans did not adequately identify and protect priority habitats. They identified sage-grouse habitat—in the process, reduced it by millions of acres from the 2013 Conservation Objectives Team (COT) Report Priority Areas for Conservation (PACs)—then divided it into three or more categories: Sagebrush Focal Areas (SFAs), Priority Habitat Management Areas (PHMAs), and General Habitat Management Areas (GHMAs). These categories are present in most Plans, while the Idaho and Southwestern Montana EIS includes Important Habitat Management Areas (IHMAs), the

Nevada and Northeastern California EIS includes Other Habitat Management Areas (OHMAs), and the Wyoming Plans identify "core" and "connectivity" PHMAs. The agencies did not include all key sage-grouse habitats within the priority habitat designations, including all PACs and winter habitats; or encompass all sage-grouse populations and subpopulations in priority habitats. Since they did not map or identify winter habitats, they also did not apply the protections the science recommended to these important habitats. In addition, they did not consider or adequately plan for connectivity between priority habitats, providing only downgraded protections to the few habitats (mostly GHMA) supposedly intended to ensure connectivity.

The DEIS states that the SFA were changed to the "appropriate HMA designation," but fails to identify how those designations were deemed more appropriate. DEIS at 2-22. There is no explanation for acres once considered to be crucial areas of sage-grouse habitat being downgraded, or any explanation of how many acres of previously-classified SFA are now anything lower than PHMA in the protective schema. A hard look at the details of these changes is warranted, and a site-specific rationale for any adjustments is required.

Now, the 2018 proposed plans seek to do away with SFA designations, reducing the protections afforded to these most significant (albeit inadequately and narrowly defined) habitats by combining them with PHMA, where waivers, exceptions and modifications to management restrictions are allowed.

The impact of removing SFA protections is inadequately analyzed in the 2018 DEIS. The DEIS takes these changes at a statewide level, but fails to assess the impacts of reducing protections in "strongholds," or areas that "have been noted and referenced as having the highest densities of greater sage-grouse and other criteria important for the persistence of the species." Great Basin ROD at 20. For example, in Nevada, where the proposed action would strip away SFA-level protections for over half a million acres (Table 201 and 202, DEIS at 2-23), the DEIS fails to really acknowledge and assess the fact that this undoes the SFA-level protections on all the northern habitats in the Humboldt-Toiyabe on the Jarbidge, Mountain City, and Santa Rosa ranger districts. Where this is framed as a fractional change statewide, it's clearly a major change for the sage-grouse habitat in these areas. Where the DEIS claims that this different management category is not substantially different ("the only difference is that PHMA allows for limited exception and the exceptions must meet a stringent set of criteria to be approved," DEIS at 4-219), the differences are significant from an enforceability standpoint. *See above.* Thus the actual differences in what could happen on the ground should the exceptions be granted is far different than the prohibition on surface disturbing activities that the SFA category provided.

3. The proposed plan changes the HMA boundaries in ways that reduce protections for sage-grouse.

The proposed plans eliminate large acreages of HMAs in Nevada, Wyoming, Idaho, and Utah.[1] DEIS at 4-216, 217.

In Nevada, PHMA acreage decreased by 99,500 acres and substantial acreage was moved from OHMA to GHMA. Table 2-1, DEIS at 2-23. PHMAs designated in the 2015 ARMPAs already were substantially reduced and fragmented from the 2013 COT Report Priority Areas for Conservation. The DEIS states this is due to current science, "including new lek locations, improved understanding of sage-grouse space-use from marked birds and modeling use, and removal of areas of non-habitat including areas near town and city centers." DEIS at 4-216. This cites to Coates et al. 2016, a reference not listed in the DEIS.[2] But the spatially explicit modeling done by Coates and others (2016) found *increases* in sage-grouse habitats across Nevada and northeastern California and decreased acreage of non-habitats. The proposed DEIS inexplicably trends the opposite direction without a sufficient site-specific analysis or rationale. GHMA are the least protected of the habitat types, so the reclassification of 298,800 acres into GHMA in Nevada reflects a serious downgrading of protections in the state.

The DEIS also says, "No impact to GRSG is anticipated from the HMA boundary adjustment." DEIS at 4-216. The levels of protections across PHMA, GHMA, and OHMA are different, and the impacts to GRSG in these habitats is necessarily variable. The conclusory "No impact" is disingenuous and undercuts the conservation pledged in the 2015 plans. Where HMA boundaries change by a great number of acres – e.g. 349,300 currently to 179,900 proposed on the Bridger-Teton NF (DEIS at 2-24) – a better explanation for why there is "no impact" is warranted. Are there no sage-grouse on the 170,000 acres no longer considered HMA on this forest?

The proposed plan for Idaho would remove the requirement to mitigate for impacts in GHMA. EIS at 4-220. This is justified by claiming that lacking a mitigation requirement would encourage development and surface disturbance outside of PHMA or IHMA. Ibid. The DEIS admits that this will increase the risk of habitat loss and displacement in GHMA. Ibid. GHMA's were defined in the 2015 ROD as, "NFS lands that are occupied seasonal or year-round habitat outside of PHMA where some special management would apply to sustain GRSG populations. The boundaries and management strategies for GHMAs are derived from and generally follow the Preliminary General Habitat boundaries." This is not unimportant. GHMA and PGH were two of the main categories of important

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[1] The DEIS says both that the proposed action includes adjustments to HMA boundaries in Idaho and that HMAs in Idaho remain the same. DEIS at 4-216. Which is it? The analysis neglects to describe or assess the changes in Idaho, and it is impossible to tell from Table 2-1 and Table 2-2 what changes are slated for Idaho. This must be reconciled in a Final EIS. Table 2-4 demonstrates that there are changes on Idaho forests, including, e.g. on the Sawtooth National Forest where HMA declined by nearly 100,000 acres. No explanation is provided in Chapter 4 of the DEIS of this change.

[2] We will herein assume that the agency means to cite to Coates, P.S., Casazza, M.L., Brussee B.E., Ricca, M.A., Gustafson, K.B., Sanchez-Chopitea, E., Mauch, K., Niell, L., Gardner, S., Espinosa, S., and Delehanty, D.J., 2016, Spatially explicit modeling of annual and seasonal habitat for greater sage-grouse (*Centrocercus urophasianus*) in Nevada and Northeastern California—An updated decision-support tool for management: U.S. Geological Survey Open-File Report 2016-1080, 160 p.,

lands to be protected; by dropping protective elements of the GHMA such as no longer requiring mitigation, the FS is now sacrificing the habitat it was supposed to be protecting. No analysis of how many leks or birds would be affected by this change is provided in the DEIS.

The reduction of acres in the HMA is attributed in part to new information. DEIS at 4-268. But the DEIS does not address how many of these boundary adjustments are due to habitat degradation that has been allowed to occur in recent years. That is, where the HMA boundaries were adjusted based on "Greater sage-grouse habitat use and distribution," (DEIS at 4-268), how many of those adjustments reflect declining populations and degrading habitats that the agency is now simply writing off for political expediency without an eye towards recovering the species.

The DEIS is unclear about the effects of changing HMAs to simply "management areas." DEIS at 2-21. There is very limited explanation of how this changes the primary administrative direction for the acreages at stake, and the impacts to sage-grouse habitat are uncertain. Forthcoming iterations of the planning documents should do a better job analyzing and disclosing the effects of such a change.

The DEIS also contains vague and unenforceable language about how often the HMA boundaries could change. In Idaho, GRSG-GEN-MA-004-Management Approach, a new provision, says that "Every five years of when a demonstrated need for change exists…" the agency will reevaluate the maps delineating HMA and Biologically Significant Units (BSUs). DEIS at 2-48. The DEIS does not identify what a "need for change" might consist of or how it would be demonstrated. It would appear that without a firm definition of those significant factors, the agency is leaving itself the discretion to adjust habitat areas on very short timeframes in response to rapid change, without accounting for sage-grouse recovery and restoration occurring on very long time scales. This creates a regulatory uncertainty in the plans and with any overarching prescriptions for HMA management.

4. The proposed plans weaken the oversight of discretionary decision-making by limiting the involvement of sage-grouse experts.

The proposed plans weaken the oversight of the implementation of the provisions of the plan by allowing much greater discretion and unilateral decision-making when it comes to offering modifications, exemptions and waivers. In Colorado, Idaho, Nevada, and Utah, a provision that required exceptions to the "No Surface Occupancy Stipulations" to achieve unanimous concurrence from "a team of agency GRSG experts from the U.S. Fish and Wildlife Service," was changed to "Exception could be granted by the authorized officer" without USFWS concurrence. DEIS at ES-7. In Nevada and Utah, the new proposed plans don't even extend the protective provisions of requiring the technical and policy teams to review the exemption. Ibid. This concentrates the power to a single person and more likely subjects the decision-making to personal agendas and/or political tampering rather than science.

Moreover, there is no analysis of the environmental impact of this significant change. Chapter 4 merely restates what the proposed action is, describing the details of what the removal of the requirement for a unanimous finding would mean. DEIS at 4-223. No analysis or discussion of how this could be influenced and subjectively distorted is provided, nor is any discussion provided of how this undermines the certainty of the plans themselves.

In Idaho, the NSO stipulations are allowed by exception "after review by the Technical and Policy Team" – without, notably, unanimous concurrence – by the authorized officer if the population trend for GRSG within the Conservation Area "is stable or increasing over a three-year period," among other criteria. DEIS at 2-70. It is not clear why the agency believes 3-years of stability or increase is sufficient in a cyclically-fluxing species like the sage-grouse, but even the DEIS contains evidence that this approach is insufficient to determine actual trends. For example, on page 3-193, Table 3-2 shows increases in sage-grouse populations (at a statewide level) over a three-year period from 2014-2016 for Idaho, Nevada, and Wyoming. In 2017, each of these states returned to pre-2015 numbers. Ibid. So, were the criterion of "increasing over a three-year period" applied, the authorized officer may have allowed an NSO exception, increasing the disturbance in sage-grouse habitats and contributing to the threats faced by the smaller population in Year 4. This proposed approach is not justified in the DEIS and no science supports decision-making based on this arbitrary timescale.

In Colorado, the FS guarantees only that exceptions to NSO stipulations could be grazing "with input" from a team of agency greater sage-grouse experts. DEIS at 2-40. "Input" doesn't equal "influence," and it's worth noting that this new "standard" requires nothing more than asking around for opinions but not necessarily following them. Regarding non-energy leasable minerals in Colorado, the agency even limits the types of recommendations it can make to BLM regarding mitigation, but specifically can no longer recommend avoidance. DEIS at 2-45.

By making certain criteria that were guidelines into "management approaches," the proposed DEIS weakens the protective methods for reducing the potential spread of West Nile virus. DEIS at 2-74. By moving the methods to accomplish this to management approaches, the likelihood of their implementation is decreased and the efficacy of their application is uncertain.

5. The proposed plan jettisons mandatory livestock grazing habitat standards, replacing them with subjective and uncertain management parameters.

From the beginning of the planning process, our organizations have been raising the issue that the livestock grazing guidelines and seasonal habitat standards have been inadequate to protect sage-grouse throughout the seasons, and we have asked numerous times for the agencies to follow the best science and secure habitat sufficiency for the bird. We were concerned that the 2015 plans contained enough vague language so as to make enforceability difficult, and we also asserted that the Forest Service should have imposed seasonal restrictions during breeding and brood-rearing along the lines of what the District of Idaho recommended: livestock grazing should be restricted in sage-grouse nesting and brood-rearing habitat to the "well established" timeframes necessary for adversely impacting sage-grouse – June 20 to August 1, and November 15 to March 1. *WWP v. Salazar*, 843 F. Supp. 2d 1105, 1123 (D. Idaho 2012). In addition, our previous comments have advocated for following the best science (Knick et al., 2005) and limiting livestock utilization to 30 percent. The Forest Service declined to adopt such scientifically-supported restrictions and instead relied on the

Habitat Guidelines of the 2015 plans to ensure that livestock weren't adversely affecting sage-grouse habitats. Now, the FS is proposing to do even less.

We have consistently reminded the agency of the recommendations of the National Technical Team (NTT) report, including, "Managing livestock grazing to maintain residual cover of herbaceous vegetation so as to reduce predation during nesting may be the most beneficial for sage-grouse populations (Beck and Mitchell 2000, Aldridge and Brigham 2003)…. Treatments used to manipulate vegetation ultimately may have far greater effect on sage-grouse through long-term habitat changes rather than direct impacts of grazing itself (Freilich et al. 2003, Knick et al. 2011)." NTT Report at 14.

Accordingly, it identified measures to benefit sage-grouse, including:

- "Within priority sage-grouse habitat, incorporate sage-grouse habitat objectives and management considerations into all BLM grazing allotments through AMPs or permit renewals";
- "Prioritize completion of land health assessments and processing grazing permits within priority sage-grouse habitat areas";
- "Manage riparian areas and wet meadows for proper functioning condition within priority sage-grouse habitats";
- "Only allow treatments that conserve, enhance or restore sage-grouse habitat (this includes treatments that benefit livestock as part of an AMP/Conservation Plan to improve sage-grouse habitat"; and
- "Maintain retirement of grazing privileges as an option in priority sage-grouse areas…."

Id. at 14-17.

The Conservation Objectives Team (COT 2013) report also included grazing management recommendations, including:

- Ensure that [grazing] allotments meet ecological potential and wildlife habitat requirements; and, ensure that the health and diversity of the native perennial grass community is consistent with the ecological site. COT report at 45.
- [Range management structures] that are currently contributing to negative impacts to either sage-grouse or their habitats should be removed or modified to remove the threat. Id. at 46.

Though not entirely consistent with the NTT, COT, and best available science, the 2015 RODs clearly pledged, "Livestock grazing will be managed to achieve or maintain desired conditions in GRSG seasonal habitats, as described in table 1 of the attached LMP amendments," (Great Basin ROD at 31, Rocky Mountain ROD at 29). The RODs continue to say that grazing will be managed to specifically <u>achieve residual grass height and cover guidelines</u>. (Emphasis ours.) The proposed 2018 plan weakens this considerably, shifting the specific guidelines to, "In GRSG habitat, if livestock grazing is limiting achievement of seasonal desired conditions, adjust livestock management, as appropriate, to address GRSG habitat requirements." DEIS at ES-9.

The agency justifies this change by saying "Based on a new understanding of habitat characteristics, plant phenology, and sampling bias (Hanser et al. 2018), the biological foundation for the development of the 2015 GRSG Plan Amendments has changed and this changed conditions warrants removal of the grazing guidelines, which are not necessary as conservation measures for sage-grouse." DEIS at 4-225. That is not the conclusion of the Hanser et al. synthesis. Instead, Hanser et al. (2018) concludes, "The absence of support for a universal effect of grass height does not imply nest concealment is unrelated to nest survival in sage-grouse (Smith and others, 2017b)," and simply emphasizes the need to correct for date-skewed data and recognize that grass height is just one indicator of habitat needs. The Forest Service seems to want to throw the baby out with the bathwater by completely jettisoning habitat guidelines (and the recommendations of the NTT and COT to set such guidelines), without acknowledging that nest success is still higher where there is more cover, and that grass height is a measurable way of limiting livestock removal of concealing factors. This is a particularly trenchant point given the significant sage-grouse population declines in areas where livestock grazing is the primary human-caused habitat impact.

Indeed, in addition to Hanser et al.'s (2018) cautions, the studies themselves don't say that grazing impacts don't affect nest survival. As noted in our scoping comments and herein, the science is much more nuanced. Gibson et al. (2016) implicitly assumes no livestock, insect, or wildlife grazing, and therefore projections of grass growth present a biased (overestimated) grass height at fledging date, were the nest successful. It calculates the estimated growth of grass from failed nest date to successful nest fledge date, assuming linear growth without interruption (herbivory). This makes an assumption every bit as unsupportable as the assumptions made by the 9-day difference between successful and unsuccessful nest measurement.

Though we may not understand exactly what causes greater sage-grouse to choose nesting sites and what makes those nests more or less successful, Gibson et al.'s conclusion that grass grows during the nesting period isn't particularly illuminating when understood within the entirety of scientific literature documenting that demonstrates non-random nest site-selection for successful nests that have significantly taller grass than nearby random sites. *See, e.g.* Hagen et al. 2007. Gibson et al. does not conclude that grass height and cover is unimportant, only that past studies may have inflated the magnitude of the effect through sampling bias.

The true test of the hypothesis would have been to measure grass heights at the nests (both failed and successful) in the field, on the same date. Studies that actually did this found significant differences (notably the seminal Oregon study, Gregg et al. 1994), while others corrected for the bias identified in Hanser et al. and still found significant differences (e.g., Doherty et al. 2014). The newest Smith et al. (2018) study (attached) parses the Doherty et al. (2014) study, plus some new datasets not tested before, and validates the Wyoming study. Smith et al. (2018) find that unpublished datasets from northeast Utah and Montana would have generated "false positives" if grass heights at failed nests were measured at failure date, without a corrective calculation applied, but this study did not undertake a "statistical power test" to determine whether nonsignificant findings for corrected data were a "false negative," (in statistics, a Type II error). With this in mind, the best one can conclude from Smith et al. (2018) is that the Doherty et al. (2014) finding ($p < 0.05$) that grass height has a significant effect on sage grouse nest survival (and indeed is the most important factor) is confirmed, while the two unpublished datasets yield ambiguous results.

It seems clear that nests are less likely to be successful when grass height is lower, whether the insufficient cover is due to phenological circumstance or herbivory; only one of these factors is within the purview of FS management and responsibility. Grass of adequate height also provides cover for grouse at other stages in their life history, such as early brood-rearing.

And even if (an "if" we do not support) grass height differences weren't explicitly responsible for nest success and failure, they are still useful proxies for addressing the impacts of grazing activities. Because livestock are known to stress out sage-grouse (Jankowski et al. 2014) cause nest abandonment, and increase nest predation (including by the cattle themselves; 75 F.R. 13940-41), the management of livestock in sage-grouse habitat must be limited during nesting and brood-rearing seasons, and grass height is a useful indicator of the intensity of grazing use. Grazing use and livestock incursions into sage-grouse habitat would increase the frequency of nest flushing, a factor also linked to nest success in observer-interaction studies. Gibson, et al. *in press.* Grass height may also have significance for foraging distance from nests, not simply cover at the nest bowl. Nest success is just one measure; nest selection and brood-rearing are others, and Coates et al. 2017 found that while nesting and non-nesting periods have some variation in spatial attributes of grass height, during both both life stages, the use of cover exemplifies the importance of structurally diverse microhabitats that consist of mixed vegetation to conceal sage-grouse nests and their chicks.

The USFS DEIS expresses a range of desired conditions for perennial grass height: In Colorado, simply "Provide overhead and lateral concealment from predators." DEIS at B-2. This statement is attributed by footnote to citations #7 and #15; #7 is Connelly et al. (2000) which recommends grass height of greater than or equal to 18 cm in breeding habitat, #15 is missing from page B-3. Idaho recommends the same qualitative grass height desired conditions (overhead and lateral concealment) and references Connelly et al. (2000) and footnotes, "Projects will be designed to provide overhead and lateral concealment of nests on a site-specific basis." DEIS at C-2 and C-3. Nevada adopts the same and cites to Connelly et al. (2000) but adds Coates et al. (2013). Utah also seeks to "provide overhead and lateral concealment from predators," and recommends, "Defer to local data whenever possible to help determine proper height." DEIS at E-2. The basis for this is Connelly et al. 2000 but Utah tacks on a citation to Smith et al (2018) which it claims, "Phenology largely explains taller grass at successful nests in greater sage-grouse." (*sic*). DEIS at E-3. Wyoming maintains the vague, "overhead and lateral concealment" desire, but cites to Stiver et al. 2015 and to a missing footnote #15.

So, while the DEIS removes specific grass height guidelines from the Desired Conditions tables, it provides various rationales for doing so, some of which support the original grass height requirements (i.e. Connelly at al. 2000) and others which are largely irrelevant (e.g. Coates et al.

2013). Stiver et al. (2015) noted that although sage-grouse may occupy areas with shorter grasses, “this is not sufficient reason to assume that the suitability indicator value for grass height should be reduced… rather this condition may indeed reflect reduced habitat suitability and likely indicates a rangeland health issue that should be addressed via … management changes.” Stiver’s HAF assessment form uses 18 cm grass height as the lower limit of habitat suitability. Stiver at al. 2015 at 37. And Smith et al. (2018) admit that the “absence of support for an effect of grass height does not imply concealment is wholly unrelated to nest survival in sage-grouse.” Smith’s study looks at three data sets (only two of which were ever published) and corrects for phenological influence; the only published data set of the three was found to maintain the nest success correlation with grass height post-correction. Thus, the evidence that USFS is relying on to knock out meaningful and quantifiable desired future conditions is scant, being overinterpreted, is itself controversial, and would not qualify as an adequate regulatory mechanism to protect nesting habitat.

Idaho’s emphasis on site-specific modifications to the desired conditions is at odds with its presentation of droop and stubble height measurements in Table 3-5. All of the Idaho samples met the 7 inch stubble height during nesting season -- it’s entirely unclear why the agency is proposing to remove this requirement if it’s possible to meet it (based on the agency’s own data and ours, as provided in early comments). (As noted below, there is not enough information to truly understand the sampling technique, the land use in at the sampling sites, the species being measures, etc.)

An additional place where the new proposals fail to comport with the recommendations of the NTT’s to “[p]rioritize completion of land health assessments and processing grazing permits within priority sage-grouse habitat areas,” is the agency’s having abandoned this approach in 2017 by ignoring the deadlines imposed by the 2015 plans and implementing grazing changes “as soon as practicable,” but without date certain.[3] The 2018 proposed action further degrades this timeline by simply stating, as au unenforceable *management approach*, “Conduct greater sage-grouse habitat assessments in allotments.” DEIS at 2-59. Nowhere is a schedule imposed or implied, meaning effectively the USFS could simply *never* get around to assessments and still be on time. Nor is there a requirement to act in a timely manner; the management approach ends by recommending that the agency “determine factors limiting achievement of desired seasonal habitat conditions.” DEIS at 2-60, GRSG-LG-MA-036. Simply knowing what the factors are doesn’t mean the agency will address them, and it’s a concern that the “management approach” is so open-ended.

In fact, this approach seems directly contrary to the explanation of the proposed alternative in the DEIS of, “revising livestock management guidelines to replace grass height requirements with standardized evaluation methods.” DEIS at 2-21. Firstly, by revising the LMPAs to remove livestock management guidelines and allow each forest to set their own is the opposite of standardized. It makes no sense, and is furthermore internally inconsistent. Secondly the DEIS appears to be conflating apples and oranges; in the first instance, the clause is referencing management guidelines and in the second

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3 https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd565108.pdf

part of the same clause, discussing methods. One is the desired outcome and one is the tool by which that is measured.

An additional flaw in the new proposed plans is the idea that grazing impacts are siloed among impacts. The plans specify, "If livestock grazing is limiting achievement of seasonal desired conditions, adjust livestock management." *See, e.g.* DEIS at 2-59 (Idaho). But this disregards the additive or multiplicative impacts grazing might have on other land uses. There may be reductions in livestock grazing that would offset other types of anthropogenic disturbance, but the new plans would preclude adjusting management proactively. This is short-sighted and unnecessarily ties the agency's hands.

The proposed plans also undercut the protective measures of the 2015 plans regarding the construction of new permanent livestock facilities. The previous plan amendments mandated that they (windmills, water tanks, corrals) not be placed within 1.2 miles of occupied leks, per Manier et al. (2013). The new plans restrict this to facilities taller than 4 feet, and limit the spatial prohibition to 1.2 miles in PHMA, .6 miles in IHMA, and .12 miles in GHMA. DEIS at 2-61 (Idaho). No explanation for this change is provided in the DEIS and no science is provided to support this adjustment. DEIS at 3-225. We note also that in both instances, the restrictions only applies to "new permanent" which fails to account for existing or temporary structures that are frequently in sage-grouse habitats. We note too that for some inexplicable reason, Nevada finds it unnecessary to specify a height requirement and simply describes livestock facilities in general, regardless of their height, but changes the application of the guideline to "active or pending leks." DEIS at 2-95. The discrepancy is indicative of the fact that this isn't based on scientific understanding of predator perch heights, or on the the differences among raptor predation in PHMA, IHMA, or GHMA. It's based on something arbitrary and inconsistent between states, and we oppose the new plans on this basis.

Finally, whereas the NTT report recommends, "Maintain retirement of grazing privileges as an option in priority sage-grouse areas…." the new proposed FS action removes the language from the 2015 plans that would emphasize the agency's authority to consider allotment closure when a permit is waived or canceled. DEIS at 2-60 (Idaho), DEIS at 2-94 (Nevada). The DEIS claims this is addressed in existing FS policy or direction, but this "direction" was actually specific to PHMA, IHMA, GHMA, and SFA, which does not exist outside of the 2015 plans. The agency is being disingenuous when it claims otherwise, as the inclusion of this language in the plan amendments made it a site-specific authority that other policy and direction do not share.

Revising the DEIS to refer to grazing "as a tool" to maintain or move towards desired habitat conditions rather than focusing on grazing management as an opportunity to maintain or move towards desired conditions is more than a simply semantic twist. DEIS at 2-32. It is the Forest Service taking what is a known threat to the sage-grouse -- livestock grazing -- and turning it on its head as a way to improve sage-grouse habitat. This is simply disingenuous and there is no evidence that livestock grazing provides a distinct benefit for sagebrush ecosystems; at best, well-managed livestock grazing does less harm to the vegetation and habitats of sage-grouse than poorly-managed livestock grazing.

And we additionally note that the use of Hanser et al. 2018 is opportunistic. The DEIS overreaches in interpreting Hanser's findings on grass height, but ignores the findings on sagebrush cover. The DEIS itself states, in contrast to the grazing claims, "Other site-scale vegetation measures, especially sagebrush cover, remain important for sage-grouse habitat use and survival and are critical for identifying desired habitat conditions." DEIS at 3-189. This makes it all the more confounding that the same DEIS proposed (for Colorado) to reduce the desired canopy cover conditions for sagebrush, from 10-30 percent to 5-25 percent. DEIS at 2-25. There is no scientific support for this change, and it is at odds with the science that the agency elsewhere cites.

Finally, the federal government's reliance on, and citation to, Hanser et al. (2018) is completely inappropriate from a scientific perspective. Hanser et al. (2018) does not undertake a comprehensive review of the state of the science and make policy recommendations as do Connelly et al. (2000) or Manier et al. (2014) on a more limited extent. Nor does it undertake a comprehensive review of some or all aspects of the sage grouse science and draw conclusions from the collective body of science that currently exists, as do Hagen et al. (2007) and NTT (2011). Instead. Hanser et al. 2018 is merely a compendium of scientific abstracts of recent works, some of which undertake statistically rigorous hypothesis testing of their own, and Hanser et al. do not draw comprehensive conclusions based on these abstracts. Therefore, in all cases where federal agencies cite to Hanser et al. (2018), they must instead cite to the original science referenced therein to fulfill NEPA's scientific integrity requirements.

6. The proposed plan relies on data that the USFS gathered, but the DEIS does not provide sufficient evidence of the methodology or sampling strategy to inform the public or the decision-makers.

The proposed plan excuses the need for standardized vegetation parameters like stubble height and droop height by citing to its own (unpublished?) studies of monitoring of sage-grouse habitats. The DEIS cited to "USDA FS 2018" in Section 4.5.7, which we assume intends to reference the USDA Forest Service 2018d, the Forest Service Greater Sage-grouse Monitoring Annual Report, Second Year Summary: October 2016-September 2017. This report is unavailable online; WWP received it upon request, but it is still not clear that this is the relevant information that USFS is citing.

The DEIS uses whatever that reference actually is to claim that, "in the majority of cases, nesting, breeding, upland summer, and winter habitats were in suitable condition with grazing being managed consistent with direction in existing land management plans." DEIS at 4-225. The DEIS appears to reference the same data set on page 3-196 where is says that the forest sampled 2,965 sites to measure droop and stubble heights. DEIS at 3-196. And despite summaries of forest-wide samples (which number from two samples on the Caribou-Targhee to 272 on the Uinta-Wasatch-Cache), no specific information was provided in the DEIS about where the samples occurred, how they were selected and distributed, the percentage of acres they relate to, whether they were measured in what was called SFA, or PHMA, GHMA, IHMA, etc. The reference cited was unable to be retrieved online or through direct request, but the DEIS lacks information as to whether these samples were collected on grazed or ungrazed habitats, at what level grazing use was implemented, and the number of acres reflected in the sample. This is important because simply saying that droop heights and brood-rearing stubble heights were more or less consistent with the science doesn't identify whether grazing management influenced the results or not. We find the results highly suspect given our observations of grass height on the national forests, and the agency's reliance on this unverified/unvalidated and not statistically supported data is problematic.

It's also a strange point for the Forest Service to be making: grass heights aren't very important anymore (per recent science), the existing forest plan standards are sufficiently protective because they are consistent with the science, and the amendments enforcing the scientifically-derived standards are unnecessary. Is the argument that the science-based standards in the existing RODs are sufficient to protect sage-grouse habitat and the amendments' inclusion is redundant? If so, what about places like the Challis National Forest Land and Resource Management Plan of 1987 that punts decision-making to AMPs? DEIS at 3-197. What is the backstop protective plan that ensures appropriate cover?

The USFS's proposed action to withdraw the grazing use restrictions of the 2015 amendments does nothing to address the high levels of use on the Humboldt National Forest. Levels up to 70 percent of use in riparian areas is downright excessive at any time, and the grazing system doesn't matter in context of their being absolutely minimal hiding cover left for young sage-grouse. DEIS at 3-197. Three inches stubble height in riparian areas is not consistent with sage-grouse science. DEIS at 3-198. In Utah, the Uinta allows riparian use to as low as 2" stubble height. Id. These are laughably low when you consider the size of a sage-grouse. The DEIS fails to retain the overarching protections promised in 2015; the proposed regulatory mechanisms to limit the harms of livestock overuse are no longer present.

An additional issue that the DEIS's deferral to site-specific planning is the lack of information about the percentage of AMPs or grazing permits that have incorporated the standards of the forest plans. For example, where the Ashley National Forest LRMP defers use levels to the establishment of AMPs, the DEIS contains no information about the number of AMPs that have been updated since the 1986 plan was adopted. In many cases, AMPs predate forest plans and/or are so dated as to not contain any contemporarily relevant management provisions. Without an assessment of how many allotments have AMPs, how many of those AMPs are consistent with current scientific understanding of sage-grouse habitat needs, or how often monitoring of those habitats is done, this information is unhelpful in the context of determining the effects of the new proposed action on sage-grouse. (We note too that no samples of summer stubble heights were collected on the Ashley NF - DEIS at 3-196.)

Additionally, a number of forest plans are under revision and will undergo entirely new planning processes under the 2012 NFMA planning rule revision. Unless and until the plans are guaranteed to include the basic habitat guidelines, the FS should not be relying on the old plans to provide sufficient protection for sage-grouse. Under the 2012 FS planning rule as amended, the responsible official must use the best available scientific information to inform the amendment process. Here, the agency is using unreliable and unvalidated sampling to inform the amendment process, jettisoning the recommendations of teams of scientists, years of research, and broadly acceptable standards that already exist within forests in the region and replacing it with vague, unenforceable and unexplained provisions like, "adequacy."

7. The proposed plan undermines the mitigation that was supposed to protect the sage-grouse from anthropogenic impacts on Forest Service lands.

The proposed plan amendments change the mitigation from "net conservation gain" to "no net habitat loss", e.g. DEIS at 2-26 (Colorado). This goes against what the best available science recommends for conserving Greater sage-grouse. The 2010 Warranted Finding for listing GRSG under the ESA was due in part to the lack of adequate regulatory mechanisms. "The combination of increased development and the inadequacy of regulatory mechanisms in <u>requiring proposed development to avoid impacts</u> is a pressing issue for sage grouse conservation. Even in areas where the primary threat is not development, providing adequate regulatory mechanisms to address anthropogenic impacts and other threats is necessary to ensure long-term protection of the species." USFWS Greater Sage-Grouse Range-Wide Mitigation Framework, emphasis added. The FWS also warned that a "No net loss" standard would be deemed less protective:

> "Mitigation programs should be strategically designed to result in net overall positive outcomes for sage-grouse. This is accomplished by employing avoidance, minimization, and compensatory mitigation actions that are based on accepted mitigation principles and standards, use best available science for sage-grouse conservation, and address population-level threats within landscape-level plans. Programs that are structured with a goal of only no net loss will be evaluated more conservatively by the Service because they are unlikely to positively influence the conservation status of the species."

Ibid. Thus, the agency's proposed changes weaken the plans' protection, putting sage-grouse (and their "Not Warranted" status) at risk. The USFS is effectively proposing to change the mitigation framework to one they have already been told is insufficient to protect the bird.

The changes to mitigation policy on the BLM lands that compose the majority of sage-grouse habitat are also detrimental to ensuring the long-term persistence of the species, and the USFS's weakening standards need to be evaluated in context of a lack of enforceable or compensatory mitigation of millions of acres of adjacent, contiguous, or connected habitats.

8. The proposed plans arbitrarily redefine protected areas and timeframes of relevant disturbance.

The plans insufficiently analyze the impacts of changing the application of standards from "occupied lek" to "active or pending lek." *See* e.g. DEIS at 2-84. An occupied lek was defined as, "A lek that has been active during at least one strutting season within the prior 10 years." DEIS at Glossary 286. An active lek is, "Any lek that has had two or more males observed at least twice in the last five years." Glossary at 280. And a pending lek is, "Any lek that has two or more males observed only once in the last five years." Glossary at 286. Thus, the agency is really reducing the number of leks to which the standards apply and allowing sparsely occupied habitat to be degraded rather than preserving its integrity for potential reoccupation or increased use under different conditions. Given the cyclicity of sage-grouse populations, a five year time-frame for jettisoning lek protections is too short. It's also inconsistent with the three-year trend on which NSO exceptions can be based, underscoring the arbitrary nature of the USFS proposed changes.

For example, GRSG-GEN-ST-006-Standard changes the standard to restrict loud noises above 10dB at an occupied lek during lekking to requiring this only for active or pending leks. DEIS at 2-83. This means that the potential habitat for reoccupation of a lek site by breeding birds is reduced by at least half due to increased disturbance being permitted in a shorter time frame. This is a significant change and one that the DEIS glosses over. Losing a lek site to new infrastructure (GRSG-LG-GL-047, -048, -046, etc.) is a significant irreversible and irretrievable commitment of resources that should be considered in the analysis, as well as an estimate of the number of leks currently (and annually for the past decade) would have moved out of the receiving these protections.

Another action of the 2018 proposed plans is to modify the application of standards to a shifting baseline, from "Do not include noise resulting from human activities that have been authorized and initiated within the past ten years in the ambient baseline measurement," to "Do not include noise resulting from human activities that have been authorized and initiated within the 10 years since the issuance of the 2015 ROD (2005) in the ambient baseline measurement." DEIS at 2-83. In the near term, the new standard is worse, allowing noise sources initiated between 2008-2015 to count toward ambient noise. After 2025, new sources can be counted into ambient noise (and only those 2015-2025 are excluded). By shifting the baseline of measurement, the agency is allowing the near term detrimental impacts of noise on sage-grouse to increase, which threatens already declining populations on a much shorter time horizon. The DEIS does not analyze the effects of this change.

In another timeline bait-and-switch, the current plan amendments for Idaho require the retrofitting of all tall structures with anti-perching devices within two years of signing the ROD (which would have been 2017). Now, the proposed action proposes this just for PHMA and "within three years of reissuing permits." DEIS at 2-52 (Idaho), at 2-119 (Utah). It is unclear how often the FS reissues permits, how many of these permits are expiring anytime soon, how long the permits are valid, and thus, the timeframe and likelihood that these anti-perch devices will be installed over, say, the next decade. This type of obfuscation renders the protectiveness of such "objectives" meaningless, and the DEIS fails to analyze and disclose relevant details that would help the decision-maker and the public best understand what the change entails. The standard that would require perch deterrents as part of protective stipulations for authorizations for infrastructure in Idaho is also modified to no longer list perch deterrents, despite their effectiveness. DEIS at 2-54, deletion of "perch deterrent installation" from list of stipulations.

The DEIS has a similar vague requirement to remove tall structures at abandoned mine sites. Whereas the 2015 plans said that in SFA, PHMA, IHMA, GHMA abandoned mine sites should be closed or mitigated to reduce predation by eliminating tall structures, the new proposed plan is simply applicable to PHMA and GHMA, and only applies when closing abandoned mine sites. DEIS at 2-78, DEIS at 20112 (Nevada). But the DEIS fails to consider the percentage of mines that are ever formally closed. Most are abandoned when they stop producing and thus the requirement to ever ensure they won't continue to harm sage-grouse is far less meaningful in the proposed action.

9. The proposed plan doesn't provide sufficient evidence that GRSG will be able to persist on national forest habitats, in violation of the agency's planning rule.

Underpinning the 2012 Planning Rule's complementary ecosystem and species-specific approach to the maintenance of plant and animal diversity and native species persistence are provisions relating to the identification and management of Species of Conservation Concern. Species of Conservation Concern (SCC) include native species occurring on the planning unit for which substantial concern exists for their ability to persist in the long term. For these species, specific plan elements must be developed when an ecosystem-scale approach to conservation is unlikely to provide adequate security from known threats to persistence. Where conditions do not exist to provide for a viable population of a Species of Conservation Concern within the plan area, plan standards must be developed to maintain or restore ecological conditions to contribute to maintaining a viable population within its range. 36 CFR 219.9.

"If species of conservation concern (SCC) have not yet been identified for a plan area and scoping or NEPA analysis for a proposed amendment reveals substantial adverse impacts to a specific species, or the proposal would substantially lessen protections for a specific species, the responsible official must determine whether that species is a potential SCC. If so, the responsible official must apply the requirements of 2012 rule with respect to that species as if it were an SCC." 81 F.R. 90726.

The Forest Service claims that for the DEIS analysis sage-grouse were treated as SCC, and concludes that "The analysis in this DEIS shows that the amendments maintain ecological conditions necessary for a viable population of greater sage-grouse in the plan area for each LMP to which the amendments would apply." (DEIS at 1-5 -- 1-6). Unfortunately, the DEIS does not support this conclusion. Given the general weakening proposed by this DEIS and the unenforceability of the proposed revisions, the agency has not demonstrated that it has met the National Forest Management Act's regulatory requirement to maintain the viability of sage-grouse populations. The burden is on these amendments to provide specific plan elements that will provide for viable populations at the forest level. The agency has not met this burden. See, e.g. DEIS at 2-80, 2-43.

Importantly, only two of the forests affected by the amendment have identified sage-grouse as an SCC. DEIS at 1-5. The DEIS claims that the analysis treats sage-grouse as an SCC and claims that the amendments would maintain conditions necessary for a viable population of greater sage-grouse for each LMP. *Ibid.* But the viability requirements will only actually apply to the forests on which the SCC determination is complete. The agency must use the amendments to formally designate sage-grouse as SCC and require the viability to be assessed on each forest going forward if the plan amendments are to be used to preclude Endangered Species Act listing.

10. The DEIS contains revisions that are removed, reworded, or rearranged in ways that are inconsistent with the original (2015) intent.

There are a number of examples in the DEIS where a provision of the 2015 plans is deleted and the clarifying text refers to another standard. For example, GRGS-LR-SUA-0-013 in Nevada is deleted because the retrofitting requirements are ostensibly included in GRSG-LR-SUA-ST-019. DEIS at 2-85. But that new standard is about locating upgrades to transmission lines. DEIS at 2-87. GRSG-LR-SUA-ST-015 in Nevada is supposedly incorporated into ST-016, but the former relates to only allowing new lands special use authorizations for infrastructure if they can be located in existing corridors and rights-of-way, and the authorization includes stipulations to protect greater sage-grouse and its habitats. DEIS at 2-86. The proposed alternative is about allowing these new or amended special land uses outside of existing corridors and designated rights-of-way plus exceptions. From restricting new infrastructure at all unless in these corridors to allowing it outside of the corridors is a different beast, one that the DEIS obscures by claiming it's "incorporated" rather than "overhauled" by the new standard. This is confusing for the general reader and dishonest in the assessment that there will be no significant impacts of this change.

Similarly, GRSG-RT-GL-082 is deleted from the proposed action for Nevada, with the claim that this is added to DC-078. DEIS at 2-103. But DC-078 is not about new roads and road realignments being designed and administered to reduce collisions with greater sage grouse, as -028 was. It just requires minimal disturbance and mortality on roads and trails. DEIS at 2-102. This is less specific than the old standard and not necessarily inherent in the revised language. It is also undermined by Guideline GRSG-RT-GL-083 that requires any seasonal closures for sage-grouse to be "demonstrably having a negative impact on GRSG breeding and nesting behavior." DEIS at 2-105. Waiting to prove disturbance is occurring before closing roads is unacceptable; protection for this species (particularly during breeding and nesting season) should be proactive.

We are concerned that new transmission lines outside of the existing designated corridors and rights-of-ways are no longer required to be buried. Tall structures, like transmission lines, interfere with the Greater Sage-Grouse ability to utilize nearby habitat. In addition, compensatory mitigation was supposed to offset any remaining residual impacts that have resulted from the authorization of land uses in priority habitat management areas. It is now unclear if that will happen.

11. The proposed plan amendments increase the inconsistency across the range of sage-grouse, undermining their ability to provide effective and certain protection for this species and its habitat.

The new proposed actions are so variable by state that it clearly points to arbitrary and politicized decision-making. For example, Nevada retains "Net Conservation Gain," whereas Idaho, Colorado, Utah, and Wyoming diminish the protection to "No Net Loss." DEIS at 4-270. As shown above, the plan is inconsistent in setting desired future conditions for sage-grouse habitat and even references different science to support its ambiguous new standards.

In one example, the Idaho proposed action is to delete Guideline GRSG-LR-SUA-GL-020 which said, "The best available science and monitoring will be used to inform infrastructure siting in greater sage-grouse habitat." DEIS at 2-55. The proposed action offers no alternative, just strikes out this requirement of the 2015 plan.

The plans also maintain and fail to correct inconsistencies in the application of lek buffers (0.6 miles in Wyoming versus 3.1 miles in other states), disturbance density calculations (5% in Wyoming and parts of Utah versus 3% in other states), and spatial designation of Priority Areas for Conservation as PHMA (a spectrum running from 100% in Colorado and Wyoming to 53% in Nevada and 30% in California)[4].

12. The proposed plan amendments violate the 2012 planning rule by failing to identify the substantive requirements of the amendments.

The DEIS acknowledges that the planning rule requires the FS to identify and apply the substantive requirements of the rule that are related to the amendment. DEIS at 1-5. The DEIS lists just three: 36 C.F.R. 219.8, 219.9(a) and (b), and 219.10((a). Ibid. But the DEIS fails to conform to 36 C.F.R. 219.9(b) which requires the responsible official to determine whether the standards and guidelines of the plan are sufficient to: "...maintain a viable population of each species of conservation concern within the plan area. If the responsible official determines that the plan components required in paragraph (a) are insufficient to provide such ecological conditions, then additional, species specific plan components, including standards or guidelines, must be included in the plan to provide such ecological conditions in the plan area." Instead, the DEIS proposes jettisoning most of the quantitative, enforceable plan components that were included in the 2015 plan.

13. The proposed plan fails to consider the alternative of correcting science-based deficiencies in the 2015 plan.

In an EIS, the Forest Service must study, develop, and describe alternatives to the proposed action, and analyze "all reasonable alternatives." 40 U.S.C. § 4332(C)(iii); 40 C.F.R. § 1502.14. The alternatives analysis is "the heart of the environmental impact statement," and the agency must "rigorously explore and objectively evaluate all reasonable alternatives." 40 C.F.R. § 1502.14. "The existence of a viable but unexamined alternative renders an environmental impact statement inadequate." *Idaho Conservation League v. Mumma*, 956 F.2d 1508, 1519 (9th Cir. 1992) (quotation omitted). NEPA's alternatives requirement also serves to inform the public of "reasonable alternatives that would avoid or minimize impacts," 40 C.F.R. § 1502.1, or that "might be pursued with less environmental harm." *Lands Council*, 395 F.3d at 1027. The EIS's alternatives analysis must "present the environmental impacts of the proposal and the alternatives in comparative form, thus sharply defining the issues and providing a clear basis for choice among options by the decision-maker and the public." 40 C.F.R. § 1502.14.

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[4] http://pdf.wildearthguardians.org/site/DocServer/The_Shrinking_Geography_of_Sage_Grouse_Conservation_Fina.pdf

In our comments, WWP et al. have consistently pointed out the inadequacies of the 2015 sage-grouse plans, the significance of the National Technical Team (2011) report, the significance of the Conservation Objectives Team (2013) report , the failure to designate spatially adequate Priority Habitat Management Areas, and failure to follow science-based habitat protections. Based on these inadequacies, the Forest Service has a responsibility to create and fully analyze a Conservation Alternative based on the best available science to correct the myriad deficiencies in the 2015 ALMPAs. This is an eminently reasonable alternative, as it follows the recommendations of the federal government's own scientific advisory teams (NTT 2011, COT 2103), would implement habitat protections based on the best available science, and maximize the potential to recover greater sage-grouse to healthy population levels such that ESA listing becomes unnecessary, a key part of the purpose and need for the original ALMPAs. To date, the Forest Service has failed to even consider such an alternative, even though it would be eminently reasonable and implementable. In failing to consider a range of reasonable alternatives, the Forest Service's environmental impacts analysis violates NEPA's 'range of alternatives' requirement.

Key deficiencies that BLM failed to correct from the 2015 ALMPAs based on NTT (2011) recommendations are as follows: Failure to apply No Surface Disturbance buffers of 4 miles around leks (these were instead set at 3.1 miles for PHMA in most states and 0.6 miles in Wyoming and parts of Utah); failure to apply a 3% disturbance cap in Wyoming and parts of Utah; failure to calculate disturbance caps and site density limitations on a per-square-mile-section basis; failure to close PHMA to fluid mineral leasing; make PHMAs exclusion areas for new rights-of-way (roads and utilities); close winter concentration areas to new surface occupancy; and find PHMAs unsuitable for future coal leasing. The conservation alternative must at minimum designate all Priority Areas for Conservation (PACs) identified in COT (2013) as PHMA; to address inadequate designation (particularly in Nevada, California, Idaho, Utah, and Montana, each of which had PHMA designations significantly smaller in acreage than the original PAC designations. The conservation alternative must apply 10 cm grass height objectives in all states; the original LRMPAs were inadequate by virtue of not applying this objective in Utah.

Another key deficiency in the original ALMPAs was the failure to prohibit vegetation treatments harmful to sagebrush and sage-grouse (Beck et al. 2012) in PHMAs. There is abundant science showing that vegetation treatments are harmful to sage-grouse and their habitats.Call and Maser (1985) asserted that the spraying of sage grouse nesting habitats is deleterious because it reduces nest cover from avian predators and suppresses forbs that are important in the sage grouse diet. According to Kerley (1994: 113), "shrub stands of 20-40% cover are needed for successful nesting and this shrub coverage should be maintained on identified breeding complexes [within 3.2 km of leks]". Wamboldt et al. (2002:24) stated:

> Natural or prescribed burning of sagebrush is seldom good for sage grouse. This assessment recommends that fires within sage grouse habitat be avoided in most cases, and should be allowed only after careful study of each local situation. The evidence also indicates that habitat

loss due to fire may well be the most serious of all the factors contributing to the decline of sage grouse.

Heath et al. (1997: 50) went even farther: "Based on our results, we recommend no reduction or control of sagebrush in areas containing between 18-30% live sagebrush canopy coverage within 4.5 km of leks." According to Beck and Braun (1980: 563), "At present we do not know the relative value of a small versus large strutting ground to the population. Therefore we should afford equal merit to all and strive to maintain the adjacent habitats, especially areas with sagebrush (*Artemesia*) suitable for nesting and brood rearing." Hess and Beck (2012) found that neither burned nor mowed areas produced suitable sage grouse habitats. Call and Maser (1985) stated that spraying should not occur within the breeding complex (which they defined as within 2 miles of a lek), and should also be forbidden in known grouse winter ranges. A new study by Shinneman et al. (2018) recognizes the habitat fragmentation and degradation problems caused by fuelbreaks in sage-grouse habitat, and surveyed the available science finding no evidence that fuelbreaks reduces the size or severity of fires in sagebrush habitat. This is significant new information that the Forest Service has not heretofore considered, and the agency should amend its plans to preclude the construction of fuelbreaks or other vegetation treatments harmful to sagebrush within PHMAs.

14. The DEIS fails to fully analyze the cumulative impacts of the proposed action in context of actions across the range of Greater sage-grouse.

We are concerned about the piecemeal review of a region-wide initiative and the potential for uneven management of Greater Sage-Grouse habitats and populations in each state; there needs to be clear and consistent approaches to grouse management. The needs of the Greater Sage-Grouse do not change from state-to-state, therefore our organization urges for more uniform management across the region, and stronger, rather than weaker conservation standards.

Because the USFS and the BLM are both undertaking management reviews and because neither process is completed, it makes a cross-tenure analysis of impacts impossible. The original ARMPAs/ALMPAs were prepared by both agencies and the current division makes consistency difficult to evaluate. The differences in sage-grouse habitat management across land ownership are clearly political and not based on science, and we object to the agency revising forest management in such an arbitrary and capricious way.

CONCLUSION

In conclusion, our organizations, having read through and analyzed the entirety of the DEIS, do not believe that the changes proposed by the USFS strengthen the protections for sage-grouse and, indeed, weaken them substantially by reducing protected acres, providing fewer enforceable standards, and including less meaningful management parameters. There are only a handful of instances where the clarifications or revisions improve habitat protection, and these are insufficient to offset the wholesale decrease in conservation the proposed plan provides. The Forest Service's draft plan does not provide adequate regulatory mechanisms to protect the Greater sage-grouse from the need for listing under the Endangered Species Act, and the DEIS fails to conform with the basic land management laws as described above.

Sincerely,

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REFERENCES CITED (provided with hard copy on DVD)

Beck, J.L., J.W. Connelly, and S.L. Wamboldt. 2012. Consequences of treating Wyoming big sagebrush to enhance wildlife habitats. Rangeland Ecol. Manage. 65: 444-455.

Beck, T.D.I., and C.E. Braun. 1980. The strutting ground count: Variation, traditionalism, and management needs. Proc. Ann. Conf. West. Assn. Fish and Wildl. Agencies 60:558-566.

Call, M.W., and C. Maser. 1985. Wildlife habitat in managed rangelands--The Great Basin of southeastern Oregon: Sage grouse. USDA Gen. Tech. Rept. PNW-187, 29 pp.

Coates, P. S., M. L. Casazza, E. J. Blomberg, S. C. Gardner, S. P. Espinosa, J. L. Yee, L. Wiechman, and B. J. Halstead. 2013. Evaluating greater sage-grouse seasonal space use relative to leks: implications for surface use designations in sagebrush ecosystems. Journal of Wildlife Management 77: 1598–1609.

Coates, P.S., Casazza, M.L., Brussee B.E., Ricca, M.A., Gustafson, K.B., Sanchez-Chopitea, E., Mauch, K., Niell, L., Gardner, S., Espinosa, S., and Delehanty, D.J. 2016b. Spatially explicit modeling of annual and seasonal habitat for greater sage-grouse (*Centrocercus urophasianus*) in Nevada and Northeastern California—An updated decision-support tool for management: U.S. Geological Survey Open-File Report 2016-1080, 160 p., http://dx.doi.org/10.3133/ofr20161080.

Coates, P.S. Brusee, B.E., Ricca, M.A., Dudko, J.E., Prochazka, B.G., Espinosa, S.P., Casazza, M.L. and Delehanty, D.J. 2017. Greater sage-grouse (*Centrocercus urophasianus*) nesting and brood rearing microhabitat in Nevada and California – Spatial variation in selection and survival patterns. U.S. Geological Survey Open-File Report 2017-1087, 79 p. https://doi.org/10.3133/ofr20171087.

Connelly, J.W., M.A. Schroeder, A.R. Sands, and C.E. Braun. 2000. Guidelines to manage sage-grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

Gibson, D., Blomberg, E.J., and Sedinger, J.S., 2016, Evaluating vegetation effects on animal demographics—The role of plant phenology and sampling bias: Ecology and Evolution, v. 6, no. 11, p. 3621–3631.

Hagen, C.A., J.W. Connelly, and M.A. Schroeder. 2007. A meta-analysis of greater sage-grouse *Centrocercus urophasianus* nesting and brood-rearing habitats. Wildlife Biology 13:42–50.

Hanser, S.E., Deibert, P.A., Tull, J.C., Carr, N.B., Aldridge, C.L., Bargsten, T.C., Christiansen, T.J., Coates, P.S., Crist, M.R., Doherty, K.E., Ellsworth, E.A., Foster, L.J., Herren, V.A., Miller, K.H., Moser, Ann, Naeve, R.M., Prentice, K.L., Remington, T.E., Ricca, M.A., Shinneman, D.J., Truex, R.L., Wiechman, L.A., Wilson, D.C., and Bowen, Z.H., 2018, Greater sage-grouse science (2015–17)—Synthesis and potential management implications: U.S. Geological Survey Open-File Report 2018–1017, 46 p., https://doi.org/10.3133/ofr20181017.

Heath, B.J., R. Straw, S.H. Anderson, and J. Lawson. 1997. Sage grouse productivity, survival, and seasonal habitat use near Farson, Wyoming. Unpublished completion report to the Wyoming Game and Fish Department.

Hess, J.E., and J.L. Beck. 2012. Burning and Mowing Wyoming Big Sagebrush: Do Treated Sites Meet Minimum Guidelines for Greater Sage-Grouse Breeding Habitats? Wildlife Society Bulletin 36: 85-93.

Kerley, L. 1994. Bird responses to habitat fragmentation caused by sagebrush management in a Wyoming sagebrush steppe ecosystem. PhD Dissertation, Univ. of Wyoming, 153 pp.

Knick, S. T., S. E. Hanser, R. F. Miller, D. A. Pyke, M. J. Wisdom, S. P. Finn, E. T. Rinkes, C. J. Henny. 2011. Ecological influence and pathways of land use in sagebrush. Pages 203-251 in S. T. Knick and J. W. Connelly (eds). GREATER SAGE-GROUSE: ECOLOGY AND CONSERVATION OF A LANDSCAPE SPECIES AND ITS HABITATS. Studies in Avian Biol. Series, vol. 38. Cooper Ornithological Society. Univ. Calif. Press. Berkeley, CA.

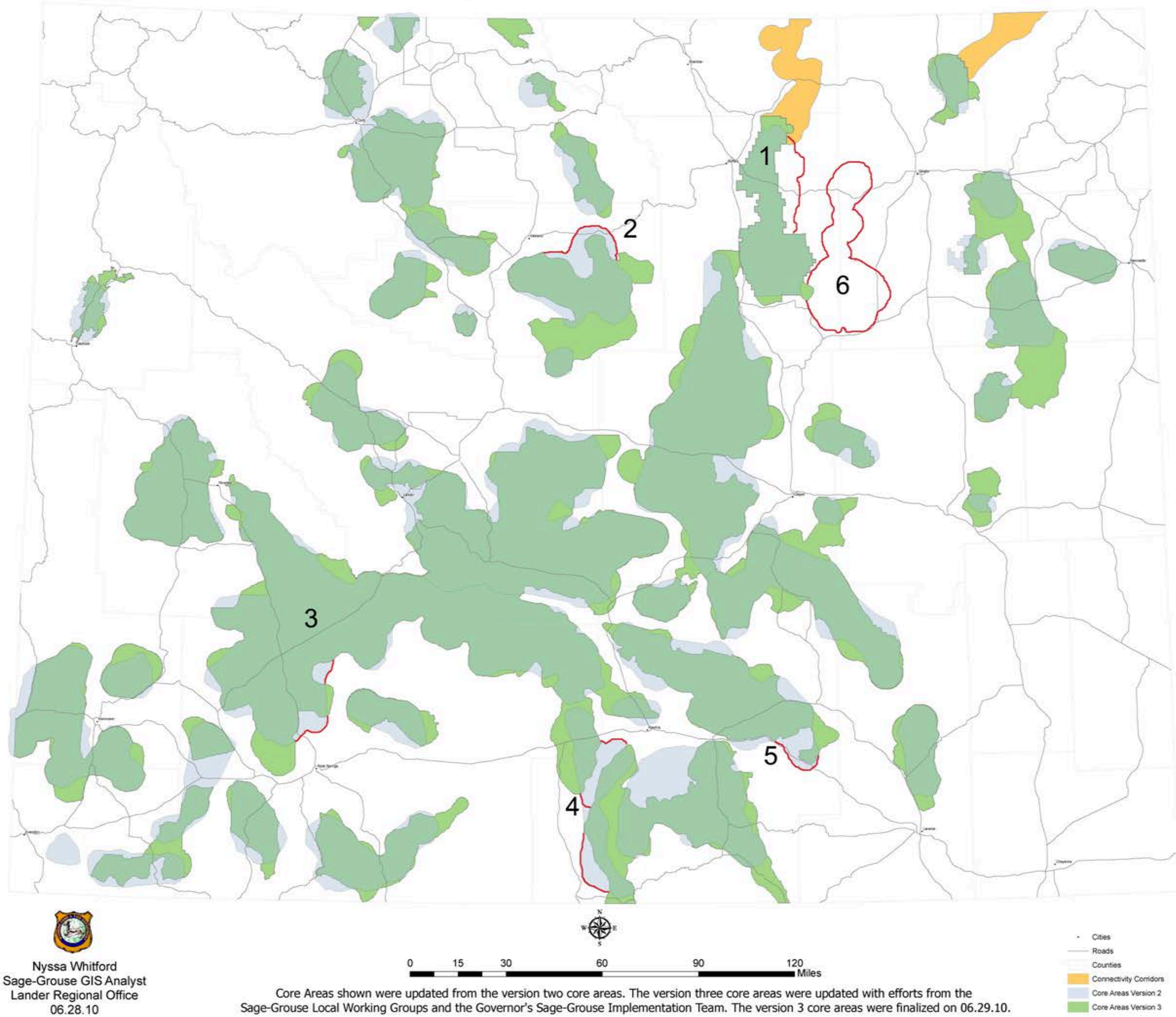
Knick, S. T., A. L. Holmes, R. F. Miller. 2005. The role of fire in structuring sagebrush habitats and bird communities. FIRE AND AVIAN ECOLOGY IN NORTH AMERICA. Studies in Avian Biology, no. 30. Page 6. Cooper Ornithological Society. Boise, ID.

Shinneman, D.J., Aldridge, C.L., Coates, P.S., Germino, M.J., Pilliod, D.S., and Vaillant, N.M., 2018, A conservation paradox in the Great Basin—Altering sagebrush landscapes with fuel breaks to reduce habitat loss from wildfire: U.S. Geological Survey Open-File Report 2018–1034, 70 p., https://doi.org/10.3133/ofr20181034.

Smith, J.T., Tack, J.D., Doherty, K.E., Allred, B.W., Maestas, J.D., Berkeley, L.I., Dettenmaier, S.J., Messmer, T.A., and Naugle, D.E., 2018, Phenology largely explains taller grass at successful nests in greater sage-grouse: Ecology and Evolution, v. 8, p. 356–364, https://doi.org/10.1002/ece3.3679.

Stiver, S.J., E.T. Rinkes, D.E. Naugle, P.D. Makela, D.A. Nance, and J.W. Karl, eds. 2015. Sage-Grouse Habitat Assessment Framework: A Multiscale Assessment Tool. Technical Reference 6710-1. Bureau of Land Management and Western Association of Fish and Wildlife Agencies, Denver, Colorado.

Version 3 Sage-Grouse Core Areas 06.29.10



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06.28.10

Core Areas shown were updated from the version two core areas. The version three core areas were updated with efforts from the Sage-Grouse Local Working Groups and the Governor's Sage-Grouse Implementation Team. The version 3 core areas were finalized on 06.29.10.