IMPACTS OF FENCES ON GREATER SAGE-GROUSE IN IDAHO: COLLISION, MITIGATION, AND SPATIAL ECOLOGY

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ABSTRACT

Conservation concerns over greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse) have drawn attention to the lack of empirical data on impacts of fences on this species. Research suggests grouse as a group may be vulnerable to collision with anthropogenic infrastructure, and sage-grouse fence collision risk has not been systematically studied in any part of their range. Therefore, I studied sage-grouse fence collision on Idaho breeding areas during spring of 2009 and 2010. I sampled fences within breeding areas to quantify relative collision frequency across the landscape, and conducted field experiments to quantify fence sampling biases and effectiveness of fence marking mitigation methods.

I used female ring-necked pheasant (*Phasianus colchicus*) carcasses as surrogates for sage-grouse to study survival and detection bias associated with avian fence collision surveys in sagebrush steppe during spring 2009. I randomly placed 50 pheasant carcasses on each of 2 study areas, estimated detection probability during fence-line surveys, and monitored survival and retention of carcasses and their sign over a 31-day period. Survival modeling suggested site and habitat features had little impact on carcass survival, and rapid scavenging resulted in estimated daily survival probabilities that ranged from 0.776-0.812. Survival of all carcass sign varied by location, and daily survival probabilities ranged from 0.863-0.988. Detection probability of carcasses during fence-line surveys was influenced by habitat type and microsite shrub height at the carcass location. Carcasses located in big sagebrush (*Artemisia tridentata*) habitats were detected at a lower rate (0.36) than carcasses in little (*A. arbuscula*) and black sagebrush (*A. nova*) habitats (0.71), and increasing shrub height reduced detection probability. Avian fence collision surveys in sagebrush-steppe should be conducted at ≤ 2-week sampling intervals to reduce the impact of sign-survival bias on collision rate estimates. Researchers should be aware that local vegetation influences detection probabilities, and apply methods to correct for detection probabilities to ensure accurate collision estimates.

I used a stratified cluster sampling design to sample fences in breeding areas (2009: $n = 16$; 2010: $n = 14$), quantify fence collision frequency, and estimate fence collision rates across the landscape. I found 86 sage-grouse collisions over 2 field seasons, and found evidence for spatial variation in fence collision rates across sampling areas (2009: *range* = 0-5.42 strikes/km; 2010: *range* = 0-2.63 strikes/km). Despite variation among sites, landscape scale sage-grouse fence collision rates corrected for detectability were consistent across years (2009: $\bar{x} = 0.70$ strikes/km; 2010: $\bar{x} = 0.75$ strikes/km). These data suggest sage-grouse fence collision during the breeding season was relatively common and widespread, and corroborate previous studies suggesting grouse are susceptible to infrastructure collision.

I collected site-scale data at random and collision fence points, quantified broad-scale attributes of sampling areas using geographic information systems (GIS), and modeled the influence of site and broad-scale features on sage-grouse fence collision. Discrimination between random and collision fence points using site-scale data suggested collision was influenced by technical attributes of the fences. Collisions were more common on fence segments bound by steel t-post and > 4 m wide, whereas random points were more common on segments with ≥ 1 wooden fence post and widths < 4 m. Broad-scale modeling suggested probability of collision was influenced by region, topography, and fence density. Probability of collision presence was greater in the Big Desert and Upper Snake regions, and lower in the

Magic Valley. Increasing terrain ruggedness reduced probability of collision presence, whereas increasing fence length per km^2 increased probability of collision. Broad-scale modeling also suggested collision counts per km^2 were influenced by distance to nearest active sage-grouse lek, where increasing distance reduced expected collision counts. These data suggest 2 km mitigation buffers around leks in high risk areas may be necessary. However, increasing topographic variation appeared to attenuate the influence of other factors, suggesting high risk areas are most likely at relatively flat sites.

I also conducted a field experiment testing effectiveness of fence marking at reducing sage-grouse collision on high risk breeding areas during spring of 2010. Using 8 study sites, I experimentally marked 3, 500-m segments of fence at each site using reflective-vinyl markers, with 3, 500-m unmarked control segments at each site, and surveyed study segments 5 times during the lekking season. Modeling suggested collision count summed over the lekking season was influenced by marking treatment, lek size, and distance to nearest lek. The top model predicted marking reduced collision counts by 74.0% at the mean lek size and distance from lek. Increasing lek size and decreasing distance to lek increased expected collision counts. Although fence marking reduced sage-grouse collision risk, expected collision counts in high risk areas (i.e., maximum lek size = 127, minimum distance = 104 m) were high (unmarked fence = 8.3 birds/500 m/season, marked fence = 2.2 birds/500 m/season), suggesting these fences may require removal to eliminate collisions. Further, expected collision counts in low risk areas (i.e., minimum lek size = 1, maximum distance = 4,650 m) were very low (unmarked fence = 0.08 birds/500 m/season, marked fence = 0.02 birds/500 m/season), suggesting not all fences require marking mitigation efforts.

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CHAPTER 1. THESIS INTRODUCTION

Greater sage-grouse (*Centrocercus urophasianus*) (hereafter sage-grouse) abundance and distributions have declined across the western United States during the last 50 years (Connelly and Braun 1997, Braun 1998, Connelly et al. 2004, Schroeder et al. 2004). Population declines may have occurred since at least the 1950s (Braun 1998), and Connelly et al. (2004) reported precipitous declines from the 1960s to 1980s. Since the 1980s some populations have been relatively stable, while others experienced declining trends (Connelly et al. 2004, Garton et al. 2011). Estimates of population declines are likely conservative as sage-grouse were not carefully monitored prior to the 1950s; thus historical population sizes are not well known (Connelly et al. 2004). Moreover, population declines have resulted in an estimated 44% range-wide reduction in the geographic range of sage-grouse (Schroeder et al. 2004).

Causes of sage-grouse population declines are varied, but generally may be related to anthropogenic-caused loss, fragmentation, and degradation of sagebrush (*Artimisia* spp.) habitats throughout the western United States (Connelly and Braun 1997, Braun 1998, Beck et al. 2003, Connelly et al. 2004). No single factor is believed responsible for degradation and elimination of sage-grouse habitats; however, numerous anthropogenic factors such as agricultural development, energy development, reservoir and road construction, urban sprawl, fences, power lines, and sagebrush control have been hypothesized (Braun 1998, Connelly et al. 2004). Sagebrush control treatments via chemical, mechanical, and prescribed burning methods have been common on western rangelands since the 1960s

(Braun 1998, Connelly et al. 2004), with detrimental effects on sage-grouse (Klebenow 1970, Martin 1970, Wallestad 1975, Nelle et al. 2000, Beck et al. 2009). Urban development of sagebrush-steppe habitats also had regional influences on sage-grouse with some Colorado counties having lost up to 50% of habitat to housing development (Braun 1998). Relationships between human-induced livestock grazing and sage-grouse population vital rates are not well understood (Beck and Mitchell 2000, Connelly et al. 2004), but evidence suggests indirect effects of grazing may be more detrimental than direct effects (Beck and Mitchell 2000).

While human activities have transformed shrub-steppe ecosystems across western North America (Connelly et al. 2004, Knick et al. 2011, Leu and Hanser 2011), natural disturbances may act synergistically with anthropogenic losses of sage-grouse habitat (Braun 1998, Connelly et al. 2004). Drought conditions have been hypothesized to supplement anthropogenic causes of nesting and brood-rearing habitat degradation (Connelly and Braun 1997, Braun 1998, Connelly et al. 2004). Changes in fire regimes occurred in xeric sagebrush habitats where non-native annual grasses such as cheatgrass (*Bromus tectorum*) have invaded native communities, shortening fire return intervals and increasing the number of fires and total area burned (Connelly et al. 2004, Baker 2006, Baker 2011, Miller et al. 2011). Burning sage-grouse nesting habitat has negative long-term consequences on sage-grouse due to long time periods required for recovery of shrub components necessary for nesting (Nelle et al. 2000). Moreover, altered fire regimes can result in conversion of sagebrush-steppe into exotic annual grasslands and eliminate sage-grouse habitat at landscape scales (Knick and Rotenberry 1997, Connelly et al. 2004, Miller et al. 2011).

Little research has addressed impacts of anthropogenic landscape features such as elevated infrastructure on sage-grouse (Braun 1998, Connelly et al. 2000b, Connelly et al. 2004, Johnson et al. 2011). Over 51,000 kilometers of fence were constructed on Bureau of Land Management properties in the western United States from 1962–1997 (Connelly et al. 2004), with an additional 1,000 kilometers added annually from 1996–2002 (Connelly et al. 2004). Moreover, fence densities exceed 2 km/km^2 in many areas occupied by sage grouse (Knick et al. 2011), but impacts of fences on sage-grouse remain unknown (Braun 1998). Sage-grouse evolved in an ecosystem with relatively few vertical obstructions, and elevated structures create potential collision risks for this species (Braun 1998, Connelly et al. 2000b, Connelly et al. 2004). Scott (1942) reported finding remains of 4 sage-grouse fence collisions near active leks in Wyoming, and witnessed a 5th sage-grouse collide with a fence during lek observations. Beck et al. (2006) reported 33% of all mortality of radio-marked juvenile sage-grouse on an Idaho study area was caused by collision with power lines. Similarly, Connelly et al. (2000a) reported 9% of adult radio-marked sage-grouse mortality in Idaho was caused by power line collision. Johnson et al. (2011) evaluated range-wide lek count trends over a ten year period (1997-2007) in relation to numbers and distance to communication towers, with general patterns of decreasing lek count trends with decreasing distance to tower or increasing the number of towers at landscape scales.

Research on grouse fence collision in North America is limited to work with lesser prairie-chickens (*Tympanuchus pallidicinctus*) (Patten et al. 2005, Wolfe et al. 2007). Collision mortality was 2nd only to avian predation in causes of mortality for radio-marked lesser prairie-chickens in Oklahoma (Wolfe et al. 2007). Of the collision mortality, 86.4%

was caused by collision with fences, while the remainder involved power lines and automobiles (Wolfe et al. 2007). Moreover, they reported 39.8% of all mortality was caused by fence collisions in Oklahoma, and significant differences existed in collision mortality between study sites (e.g., greater mortality in Oklahoma than New Mexico) and sex of the birds (e.g., greater mortality for females than males).

Although little research exists on the impacts of fences on North American grouse, substantial research exists on collision of European grouse species (Catt et al. 1994, Baines and Summers 1997, Bevanger and Brøseth 2000, Moss et al. 2000, Moss 2001, Baines and Andrew 2003). European fences designed to limit over-browsing by red deer (*Cervus elaphus*) have been a significant mortality factor for capercaillie (*Tetrao urogallus*) in Scotland (Catt et al. 1994, Moss et al. 2000, Moss 2001). Catt et al. (1994) reported a 32% annual collision mortality for radio-marked capercaillie in Scotland, and suggested collision rate may be influenced by local vegetation conditions. More juvenile capercaillie in Scotland are believed to die from fence collision than any other source, and collision may be contributing to capercaillie population declines in that country (Moss et al. 2000, Moss 2001).

Although fence collisions may be limiting capercaillie populations in parts of Europe (Moss et al. 2000, Moss 2001), collision is also prevalent for more abundant species of grouse (Baines and Summers 1997, Bevanger and Brøseth 2000, Baines and Andrew 2003). Baines and Summers (1997) monitored 135 km of fences in Scotland for collisions over a 2-year period, and two-thirds of 281 fence collision victims reported were red grouse (*Lagopus lagopus scoticus*). Baines and Andrew (2003) monitored 80 km of fences in Scotland for collisions over a 2-year period, and reported 42% and 29% of 437 fence collision victims were red grouse and black grouse (*Tetrao tetrix*), respectively. Bevanger and Brøseth (2000) monitored 71.1 km of fences on 12 sections over a 4-year period in Norway and reported 253 avian fence-collision victims, 85% of which were either willow (*Lagopus lagopus*) or rock (*Lagopus mutus*) ptarmigan. These studies indicated that grouse as a group are highly susceptible to fence collision, as grouse commonly comprise more than 90% of observed collision species (Baines and Summers 1997, Bevanger and Brøseth 2000, Baines and Andrew 2003).

To date there has been little research addressing impacts of fences on sage-grouse in any part of their range (Connelly et al. 2000b, Connelly et al. 2004). Moreover, greater sage-grouse were recently listed by the U. S. Fish and Wildlife Service as warranted but precluded under the Endangered Species Act (United States Department of the Interior 2010), and a future listing will have land-use implications for a large portion of the western United States. Anecdotal evidence with sage-grouse and published studies with other grouse species suggest fence collision may be a widespread and common phenomenon. Therefore, this research was pursued to evaluate impacts of fences on sage-grouse in Idaho. In this research I attempted to answer 4 primary questions related to sage-grouse fence collision: 1) is sage-grouse fence collision common and widespread across southern Idaho sage-grouse breeding habitats, 2) what factors influence survival and detection bias associated with fence sampling methods, 3) what factors are related to sage-grouse fence collision risk at the collision site and broader spatial scales, and 4) is fence marking a viable and effective mitigation strategy to reduce collisions in high risk areas.

Quantifying collision frequency and risk with linear infrastructure requires hiking along infrastructure corridors while searching for evidence of collision (Bevanger 1999). Sampling linear infrastructure to estimate avian collision rates is accompanied by several sources of bias (Bevanger 1999). Chapter 2 of this thesis describes a field experiment used to quantify sampling bias due to survival of collision evidence over time, and detectability of collision evidence present. I developed a probability sampling framework for estimating avian fence collision rates at landscape scales, and chapter 3 of this thesis describes application of these methods to quantify broad-scale sage-grouse fence collision frequency on breeding areas of southern Idaho. I evaluated factors influencing sage-grouse fence collision risk in breeding areas at multiple spatial scales, and chapter 4 of this thesis describes analyses of factors influencing collision risk at the fence collision site and broader spatial scales. Lastly, I modified fence-marker mitigation methods currently used for lesser prairie-chickens, and chapter 5 describes the first experimental test of effectiveness of fence-marker mitigation on collision risk for North American grouse.

LITERATURE CITED

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 941-948.

Baker, W. L. 2006. Fire and restoration of sagebrush ecosystems. Wildlife Society Bulletin 34:177-185.

Baker, W. L. 2011. Pre-Euro-American and recent fire in sagebrush ecosystems. *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, Volume 38, University of California Press, Berkeley, California, 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., 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, 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 34:1070-1078.

Bevanger, K. 1999. Estimating bird mortality caused by collision with power lines and electrocution: a review of methodology. Pages 29-56 *in* M. Ferrer and G. F. E. Janss, editors. Birds and Power Lines: Collision, Electrocution and Breeding. Quercus, Madrid, Spain.

Bevanger, K., and H. Brøseth. 2000. Reindeer *Rangifer tarandus* fences as a mortality factor for ptarmigan *Lagopus* spp. Wildlife Biology 6:121-127.

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.

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., 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., 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., 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.

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.

Garton, E. O., J. W. Connelly, C. A. Hagen, J. S. Horne, A. Moser, and M. A. Schroeder. 2011. Greater sage-grouse population dynamics and probability of persistence. *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, Volume 38, University of California Press, Berkeley, California, 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, 1997-2007. *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, Volume 38, University of California Press, Berkeley, California, USA.

Klebenow, D. A. 1970. Sage grouse versus sagebrush control in Idaho. Journal of Range Management 233:396-400.

Knick, S. T., and J. T. Rotenberry. 1997. Landscape characteristics of disturbed shrubsteppe habitats in southwestern Idaho (U.S.A.). Landscape Ecology 12:287-297.

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. *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, Volume 38, University of California Press, Berkeley, California, USA.

Leu, M., and S. E. Hanser. 2011. Influences of the human footprint on sagebrush landscape patterns: implications for sage-grouse conservation. *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, Volume 38, University of California Press, Berkeley, California, USA.

Martin, N. S. 1970. Sagebrush control related to habitat and sage grouse occurrence. Journal of Wildlife Management 34:313-320.

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, editors. Greater Sage-Grouse: Ecology and Conservation of a Landscape Species and its Habitats. Studies in Avian Biology Series, Volume 38, University of California Press, Berkeley, California, USA.

Moss, R. 2001. Second extinction of capercaillie (*Tetrao urogallus*) in Scotland? Biological Conservation 101:255-257.

Moss, R., N. Picozzi, R. W. Summer, and D. Baines. 2000. Capercaillie *Tetrao urogallus* in Scotland – demography of a declining population. Ibis 142:259-267.

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.

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.

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.

Scott, J.W. 1942. Mating behavior of the sage grouse. Auk 59:477-498.

United States Department of the Interior. 2010. Endangered and threatened wildlife and plants; 12-month finding for petitions to list greater sage-grouse as threatened or endangered. Federal Register 75:3909-14014.

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.

Wallestad, R. O. 1975. Male sage grouse responses to sagebrush treatment. Journal of Wildlife Management 39:482-484.

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 (Suppl. 1):95-104.

CHAPTER 2. SURVIVAL AND DETECTABILITY BIAS OF AVIAN FENCE COLLISION SURVEYS IN SAGEBRUSH STEPPE

INTRODUCTION

Avian collision with anthropogenic infrastructure has received considerable attention in recent years (Wolfe et al. 2007, Drewitt and Langston 2008, Smallwood et al. 2009, Gehring et al. 2009). Elevated structures known to cause avian collision mortality include fences (Baines and Summers 1997, Bevanger and Brøseth 2000, Wolfe et al. 2007), power lines (Bevanger 1995, Janss and Ferrer 2000), wind power turbines (Smallwood 2007, Smallwood and Thelander 2008, Smallwood et al. 2009), and communication towers (Avery et al. 1978, Gehring et al. 2009). Fence collision has been identified as a substantial source of mortality in some areas for lesser prairie chickens (*Tympanuchus pallidicinctus*; Wolfe et al. 2007). Although elevated infrastructures such as fences and power lines are abundant on western rangelands, their impacts on sagebrush obligates like greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse) are not well understood (Connelly et al. 2000, 2004). Connelly et al. (2000) suggested flagging fences to increase their visibility near sage-grouse habitats if the fences appear hazardous, but no studies have evaluated the degree of hazard fences present to sage-grouse. Many populations of shrub-steppe birds are believed to be declining (Knick et al. 2003), including sage-grouse, and elevated structures have been hypothesized to contribute to sage-grouse declines (Braun 1998, Connelly et al. 2004). Despite the lack of information concerning collision risk of sage-grouse, collision with fences and power lines has been documented (Beck et al. 2006, Flake et al. 2010), and both fences and power lines have proven a significant source of mortality for other species of tetraonids (Baines and Summers 1997, Bevanger and Brøseth 2000, Moss et al. 2000, Wolfe et al. 2007). Conservation concerns for sage-grouse have made it necessary to estimate avian collision rates in sagebrush habitats and to quantify collision risk across the landscape.

A commonly used method of estimating collision rates of birds with elevated infrastructure involves searchers walking along power line corridors or fence sections to locate collision sites (Baines and Summers 1997, Bevanger and Brøseth 2004). Although this method is effective for determining collision rates over large areas, there are 2 major biases associated with searching for collisions in this manner: detectability and scavenging bias (Bevanger 1999).

Detectability bias is common in mortality studies and may be influenced by factors such as meteorological conditions, snow cover, size of the bird under study, local vegetation, and the ability of the observers (Bevanger 1999). Scavenging bias refers to survival of the collision evidence (i.e. temporal longevity of evidence at a collision site), which is often removed by scavenging animals or weather prior to observation during sampling (Bevanger 1999, Smallwood 2007). Because survival of collision evidence depends on more than scavenging alone, we refer to this as survival bias.

Numerous attempts to quantify carcass retention rates have been made and usually involve monitoring collision victims or planted carcasses (James and Haak 1979, Bevanger et al. 1994, Savereno et al. 1996). Most previous studies of carcass retention suggest birds were placed randomly; however, these studies often lacked a strong experimental design or presented vague descriptions of their methods (e.g. Pain 1991, Bevanger and Brøseth 2004).

For example, Baines and Summers (1997) reported that 18 of 20 red grouse (*Lagopus lagopus scoticus*) carcasses placed near fences were gone after 1 month, however those authors did not provide a detailed description of field placement methods. Accuracy and applicability of scavenging rates calculated in many studies are also hindered by small sample sizes ($n \leq 20$; e.g. Savereno et al. 1996, Baines and Summers 1997). In addition to small sample sizes and vague descriptions of methodology, Smallwood (2007) suggested that much of the previous research has not asked the correct questions, often not attempting to address factors influencing carcass survival.

Although most studies estimating carcass retention only considered carcass survival as a function of time, several studies have evaluated the influence of other factors. Bumann and Stauffer (2002) suggested that birds with exposed viscera likely provided stronger olfactory stimulus to scavengers. One study using songbird carcasses reported differences in carcass retention among different habitat types (Kostecke et al. 2001). Furthermore, Smallwood (2007) found evidence for seasonal variation in scavenging rates of birds killed by wind facilities, with the greatest scavenging rates occurring in fall.

Much like estimates of carcass retention, estimates of carcass detectability commonly accompany avian collision mortality studies (Anderson 1978, Savereno et al. 1996, Bevanger 1999). Although most of these studies have reported detectability rates to correct mortality estimates, many have failed to quantify the factors that influence detectability. Osborn et al. (2000) evaluated the influence of season, vegetation height, and snow cover on detectability of bird carcasses on wind facilities in Minnesota. Detectability was only influenced by size of the bird, with large birds having a higher detection rate (92.3%) than small birds (68.7%;

Osborn et al. 2000). Additionally, Smallwood (2007) examined influences of vegetation height and bird group on detectability of wind turbine collision victims and found that detectability varied by bird group (e.g. large raptors, large non-raptors), whereas vegetation height only influenced detection for small non-raptor birds.

Conservation concerns associated with elevated infrastructure and greater sage-grouse have made it desirable to quantify avian fence collision rates in sagebrush-steppe habitats. Proper estimation of collision rates requires quantifying various sources of bias associated with searching for and detecting collision victims along infrastructure (Bevanger 1999, Kuvlesky et al. 2007). Therefore, my objectives were to: 1) estimate surrogate sage-grouse carcass and collision sign survival and detectability in sagebrush-steppe habitats and 2) determine the influence of habitat composition, distance of surrogate sage-grouse carcass from fence, and road presence on carcass and collision sign survival and detectability.

STUDY AREA

I replicated this study on 2 geographic regions of southern Idaho (Fig. 1). I initiated studies to determine retention of pheasant carcasses on 27 March 2009 and 7 April 2009 on the Browns Bench and Upper Snake study areas. The Browns Bench region was in southern Twin Falls County and bordered Nevada. Elevations on Browns Bench ranged from approximately 1,450-1,850 m, and vegetation was dominated by little sagebrush (*Artemisia arbuscula*) and black sagebrush (*A. nova*) in the southern portion of the study area and big sagebrush (*A. tridentata*) in the northern areas. However, habitat conditions were variable and ranged from dense stands of sagebrush to bare pasture and large stands of crested wheatgrass (*Agropyron cristatum*). Cheatgrass (*Bromus tectorum*) was also common on the study area. Additionally, riparian areas in the southern portion of Browns Bench commonly contained stands of big sagebrush.

The Upper Snake study site occurred on the Table Butte and Crooked Creek areas of the Upper Snake River Plain, in Clark and Jefferson counties. I selected the Table Butte area for big sagebrush habitat and the Crooked Creek area for little sagebrush habitat because I could not locate big and little sagebrush-dominated areas at the same site in Upper Snake region. The Table Butte and Crooked Creek areas were approximately 10 km apart, with similar elevations ranging from approximately 1,520-1,825 m. The big sagebrush study site was directly north of Table Butte, bounded on the east by United States Highway 15, and extended to the north and west of Table Butte by approximately 2.5 km and 8 km, respectively. This site was dominated by big sagebrush, with an approximately 900-ha stand of crested wheatgrass. The little sagebrush site in the Crooked Creek drainage was bounded on the south by Idaho Highway 22 and on the north, east, and west by the Beaverhead Mountains. This site was dominated by little sagebrush, with some pasture and grassy areas intermixed. Furthermore, unlike the Browns Bench study site, habitat types in the Upper Snake study area were distinctly separated, such that carcasses placed in big sagebrush were not as near to those placed in little sagebrush as on the Browns Bench study site.

Avian, mammalian, and arthropod scavengers were common on the study areas. Common scavengers previously identified in the vicinity of our study sites included coyotes (*Canis latrans*), American badgers (*Taxidea taxus*), weasels (*Mustela* spp.), common ravens (*Corvus corax*), American crows (*Corvus brachyrhynchos*), and black-billed magpies (*Pica hudsonia*; Coates et al. 2008). Small mammal species previously identified included least chipmunks (*Tamias minimus*), Wyoming ground squirrels (*Spermophilus elegans*), Piute ground squirrels (*S. mollis*), northern pocket gophers (*Thomomys talpoides*), Great Basin pocket mice (*Perognathus parvus*), deer mice (*Peromyscus maniculatus*), and sagebrush voles (*Lemmiscus curtatus*; Coates et al. 2008). I observed additional predators on my study sites, including golden eagles (*Aquila chrysaetos*) and multiple species of hawks (*Buteo* spp.) and owls (*Asio* spp., *Athene* sp.). On the Upper Snake study site I identified bobcat (*Lynx rufus*) tracks near carcass locations and commonly observed scavenging by arthropods (Orders Coleoptera, Diptera, and Hymenoptera).

METHODS

Field Methods

I used pen-raised female ring-necked pheasants (*Phasianus colchicus*) as experimental units to evaluate factors influencing survival and detectability of fence collision victims in sagebrush-steppe habitats. I used a completely randomized design (Ott and Longnecker 2001) with 2 levels of treatment effects for habitat type (big sagebrush or little and black sagebrush) and carcass distance from the fence as a covariate, with carcasses placed at random distances of 0-15 m from a fence. I euthanized all birds using cervical dislocation, with approval from the University of Idaho Animal Care and Use Committee (Protocol 2009-21).

I banded all carcasses for individual identification prior to field placement to aide in monitoring. To prevent scavenging bias associated with unrealistic carcass presentation (Bumann and Stauffer 2002), I removed feathers from the front of the breast of each carcass and made 2 perpendicular 4-cm incisions, centered where the feathers were removed from the breast, to simulate collision with a barbed-wire fence. I placed carcasses in coolers on ice until field placement. A technician wearing rubber boots and gloves to minimize human scent on the carcasses (Whelan et al. 1994) placed each carcass and feather pile (*n* = 10-15 feathers) in the field at night to minimize diurnal scavenger detection by simply observing field workers. I placed each carcass at random distances perpendicular to the center point of the randomly selected fence segment to facilitate site relocation by researchers. Additionally, I placed 10-15 feathers immediately around each carcass. I also placed lone piles of 10-15 breast feathers to determine feather-pile detectability, as it could differ from carcass detectability.

I measured the influence of habitat type and distance from fence on detectability and longevity for the entire collision sign in addition to carcasses. Placing feathers around each carcass prevented it from being removed and leaving no visible sign, an unlikely scenario with collision victims, as feathers will likely fall when birds strike fences (Flake et al. 2010). I replicated this design on both the Browns Bench and Upper Snake study areas to allow detection of regional differences in carcass survival. I only measured detectability on the Browns Bench site due to a lack of field personnel available to conduct detectability trials on the Upper Snake study area.

I quantified available fences in each habitat type on each study area using ground searches, handheld Global Positioning System (GPS) units and ArcGIS Geographic Information System (GIS) software. I mapped fence sections in each habitat type by taking GPS waypoints along and at the ends of each fence section, then digitizing the fence segments in GIS. Because sagebrush-steppe habitats often have a patchy mosaic of sagebrush, pasture, and grassy areas, I only included fence segments traversing areas dominated on both sides by the desired habitat type. I did not differentiate between little sagebrush and black sagebrush, and I grouped both species together in the little sagebrush level of habitat treatment.

Once I digitized fence sections, I used the GIS extension Hawth's Tools (Hawth's analysis tools for ArcGIS version 3, <http://www.spatialecology.com/htools.>, accessed 12 Jan 2009) to generate random fence points >200 m apart in each habitat type. Bumann and Stauffer (2002) placed ruffed grouse carcasses >100 m apart in their Appalachian study, however, sagebrush-steppe habitats are more open than deciduous forest, so larger inter-carcass distances are likely necessary. Therefore, I ensured that carcass locations were ≥ 200 m apart. I placed 50 bird carcasses on each study area with 25 replicates for each treatment level (big and little sagebrush). In addition to the 50 points generated on each area for placement of pheasant carcasses, I similarly generated 50 points (25 for each treatment level) on the Browns Bench study area for placement of feather piles used in detectability trials. I ground verified all randomly generated points were in the desired habitat type prior to field placement, and discarded points not in the desired habitat. I placed carcasses and feather piles on the Browns Bench site randomly along 31.7 km of fence (little sagebrush = 18.2 km, big sagebrush = 13.5 km). Less fence was available on the Upper Snake site, so I placed carcasses randomly along 22.9 km of fence (little sagebrush = 11.2 km, big sagebrush = 11.7 km). I did not use fence sections running along paved roads, which could bias results if scavengers used or avoided improved road corridors. I did include fence sections running along unimproved 2-track dirt or gravel roads due to the abundance of these roads on the study areas, and treated road presence as a random covariate in this analysis. I placed 32 carcasses along fences with a road present (Upper Snake = 21, Browns Bench = 11), with more in little sagebrush (*n* = 23) than in big sagebrush habitats (*n* = 9).

The day following placement of carcasses on the Browns Bench site, observers searched all study fence segments to estimate detection probabilities. Two observers walked each fence section (1 on each side of the fence or 1 observer walked both sides in turn) searching for bird carcasses and sign within approximately 15 m of the fence, while monitoring the fence itself for presence of feathers or bird parts. Field observers searched all potential fence sections digitized for the study, both with and without planted carcasses and feather piles, to eliminate detection bias by workers expecting bird locations. Furthermore, field observers searched extra fence sections not included in the random point generation without knowledge of which sections were included in the study. The technician who planted the birds was not involved in searches but did verify the presence of birds not detected by field searchers on day one.

After initial searches I monitored carcasses and their remaining sign every 1-3 days until removal, for a maximum of 31 days. During each monitoring period the observer recorded one of the following: a) intact carcass, b) carcass scavenged but present, c) carcass removed but feathers or sign still present, or d) all collision evidence removed. Additionally, the observer qualitatively described the carcass sign and remaining feathers over time within approximately 5 m of the original carcass location. The observer also recorded presence of any precipitation events that could influence carcass retention or detection of remaining sign, such as snow or rain, at the start of each search. Observers also noted any obvious scavenger sign or individual scavengers detected near carcasses during each search.

I recorded microhabitat characteristics of carcass placement sites after carcass removal to determine influences of vegetation on survival and detectability. I measured grass height and shrub height at the carcass location and 1 m from the carcass location in each cardinal direction (Hausleitner et al. 2005). I measured shrub canopy coverage on 2 perpendicular 4-m transects centered on the carcass location and oriented in each cardinal direction using the line-intercept method (Canfield 1941). Additionally, I used a 12 × 12 cm coverboard to estimate percent visual concealment at heights of 1.5 m and 1.0 m, at a distance of 10 m in each cardinal direction from the carcass location (Jones 1968, Hausleitner et al. 2005). Due to observer error, sign from one carcass on the Browns Bench study area was not monitored until complete removal occurred; therefore all calculations regarding sign survival are with a sample size of 99 birds, whereas carcass survival analysis included all 100 experimental units.

Statistical Methods

I conducted survival analysis for hypothetical collision victims using the nest survival module in Program MARK (White and Burnham 1999). Because I monitored survival of carcasses analogously to monitoring of avian nests, with variable time intervals and binary Bernoulli trials for each monitoring event, the nest survival module was an appropriate model for rigorous survival estimation. I followed the terminology of Dinsmore et al. (2002), where daily survival rate is the probability the carcass will survive one day, and survival probability is the probability of survival over the entire study period. I generated all survival models using the logit link function, such that

$$\operatorname{logit}\left(\widehat{S}\right) = \ln\left(\frac{\widehat{S}}{1-\widehat{S}}\right) = \widehat{\beta}_0 + \widehat{\beta}_1\left(X_1\right) + \ldots + \widehat{\beta}_k\left(X_k\right)$$

where $\widehat{S}$ is estimated daily survival probability, $\widehat{\beta}_i$ are linear model coefficients, and X_i are independent predictor variables. Furthermore, I calculated reconstituted daily survival rate estimates by back transforming the given model, where

$$\widehat{S} = \frac{e^{\widehat{\beta}_0 + \widehat{\beta}_1(X_1) + \ldots + \widehat{\beta}_k(X_k)}}{1 + e^{\widehat{\beta}_0 + \widehat{\beta}_1(X_1) + \ldots + \widehat{\beta}_k(X_k)}}$$

and the linear model is the generalized linear model under consideration. Lastly, I calculated variances of reconstituted survival rates using the delta method (Seber 1982).

I conducted all modeling within an information-theoretic model selection framework (Burnham and Anderson 2002). I ranked hypothesized models using Akaike's Information Criterion corrected for small sample sizes ($\mathrm{AIC_c}$; Burnham and Anderson 2002). I estimated overdispersion ($\widehat{c}$) in carcass survival models by the deviance to deviance degrees of freedom ratio for the global model. This estimate is often biased high for small sample sizes (McCullagh and Nelder 1989) but is currently the only way to assess goodness-of-fit for the nest survival model in Program MARK (Dinsmore et al. 2002). When modeling indicated overdispersion ($\widehat{c} > 1$) I ranked models using quasi-AIC corrected for small sample size ($\mathrm{QAIC_c}$; Burnham and Anderson 2002). I used information-theoretic methods to compare models instead of likelihood-ratio testing to permit comparison of non-nested models. Additionally, I used normalized Akaike model weights (w_i) as a measure of strength of evidence for a given model and generated model averaged daily survival rates and survival probabilities to account for uncertainty in model selection procedures (Burnham and Anderson 2002). I calculated unconditional variances (Buckland et al. 1997) for model averaged survival rates in Program MARK.

I constructed survival models for both planted carcasses and collision sign. I determined survival for a carcass by the length of time until a planted carcass was first found scavenged. Similarly, I determined survival for all collision sign by the length of time until < 5 feathers were found present at the original carcass location. Daily survival rate of all collision sign was of interest because it has direct application to the appropriate time-interval lengths between fence-line surveys for avian collision sites.

I constructed survival models using a priori hypothesized local-scale and microhabitat characteristics (Table 1). Local-scale factors used in model building included the habitat type treatment and the random covariates for road presence and distance of the carcass from the fence. Site was also a 2-level factor included in the local-scale models because I replicated the field experiment on 2 study areas. Survival modeling for local-scale factors included comparison of 16 additive models, using all combinations of the 4 independent variables, as well as the constant survival model (Appendix A). Furthermore, I hypothesized 3 biologically plausible 2-way interactions for these models (site × habitat, site × road, and habitat × road) and added them only when the terms in the interaction were in a model together among the top group of models ($\Delta AIC_c \approx \leq 2$; Table 1).

I used a separate group of hypotheses and models to evaluate the importance of features at the microhabitat scale at carcass locations. Because these factors represented a different scale, and were not factors we directly considered in the design of the study, these models represented a separate group of hypotheses that we did not directly compare to the local-scale models. Variables included in microhabitat scale modeling were average grass height, average shrub height, percent visual concealment (measured as total proportion of coverboard blocks concealed), and average percent canopy coverage (Table 1). I tested for correlation between microhabitat predictor variables using correlation t-tests. I did not include microhabitat predictor variables that were significantly correlated ($P < 0.05$) together in the same model. I hypothesized 2 biologically plausible 2-way interactions in microhabitat models (grass height × shrub height, grass height × canopy coverage) and again added these terms only when the individual terms were in a model together among the top models (Table 1). Correlation in microhabitat variables resulted in comparison of 10 models at this scale, with the global model included to estimate overdispersion in the model set (Appendix A). Candidate model sets I used for each group of hypotheses were identical for both carcass survival and sign survival modeling.

I conducted logistic regression modeling to determine the influence of features on carcass detectability using the known fate model in Program MARK. Because this model assumes perfect detection of individuals, known fate survival estimation using only 1 time interval is identical to estimating success probability from a binomial likelihood model (i.e. probability given event occurred). For example, the maximum likelihood estimate of survival for a known fate model with 1 time interval is

$$\widehat{S} = \frac{x}{n}$$

where *n* is the number of individuals alive at the start of the interval and *x* is the number of individuals alive at the end of the interval. This is identical to the maximum likelihood estimator for a binomial model, and as such, when used in conjunction with the logit link function, is identical to logistic regression in standard statistical software. I specifically used Program MARK in this analysis to facilitate model selection and calculation of model weights, model averaged parameter estimates, and reconstituted parameter estimates.

I generated logistic regression models of carcass detection probabilities using both local-scale and microhabitat variables. Local-scale features hypothesized to influence detection probability during fence-line surveys included habitat type, distance of the carcass from the fence, and observer experience (experienced vs. inexperienced; Table 1). Additionally, a snowstorm abruptly developed during one detectability trial, severely limiting visibility. Therefore, I included a parameter for snow conditions to accommodate this confounding factor. I compared all 15 combinations of additive models and the constant detectability model for the experimental factors, and I did not consider any interaction terms in this group of hypotheses due to small sample sizes (Appendix B). Microhabitat characteristics I used in modeling were identical to those used in survival analysis (Table 1). I compared 9 models representing features at the microhabitat scale (Appendix B). Once again I constructed these models such that we did not include strongly correlated predictor variables in the same model, and included 2 hypothesized interaction terms (grass height × shrub height, grass height × canopy coverage) when their constituent terms were together among the top group of models (Table 1). I calculated generalized likelihood-ratio r-squared values (R_L^2) for each survival and logistic regression model to determine the performance of individual models relative to the null constant models (Menard 2000). Lastly, I estimated prediction success for each model using crossvalidation procedures in SAS (SAS Version 9.2, Cary, NC). Crossvalidation procedures re-fit each model dropping each data point in turn and subsequently predicted probability of detection for each dropped data point with the re-fit model. If probability of detection was > 0.5 the point was predicted to be observed, and the reported success probability is the proportion of correct classifications in this analysis.

RESULTS

Survival

I monitored persistence of 100 female pheasant carcasses over 2 study areas, and measured local-scale and microhabitat characteristics of the carcass locations. Mean grass height at carcass locations was 10.5 cm (SD = 8.8, *n* = 99), mean shrub height was 23.6 cm (SD = 21.0, *n* = 100), and mean canopy coverage was 22.5% (SD = 13.9%, *n* = 100). For survival models I used visual concealment at a height of 1.0 m, which averaged 88.4% (SD = 16.7%, *n* = 99), whereas visual concealment at 1.5 m, which was used in detectability modeling, averaged 71.0% (SD = 26.0%, *n* = 45). One microhabitat measurement was missing each for grass height and visual concealment, thus for those factors *n* = 99. Shrub height was correlated with canopy coverage (*r* = 0.71), and visual concealment was correlated with all other measurements (*r* = 0.24-0.40), therefore I did not include these variables together in modeling.

Average time to detection of first scavenging of carcasses was 5.8 days (SD 2.9, *n* = 100) for both areas combined and appeared similar between sites (Browns Bench: $\bar{x}$ = 5.6, SD = 3.0, n = 50; Upper Snake: $\bar{x}$ = 6.0, SD = 2.8, n = 50). Average time to detection of first scavenging was 5.1 days for big sagebrush (SD = 2.6, n = 50) and 6.6 days for little sagebrush (SD = 3.0, n = 50) habitats. I observed differences between study sites in persistence of collision sign. Average number of days until sign was no longer detected on the Upper Snake site ($\bar{x}$ = 27.2, SD = 6.8, n = 50) was greater than for the Browns Bench site ($\bar{x}$ = 8.8, SD = 2.0, n = 49), due to many carcasses with sign surviving the entire study period on the Upper Snake site (n = 32; little sagebrush = 18, big sagebrush = 14). In contrast, none of the Browns Bench carcasses had sign persist the entire 31-day sampling period. Due to many carcasses with sign persisting the entire study period, average sign persistence calculated for the Upper Snake site was biased low, and the true average length of sign persistence is unknown.

Pheasant carcasses in little sagebrush habitats or on the Browns Bench study area were more likely to be directly removed during initial scavenging, whereas carcasses in big sagebrush habitats and those on the Upper Snake site were more likely to be scavenged in their original location and not directly removed. On the Browns Bench site 80% of carcasses were directly removed during initial scavenging (big sagebrush = 68%, little sagebrush = 92%), whereas 82% of carcasses on the Upper Snake site were scavenged in their original location (big sagebrush = 88%, little sagebrush = 76%). Furthermore, of the Upper Snake carcasses that had sign persist the entire sampling period, 100% of those located in big sagebrush and 78% of those located in little-sagebrush habitats were first scavenged in their original location and not directly removed. I found evidence of small mammal scavenging at 9 carcasses (Browns Bench = 3, Upper Snake = 6), commonly in the form of tracks and scat on or near the carcass. Carcasses with apparent small mammal scavenging also had patches of feathers plucked or removed, and the underlying tissue appeared gnawed upon. I observed arthropods scavenging at 26 of the Upper Snake carcasses (big sagebrush = 19, little sagebrush = 7), whereas I did not observe scavenging by arthropods at the Browns Bench site. I also observed arthropod scavengers carrying feather evidence away from carcass locations on the Upper Snake site.

None of the hypothesized local-scale parameters received strong support in the carcass survival modeling, and the null constant survival model was most supported by the data ($\Delta\text{QAIC}_\text{c} = 0$, $w_i = 0.238$; Table 2). I found minimal support for both habitat type and road presence influences on carcass survival ($\Delta\text{QAIC}_\text{c} < 2$), however, these models performed nearly identical to the constant survival model in terms of proportional increase in likelihood ($R_L^2 = 0.005\text{-}0.009$). Model averaged survival estimates showed slightly lower carcass daily survival in big sagebrush habitats ($\Delta\text{DSR} = 0.022$) and areas with no roads present ($\Delta\text{DSR} = 0.013\text{-}0.014$) regardless of habitat, however, precision was low and confidence intervals were wide and overlapping (Table 3). Reconstituted daily survival rates for the constant survival model showed low daily survival rates (DSR = 0.794, 95% CI = 0.721-0.851), resulting from the speed at which the carcasses were scavenged.

Similarly, I generated carcass survival models to evaluate the influence of microhabitat characteristics on survival, and these factors received little support from the data (Table 2). The top microhabitat model suggested constant carcass survival ($\Delta\text{QAIC}_\text{c} = 0$, $w_i = 0.318$), and I found weak support for grass height, visual concealment, and shrub canopy coverage influences on carcass survival ($\Delta\text{QAIC}_\text{c} \leq 2$). Again, these models performed nearly identical to the null constant survival model in terms of proportional increase in likelihood (R_L^2 = 0-0.007).

Local-scale sign survival modeling suggested survival was most influenced by study site (ΔQAIC$_c$ = 0, w_i = 0.341), and the site effect model was approximately 2.4 times more likely than the second best model (Table 4). I also found minimal support for both habitat type and distance of the carcass from the fence effects on sign survival in addition to study area effects (ΔQAIC$_c$ < 2). Local-scale sign survival models that did not contain a site effect parameter received virtually no support in this analysis (ΔQAIC$_c$ > 25), and all of the top models performed better than the null constant survival model in terms of proportional increase in likelihood (R_L^2 = 0.226-0.228). Model averaged sign survival rates showed strong differences between study sites, with daily survival rates ranging from 0.987-0.988 on the Upper Snake site and from 0.863-0.872 on the Browns Bench site (Table 5). Differences in model averaged sign daily survival rates resulted in reduced model averaged sign survival probabilities for the entire 31-day period from the Upper Snake (0.673-0.699) to Browns Bench (0.011-0.015) study sites. Regression coefficient estimates for the site effect model again showed the strong influence of site on sign daily survival rates (β_1 = -2.528, 95% CI = -3.566 - -1.490), with lower survival at the Browns Bench site.

Similarly, sign survival models evaluated the influence of fine scale microhabitat features on daily survival rates. None of the hypothesized microhabitat features received strong support in this analysis, and the null constant survival model was most supported by the data (ΔQAIC$_c$ = 0, w_i = 0.228; Table 4). I found weak support for the influence of all microhabitat variables on sign survival, but again these models performed similarly to the null model in terms of proportional increase in likelihood (R_L^2 = 0.003-0.023; Table 4).

Detection

Because I planted carcasses the night preceding detection trials, some carcasses were not present during trials. Three carcasses were removed prior to detection trials (2 in big sagebrush, 1 in little sagebrush), and 1 carcass in big sagebrush was buried under a snow drift by the time of the trial, therefore we excluded 4 carcasses from our calculations. Furthermore, I did not use 10 of the original feather piles placed, which were either blown away or covered with snow prior to initiating detection trials; however, the 3 carcasses removed all left feather piles and I used these feathers in feather pile detection calculations. Detection probability for feather piles was extremely low, only 1 of 43 (2.3%) feather piles present was located. I constructed no detectability models for feather piles because such a small proportion was located. The total proportion of detected carcasses was 0.54 (*n* = 46) and was higher in little sagebrush (0.71) than in big sagebrush (0.36) habitats. Detection probability for 6 of the 22 carcasses in big sagebrush and 1 of the 24 carcasses in little sagebrush may have been confounded by snowfall during sampling. Excluding these carcasses, the proportion of carcasses in big sagebrush detected rises to 0.44. The proportion of detected carcasses was similar for experienced (54.8%, *n* = 31 carcasses) and inexperienced (53.3%, *n* = 15 carcasses) field searchers.

I used logistic regression models to evaluate local-scale and microhabitat factors influencing carcass detection probability during fence-line surveys. Habitat type influences on detectability were most supported by the data at the local scale ($\Delta AIC_c = 0$, $w_i = 0.306$;

Table 6), however, I found additional weak support for the influence of snow presence and carcass distance from the fence on detectability ($\Delta AIC_c < 2$). The top local-scale models all predicted success moderately (0.674; Table 6), and all 3 top models performed identically in this measure. Model averaged detection probabilities were 0.67 for little sagebrush (95% CI = 0.43-0.85) and 0.40 for big sagebrush (95% CI = 0.20-0.65) habitats, and the regression coefficient from the habitat type model showed a positive influence of little sagebrush habitats on detection probability ($\beta_1 = 1.447$, 95% CI = 0.210-2.684).

In the analysis of microsite habitat characteristics on detection of carcasses during fence-line surveys the influence of shrub height on detectability was most supported by the data ($\Delta AIC_c = 0$, $w_i = 0.323$; Table 6). I found less support for the influence of shrub canopy cover, and grass height in addition to shrub height, on detectability ($\Delta AIC_c < 2$; Table 6). Both shrub height and canopy cover reduced detection probability (Fig. 2), and the confidence interval for the regression coefficient from the shrub height model did not include zero ($\beta_1 = -0.039$, 95% CI = -0.077 - -0.002). Both shrub height and canopy cover predicted detection success moderately well, however, the canopy cover model performed slightly better (0.652) than the shrub height model (0.609).

DISCUSSION

Survival

Carcasses experienced rapid scavenging on both study areas, which produced low daily survival rates. Although the speed at which carcasses were scavenged was similar between study sites, the manner of scavenging, daily survival rate of the collision sign as a whole, and subsequent longevity of the collision sign differed between study areas.

Additionally, microhabitat characteristics performed poorly in both carcass and sign survival models, suggesting broad scale site or landscape features may have a stronger influence on survival of collision evidence.

Rapid scavenging occurred in many scavenging studies, and is common for both avian (Crawford 1971, Houston 1986, Peterson et al. 2001) and mammalian carcasses (Heinrich 1988, Travaini et al. 1998). Crawford (1971) reported 93% of bird carcasses planted around a television tower in Florida were scavenged during the first night of observation. Houston (1986) studied scavenging by turkey vultures (*Cathartes aura*) in tropical forests and reported 96% of chicken carcasses were scavenged within 3 days. In British Columbia, 52 of 54 waterfowl carcasses placed in agriculture fields to simulate poisoning were removed within 72 hours (Peterson et al. 2001). The ability to locate and consume carrion rapidly provides benefits to many predator and scavenger species, and most vertebrate predators will opportunistically scavenge fresh carrion when available (Devault et al. 2003). Both coyotes and common ravens are opportunistic scavengers common to shrub-steppe dominated landscapes (Hilton 1978, Heinrich 1988, Coates et al. 2008). Thus, the rapid scavenging observed should not be surprising and likely occurs on many areas within shrub-steppe habitats.

In contrast to the rapid rate of initial scavenging on both study areas, overall sign survival rates differed between study sites. Similarly, 50% of planted bobwhite quail (*Colinus virginianus*) remains were completely removed within 4 days in Alabama, whereas only 13% of bobwhite quail remains in Texas were completely removed in that period (Rosene and Lay 1963). In addition to differences in sign survival rates, I recorded differences in the way carcasses were scavenged between study sites, resulting in differences in the remaining carcass evidence. Carcasses that were directly removed often had few or no feather evidence remaining at the site, whereas carcasses scavenged at their initial location often had large feather piles (≥100 feathers) and bird pieces scattered around the site. Similarly, Rosene and Lay (1963) found that large feather piles disappeared at a slower rate than small feather piles, which is consistent with my observations and suggests the way a carcass is scavenged may influence overall survival of sign at the original location.

Although site scale differences between study areas had a large influence on sign survival, microhabitat characteristics performed poorly in both carcass and sign survival models. Previous research has produced variable results with respect to the influence of habitat features on avian carcass persistence. Pain (1991) reported mallard (*Anas platyrhynchos*) carcass longevity was significantly lower for exposed carcasses than those concealed by vegetation. In contrast, Bumann and Stauffer (2002) found no relationships between scavenging of ruffed grouse (*Bonasa umbellus*) carcasses and habitat characteristics.

Detection

In contrast to survival models, habitat characteristics did influence detection probability of carcasses during fence-line surveys. Similarly, Tobin and Dolbeer (1990) indicated that the lowest detection rate (50%) for songbird carcasses in New York fruit orchards occurred at the site with the greatest ground cover. Smallwood (2007) summarized results from 10 unpublished reports at wind facilities and found detection appeared to vary by bird group (e.g. large raptors, large non-raptors), whereas vegetation only influenced detection for birds classified as small birds. Overall, my detection rate (0.53) appears low compared to previous published studies in other habitats. Savereno et al. (1996) found 66% and 73% of planted bird carcasses in a power-line corridor in coastal South Carolina, and Osborn et al. (2000) reported detection rates varied from 68.7% for small birds to 92.3% for large birds on a Minnesota wind facility.

My detection rate of 2.3% for feather piles may be unrealistically low, as I located 4 actual avian collision sites during trials (B.S. Stevens, University of Idaho, unpublished data), 3 of which were feather piles. Alternatively, if my measured detection probability for feather piles does accurately represent true feather pile detection, it suggests presence of many collision sites.

The applicability of my results to fence-line surveys for sage-grouse collision victims relies on the assumption that detection and survival probabilities are similar for sage-grouse and female pheasants. Smallwood (2007) suggested researchers use the species of interest to avoid misleading results and application. However, Gehring et al. (2009) successfully used surrogate songbird carcasses when evaluating avian collision with communication towers in Michigan. Conservation concerns surrounding sage-grouse prevented us from obtaining 100 grouse carcasses, and greater sage-grouse were recently listed as warranted but precluded under the Endangered Species Act by the United States Fish and Wildlife Service (United States Department of the Interior 2010). Pheasant carcasses used were similar to female sage-grouse in body size and cryptic plumage, which should eliminate potential bias caused by these factors (Osborn et al. 2000, Smallwood 2007). Although a potential source of error could arise from differences in coloration between female pheasants and male sage-grouse, I am unaware of any studies that fully quantify the effects of small changes in coloration on avian carcass detection probability. Linz et al. (1991) reported significantly more male (83%) than female (78%) red-winged blackbirds (*Agelaius phoeniceus*) were found during carcass searches in cattail marshes, however, these results were not consistent across trials.

Sample sizes and length of survival trials used in carcass studies can also introduce bias in survival estimates due to predator swamping and data censoring (Smallwood 2007, Smallwood et al. 2010). I distributed carcasses across large geographic areas, which should have reduced the potential for predator swamping. Further, the ability of mammalian predators to remove and cache carcasses (Stoddart 1970, Prior and Weatherhead 1991) should reduce the influence of predator swamping on carcass removal rates. Sign survival on the Upper Snake site was high due to many carcasses with evidence persisting throughout the entire study. Censoring sign survival data at the end of the study could lead to biased survival estimates (Smallwood 2007), and caution should be used extrapolating our results past the 31-day sampling interval length. Smallwood (2007) suggested carcass survival studies monitor all carcasses on a daily basis. However, rigorous statistical models are available for unbiased estimation of daily survival rates for variable time interval monitoring (Dinsmore et al. 2002, Rotella et al. 2004, Shaffer 2004), and longer intervals between monitoring periods would likely reduce the chances for observer effects on daily survival rates (e.g. Rotella et al. 2000).

MANAGEMENT IMPLICATIONS

I provide the first estimates of avian carcass survival and detection probabilities associated with fence-line surveys in sagebrush-steppe habitats. Recommendations for standardized searches are difficult given the regional variability in collision sign survival documented. However, low daily survival rates for carcasses and collision sign suggest time-interval lengths between surveys should be ≤ 1-2 weeks to avoid potential negative effects of survival bias on collision rate estimation. For small-scale studies it may be possible to sample on a weekly or bi-weekly basis, however, studies estimating collision rates at landscape scales may not be able to sample at such frequencies. Regional variation in sign survival also suggests the need to estimate survival on all study areas. Survey protocols should be standardized to avoid sampling when weather conditions are poor and could influence detection probabilities (e.g. snow cover, extreme wind or rain), and collision rates should be corrected based on identified site attributes influencing detection. Moreover, caution is warranted when aggregating or comparing un-corrected collision data from sites with varying vegetation characteristics, as detection probabilities are likely different between sites.

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LITERATURE CITED

Avery, M. L., P. F. Springer, and J. F. Cassel. 1978. The composition and seasonal variation of bird losses at a tall tower in southeastern North Dakota. American Birds 32:1114-1121.

Anderson, W. L. 1978. Waterfowl collisions with power lines at a coal-fired power plant. Wildlife Society Bulletin 6:77-83.

Baines, D., and R. W. Summers. 1997. Assessment of bird collisions with deer fences in Scottish forests. Journal of Applied Ecology 34:941-948.

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.

Bevanger, K. 1995. Estimates and population consequences of tetraonid mortality caused by collisions with high tension power lines in Norway. Journal of Applied Ecology 32:745-753.

Bevanger, K. 1999. Estimating bird mortality caused by collision with power lines and electrocution: a review of methodology. Pages 29-56 *in* M. Ferrer and G. F. E. Janss, editors. Birds and Power Lines: Collision, Electrocution and Breeding. Quercus, Madrid, Spain.

Bevanger, K., and H. Brøseth. 2000. Reindeer *Rangifer tarandus* fences as a mortality factor for ptarmigan *Lagopus* spp. Wildlife Biology 6:121-127.

Bevanger, K., and H. Brøseth. 2004. Impact of power lines on bird mortality in a subalpine area. Animal Biodiversity and Conservation 27:67-77.

Bevanger, K., Ø. Bakke, and S. Engen. 1994. Corpse removal experiments with willow ptarmigan (*Lagopus lagopus*) in power-line corridors. Ökologie der Vögel 16:597-607.

Bradley, L. C., and D. B. Fagre. 1988. Coyote and bobcat responses to integrated ranch management practices in south Texas. Journal of Range Management 41:322-327.

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.

Buckland, S. T., K. P. Burnham, and N. H. Augustin. 1997. Model selection: an integral part of inference. Biometrics 53:603-618.

Bumann, G. B., and D. F. Stauffer. 2002. Scavenging of ruffed grouse in the Appalachians: influences and implications. Wildlife Society Bulletin 30:853-860.

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.

Canfield, R. H. 1941. Application of the line interception method in sampling range vegetation. Journal of Forestry 39:388-394.

Coates, P. S., and D. J. Delehanty. 2010. Nest predation of greater sage-grouse in relation to microhabitat factors and predators. Journal of Wildlife Management 74:240-248.

Coates, P. S., J. W. Connelly, and D. J. Delehanty. 2008. Predators of greater sage-grouse nests identified by video monitoring. Journal of Field Ornithology 79:421-428.

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, Cheyenne, Wyoming, USA.

Crawford, R. L. 1971. Predation on birds killed at a TV tower. Oriole 36:33-35.

DeVault, T. L., O. E. Rhodes and J. A. Shivik. 2003. Scavenging by vertebrates: behavioral, ecological, and evolutionary perspectives on an important energy transfer pathway in terrestrial ecosystems. Oikos 102:225-234.

Dinsmore, S. J., G. C. White, and F. L. Knopf. 2002. Advanced techniques for modeling avian nest survival. Ecology 83:3476-3488.

Drewitt, A. L., and R. H. W. Langston. 2008. Collision effects of wind-power generators and other obstacles on birds. Year in Ecology and Conservation Biology 2008 *in* Annals of the New York Academy of Sciences 1134:233-266.

Flake, L. D., J. W. Connelly, T. R. Kirschenmann, and A. J. Lindbloom. 2010. Grouse of plains and mountains: the South Dakota story. South Dakota Department of Game, Fish, and Parks, Pierre, South Dakota, USA.

Gehring, J., P. Kerlinger, and A. M. Manville. 2009. Communication towers, lights, and birds: successful methods of reducing the frequency of avian collisions. Ecological Applications 19:505-514.

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.

Hausleitner, D., K. P. Reese, and A. D. Apa. 2005. Timing of vegetation sampling at greater sage-grouse nests. Rangeland Ecology and Management 58:553-556.

Heinrich, B. 1988. Winter foraging at carcasses by three sympatric corvids, with emphasis on recruitment by the raven, *Corvus corax*. Behavioral Ecology and Sociobiology 23:141-156.

Hilton, H. 1978. Systematics and ecology of the eastern coyote. Pages 220-227 *in* M. Bekoff, editor. Coyotes: Biology, Behavior, and Management. The Blackburn Press, Caldwell, New Jersey, USA.

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.

Houston, D. C. 1986. Scavenging efficiency of turkey vultures in tropical forest. Condor 88:318-323.

James, B. W., and B. A. Haak. 1979. Factors affecting avian flight behavior and collision mortality at transmission lines. Bonneville Power Administration, Portland, Oregon, USA.

Janss, G. F. E., and M. Ferrer. 2000. Common crane and great bustard collision with power lines: collision rate and risk exposure. Wildlife Society Bulletin 28:675-680.

Jones, R. E. 1968. A board to measure cover used by prairie grouse. Journal of Wildlife Management 32:28-31.

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.

Kostecke, R. M., G. M. Linz, and W. J. Bleier. 2001. Survival of avian carcasses and photographic evidence of predators and scavengers. Journal of Field Ornithology 72:439-447.

Kuvlesky, W. P., L. A. Brennan, M. L. Morrison, K. K. Boydston, B. M. Ballard, F. C. Bryant. 2007. Wind energy development and wildlife conservation: challenges and opportunities. Journal of Wildlife Management 71:2487-2498.

Linz, G. M., J. E. Davis Jr., R. M. Engeman, D. L. Otis, and M. L. Avery. 1991. Estimating survival of bird carcasses in cattail marshes. Wildlife Society Bulletin 19:195-199.

McCullagh, P., and J. A. Nelder. 1989. Generalized linear models. Second edition. Chapman and Hall, New York, New York, USA.

Menard, S. 2000. Coefficients of determination for multiple logistic regression analysis. American Statistician 54:17-24.

Moss, R., N. Picozzi, R. W. Summers, and D. Baines. 2000. Capercaillie *Tetrao urogallus* in Scotland – demography of a declining population. Ibis 142:259-267.

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.

Osborn, R. G., K. F. Higgins, R. E. Usgaard, C. D. Dieter, and R. D. Neiger. 2000. Bird mortality associated with wind turbines at the Buffalo Ridge Wind Resource Area, Minnesota. American Midland Naturalist 143:41-52.

Ott, R. L., and M. Longnecker. 2001. An introduction to statistical methods and data analysis. Fifth edition. Wadsworth Group-Thompson Learning Inc., Pacific Grove, California, USA.

Pain, D. J. 1991. Why are lead-poisoned waterfowl rarely seen? the disappearance of waterfowl carcasses in the Camargue, France. Wildfowl 42:118-122.

Peterson, C. A., S. L. Lee, and J. E. Elliot. 2001. Scavenging of waterfowl carcasses by birds in agricultural fields of British Columbia. Journal of Field Ornithology 72:150-159.

Prior, K. A., and P. J. Weatherhead. 1991. Competition at the carcass: opportunities for social foraging by turkey vultures in southern Ontario. Canadian Journal of Zoology 69:1550-1556.

Rosene, W., and D. W. Lay. 1963. Disappearance and visibility of quail remains. Journal of Wildlife Management 27:139-142.

Rotella, J. J., M. L. Taper, and A. J. Hansen. 2000. Correcting nest-success estimates for observer effects: maximum-likelihood estimates of daily survival rates with reduced bias. Auk 117:92-109.

Rotella, J. J., S. J. Dinsmore, and T. L. Shaffer. 2004. Modeling nest-survival data: a comparison of recently developed methods that can be implemented in MARK and SAS. Animal Biodiversity and Conservation 27:187-205.

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.

Seber, G. A. F. 1982. The estimation of animal abundance and related parameters. Second edition. Macmillan Book, New York, New York, USA.

Shaffer, T. L. 2004. A unified approach to analyzing nest success. Auk 121:526-540.

Smallwood, K. S. 2007. Estimating wind turbine-caused bird mortality. Journal of Wildlife Management 71:2781-2791.

Smallwood, K. S., and C. Thelander. 2008. Bird mortality in the Altamont Pass Wind Resource Area, California. Journal of Wildlife Management 72:215-223.

Smallwood, K. S., L. Rugge, and M. L. Morrison. 2009. Influence of behavior on bird mortality in wind energy developments. Journal of Wildlife Management 73:1082-1098.

Smallwood, K. S., D. A. Bell, S. A. Snyder, and J. E. DiDonato. 2010. Novel scavenger removal trials increase wind turbine-caused avian fatality estimates. Journal of Wildlife Management 74:1089-1097.

Stoddart, L. C. 1970. A telemetric method for detecting jackrabbit mortality. Journal of Wildlife Management 34:501-507.

Tobin, M. E., and R. A. Dolbeer. 1990. Disappearance and recoverability of songbird carcasses in fruit orchards. Journal of Field Ornithology 61:237-242.

Travaini, A., J. A. Donázar, A. Rodríguez, O. Ceballos, M. Funes, M. Delibes, and F. Hiraldo. 1998. Use of European hare (*Lepus europaeus*) carcasses by an avian scavenging assemblage in Patagonia. Journal of Zoology 246:175-181.

United States Department of the Interior. 2010. Endangered and threatened wildlife and plants; 12-month finding for petitions to list greater sage-grouse as threatened or endangered. Federal Register 75:3909-14014.

Whelan, C. J., M. L. Dilger, D. Robson, N. Hallyn, and S. Dilger. 1994. Effects of olfactory cues on artificial-nest experiments. Auk 111:945-952.

White, G. C., and K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46 Supplement:120-138.

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 (Suppl. 1):95-104.

Table 1. Parameters used, and justification for each parameter, in survival and detectability modeling for female pheasant carcasses placed along fences on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009.

Parameter	Justification for hypothesized parameter
Local-scale models	
Site[a]	Differences in landscape context
Habitat type[a,b]	Structural, concealment, ht differences
Road presence[a]	Potential predator space use[d]
Distance of carcass from fence[a,b]	Potential predator space use[d]
Snow presence[b]	Covers collision evidence
Observer[b]	Experience during surveys may influence efficiency
Site × habitat[a,c]	Landscape effects may vary by vegetation
Site × road[a,c]	Road effects may vary by local community
Habitat × road[a,c]	Road effects may vary by habitat type
Microhabitat models	
Shrub ht[a,b]	Effects on predation[e], visual obstruction
Shrub canopy coverage[a,b]	Effects on predation[f,g], visual obstruction
Grass ht[a,b]	Effects on predation[f,h], visual obstruction
Visual concealment[a,b]	Effects on predation[g], visual obstruction
Grass ht × shrub ht[a,b,c]	Shrub concealment effects may vary by grass ht
Grass ht × canopy coverage[a,b,c]	Shrub coverage effects may vary by grass ht

[a] Parameter used in survival modeling.

[b] Parameter used in logistic regression modeling.

[c] We added interaction terms when individual terms were in a model together among the top group of models ($\Delta \mathrm{AIC}_c \approx \leq 2$).

[d] Bradley and Fagre (1988).

[e] Gregg et al. (1994).

[f] Holloran et al. (2005).

[g] Coates et al. (2010).

[h] Moynahan et al. (2007).

Table 2. Top models of daily survival rate of female pheasant carcasses placed along fences as a function of local-scale and microhabitat characteristics on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009. I ranked and compared models using quasi-AIC corrected for small sample sizes ($QAIC_C$) and normalized Akaike model weights (w_i; Burnham and Anderson 2002).

Model	QDeviance[a]	K[b]	QAICc	Δ QAICc	w_i	Likelihood[c]	R_L^{2}[d]
Local-scale models[e]							
$S_{(.)}$	96.135	2	100.165	0.000	0.238	1.000	N/A
$S_{(Habitat)}$	95.233	3	101.293	1.128	0.135	0.569	0.009
$S_{(Road)}$	95.637	3	101.696	1.532	0.111	0.465	0.005
$S_{(Site)}$	96.129	3	102.188	2.023	0.087	0.364	0.000
$S_{(Dist)}$	96.135	3	102.194	2.030	0.086	0.363	0.000
Microhabitat models[f]							
$S_{(.)}$	95.720	2	99.749	0.000	0.318	1.00	N/A
$S_{(GH)}$[g]	95.041	3	101.100	1.351	0.163	0.509	0.007
$S_{(VC)}$[h]	95.614	3	101.673	1.924	0.121	0.382	0.001
$S_{(CC)}$[i]	95.690	3	101.749	2.000	0.117	0.368	0.000
$S_{(SH)}$[j]	95.719	3	101.778	2.029	0.115	0.363	0.000

[a] QDeviance = quasi-deviance (Burnham and Anderson 2002).

[b] K = no. of parameters in model.

[c] Likelihood = w_i/w_{top}, where w_i = normalized Akaike model weight for model of interest, and w_{top} = normalized Akaike model weight for the top model (i.e. Δ QAICc = 0; Burnham and Anderson 2002).

[d] $R_L^2 = 1-(\ln(L_m)/\ln(L_o))$, where L_m = maximized likelihood for model of interest, and L_o = maximized likelihood for intercept only model (Menard 2000).

[e] I estimated overdispersion ($\hat{c}$) for this group of models as the deviance divided by the deviance df. For this group of models $\hat{c} = 3.141$.

[f] I estimated $\hat{c}$ for this group of models as the deviance divided by the deviance df. For this group of models $\hat{c} = 3.155$.

[g] GH = grass ht microhabitat variable.

[h] VC = visual concealment microhabitat variable.

[i] CC = % shrub canopy coverage microhabitat variable.

[j] SH = shrub ht microhabitat variable.

Table 3. Model averaged estimates of female pheasant carcass daily survival rates on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009. Groups represent 8 combinations of 3 binary classification variables, representing site (BB = Browns Bench, US = Upper Snake), habitat type (LS = little sagebrush, BS = big sagebrush), and road presence (NR = no road, RP = road present).

		95% CI	
Group	Daily survival	Lower	Upper
BB LS NR	0.798	0.694	0.873
BB LS RP	0.811	0.699	0.888
BB BS NR	0.776	0.661	0.860
BB BS RP	0.789	0.655	0.881
US LS NR	0.798	0.691	0.875
US LS RP	0.812	0.703	0.887
US BS NR	0.777	0.661	0.861
US BS RP	0.790	0.661	0.879

Table 4. Top models of daily survival rate of female pheasant sign placed along fences as a function of local-scale and microhabitat characteristics on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009. I ranked and compared models using quasi-AIC corrected for small sample sizes ($QAIC_C$) and normalized Akaike model weights (w_i; Burnham and Anderson 2002).

Model	QDeviance[a]	K[b]	QAICc	Δ QAICc	w_i	Likelihood[c]	R_L^{2}[d]
Local-scale[e]							
$S_{(\text{Site})}$	94.597	3	100.612	0.000	0.341	1.000	0.226
$S_{(\text{Site+Dist})}$	94.298	4	102.322	1.711	0.145	0.425	0.228
$S_{(\text{Site+Habitat})}$	94.322	4	102.347	1.735	0.143	0.420	0.228
$S_{(\text{Site+Road})}$	94.596	4	102.621	2.009	0.125	0.366	0.226
$S_{(\text{Site+Habitat+Dist})}$	94.002	5	104.039	3.427	0.061	0.180	0.231
Microhabitat [f]							
$S_{(.)}$	97.444	2	101.451	0.000	0.228	1.000	N/A
$S_{(\text{GH})}$[g]	95.933	3	101.948	0.496	0.178	0.780	0.016
$S_{(\text{CC})}$[h]	96.348	3	102.363	0.912	0.145	0.634	0.011
$S_{(\text{SH})}$[i]	97.061	3	103.076	1.624	0.101	0.444	0.004
$S_{(\text{VC})}$[j]	97.173	3	103.188	1.736	0.096	0.420	0.003
$S_{(\text{GH+CC})}$	95.233	4	103.258	1.806	0.093	0.405	0.023

[a] QDeviance = quasi-deviance (Burnham and Anderson 2002).

[b] K = no. of parameters in model.

[c] Likelihood = w_i/w_{top}, where w_i = normalized Akaike model weight for model of interest, and w_{top} = normalized Akaike model weight for the top model (i.e. Δ QAICc = 0; Burnham and Anderson 2002).

[d] $R_L^2 = 1-(\ln(L_m)/\ln(L_o))$, where L_m = maximized likelihood for model of interest, and L_o = maximized likelihood for intercept only model (Menard 2000).

[e] I estimated overdispersion ($\hat{c}$) for this group of models as the deviance divided by the deviance df ($\hat{c}$ = 3.216).

[f] I estimated $\hat{c}$ for this group of models as the deviance divided by the deviance df ($\hat{c}$ = 4.032).

[g] GH = grass ht microhabitat variable.

[h] CC = % shrub canopy coverage microhabitat variable.

[i] SH = shrub ht microhabitat variable.

[j] VC = visual concealment microhabitat variable.

Table 5. Model averaged estimates of female pheasant sign daily survival rates on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009. Groups represent 8 combinations of 3 binary classification variables, representing site (BB = Browns Bench, US = Upper Snake), habitat type (LS = little sagebrush, BS = big sagebrush), and road presence (NR = no road, RP = road present).

		95% CI	
Group	Daily survival	Lower	Upper
BB LS NR	0.872	0.783	0.928
BB LS RP	0.872	0.764	0.935
BB BS NR	0.863	0.770	0.922
BB BS RP	0.863	0.746	0.931
US LS NR	0.988	0.970	0.996
US LS RP	0.988	0.969	0.996
US BS NR	0.987	0.967	0.995
US BS RP	0.987	0.966	0.995

Table 6. Top logistic regression models of female pheasant carcass detection probability during fence collision surveys as a function of local-scale and microhabitat characteristics on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009. I ranked and compared models using Akaike's Information Criteria corrected for small sample sizes (AIC_C) and normalized Akaike model weights (w_i; Burnham and Anderson 2002).

Model	Deviance	K[a]	AICc	Δ AICc	w_i	Likelihood[b]	Prediction Success[c]
Local-scale							
$P_{(Habitat)}$	57.816	2	62.095	0.000	0.306	1.000	0.674
$P_{(Habitat+Snow)}$	57.146	3	63.717	.623	0.136	0.444	0.674
$P_{(Habitat+Dist)}$	57.424	3	63.996	1.901	0.118	0.387	0.674
$P_{(Habitat+Observer)}$	57.795	3	64.367	2.272	0.098	0.321	0.674
$P_{(Snow)}$	61.178	2	65.457	3.362	0.057	0.186	0.609
Microhabitat							
$P_{(SH)}$[d]	57.329	2	61.608	0.000	0.323	1.000	0.609
$P_{(CC)}$[e]	57.931	2	62.210	0.602	0.239	0.740	0.652
$P_{(GH+SH)}$[f]	56.980	3	63.551	1.943	0.122	0.379	0.587
$P_{(GH+CC)}$	57.332	3	63.904	2.296	0.102	0.317	0.652
$P_{(VC)}$[g]	60.385	2	64.664	3.056	0.070	0.217	0.565

[a] K = no. of parameters in model.

[b] Likelihood = w_i/w_{top}, where w_i = normalized Akaike model weight for model of interest, and w_{top} = normalized Akaike model weight for the top model (i.e. Δ QAICc = 0; Burnham and Anderson 2002).

[c] I calculated prediction success via crossvalidation in PROC LOGISTIC, PREDPROBS=CROSSVALIDATE output statement (SAS Version 9.2, Cary, NC).

[d] SH = shrub ht microhabitat variable.

[e] CC = % shrub canopy coverage microhabitat variable.

[f] GH = grass ht microhabitat variable.

[g] VC = visual concealment microhabitat variable.

Figure 1. Southern Idaho, USA, study areas, where I studied survival and detectability of female pheasant carcasses planted as hypothetical collision victims. Dashed lines represent my 2 study regions (BB = Browns Bench, US = Upper Snake) during spring of 2009.

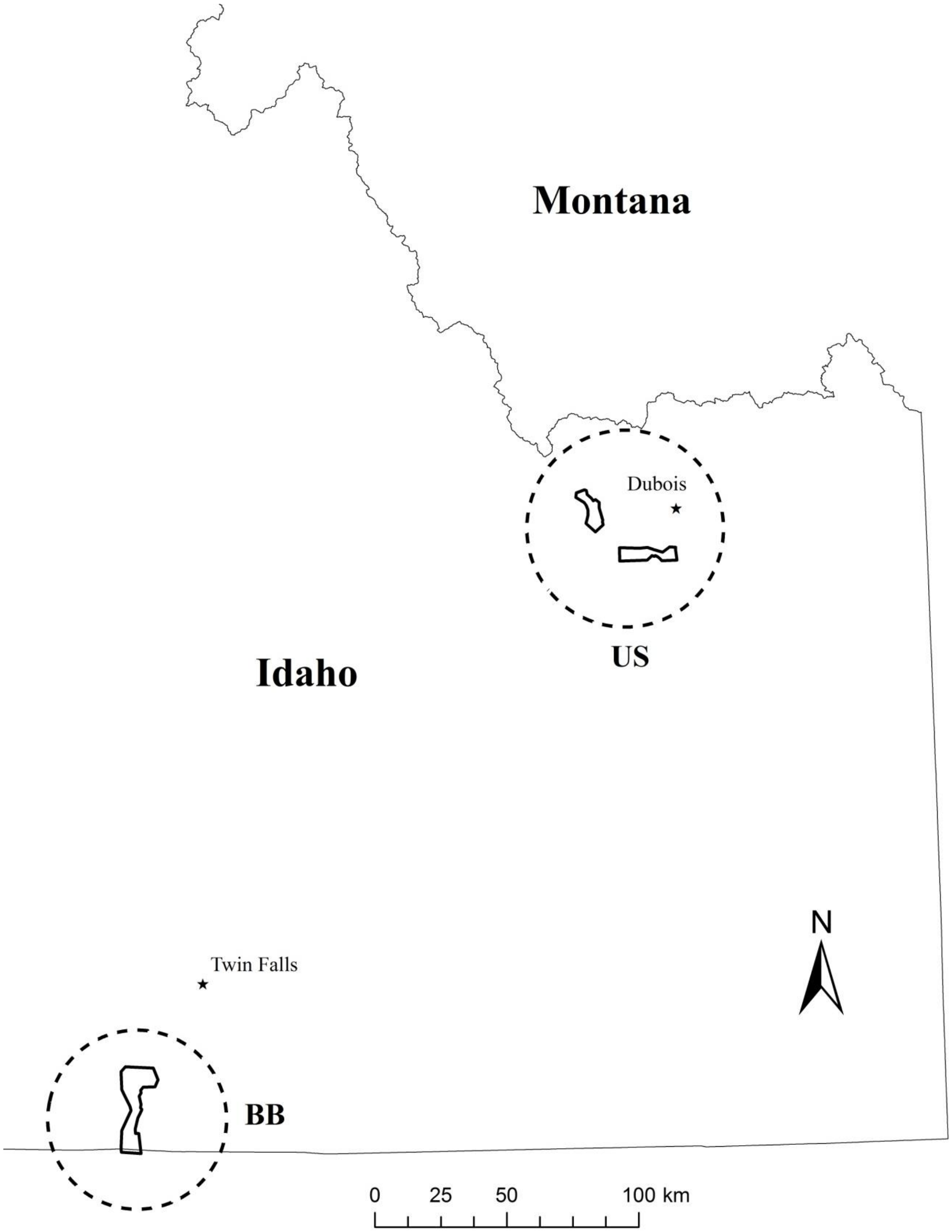
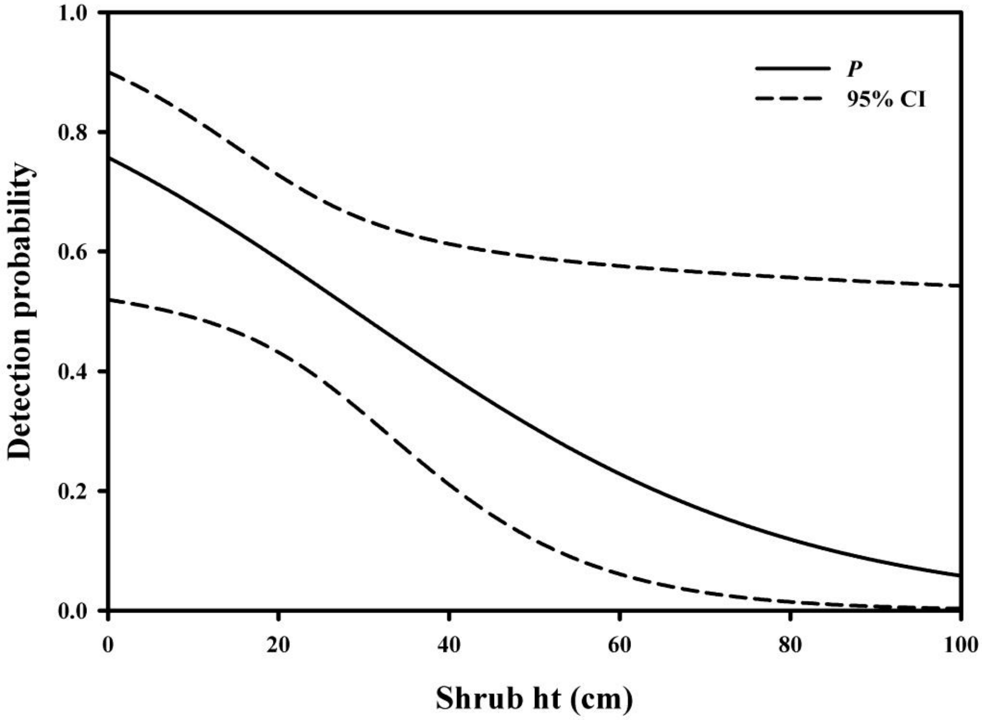
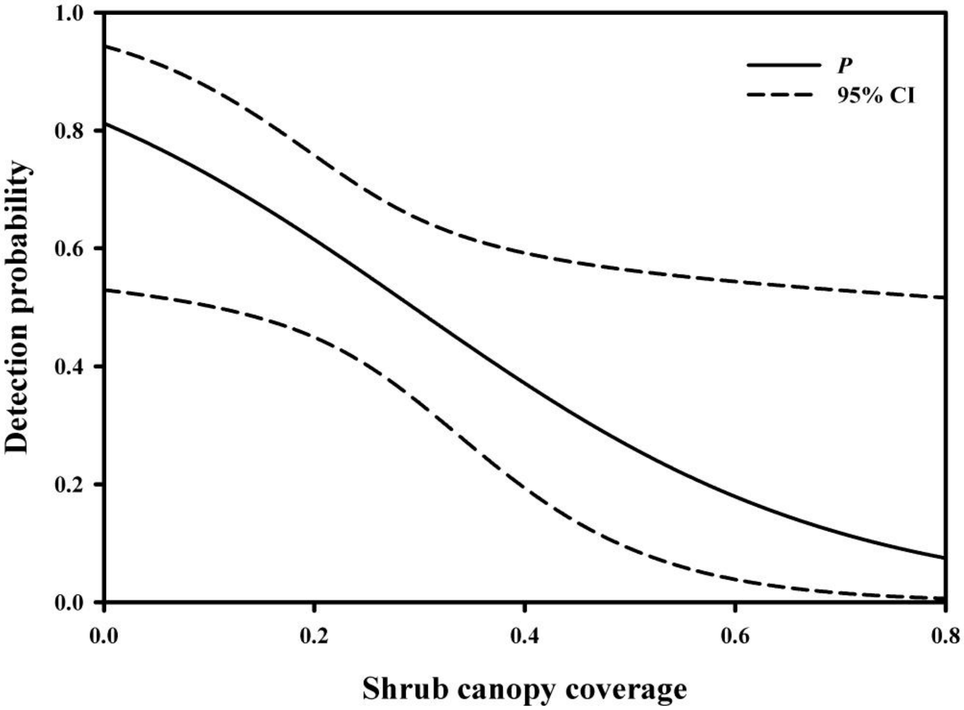


Figure 2. Plots of female pheasant carcass detection probability during fence surveys as a function of microhabitat characteristics on the Browns Bench region of Idaho, USA, during 2009. (a) Carcass detection probability as a function of shrub height from the top microhabitat detectability model. (b) Carcass detection probability as a function of shrub canopy coverage from the second best microhabitat detectability model.

a)



b)


1.0
0.8
0.6
0.4
0.2
0.0
Detection probability
0.0
0.2
0.4
0.6
0.8
Shrub canopy coverage
P
95% CI

CHAPTER 3. ESTIMATING AVIAN FENCE-COLLISION RATES IN GREATER SAGE-GROUSE BREEDING AREAS: A PROBABILITY SAMPLING APPROACH

INTRODUCTION

Collision with linear infrastructure is widespread and common among many avian species (Morkill and Anderson 1991, Bevanger 1994, Brown and Drewien 1995, Baines and Summers 1997). Grouse as a group appear highly susceptible to infrastructure collision (e.g., Bevanger 1995, Baines and Summers 1997, Wolfe et al. 2007), which may be related to morphological factors such as large body weight and high wing loading (Bevanger 1998, Janss 2000). Research in Europe suggests fences and power lines are common sources of collision mortality for a variety of grouse, including capercaillie (*Tetrao urogallus*), black grouse (*Tetrao tetrix*), red grouse (*Lagopus lagopus scoticus*), and ptarmigan (*Lagopus* spp.; Bevanger 1990, Catt et al. 1994, Baines and Summers 1997, Bevanger and Brøseth 2000, Baines and Andrew 2003). Although research concerning grouse-infrastructure collision in North America is limited, evidence suggests fence collision mortality may pose a serious threat to lesser prairie-chicken (*Tympanuchus pallidicinctus*) populations in the southern great plains (Patten et al. 2005, Wolfe et al. 2007). Moreover, infrastructure-collision has been documented in multiple locations for greater sage-grouse (hereafter sage-grouse; *Centrocercus urophasianus*), a sagebrush (*Artemisia* spp.) obligate and species of concern in the western United States (Scott 1942, Beck et al. 2006, Flake et al. 2010).

Quantifying collision frequency for widely dispersed species such as grouse requires traversing linear infrastructure and recording the number of collision remains located

(Bevanger 1999). This approach has inherent biases associated with detection probabilities and survival of collision remains, and several published studies have focused on methods to quantify these sources of bias (Bevanger et al. 1994, Smallwood 2007, Stevens et al. 2011). Smallwood (2007) reviewed unpublished manuscripts from avian wind power collision studies to evaluate sources of detection error in avian collision surveys. Similarly, Stevens et al. (2011) quantified the influence of microsite and broad scale factors on survival and detection of collision evidence in sagebrush-steppe systems.

Although sampling biases associated with searching for collision evidence in linear-infrastructure corridors are well known, biases due to non-random allocation of sampling efforts and unwarranted extrapolation have not been addressed. Much of the previous research has focused on worst-case-scenario studies, where observations are made at locations of known high impact. High impact areas are necessary to obtain sample sizes required to evaluate effectiveness of marker mitigation, and such markers are targeted for high risk areas and thus inferences drawn are appropriate. Also common, however, is arbitrary or convenience selection of study fence or power-line segments and unwarranted extrapolation of collision rate estimates to larger areas or landscapes (e.g., Bevanger 1995, Shaw et al. 2010). Unfortunately, arbitrary selection of sampling units (e.g., fence or power-line segments) is subject to inherent researcher bias which leaves study results questionable and overall extrapolation of inference to the landscape at large limited (Anderson 2001, Scheaffer et al. 2006). Fortunately, a diverse and accessible statistical literature exists to aide field ecologists in probability based sampling design and estimation (Garton et al. 2005, Scheaffer et al. 2006).

Conservation concerns about sage-grouse throughout the species' range (Braun 1998, Connelly et al. 2004) have made the lack of empirical data on North American grouse collisions apparent. That sage-grouse collide with fences has been known since at least the 1940's (Scott 1942), however, sage-grouse fence collision has never been systematically studied. Moreover, the distribution of fences in sagebrush habitats has increased over several decades, and multiple authors have hypothesized impacts on sage-grouse (Braun 1998, Connelly et al. 2000, Connelly et al. 2004). Previous collision research often provided temporally intensive studies over spatially restricted sites with limited application to other areas, and I am not aware of any studies using rigorous probability sampling designs to quantify collision frequency at landscape scales. Therefore, this research was pursued with the objective of using probability sampling methods to estimate landscape scale avian fence collision rates in sage-grouse breeding habitats across southern Idaho.

STUDY AREA

I conducted fence collision surveys in 16 greater sage-grouse breeding areas across 4 distinct geographic regions of southern Idaho (Fig. 1). Each breeding area represented 1 lek complex, defined as a group of leks in relatively close spatial proximity believed to represent part or all of a single breeding population (Connelly et al. 2003) and adjacent sagebrush-grass habitat. I sampled fences within 2 lek complexes in the East Jarbidge region (hereafter EJ), the Antelope Pocket and Browns Bench areas, respectively. In the northern Magic Valley region (hereafter NMV) I sampled fences in 4 lek complexes, the North Shoshone, Picabo Hills, Timmerman, and Paddelford Flats sites. In the Big Desert region (hereafter BD) I sampled fences in 4 lek complexes, Big Desert sites 1, 3, and 5, and the Fingers Butte site. Lastly, I sampled fences in 6 lek complexes in the Upper Snake region (hereafter US) of southeast Idaho, the Crooked Creek, Lidy, Table Butte, Medicine Lodge, Plano, and Red Road sites. In order to sample more intensively at each study site I did not sample fences from the Timmerman or Big Desert 5 sites in 2010, and thus only sampled fences in 14 lek routes. Elevations on sampling areas ranged from approximately 1,450 m on Browns Bench to approximately 2,000 m on the northern portions of the Red Road and Medicine Lodge sites. Habitat types on sampling areas varied considerably, and included large stands of big (*Artemisia tridentata*), little (*A. arbuscula*), black (*A. nova*), three-tip (*A. tripartita*), and mixed sagebrush types, as well as grasslands, pasture, and burned areas. Non-native grasses such as cheatgrass (*Bromus tectorum*) and crested wheatgrass (*Agropyron christatum*) were common on many study sites. Since there was considerable spatial variation in habitat types, these sites were representative of the variety of habitat conditions on southern Idaho rangelands.

METHODS

I used cluster sampling to estimate avian fence collision rates in sage-grouse breeding habitat. Specifically, lek routes monitored by Idaho Department of Fish and Game (hereafter IDFG) were selected (2009: n = 16; 2010: n =14) for inclusion in the study based on accessibility and breeding bird use, and lek routes were treated as strata in sampling and estimation. I buffered each lek in a route with ≥ 1 displaying male documented the previous year (2008 or 2009) by 1.5 km using ArcGIS software (hereafter GIS). I then overlaid a 1x1 km spatial grid over the buffered leks within each route using GIS. Grid cells that intersected with (i.e., contained) United States Bureau of Land Management (hereafter BLM) pasture boundaries (our surrogate for fence) were used to define the sampling frame (Fig. 2). In this design the 1x1 km cells each represented 1 sampling unit or cluster. I randomly selected a stratified cluster sample of 1x1 km grid cells using the GIS extension Hawth's Tools (Hawth's analysis tools for ArcGIS version 3, <http://www.spatialecology.com/htools.>, accessed 12 Jan 2009). I allocated the sample (2009: n = 60; 2010: n =80) to each stratum in proportion to the number of cells in the stratum. For example, if one strata (i.e., lek route) contained 10% of the total number of cells in the sampling frame in 2009, then 6 sample cells (10% of 60) were allocated to that strata. I selected the number of sampling cells each year as the estimated maximum number of cells that could be sampled in a one month period given time and logistical constraints, and increased sampling for the 2nd year after we had experienced time requirements for sampling. I was only able to sample a portion of lek routes on ≥ 1 occasion each year due to logistical constraints and weather related limited access. In 2009 we sampled 8 of 16 lek routes ≥ 2 occasions, and 2 of these 8 lek routes were sampled 3 times. I sampled 10 of 14 lek routes twice in 2010.

Within each sampling unit I searched all fence sections (i.e., sampling elements) for avian collision evidence in the form of carcasses or sign and feather tufts on the barbed wire using 1-2 searchers (1 on each side of the fence, or 1 searcher sampling each side in turn). Fence searchers walked approximately 1-3 m from the fence on either side during surveys, and monitored the area up to approximately 15 m from the fence for carcasses or collision evidence. When feather evidence was located I searched more intensively for carcasses within approximately 30-50 m of the site. All fence segments and fence types inside our spatial clusters (i.e., cells) were sampled, and digitized using handheld GPS units and GIS. I sampled fence on sage-grouse breeding areas from 5 March – 19 May 2009, and 15 March – 18 May 2010.

Statistical estimation of the average number of collisions per linear kilometer of fence and total number of collisions located on each lek route followed from elementary survey sampling (Scheaffer et al. 2006). Specifically, estimation of collision rates ($\bar{x} = \hat{\mu}$) and variance of the rates ($\hat{V}(\bar{x})$) on each study area were calculated according to 1-stage cluster sampling:

$$\bar{x} = \frac{\sum_{i=1}^{n} x_i}{\sum_{i=1}^{n} m_i} \tag{1}$$

$$\hat{V}(\bar{x}) = \left(\frac{N-n}{Nn\bar{m}^2}\right) s_r^2 \tag{2}$$

Where:

$$s_r^2 = \frac{\sum_{i=1}^{n}(x_i - \bar{x}m_i)^2}{n-1} \tag{3}$$

Notation in these formulas represents the following values: N = the total number of clusters in the lek route; n = the number of clusters sampled; m_i = the length of fence in kilometers in cluster i; $\bar{m}$ = the average length of fence in kilometers per cluster in the sample; and x_i = the number of collision sites located in the i^{th} cluster. Since the true average cluster size for the population ($\bar{M}$) is unknown, $\bar{m}$ was used in the variance formula (Scheaffer et al. 2006). Furthermore, estimation of the total number of collision sites on each study area ($\hat{\tau}$) and variance of this estimate ($\hat{V}(\hat{\tau})$) were calculated as follows:

$$\hat{\tau} = \frac{N}{n}\sum_{i=1}^{n} x_i \tag{4}$$

$$\hat{V}(\hat{\tau}) = N^2\left(\frac{N-n}{Nn}\right)s_t^2 \tag{5}$$

Where:

$$s_t^2 = \frac{\sum_{i=1}^{n}(x_i - \bar{x}_t)^2}{n-1} \tag{6}$$

I emphasize that $\bar{x}_t$ (the average number of collision sites located in the n clusters sampled) is not the same as $\bar{x}$ (the average number of collision sites located per linear kilometer of fence) as calculated in equation one.

A collision in this study was defined as detection of a whole carcass or a feather pile (≥ 5 feathers) within 15 m of the fence, or detection of feathers stuck in the fence. Despite this definition of a collision I was cautious when only feather sign was detected, and if a likely raptor plucking post was present we were conservative and did not classify these sites as collision locations. Plucking posts were common in some areas (mostly for passerine prey species) and were usually located at large wooden fence-posts, with the resulting feather piles scattered from the base of the post in the prevailing wind direction. In contrast, sites deemed collision locations based solely on feather-pile evidence commonly contained large numbers of feathers scattered in the prevailing wind direction from under the fence itself, or very close to the fence (typically < 1 m). Given this definition of a fence collision, the only victims not accounted for were birds flying into fences and leaving no feathers either on the fence or on the ground, and no carcass, or victims whose evidence was removed prior to sampling. I sent collision evidence from unknown avian species to the Feather Identification Laboratory at the

Smithsonian Institution, where whole feather characteristics (Sabo and Laybourne 1994, Woodman et al. 2005), microscopic feather characteristics (Dove and Koch 2010), and DNA barcoding (Dove et al. 2007) were used to to identify species of individual collision victims.

I counted feather tufts on fences and feather piles as collisions during searches with no knowledge of fate of the collision victim. Therefore my estimates are of number of collision sites present at the time of sampling, and not of collision mortalities, as we had no way to assess the crippling bias caused by individual birds flying into fences and dying at a later time or in a different area (Bevanger 1999). Thus, the relationship between the collision itself and the extent of negative effects on individual birds was left unstudied, as this is difficult to accurately assess (Bevanger 1999).

Statistical estimation of global collision rates and total number of collision sites in the breeding areas across the landscape once again followed from elementary survey sampling (Scheaffer et al. 2006). The global estimate of collision rates over all strata was analogous to a combined form ratio estimate, consisting of the ratio of the estimated cluster total (i.e., total number of collisions present) to an estimator of the total cluster size (i.e., total length of fence in kilometers). Estimates of the population average collision rate per linear kilometer of fence ($\bar{x}_c$) and its variance ($\hat{V}(\bar{x}_c)$) were calculated according to stratified cluster sampling:

$$\bar{x}_c = \frac{\sum_{i=1}^{L} N_i \bar{x}_{ti}}{\sum_{i=1}^{L} N_i \bar{m}_i} \tag{7}$$

$$\hat{V}(\bar{x}_c) = \frac{1}{\widehat{M}^2}\left\{\sum_{i=1}^{L} \frac{N_i(N_i - n_i)}{n_i} S_{ci}^2\right\} \tag{8}$$

Where:

$$S_{ci}^2 = \frac{\sum_{j=1}^{n_i}\left[\left(x_{ji} - \bar{x}_c m_{ji}\right) - \frac{1}{n_i}\sum_{j=1}^{n_i}\left(x_{ji} - \bar{x}_c m_{ji}\right)\right]^2}{n_i - 1} \tag{9}$$

$$\widehat{M}^2 = \left\{\sum_{i=1}^{L} N_i \bar{m}_i\right\}^2 \tag{10}$$

Notation used in these formulas represents the following values: N = total number of clusters in the L strata; N_i = total number of clusters in the ith strata; n_i = the number of clusters sampled in the ith strata; x_{ji} = the number of collisions detected in the jth cluster of the ith strata, where $j = 1, \dots, n_i$ for each strata; m_{ji} = the kilometers of fence in the jth cluster of the ith strata; $\bar{x}_{ti}$ = average number of collision sites located in clusters of the ith strata (total number of collisions located / total number of clusters sampled); and $\bar{m}_i$ = average kilometers of fence of clusters in the ith strata (total length of fence / total number of clusters sampled). In equation 8 the term $\widehat{M}^2$ was used because the total length of fence in the population of clusters (M) was unknown (Scheaffer et al. 2006).

I corrected collision counts for detectability based on the following model:

$$Corrected\ Count = \frac{1}{\hat{P}} \tag{11}$$

where $\hat{P}$ is estimated detection probability for a given collision point, estimated from the intercept only (i.e., mean) logistic regression model developed using hen ring-necked pheasant (*Phasianus colchicus*) carcasses presented by Stevens et al. (2011):

$$logit(\hat{P}) = \ln\left(\frac{\hat{P}}{1 - \hat{P}}\right) = \hat{\beta}_0 \tag{12}$$

and

$$\hat{P} = \frac{1}{1 + e^{-\{\hat{\beta}_0\}}} \tag{13}$$

where $\hat{\beta}_0$ = 0.174 was the linear model intercept coefficient. Therefore, I assumed constant detection probability for all collisions across space and time, which may have resulted in inaccurate detection probability corrections for non-target avian species. I did not correct individual collision points for detectability based on microsite vegetation characteristics (e.g., Stevens et al. 2011) because these microsite vegetation measurements were only made in 2010. Therefore, I used corrected collision counts at each sampling unit to calculate collision rates in equations 1-10. I did not correct counts for sign-survival bias since I was unable to measure collision sign longevity at each study site, and because my sampling methodology sacrificed intensive temporally repeated sampling for a broader spatial extent. As such, my collision counts should be considered as likely biased low for the entire lekking season due to the removal of collision evidence at sites and the point-in-time sampling approach taken (Smallwood 2007, Stevens et al. 2011).

Sampling methods used limited inference of my estimates to fence collision rates during the breeding season within approximately 2.5 km of leks, however, this allowed me to sample a larger geographic extent across southern Idaho. I assessed collision over a broad geographic extent because collision risk across a landscape for sage-grouse has never been investigated, although this necessarily limited the intensity with which we could sample spatially and temporally at any given site.

RESULTS

I sampled 129.5 km of fence in 140 1x1 km sampling units across south-central and southeast Idaho ≥ 1 time during spring of 2009 and 2010 (Appendix C). I sampled 23 1x1 km cells in the US region twice each, and 15 cells in the EJ region 3 times each in 2009, and in 2010 I sampled all but 3 sites twice each. I sampled Crooked Creek, Medicine Lodge, and Red Road sites once each in 2010 because weather prevented access early in the sage-grouse breeding season.

I found 111 (2009: *n* = 61; 2010: *n* = 50) avian collision sites during sampling, 75 (2009: *n* = 37; 2010: *n* = 38) in randomly selected sampling units and 36 (2009: *n* = 24; 2010: *n* = 12) opportunistically while traversing study areas. I found 86 (2009: *n* = 48; 2010: *n* = 38) sage-grouse fence collisions, 58 (2009: *n* = 28; 2010: *n* = 30) in randomly selected sampling areas. Other fence-collision species located during fieldwork included grey partridge (*Perdix perdix*; n = 4), sharp-tailed grouse (*Tympanuchus phasianellus*; *n* = 1), chukar (*Alectoris chukar*; *n* = 1), short-eared owl (*Asio flammeus*; *n* = 3), long-eared owl (*Asio otus*; *n* = 1), great-horned owl (*Bubo virginianus*; *n* = 1), Ferruginous hawk (*Buteo regalis*; *n* = 1), American robin (*Turdus migratorius*; *n* = 1), rock dove (*Columba livia*; *n* = 1), Brewer's sparrow (*Spizella breweri*; *n* = 1), horned lark (*Eremophilia alpestris*; *n* = 2), western meadowlark (*Sturnella neglecta*; *n* = 6), an unidentified owl (Family Strigidae; *n* = 1), and an unidentified dabbling duck (Subfamily Anatinae; *n* = 1). I did not use collisions located opportunistically outside of randomly selected sampling areas in collision rate estimation.

Composition of evidence types found at collision sites was dominated by feather piles (53.6%, *n* = 60), with lesser amounts of feather piles and feathers lodged in the fence (33.0%, *n* = 37), and only feathers lodged in the fence (12.5%, *n* = 14). I found 1 unscavenged fence collision carcass, a western meadowlark. I was able to determine sex from 48.3% of sage-grouse collision remains (*n* = 42), resulting in 36 male and 6 female known sex collisions. However, ability to determine sex in sage-grouse collision remains appears to be male biased due to presence of filoplume and air-sac feathers.

Estimated sage-grouse fence collision rates showed spatial variation among study areas both among and within regions in 2009 (Table 2) and 2010 (Table 3). Uncorrected estimates of sage-grouse fence collision rates ranged from 0-2.94 strikes/km in 2009, and 0-1.43 strikes/km in 2010. Applying the simple mean detectability correction model assuming constant detection across space and time essentially doubled collision rate estimates. Corrected sage-grouse collision rate estimates ranged from 0-5.42 strikes/km in 2009, and 0-2.63 strikes/km in 2010. In general I found highest collision rates in the Big Desert and Upper Snake regions, and lower collision rates in the EJ and NMV regions in both 2009 and 2010 (Table 2, 3). I found evidence for site-scale variability in collision rates between years. Both Plano and Paddelford Flat had low estimated collision rates in 2009 (0-0.30 strikes/km), but had among the largest estimated collision rates during the first sampling occasion in 2010 (1.87-2.31 strikes/km). Moreover, at least 1 site in each region had no sage-grouse collisions detected (Table 2, 3).

Estimated avian fence collision rates for all species also showed spatial variation both among and within regions in 2009 (Table 2) and 2010 (Table 3). Since most avian-fence collisions detected were sage-grouse, fence collision rates for all species were similar to sage-grouse fence collision rates at most sites. However, Paddelford Flat (0.61 strikes/km) and North Shoshone (0.92 strikes/km) had relatively large avian collision rates in 2009 despite no sage-grouse collisions detected. Uncorrected estimates of avian fence collision rates for all species ranged from 0-2.94 strikes/km in 2009, and 0-1.90 strikes/km in 2010. Avian fence collision rate estimates for all species corrected for detectability ranged from 0-5.42 strikes/km in 2009, and 0-3.50 strikes/km in 2010 (Table 2, 3).

Estimated total number of fence collisions present at the time of sampling for each site reflected both mean collision rate and estimated total length of fence on each area (Table 4, 5). Estimated collision totals again showed variation both among and within regions. Despite variation between sites, estimated sage-grouse collision totals for both 2009 and 2010 showed similar patterns as estimated collision rates, and were highest in the Big Desert and Upper Snake regions, and lowest in the East Jarbidge and North Magic Valley regions. However, several differences existed in relative risk between sites as measured by estimated collision totals compared to collision rates. In 2009 Crooked Creek had among the highest estimated sage-grouse collision rates but only moderate estimated collision totals, and a similar pattern emerged for Paddelford Flat in 2010 (Table 4, 5). Both Crooked Creek and Paddelford Flat had relatively low estimated total fence lengths, reducing the estimated number of sage-grouse collision sites present during sampling.

Estimated collision totals for all species were similar to estimated collision totals for sage-grouse. However, North Shoshone ($\hat{\tau}$ = 12.88 strikes) and Paddelford Flat ($\hat{\tau}$ = 10.12 strikes) had moderate collision total estimates in 2009 despite no sage-grouse fence collisions detected (Table 4, 5). I calculated a conservative estimate of number of avian collision strikes over all study areas by summing estimated totals for each site (i.e., Table 3, 4). I estimated 337.1 total avian fence collisions at the time of first sampling on 16 sites in 2009, and 227.6 fence collisions on 14 sites in 2010 (Table 4, 5). I estimated 261.62 total sage-grouse fence collisions at the time of first sampling on 16 sites in 2009, and 197.77 total sage-grouse fence collisions on 14 sites in 2010 (Table 4, 5).

Although there was evidence for temporal variation in collision rates between years within sites, global collision rates at landscape scales were consistent across years (Table 6). Uncorrected landscape-scale sage-grouse collision rates ranged from 0.06-0.38 strikes/km in 2009, and 0.10-0.41 strikes/km in 2010 (Table 6). Applying the mean detectability model again approximately doubled collision rates, and 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. Despite consistent landscape-scale collision rate estimates across years, I found evidence for reduced collision frequency during the second half of the lekking season (i.e., during sampling occasions 2 and 3). Although large time intervals between sampling occasions (e.g., ≥ 1 month) precluded accurate assessment of temporal dynamics of collision risk during the lekking season, both the maximum estimated collision rates reported for each site sampled ≥ 1 time and the maximum landscape-scale collision rates were observed during the first sampling occasion. Further, landscape-scale collision rate estimates during the first sampling occasion were 5.06 times greater than the second sampling occasion in 2009, and 4.08 greater during 2010.

DISCUSSION

I found spatial variation in collision frequency for sage-grouse and all species combined, with variation both among and within regions during the breeding season. Spatial variation both locally and regionally is common in collision studies, and is likely influenced by factors such as regional or local abundance (Baines and Andrew 2003, Barrios and

Rodríguez 2004), space-use and distribution of habitats (Baines and Summers 1997, Shaw et al. 2010), and unknown factors. Shaw et al. (2010) found collision frequency of blue cranes in South Africa varied by region and amount of preferred foraging habitats. Baines and Summers (1997) reported widespread grouse collision in Scotland that appeared related to foraging habitat, but regional variability was high with a 7-fold range in collision rates among sites. Baines and Andrew (2003) indicated collision rates correlated with indices of local black grouse abundance in Scotland. Janss (2000) found 88% of common crane power line collisions in Spain on only 37.5% of line segments. Similarly, research in Norway indicated ptarmigan collision rates varied significantly among segments for both fences and power lines (Bevanger and Brøseth 2000, Bevanger and Brøseth 2004). Observed variability may have also been influenced by small sample sizes at some sites; I only sampled 1 cluster at Crooked Creek in both 2009 and 2010 due to the relatively low abundance of known fences in close proximity to leks.

Despite high spatial variability, collision was widespread for sage-grouse and all species combined, and landscape scale collision rates were similar across years. Most avian collision species detected were sage-grouse (77.5%), indicating collision vulnerability was relatively high for this species compared to other avian species using sagebrush-grass habitats. These results corroborate European studies suggesting grouse as a group are highly susceptible to infrastructure collision. Bevanger and Brøseth (2000) found 253 fence collisions from 1991-1994 in Norway, 85% of which were ptarmigan. Bevanger and Brøseth (2004) studied power-line collision over 6 years in Norway, and 80% of recorded collisions were ptarmigan. Similarly, Baines and Summers (1997) found 93% of avian collisions with

Scottish deer fences were grouse (*n* = 261), and 91% of collisions in a fence marking mitigation study were also grouse (*n* = 399; Baines and Andrew 2003). These are comparable to my findings, where 83% of observed avian fence collisions were upland gamebird species. My landscape scale collision rates are within the range of those reported by European-grouse fence collision studies (e.g., Baines and Summers 1997, Bevanger and Brøseth 2000, Baines and Andrew 2003), however, differences in sampling methodology and intensity, bias correction, and scale preclude meaningful direct comparisons. Although not directly estimating collision frequency, Wolfe et al. (2007) reported almost 40% of radiomarked lesser-prairie chicken mortality in Oklahoma was caused by collision with stock fences. Clearly, grouse are highly susceptible to fence collision and frequency of collision is likely more widespread than previously acknowledged in North America.

Landscape-scale fence collision rates show avian collision to be widespread and relatively common, but my estimates should still be considered conservative. Logistical constraints prevented us from estimating survival of collision evidence on all study sites, thus I did not correct estimates for sign-survival bias. Existing research suggests high variability of sign-survival across study sites and regions, which could have contributed to observed spatial variability in collision rate estimates (Smallwood 2007, Stevens et al. 2011). My broad-scale sampling precluded temporally intensive repeat sampling at any given site, and resulted in point-in-time estimates of the number of collisions present during sampling. However, my study objectives favored broad-scale surveys to quantify collision across the landscape, while admittedly sacrificing temporal dynamics of collision throughout the sage-grouse breeding season. Moreover, I chose not to correct estimates with previously published sign-survival rates (e.g., Bevanger 1995, Janss and Ferrer 2000, Smallwood et al. 2007) because that is speculative and imprecise, and estimators are sensitive to small changes in bias corrections (Bevanger 1995).

Although widespread collision of birds with linear infrastructure such as power lines and fences has been documented, our understanding, estimation, and prediction of such phenomenon at broad spatial scales has been hindered by non-random allocation of sampling efforts and studies with limited scope. Many previous studies were designed as worst-case-scenarios in locations of known or hypothesized high impact (e.g., Catt et al. 1994, Janss and Ferrer 2000, Bevanger and Brøseth 2004). These studies often provided temporally intensive efforts over small numbers of sites which resulted in abundant information for localized areas, but limited generalization of results to other sites or regions (Bevanger 1998). Moreover, studies conducted at broad spatial scales were often limited in their ability to extrapolate results through subjective allocation of sampling efforts (e.g., Bevanger 1995, Shaw et al. 2010). Shaw et al. (2010) arbitrarily selected and sampled 199 of 4671 km of power lines on their South African study site for the presence of avian collisions, and subsequently extrapolated mean results to the entire region to estimate total number of collisions. Similarly, Bevanger (1995) calculated collision rates on 11 power line segments sampled, and extrapolated results across the entire country of Norway to estimate annual grouse mortalities in that country.

Studies using arbitrary sample allocation often suggested their sampling efforts were designed to be representative (e.g., Baines and Summers 1997, Shaw et al. 2010), however, others provided no clear description of site selection or methods and reasoning used to allocate sampling efforts (Bevanger and Brøseth 2000, Bevanger and Brøseth 2004). Critical assumptions in inference from these studies were that sampled fences or power lines were both representative of conditions across the landscape and unbiased with respect to potentially confounding factors influencing collision, such as local population sizes. However, such assumptions are rarely tested and often unmet, and arbitrary sample allocation is not a basis for rigorous estimation and inference (Anderson 2001, Scheaffer et al. 2006). Moreover, only sampling fences or power lines in areas of high population density suggests low ability to extrapolate collision rate estimates to areas of low density.

In contrast to arbitrary allocation of sampling efforts, probability or model based sampling provides a foundation for rigorous estimation, extrapolation, and predictive model building (Anderson 2001, Garton et al. 2005, Scheaffer et al. 2006). Probability sampling requires formal declaration of a statistical population (i.e., sampling frame), followed by randomized selection of sampling units to help ensure samples are unbiased and representative of the population under study. Predictive models built using arbitrarily sampled areas may have different interpretation and predictive capabilities than those built using probability based designs (Edwards et al. 2006). Edwards et al. (2006) compared models to predict presence of 4 forest lichens using data collected with probability and purposive (defined as sampling where researchers believed lichen would occur) sampling methods. They found predictive accuracy with both cross-validation and independent test data was greater for the model constructed using probability sampling data, suggesting a higher degree of generality in the model (Edwards et al. 2006). Similarly, Tintó et al. (2010) randomly selected power line pylons in Spain to survey for avian electrocution, and subsequently built models that successfully predicted electrocution based on biological, technical, and topographic features. Several authors have suggested high variability in collision frequency leaves collision unpredictable (e.g., Brown and Drewien 1995, Shaw et al. 2010), however, lack of successful prediction across the landscape may also be a function of unrepresentative or biased data collected through non-randomized methods or over a limited range of conditions.

Sampling linear infrastructure to estimate avian collision rates requires a trade-off between temporal precision and spatial extent. Temporally intensive repeat sampling over limited spatial extents provides valuable data that are likely more accurate on a site-specific basis. Unfortunately, broad-scale studies over large spatial scales often limit temporal sampling frequency, and thus leave results subject to potentially greater sign-survival bias if the objective is to quantify seasonal variation in collision (e.g., Bevanger et al. 1994, Smallwood 2007, Stevens et al. 2011). Despite limited temporal precision, however, broad-scale sampling covers a wider range of conditions and likely provides more generality in modeling and inference, particularly if sampling is representative and unbiased (Edwards et al. 2006, Braunisch and Suchant 2010). Thus, landscape-scale probability based designs should be considered for studies where broad-scale inference and predictive modeling are research objectives.

MANAGEMENT IMPLACTIONS

I found sage-grouse fence collision was common and widespread in breeding areas across southern Idaho rangelands. Widespread sage-grouse fence collision and previously published studies suggest grouse are highly susceptible to infrastructure collision, and grouse collision with elevated infrastructure may be under-reported in North America. Despite strong spatial variability in observed collision, collision frequency was consistently higher at some sites and regions, suggesting high risk in those areas. Mitigation of collision through fence removal or marking (e.g., Baines and Andrew 2003, Wolfe et al. 2009) may be an appropriate management option where sage-grouse fence collision risk in lekking areas appears high. I also provided a novel application of a probability based sampling design to quantify collision frequency across the landscape. Such designs should be considered when landscape-level inference or predictive models are research objectives, and I suggest future avian-collision research invest more time during study design phases for more generalized inferences. However, such designs may be less appropriate when objectives are focused on temporal dynamics of collision at individual sites, as a trade-off exists between spatial extent and temporal precision of collision sampling efforts. Future research should also investigate sage-grouse fence collision risk during other seasons and seasonal habitats (e.g., high density winter range), as my inferences are limited to areas in relative close proximity to active sage-grouse leks during the breeding season.

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LITERATURE CITED

Anderson, D. R. 2001. The need to get the basics right in wildlife field studies. Wildlife Society Bulletin 29:1294-1297.

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 941-948.

Barrios, L., and A. Rodríguez. 2004. Behavioural and environmental correlates of soaring-bird mortality at on-shore wind turbines. Journal of Applied Ecology 41:72-81.

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.

Bevanger, K. 1990. Topographic aspects of transmission wire collision hazards to game birds in central Norwegian coniferous forest. Fauna Norvegica Serie C, Cinclus 13:11-18.

Bevanger, K. 1994. Bird interactions with utility structures: collision and electrocution, causes and mitigating measures. Ibis 136:412-425.

Bevanger, K. 1995. Estimates and population consequences of tetraonid mortality caused by collisions with high tension power lines in Norway. Journal of Applied Ecology 32:745-753.

Bevanger, K. 1998. Biological and conservation aspects of bird mortality caused by electricity power lines: a review. Biological Conservation 86:67-76.

Bevanger, K. 1999. Estimating bird mortality caused by collision with power lines and electrocution: a review of methodology. Pages 29-56 *in* M. Ferrer and G. F. E. Janss, editors. Birds and Power Lines: Collision, Electrocution and Breeding. Quercus, Madrid, Spain.

Bevanger, K., and H. Brøseth. 2000. Reindeer *Rangifer tarandus* fences as a mortality factor for ptarmigan *Lagopus* spp. Wildlife Biology 6:121-127.

Bevanger, K., and H. Brøseth. 2004. Impact of power lines on bird mortality in a subalpine area. Animal Biodiversity and Conservation 27:67-77.

Bevanger, K., Ø. Bakke, and S. Engen. 1994. Corpse removal experiments with willow ptarmigan (*Lagopus lagopus*) in power-line corridors. Ökologie der Vögel 16:597-607.

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.

Braunisch, V., and R. Suchant. 2010. Predicting species distributions based on incomplete survey data: the trade-off between precision and scale. Ecography 33:826-840.

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., K. P. Reese, and M. A. Schroeder. 2003. Monitoring of greater sage-grouse habitats and populations. College of Natural Resources Experiment Station, Contribution No. 979, 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. Unpublished Report. 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.

Dove, C. J., and S. L. Koch. 2010. Microscopy of feathers: a practical guide for forensic feather identification. Journal of American Society of Trace Evidence Examiners 1:15-61.

Edwards, T. C., D. R. Cutler, N. E. Zimmermann, L. Geiser, and G. C. Moisen. 2006. Effects of sample survey design on the accuracy of classification tree models in species distribution models. Ecological Modelling 199:132-141.

Flake, L. D., J. W. Connelly, T. R. Kirschenmann, and A. J. Lindbloom. 2010. Grouse of plains and mountains: the South Dakota story. South Dakota Department of Game, Fish, and Parks, Pierre, South Dakota, USA.

Garton, E. O., J. T. Ratti, and J. H. Giudice. 2005. Research and experimental design. Pages 43-71 *in* C. E. Braun, editor. Techniques for Wildlife Investigation and Management. Sixth edition. The Wildlife Society, Bethesda, Maryland, USA.

Janss, G. F. E. 2000. Avian mortality from power lines: a morphologic approach of a species-specific mortality. Biological Conservation 95:353-359.

Janss, G. F. E., and M. Ferrer. 2000. Common crane and great bustard collision with power lines: collision rate and risk exposure. Wildlife Society Bulletin 28:675-680.

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 persistance. Evolutionary Ecology Research 7:235-249.

Sabo, B. A., and R. C. Laybourne. 1994. Preparation of avian material recovered from pellets and as prey remains. Journal of Raptor Research 28:192-193.

Scheaffer, R. L., W. Mendenhall III, and R. L. Ott. 2006. Elementary survey sampling. 6th Edition. Duxbury Advanced Series, Thompson Brooks/Cole, Belmont, California, USA.

Scott, J. W. 1942. Mating behavior of the sage-grouse. Auk 59:477-498.

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.

Smallwood, K. S., C. G. Thelander, M. L. Morrison, and L. M. Rugges. 2007. Burrowing owl mortality in the Altamont Pass wind resource area. Journal of Wildlife Management 71:1513-1524.

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.

Tintó, A., J. Real, and S. Mañosa. 2010. Predicting and correcting of electrocution of birds in Mediterranean areas. Journal of Wildlife Management 74:1852-1862.

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(Suppl. 1):95-104.

Woodman, N., C. J. Dove, and S. C. Peurach. 2005. A curious pellet from a great horned owl (*Bubo virginianus*). Northeastern Naturalist 12:127-132.

Table 1. Species composition of upland bird (Order Galliformes) avian collision sites found during fence collision surveys in southern Idaho, USA, during spring 2009 and 2010. Species include: greater sage-grouse (GRSG), grey partridge (GP), sharp-tailed grouse (STG), and chukar (CKR).

Region/Site	GRSG	GP	STG	CKR	Total
East Jarbidge					
Antelope Pocket	2	1	-	1	4
Browns Bench	8	-	-	-	8
North Magic Valley					
North Shoshone	-	-	-	-	0
Paddelford Flat	3	-	-	-	3
Picabo Hills	1	-	-	-	1
Timmerman[a]	-	-	-	-	0
Big Desert					
Big Desert #1	3	-	-	-	3
Big Desert #3	7	-	-	-	7
Big Desert #5[a]	-	-	-	-	0
Fingers Butte	13	-	-	-	13
Upper Snake					
Crooked Creek	4	1	-	-	5
Lidy	11	1	-	-	12
Medicine Lodge	-	-	-	-	0
Plano	9	-	1	-	10
Red Road	13	-	-	-	13
Table Butte	12	1	-	-	13
Total	86	4	1	1	93

[a] These sites were only sampled during the 2009 field season.

Table 2. Avian fence collision rates for all species and greater sage-grouse (GRSG) found during fence collision surveys in southern Idaho, USA, during spring 2009. All collision rates are expressed in number of fence strikes per km of fence, and were corrected for detection probability of collision evidence during fence surveys.

Lek Route/Area	n[a]	N[b]	Collision rate (all species)[c]	Collision rate (GRSG)[c]
East Jarbidge				
Antelope Pocket	8	42	0	0
Browns Bench	7	37	0.20-0.41	0-0.20
North Magic Valley				
North Shoshone	2	14	0.92	0
Paddleford Flats	4	22	0.61	0
Picabo Hills	4	22	0.98	0.33
Timmerman	2	13	0	0
Big Desert				
Big Desert #1	1	7	1.68	1.68
Big Desert #3	1	8	5.42	5.42
Big Desert #5	3	17	0	0
Fingers Butte	5	28	1.34	1.34
Upper Snake				
Crooked Creek	1	3	0-2.17	0-2.17
Lidy	3	18	0-1.75	0-1.17
Medicine lodge	4	22	0	0
Plano	4	25	0-0.60	0-0.30
Red Road	8	45	1.24-1.95	0.93-1.95
Table Butte	3	17	0-1.68	0-1.68

[a] n = number of 1x1 km clusters randomly selected and sampled in each lek route stratum.

[b] N = Total number of 1x1 km clusters in sampling frame for each lek route stratum.

[c] Range of estimates represented the minimum-maximum estimated collision rates for sites sampled ≥ 1 time during the field season.

Table 3. Avian fence collision rates for all species and greater sage-grouse (GRSG) found during fence collision surveys in southern Idaho, USA, during spring 2010. All collision rates are expressed in number of fence strikes per km of fence, and were corrected for detection probability of collision evidence during fence surveys.

Lek Route/Area	n[a]	N[b]	Collision rate (all species)[c]	Collision rate (GRSG)[c]
East Jarbidge				
Antelope Pocket	10	42	0.51	0-0.25
Browns Bench	9	37	0	0
North Magic Valley				
North Shoshone	4	14	0	0
Paddleford Flats	7	29	0-2.31	0-2.31
Picabo Hills	5	19	0	0
Big Desert				
Big Desert #1	2	7	0	0
Big Desert #3	2	8	0.88-3.50	0-2.63
Fingers Butte	8	31	0-0.98	0-0.98
Upper Snake				
Crooked Creek	1	4	0	0
Lidy	5	19	1.03-1.38	1.03
Medicine lodge	8	32	0	0
Plano	5	22	0.53-1.87	0.27-1.87
Red Road	10	42	1.48	1.18
Table Butte	4	17	0-0.87	0-0.87

[a] n = number of 1x1 km clusters randomly selected and sampled in each lek route stratum.

[b] N = Total number of 1x1 km clusters in sampling frame for each lek route stratum.

[c] Range of estimates represented the minimum-maximum estimated collision rates for sites sampled ≥ 1 time during the field season.

Table 4. Estimated total number of avian fence collision sites present at the time of sampling on each site for all species and greater sage-grouse (GRSG) found during fence collision surveys in southern Idaho, USA, during spring 2009. All collision total estimates were corrected for detection probability of collision evidence during fence surveys.

Lek Route/Area	n[a]	N[b]	Estimated total (all species)[c]	Estimated total (GRSG)[c]
East Jarbidge				
Antelope Pocket	8	42	0	0
Browns Bench	7	37	9.73-19.45	0-9.73
North Magic Valley				
North Shoshone	2	14	12.88	0
Paddleford Flats	4	22	10.12	0
Picabo Hills	4	22	30.36	10.12
Timmerman	2	13	0	0
Big Desert				
Big Desert #1	1	7	12.88	12.88
Big Desert #3	1	8	44.16	44.16
Big Desert #5	3	17	0	0
Fingers Butte	5	28	41.22	41.22
Upper Snake				
Crooked Creek	1	3	0-16.56	0-16.56
Lidy	3	18	0-33.12	0-22.08
Medicine lodge	4	22	0	0
Plano	4	25	0-23.00	0-11.50
Red Road	8	45	41.40-62.10	31.05-62.10
Table Butte	3	17	0-31.28	0-31.28

[a] n = number of 1x1 km clusters randomly selected and sampled in each lek route stratum.

[b] N = Total number of 1x1 km clusters in sampling frame for each lek route stratum.

[c] Range of estimates represented the minimum-maximum estimated collision totals for sites sampled ≥ 1 time during the field season.

Table 5. Estimated total number of avian fence collision sites present at the time of sampling on each site for all species and greater sage-grouse (GRSG) found during fence collision surveys in southern Idaho, USA, during spring 2010. All collision total estimates were corrected for detection probability of collision evidence during fence surveys.

Lek Route/Area	n[a]	N[b]	Estimated total (all species)[c]	Estimated total (GRSG)[c]
East Jarbidge				
Antelope Pocket	10	42	15.46	0-7.73
Browns Bench	9	37	0	0
North Magic Valley				
North Shoshone	4	14	0	0
Paddleford Flats	7	29	0-15.25	0-15.25
Picabo Hills	5	19	0	0
Big Desert				
Big Desert #1	2	7	0	0
Big Desert #3	2	8	7.36-29.44	0-22.08
Fingers Butte	8	31	0-28.52	0-28.52
Upper Snake				
Crooked Creek	1	4	0	0
Lidy	5	19	20.98-27.97	20.98
Medicine lodge	8	32	0	0
Plano	5	22	16.19-56.67	8.10-56.67
Red Road	10	42	38.64	30.91
Table Butte	4	17	0-15.64	0-15.64

[a] n = number of 1x1 km clusters randomly selected and sampled in each lek route stratum.

[b] N = Total number of 1x1 km clusters in sampling frame for each lek route stratum.

[c] Range of estimates represented the minimum-maximum estimated collision totals for sites sampled ≥ 1 time during the field season.

Table 6. Landscape-scale avian fence collision rates for all species and greater sage-grouse (GRSG) found during fence collision surveys in southern Idaho, USA, during spring 2009 and 2010, treating individual lek routes as sampling strata. All collision rates are expressed in number of fence strikes per km of fence.

Data type / year / round	Lek routes sampled[a]	Collision rate (all species)	95 % CI	Collision rate (GRSG)	95% CI
Corrected[b]					
2009					
Round 1	16	0.91	0.56-1.26	0.70	0.39-1.02
Round 2	8	0.23	0.05-0.41	0.14	0.02-0.26
Round 3	2	0.12	-0.10-0.33	0.12	-0.10-0.33
2010					
Round 1	14	0.89	0.55-1.24	0.75	0.45-1.05
Round 2	11	0.30	0.12-0.48	0.18	0.03-0.34
Uncorrected[c]					
2009					
Round 1	16	0.49	0.30-0.68	0.38	0.21-0.55
Round 2	8	0.12	0.03-0.22	0.08	0.01-0.14
Round 3	2	0.06	-0.05-0.18	0.06	-0.05-0.18
2010					
Round 1	14	0.49	0.30-0.67	0.41	0.24-0.57
Round 2	11	0.16	0.07-0.26	0.10	0.02-0.18

[a] Lek routes sampled = number of strata sampled for a given sampling round.

[b] Corrected = data corrected for detection probability prior to collision rate estimation.

[c] Uncorrected = data not corrected for detection probability prior to collision rate estimation.

Figure 1. Distribution of study areas in southern Idaho, USA, that were sampled to estimate landscape-scale avian and sage-grouse fence collision rates in breeding areas during spring of 2009 and 2010. Study sites are grouped by region, where regions were: East Jarbidge (EJ), North Magic Valley (NMV), Big Desert (BD), and Upper Snake (US). Leks represent known and active sage-grouse leks on each study site.

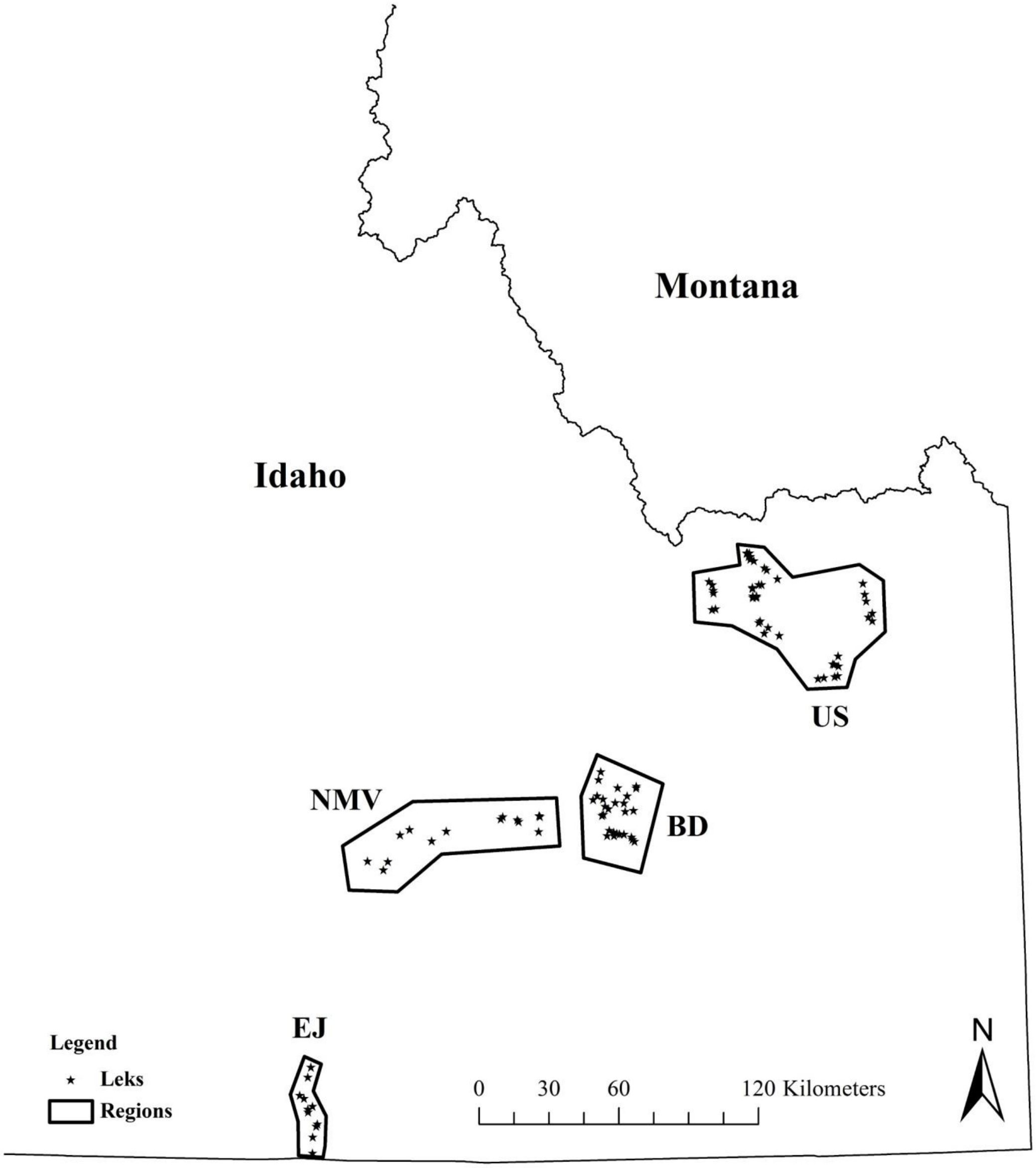
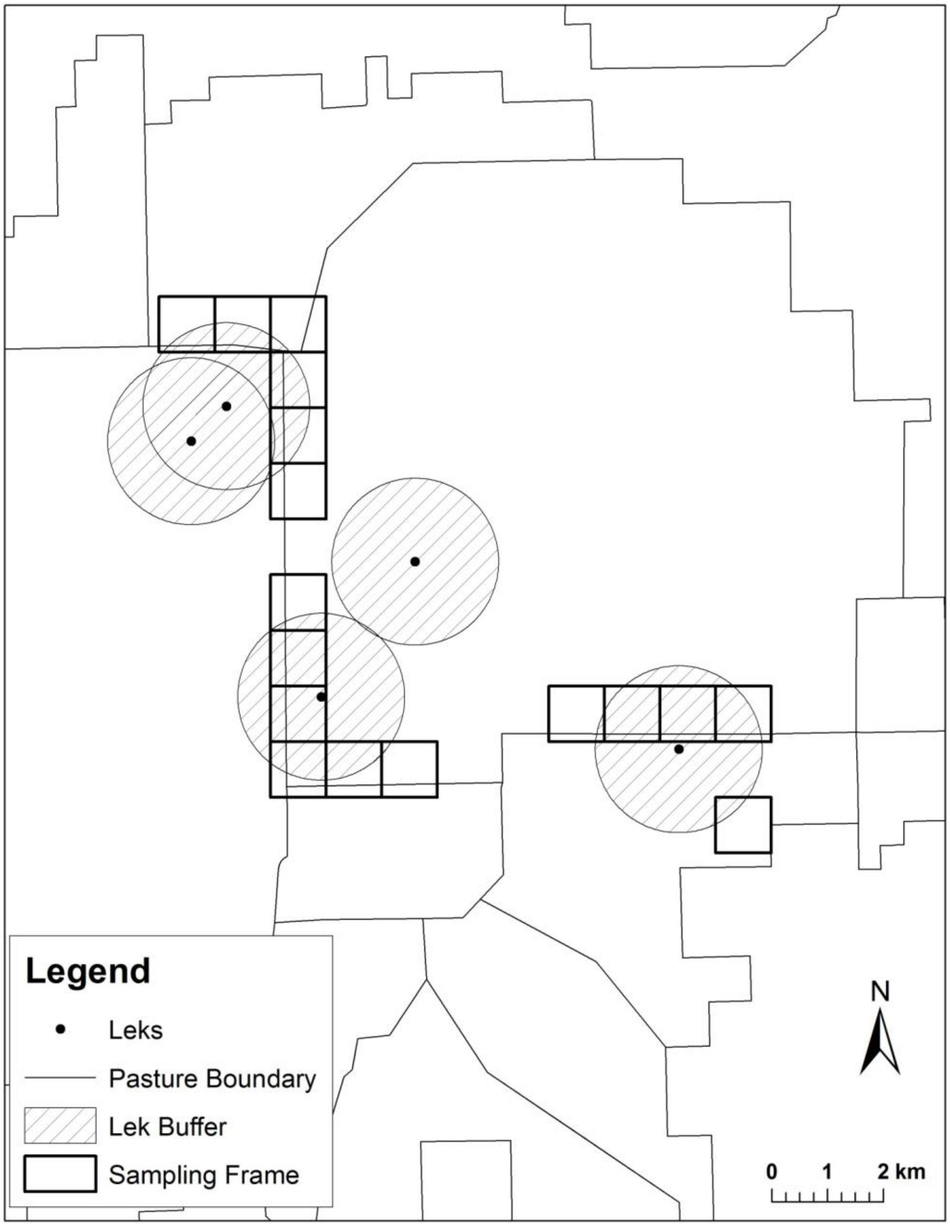


Figure 2. Example of sampling frame from 1 strata (Table Butte lek route) used in stratified cluster sampling design to estimate avian and sage-grouse fence collision frequency across the southern Idaho landscape. The sampling frame for each strata consisted of all 1x1 km clusters that intersected with both a 1.5 km buffer around known active sage-grouse leks and existing pasture boundary GIS layers (fence surrogate).



CHAPTER 4. GREATER SAGE-GROUSE FENCE COLLISION: A MULTISCALE ASSESSMENT OF COLLISION RISK AS A FUNCTION OF SITE AND BROAD-SCALE FACTORS

INTRODUCTION

Collision with infrastructure is a widespread and common phenomenon among European grouse species (Bevanger 1990, Catt et al. 1994, Bevanger 1995b, Baines and Summers 1997, Moss et al. 2000). Research in Scotland and Norway suggests collision with fences and power lines is a mortality source for a variety of grouse, including capercaillie (*Tetrao urogallus*), black grouse (*Tetrao tetrix*), red grouse (*Lagopus lagopus scoticus*), and ptarmigan (*Lagopus* spp.; Bevanger 1990, Catt et al. 1994, Baines and Summers 1997, Bevanger and Brøseth 2000, Baines and Andrew 2003). Although population level consequences are not well understood in most areas, capercaillie fence collision may have contributed to population declines in Scotland (Moss et al. 2000, Moss 2001), and grouse collision mortality may approach harvest mortality in some areas (Bevanger 1995b). Moreover, grouse may be morphologically predisposed to collision due to high wing loading and heavy body weight (Bevanger 1998, Janss 2000).

Research concerning relative extent and impact of collision on North American grouse is limited. Wolfe et al. (2007) studied mortality patterns of lesser prairie-chickens (*Tympanuchus pallidicinctus*) in Oklahoma and New Mexico, and found 39.8% of all mortality in Oklahoma was caused by fence collision. Similarly, Patten et al. (2005) concluded fragmentation caused by fences, power lines, and roads in Oklahoma has resulted in higher mortality rates for female lesser prairie-chickens in Oklahoma than in New Mexico. This increased female mortality may result in more variable nesting strategies and increased vulnerability to stochastic population fluctuations (Patten et al. 2005). Additionally, Beck et al. (2006) reported 33% of the juvenile mortality of radio-marked greater sage-grouse (*Centrocercus urophasianus;* hereafter sage-grouse) on an Idaho study area was due to collision with power lines.

In a review of interactions of birds with utility infrastructure, Bevanger (1994) suggested that factors influencing collision can be classified as biological, topographical, meteorological, or technical. Biological factors that influence collision risk include morphology (e.g., heavy weight, short and broad wings; Bevanger 1998, Janss 2000), vision and visual perception (Martin and Shaw 2010, Martin 2011), crepuscular or nocturnal activity patterns (Avery et al. 1978), local or migratory movement patterns (Avery et al. 1978, Malcom 1982, Cooper and Day 1998), space and habitat use (Baines and Summer 1997, Rollan et al. 2010), local bird densities (Anderson 1978, Baines and Andrew 2003, Bevanger and Brøseth 2004), and structure and height of local vegetation (Bevanger 1990, Catt et al. 1994). Several studies have also reported sex-, season-, and region-specific variation in collision risk (Bevanger 1995a, Baines and Summers 1997, Bevanger and Brøseth 2004, Wolfe et al. 2007). Topographical features may also influence avian-infrastructure collision, including mountain valleys, coastlines or other geomorphic forms that influence flight corridors (Cooper and Day 1998), as well as sloping terrain and ridges hypothesized to influence flight altitude (Bevanger 1990, Rollan et al. 2010).

In addition to influence of biological and topographical features on collision, risk may be influenced by meteorological conditions and infrastructure design (Bevanger 1994). Meteorological conditions influencing collision risk include fog, precipitation, wind speed and direction, and weather conditions influencing bird flight intensity and behavior (Bevanger 1994). Although there is evidence for changes in flight behavior due to weather conditions, evidence for increased collision risk due to weather conditions is mostly anecdotal (Brown and Drewien 1995, Savereno et al. 1996). However, Brown and Drewien (1995) reported increased collision rates during periods of strong gusty winds. Technical designs related to infrastructure construction may also influence collision risk (Bevanger 1994, Bevanger and Brøseth 2004). Infrastructure density on the landscape has been hypothesized to influence avian collision risk (Wolfe et al. 2007), however, this technical attribute is often not measured in avian collision studies.

Concerns about impacts of elevated infrastructure on sage-grouse have highlighted the lack of empirical data concerning collision frequency and grouse in North America. The spatial extent of fences and other elevated structures has increased dramatically in sagebrush habitats during the last 50 years, and potential impacts on sage-grouse have been hypothesized (Braun 1998, Connelly et al. 2000, Connelly et al. 2004, Johnson et al. 2011). Previous studies involving collisions have often focused on presumed high risk areas (e.g., Morkill and Anderson 1991, Brown and Drewien 1995, Savereno et al. 1996), and many studies have provided temporally intensive surveys over spatially restricted sites with limited application to other areas. Thus, few studies have evaluated collision and its contributing factors over large geographic areas or across multiple spatial scales, further limiting our predictability of collision risk across the landscape. Therefore, this research was pursued with the objective of evaluating relationships between sage-grouse collision risk and biological, topographical, and technical features in southern Idaho at multiple spatial scales.

STUDY AREA

I conducted fence collision surveys in greater sage-grouse breeding areas (2009: *n* = 16; 2010: *n* = 14) across 4 geographic regions of southern Idaho. Each breeding area represented 1 lek complex, defined as a group of leks in relatively close spatial proximity believed to represent part or all of a single breeding population (Connelly et al. 2003). In 2009 I sampled fences within 2 lek complexes in the East Jarbidge region, 4 lek complexes in the northern Magic Valley region, 4 lek complexes in the Big Desert region, and 6 lek complexes in the Upper Snake region of southern Idaho (See chapter 2 for names and locations of specific lek complexes). In order to sample more intensively at each site, I did not sample fence from 1 north Magic Valley lek complex and 1 Big Desert lek complex in 2010, and thus only sampled fence in 14 lek complexes. Elevations on sampling areas ranged from approximately 1450-2000 m, and habitat types on sampling areas varied considerably. Study area habitats included stands of big (*Artemisia tridentata*), little (*A. arbuscula*), black (*A. nova*), three-tip (*A. tripartita*), and mixed sagebrush types, as well as grasslands, pasture, and burned areas. Non-native grasses such as cheatgrass (*Bromus tectorum*) and crested wheatgrass (*Agropyron christatum*) were common on many study sites. There was considerable spatial variation in habitat conditions, thus these sites were representative of the variety of habitats on southern Idaho rangelands.

METHODS

Fence Collision Surveys

I used a stratified cluster sampling design to survey fences in sage-grouse breeding habitat. I treated lek routes (i.e., lek complexes) monitored by Idaho Department of Fish and Game (hereafter IDFG) as strata in sampling, and selected lek routes (2009: n = 16; 2010: n =14) for inclusion in the study based on accessibility and breeding bird use. I buffered each lek by 1.5 km, overlaid the United States Bureau of Land Management's pasture boundary layer (our surrogate for fence), and overlaid a 1x1 km spatial grid over each site. I used 1x1 km spatial grid cells over each lek route as sampling units (i.e., clusters), and randomly selected a stratified cluster sample from those cells intersecting with both the lek buffer and pasture boundary layer using the ArcGIS (GIS) extension Hawth's Tools (Hawth's analysis tools for ArcGIS version 3, <http://www.spatialecology.com/htools.>, accessed 12 Jan 2009). In 2009 I sampled 8 of 16 lek routes ≥ 2 occasions and 2 of those lek routes 3 times, and in 2010 I sampled 10 of 14 lek routes twice from March-May. I located avian fence collisions opportunistically while traversing study sites in addition to those located during randomized surveys. Methods used limited inference to breeding areas within approximately 2.5 km of leks at each site; however, this allowed me to sample a larger geographic extent across southern Idaho. I covered a broad spatial extent because collision risk for sage-grouse has never been systematically investigated. See chapter 3 for a full description of sampling frame construction, sample allocation to each strata, and sampling methodologies.

Characterizing Site-scale Attributes of Collision Locations

I measured vegetation, lek, technical, and topographical and characteristics at sage-grouse fence collision locations. At fence collision points I measured heights of the closest shrubs not intersecting the plane of the fence (i.e., lateral shrub height), and distances to the closest non-intersecting shrubs. I also measured canopy coverage along 10 m transects in the cardinal directions and centered on collision locations (Canfield 1941). I measured height of the closest shrub growing directly along the fence (i.e., longitudinal shrub height intersecting the plane of the fence) within 5 m of the collision site in both directions to evaluate the influence of vegetation in the longitudinal direction of the fence. I recorded maximum lek count and distance to the nearest known lek for each site in GIS using annual lek count data provided by IDFG (Jenni and Hartzler 1978, Connelly et al. 2003). For these analyses I used only leks where ≥ 1 male was counted displaying in 2009-2010, and deleted all leks with zero counts in both years and all historical leks with unknown status.

I collected technical data at each collision site, including fence height, fence type, and the distance between fence posts for each segment. I recorded types of posts bounding fence collision segments (i.e., wooden, steel t-post) at a later date from pictures recorded at each site. I also calculated differences between fence height and vegetation height both laterally and longitudinally to the fence. I collected topographic data including UTM location, and slope both across and along the fence (measured over 20 m centered on the collision site). Additionally, I obtained aspect at each collision site using a digital elevation model (hereafter DEM) in GIS. If collision sign was located without evidence present on the fence, all collision site measurements were made at the fence location perpendicular to the closest collision evidence. For example, feather pile evidence was often scattered by the wind from a location under or very close to fence, and the closest evidence (i.e., feather or group of feathers) was used to determine the point used for measurements.

I selected random fence points on each study area to assess the significance of features recorded at collision locations for site scale analysis. I randomly generated 1 spatial location for each collision point found on each study area within the sampling frame using the GIS extension Hawth's Tools (Hawth's analysis tools for ArcGIS version 3, <http://www.spatialecology.com/htools.>, accessed 12 Jan 2009), and used the closest fence segment to this location on each area to measure site-scale variables. I selected random points from all available locations within the sampling frame of each sampled lek route, and recorded data at the closest fence segment to this point. Random fence points were not specifically paired with collision fence points, and no minimum distances were enforced between random points and the nearest collision fence point. Biological, topographic, and technical characteristics measured at random fence points were identical to those measured at fence collision locations. Spatial autocorrelation in covariate data at random and collision fence points was not specifically addressed, however, classification methods described below often show increased predictive accuracy with spatially correlated data (Marmion et al. 2009).

I used multivariate machine learning methods to classify random and collision fence points based on measured site-scale characteristics. Specifically, I used classification and regression tree (hereafter CART; Breiman et al. 1984, De'ath and Fabricius 2000) models to discriminate random and collision fence points based on site attributes. The fence post type covariate had multiple missing values due to incomplete picture data (*n* = 15 missing values), therefore, CART provided an analytical tool capable of accommodating missing values in covariate data using surrogate splits (De'ath and Fabricius 2000). I built all CART models using the rpart package in the R statistical computing language (R Core Development Team 2006), and maximized between group differences at each split by minimizing the Gini index at each node (Breiman et al. 1984; De'ath and Fabricius 2000). I used 10-fold cross-validation to estimate predictive accuracy and determine optimal tree size. Ten-fold cross-validation divided point data into 10 random subsets of approximately equal size, and subsequently used 9 subsets as training data to build each model. The remaining subset was used as test data to estimate model prediction error, and the process was repeated leaving out each subset in turn (Breiman et al. 1984, De'ath and Fabricius 2000). I repeated this process 1,000 times because optimal tree size can vary under different cross-validations. I selected optimal tree size at each repetition of the analysis as the smallest model with estimated prediction error within 1 standard error of the minimum (1-SE Rule; Breiman et al. 1984, De'ath and Fabricius 2000). This process resulted in a distribution of optimal tree sizes, from which I selected the modal tree size (i.e., most common) as the final model (De'ath and Fabricius 2000).

Because CART analyses often have low predictive success relative to other machine learning methods (Cutler et al. 2007, Kampichler et al. 2010), I also used random forest models to evaluate classification ability of site-scale fence-point characteristics (Breiman 2001, Cutler et al. 2007). Random forest methods build classification tree models similar to CART, however, instead of choosing the optimal split at each tree node based on all covariates, the optimal split at each node is performed based on covariates randomly chosen at each step. Since choosing the optimum split at each node from all covariates (i.e., CART) can result in an overall tree that is not optimal, random forest models can improve predictive accuracy over CART methods (Breiman 2001, Cutler et al. 2007, Kampichler et al. 2010). Further, random forest models provide relative measures of importance for all covariates, based on mean decrease in predictive accuracy from permuting the covariate, which provides a means to quantify importance of variables that may not be included in final CART models. I evaluated random forest model prediction success with out-of-bag (hereafter OOB) error estimates that are essentially cross-validated prediction error estimates from random subsets of the data excluded from model fitting processes (Breiman 2001, Cutler et al. 2007). I fit random forest models to classify random and collision fence points using 10,000 iterations of the randomForest package (Liaw and Wiener 2002) in the R statistical computing language (R Core Development Team 2006). I constructed both CART and random forest models using 16 site-scale covariates (Appendix D).

Predicting Collision Risk at Broad Spatial Scales

I evaluated influence of patch and landscape-scale characteristics on observed collision rates. I defined a patch as each of the 1x1 km grid cells used as sampling units in the study where we recorded collision counts. I also buffered each lek at each site by 8.5 km to quantify landscape attributes on each study area. Holloran and Anderson (2005) reported 92.5% of all sage-grouse nests in Wyoming were within of 8.5 km of a lek, thus 8.5 km was selected to encompass the majority of broader breeding habitat on each area. I quantified biological, technical, and topographic variables on each study area at the patch scale. I quantified vegetation and topography at each site using 30x30 m raster grid data. I quantified topographic heterogeneity on each area using the terrain ruggedness index (hereafter TRI; Riley et al. 1999), which calculates the average change in elevation (m) between a given 30x30 m raster pixel and its surrounding cells. I downloaded United States Geological Survey 30 m DEM's (www.seamless.usgs.gov, accessed 9/24/2010) for each area. I used an ArcInfo script to quantify TRI for each 30 m pixel of each sampled patch, and calculated mean TRI for each 1x1 km cluster using GIS. I quantified fence length over each sampled patch using digitized sampled fences in GIS.

I quantified vegetation characteristics of each 1x1 km cluster using data obtained from the national Landfire database (www.landfire.gov, accessed 9/24/2010). I used 30 m pixel Landfire raster data to quantify sagebrush cover type, vegetation height, and shrub canopy cover on each site. I quantified vegetation cover types using the Landfire Veg Type dataset, vegetation height using the Landfire Veg Height dataset, and canopy cover using the Existing Veg Cover dataset, respectively, all downloaded separately for each site. I quantified proportion of each sampled patch covered by reclassified raster values for each area using GIS. Numerous studies at multiple spatial scales found sage-grouse habitat-use during the nesting season was influenced by sagebrush presence, canopy cover, and shrub height (Schroeder et al. 1999, Connelly et al. 2000, Aldridge and Brigham 2002, Holloran et al. 2005, Doherty et al. 2010). Moreover, grouse-infrastructure collision studies have suggested collision may be related to vegetation height characteristics (Bevanger 1990, Catt et al. 1994). Therefore, I reclassified Landfire cover type values into big sagebrush, and low sagebrush cover types, vegetation height values into 0-0.5 m, 0.5-1.0 m, and > 1 m height classes, and shrub canopy cover values into 10-20%, 20-30%, 30-40%, and 40-50% cover values. No other Landfire cover type, vegetation height, or canopy cover categories were used in our analyses.

I quantified lek distribution and count data relative to our study sites using annual lek count survey and location data provided by IDFG (Jenni and Hartzler 1978, Connelly et al. 2003). I calculated distance to nearest lek, and number of leks within 2 km from each sampled cluster's centroid at the patch scale using GIS. For each sampled patch I also recorded maximum count at the closest lek, and summed the maximum counts of all leks within 2 km of the centroid. At landscape scales I calculated the number of leks and summed lek count over each area.

I used logistic regression to model probability of sage-grouse collision site presence for a survey as a function of covariates using the known fate model and logit link function in Program MARK (White and Burnham 1999). I used an information-theoretic model selection framework to rank and compare models using Akaike's Information Criterion corrected for small samples sizes ($\mathrm{AIC_C}$) and normalized Akaike model weights (w_i; Burnham and Anderson 2002). I used information-theoretic model selection methods to facilitate comparison of non-nested models, and used Program MARK in this analysis to facilitate use of information theoretic methods and calculation of model weights. Since many sites were only sampled once per year, and very few collisions were located during subsequent sampling rounds on sites sampled > 1 time per year, I only used data from the first sampling occasion for broad-scale analyses.

I constructed logistic regression models in this analysis using 17 covariates hypothesized to influence collision a priori, all of which were hypothesized based on avian collision or sage-grouse literature (Appendix D). For this analysis I grouped sampled clusters by region, with sites from East Jarbidge and northern Magic Valley in the Magic Valley region (hereafter MV), and Upper Snake (hereafter US) and Big Desert (hereafter BD) as stand-alone regions. I tested for correlation between covariates using correlation t-tests, and no significantly correlated ($P < 0.05$) variables were included together in a model (Appendix E). I used a two stage modeling process to select a final reduced group of candidate models due to the large number of covariates and covariate combinations. I first constructed models using individual covariates divided into 3 groups: vegetation covariates, lek distribution and count covariates, and all other remaining covariates (i.e., region, year, TRI, and fence length; Appendix F). I then incorporated combinations of covariates from the top models (i.e., $\Delta AIC_C < 2$) from each group into a final model suite of 23 models (Appendix F).

I evaluated goodness-of-fit, discrimination ability, and classification success for the top logistic regression model. I used the Hosmer and Lemeshow test to evaluate goodness-of-fit for the top model, using a P-value of 0.05 to test the null hypothesis that the model fits the observed data (Hosmer and Lemeshow 2000). I used area under the Receiver Operating Characteristic curve (hereafter ROC) to evaluate ability of the top model to discriminate between patch-scale surveys with and without sage-grouse collision sites present (Hosmer and Lemeshow 2000). Area under the ROC curve required pairwise comparison of all surveys where collision was detected with all surveys where collision was not detected. For each pairwise comparison the model was used to predict probability of collision occurrence for each survey, and area under the ROC curve represents the proportion of pairwise comparisons where the survey with collision detected had a higher predicted probability than the survey without a collision detected (Hosmer and Lemeshow 2000). I conducted goodness-of-fit testing and estimated area under the ROC curve using SAS Version 9.2 (SAS, Cary, NC).

I used classification tables to evaluate classification success for the top logistic regression model (Hosmer and Lemeshow 2000). In logistic regression analysis evaluation, sensitivity is defined as the probability of correctly predicting an event when it occurs, and specificity is defined as the probability of correctly predicting absence of an event when it does not occur (Hosmer and Lemeshow 2000). Logistic regression classification commonly uses a prediction threshold of 0.5 (i.e., $P(y = 1) \geq 0.5$ = predicted event occurrence), however, this is only optimal when probability of event occurrence and probability of event not occurring are approximately equal (Hosmer and Lemeshow 2000). Therefore, I calculated an optimum prediction threshold for the top model as the point where sensitivity and specificity are approximately equal, thereby jointly minimizing both types of prediction error (Hosmer and Lemeshow 2000). I used the optimum prediction threshold to construct classification tables used to estimate classification success, and calculated optimum prediction thresholds and classification tables using the R statistical computing language (R Core Development Team 2006).

I summarized collision count data over each sampled patch to evaluate influence of patch-scale characteristic covariates on expected collision counts. I corrected collision counts for undetected collisions using the mean or intercept-only logistic regression model from detectability trial experiments, where each observed collision represented $1/0.54$ corrected collisions, and 0.54 was the proportion of carcasses detected during field experiments (Stevens et al. 2011). Since count based modeling requires integer-valued random variables, I summed corrected counts on each patch, and rounded corrected collision counts for each patch to the nearest whole number. I used zero-inflated Poisson regression to model the influence of covariates on corrected collision counts (Lambert 1992, Martin et al. 2005). Zero-inflated Poisson models (hereafter ZIP models) are statistical mixture models, where observed counts are treated as a binomial mixture of a Poisson random variable with a point mass at zero. As such, ZIP models take the following form:

$$P(Y=0) = 1 - \hat{p} + \hat{p} \times e^{-\hat{\lambda}} \tag{1}$$

$$P(Y=r) = \hat{p}\frac{e^{-\hat{\lambda}}\hat{\lambda}^{r}}{r!}\ r = 1,2,\dots, \tag{2}$$

Where:

$$logit(\hat{p}) = \hat{\beta}_0 + \hat{\beta}_1(X_1) + \dots + \hat{\beta}_k(X_k) \tag{3}$$

$$log(\hat{\lambda}) = \hat{\beta}_0 + \hat{\beta}_1(X_1) + \dots + \hat{\beta}_k(X_k). \tag{4}$$

In this analysis r represents integer valued corrected-count data, $\hat{\lambda}$ represents the mean number of collisions at a given 1x1 km cluster, and $\hat{p}$ represents the binomial mixture probability (i.e., probability of the data coming from the Poisson distribution, where 1 - $\hat{p}$ is probability of data coming from a point mass at zero; Lambert 1992, Martin et al. 2005). As such, ZIP models are a type of generalized linear model that facilitate modeling expected counts, probability of an event occurring, and source of zero inflation simultaneously as a function of covariates. I built and compared ZIP models using an information-theoretic model selection framework to facilitate comparison of non-nested models (Burnham and Anderson 2002), and this analysis was completed using the pscl package and zeroinfl function (Zeileis et al. 2008) in the R statistical computing language (R Core Development Team 2006).

I constructed ZIP models in this analysis using the 17 covariates hypothesized to influence collision a priori, and covariates used were identical to those used in logistic regression (Appendix D). Since logistic regression modeling was used to model the influence of covariates on collision presence, I used the top logistic regression model to explicitly model the binomial mixture probabilities of ZIP models. I used the same two-stage modeling process previously described for logistic regression analysis for ZIP modeling, which resulted in comparison of 21 models representing combinations of site and regional covariates in the final model suite (Appendix G).

I evaluated goodness-of-fit and prediction success for the top ZIP model with parametric bootstrap and cross-validation procedures (Efron and Tibshirani 1994). I used 1,000 parametric bootstrap samples of the Pearson χ^2 statistic to test goodness-of-fit for the top ZIP model, using a P-value of 0.05 to test the null hypothesis that the model fits the observed data. I used leave-one-out cross-validation to estimate prediction success for ZIP models. Cross-validation procedures re-fit each model leaving out each data point in turn, and subsequently calculated the expected collision count for the dropped point with the re-fit model. Root-mean-squared error was calculated for each data point, and mean root-mean-squared error was calculated for each model to show the average error between predicted and observed collision counts. I conducted all bootstrap and cross-validation analyses using the R statistical computing language (R Core Development Team 2006).

RESULTS

Site-scale Analysis

I sampled 129.5 km of fence in 140 1x1 km sampling units across south-central and southeast Idaho ≥ 1 time during spring of 2009 and 2010. I detected 86 (2009: n = 48; 2010: n = 38) sage-grouse fence collisions, 58 (2009: n = 28; 2010: n = 30) were in randomly selected sampling areas and 28 (2009: n = 20; 2010: n = 8) were found opportunistically. I collected site-scale covariate data at 172 fence locations (collision sites: n = 86; random points: n = 86). Continuous covariate data were relatively similar for collision and random fence points (Table 1). Local topography and fence characteristic data were also relatively similar for random and collision fence points. I categorized aspect data for site-scale analyses based on their location in a Cartesian plane, with categories for northeast ($0 \leq x < 90$; Collision: n = 11; Random: n = 17), southeast ($90 \leq x < 180$; Collision: n = 19; Random: n = 26), southwest ($180 \leq x < 270$; Collision: n = 29; Random: n = 26), and northwest ($270 \leq x < 0$; Collision: n = 20; Random: n = 14), and those sites too flat for an aspect calculation were placed in a 5th category (Collision: n = 7; Random: n = 3). I also treated fence type categorically, with 3-strand barbed-wire (Collision: n = 21; Random: n = 21), 4-strand barbed-wire (Collision: n = 39; Random: n = 47), 5-strand barbed-wire (Collision: n = 26; Random: n = 12), and other (i.e., split rail, woven wire; Collision: n = 0; Random: n = 6) categories. Lastly, 92 points had only steel t-post's bounding their segment (Collision: n = 63; Random: n = 29), 65 points had wooden fence posts present (Collision: n

= 19; Random: *n* = 46), and fence post data was not available for 15 points (Collision: *n* = 4; Random: *n* = 11) due to lack of site photographs.

I used CART and random forest modeling to classify collision and random fence points based on site-scale data. One-thousand replicates of 10-fold cross-validation procedures resulted in an optimal CART tree size with 1 split (*n* = 574) using the 1-SE rule. The optimal CART model suggested point type was influenced by presence of wooden fence posts, and classified 67 points as random (*n* = 19 collisions misclassified as random, 28.4% error rate) with ≥ 1 wooden post present, and 105 points as collisions (*n* = 38 random points misclassified as collisions, 36.2% error rate) with no wooden posts present. Distance between fence posts was the best surrogate split for wooden post presence (78.5% agreement in predictions) and was used to classify points missing covariate values for wooden post presence (*n* = 15). Distance between posts predicted collision sites with values > 3.94 m, and random sites with values < 3.94 m, and mean prediction success for 1-split models under 10-fold cross-validation was 63.6%. Random forest modeling increased prediction success slightly, with an OOB prediction success estimate of 67.4% (collision sites: 72.1% success; random sites: 62.8% success). Random forest modeling variable importance suggested wooden post presence, distance between posts, and distance to lek were most important in discriminating collision and random points, and permuting these variables in OOB test data resulted in mean accuracy decreases of 7.2%, 4.5%, and 3.6%, respectively (Fig. 1). Partial dependence plots show the probability of a point being random increases with number of wooden posts, decreases sharply at distance between posts > 4 m, and increases with increasing distance to lek (Fig. 2).

Broad-scale Analysis

I used 1x1 km spatial clusters as sampling units to evaluate factors influencing collision risk at patch scales. I sampled 140 1x1 km clusters during spring of 2009 and 2010, however, 17 clusters had no fence present, thus all patch-scale modeling and analyses used 123 sampling units. Average fence length per cluster was 1.05 km (SD = 0.56, n = 123). Mean distance to nearest lek from cluster centroids was 1.37 km (SD = .55, n = 123), and mean size of nearest lek was 16.2 birds (SD = 16.0, n = 123). Average number of leks within 2 km of cell centroids was 1.3 (SD = 1.05, n = 123), and mean sum of lek counts within 2 km was 22.0 birds (SD = 31.9, n = 123). All patch-scale proportion of vegetation cover data was arcsine square-root transformed prior to analyses, and cover of vegetation < 0.5 m tall averaged 0.62 (SD = 0.43, n = 123), cover of vegetation > 0.5 m but ≤ 1.0 m tall averaged 0.79 (SD = 0.41, n = 123), and cover of vegetation > 1.0 m tall averaged 0.18 (SD = 0.28, n = 123). Sagebrush canopy cover of 20-30% had the highest mean coverage ($\bar{x}$ = 1.17, SD = 0.39, n = 123), followed by 10-20% sagebrush canopy cover ($\bar{x}$ = 0.18, SD = 0.23, n = 123), 30-40% sagebrush canopy cover ($\bar{x}$ = 0.10, SD = 0.24, n = 123), and 40-50% sagebrush canopy cover ($\bar{x}$ = 0.02, SD = 0.08, n = 123). Cover of big sagebrush vegetation type averaged 1.28 (SD = 0.31, n = 123), and low sagebrush cover averaged 0.14 (SD = 0.24; n = 123). Mean TRI values averaged 4.33 m (SD = 3.83, n = 123) over sampled 1x1 km clusters. Multiple broad-scale covariates were correlated (Appendix E); however, I did not include significantly correlated predictors together in the same model. I quantified landscape scale lek distribution and count on sampling areas within 8 km of leks on a regional basis. Number of leks with ≥ 1 displaying male in 2009 or 2010 within 8 km of sampled lek routes was relatively similar among regions (MV: *n* = 45; BD: *n* = 52; US: *n* = 50). In contrast, sum of lek counts within 8 km for each region was highest for the BD (2009: *n* = 627; 2010: *n* = 1333) and US regions (2009: *n* = 867; 2010: *n* = 999), and lower for the MV region (2009: *n* = 415; 2010: *n* = 256).

Logistic regression modeling suggested probability of collision presence in a 1x1 km cluster was influenced by region, TRI, and fence length ($\Delta AIC_c = 0$, $w_i = 0.349$; Table 2). I also found weak evidence for the influence of proportion of area covered by vegetation > 1.0 m tall in addition to the previously mentioned covariates on probability of sage-grouse collision presence ($\Delta AIC_c = 1.509$, $w_i = 0.164$; Table 2). The top logistic regression model suggested sage-grouse collision probability was lower in the MV region (β = -2.88, 95% CI = -4.96 to -0.79; Fig. 3), as compared to the BD (β = 1.73, 95% CI = 0.18-3.28; Fig. 3) and US regions (β = 2.15, 95% CI = 0.78-3.53; Fig. 3). Moreover, the top model suggested sage-grouse collision probability decreased considerably with increasing mean TRI (β = -0.33, 95% CI = -0.65 to -0.01; Fig. 3), and increased with increasing fence length over the 1x1 km area (β = 1.10, 95% CI = 0.09-2.12; Fig. 3). The second best model suggested increasing the proportion of area covered by vegetation > 1.0 m tall increased probability of collision presence (β = 0.67, 95% CI = -0.92-2.26), however this parameter was estimated imprecisely and the confidence interval overlapped zero. Hosmer and Lemeshow goodness-of-fit testing failed to reject the null hypothesis that the top model fits the data (P =0.83 , χ^2 = 4.26, DF = 8), and area under the ROC curve for the top model was 0.82, suggesting excellent ability to discriminate between 1x1 km clusters with and without sage-grouse collisions present (Hosmer and Lemeshow 2000). Lastly, jointly maximizing sensitivity (0.75) and specificity

(0.77) resulted in an optimum prediction threshold of 0.32 for the top model, and overall classification success of 0.76 using the optimum prediction threshold suggests good predictive ability.

I used ZIP models to evaluate the influence of broad-scale covariates on corrected collision counts for each 1x1 km cluster, conditional on the top logistic regression model for the binomial mixture probability. Therefore, all ZIP models were constructed with the binomial mixture probability modeled as a function of region, TRI, and fence length at each site. The influence of distance to lek on corrected collision counts was most supported by the data ($\Delta \mathrm{AIC}_c = 0$, $w_i = 0.181$; Table 3). The top ZIP model suggested a decrease in expected collision counts with increasing distance to lek ($\beta = -0.0006$, 95% CI = -0.0008 to -0.0003; Fig. 4). ZIP model selection uncertainty was high and weak evidence for the influence of sagebrush canopy cover and fence length was found in addition to distance to lek ($\Delta \mathrm{AIC}_c < 2$, Table 3). However, there was redundancy of model parameters among the top models (i.e., D2L in all models with $\Delta \mathrm{AIC}_c < 2$), and the additional sagebrush canopy cover and fence length terms in the top group of model were all estimated imprecisely and had 95% confidence intervals that were wide and overlapped zero. Moreover, cross-validated prediction error was similar among the top models, suggesting weak evidence for the influence of parameters in addition to distance to lek on expected collision counts (*range* = 1.51-1.61; Table 3). Parametric bootstrap goodness-of-fit testing failed to reject the null hypothesis that the top ZIP model fit the data ($P = 0.83$).

DISCUSSION

Site-scale Factors

I found evidence for the influence of several factors on sage-grouse fence collision risk at multiple spatial scales in breeding areas of southern Idaho. At the site-scale I found evidence for the influence of technical factors, and analyses of factors discriminating sage-grouse collision and random fence points suggest fence-post type and width of fence segments influence sage-grouse fence-collision risk in breeding areas. Since most previous studies focused on the influence of site-scale parameters, several studies found significant factors discriminating random and collision infrastructure points (Catt et al. 1994, Baines and Summers 1997, Bevanger and Brøseth 2004). Catt et al. (1994) reported increased distance to nearest tree > 2 m tall reduced capercaillie collision risk in Scotland. Baines and Summers (1997) found collision risk for both capercaillie and black grouse was influenced by vegetation composition at collision locations. Bevanger and Brøseth (2004) indicated tetraonid power line collision points were more likely at sites with lower site-scale tree height, and Bevanger (1990) reported collision risk was influenced by site-scale topography in Norway. Although several studies identified relationships between site-scale attributes and collision risk, these studies reported no relationship between site-scale fence attributes and collision risk of European grouse (Catt et al. 1994, Baines and Summers 1997, Bevanger and Brøseth 2000). In contrast, Smallwood et al. (2007) reported burrowing owl (*Athene cunicularia*) collision risk at wind power facilities in California varied as a function of turbine design, with vertical axis turbines and turbines on tubular towers causing disproportionately more fatalities.

Broad-scale Factors

I found evidence for the influence of biological, technical, and topographic features on sage-grouse fence collision risk at broad-spatial scales. Probability of sage-grouse fence collision presence per square-km was related to region, topographic ruggedness, and fence length, and conditional on these factors sage-grouse collision counts per square-km were influenced by distance to nearest lek. In this study collision risk appeared to be much greater in the BD and US regions relative to the MV region. Several previous grouse-collision studies sampled fences over broad spatial scales (Baines and Summers 1997, Bevanger and Brøseth 2000). Although these studies often sampled over broad geographic areas, measurement and analysis of predictor variables usually focused on site-scale covariates, limiting understanding of collision risk at broad spatial-scales. Moreover, I measured, and found relationships of collision with covariates in 3 of 4 groups of factors influencing avian-infrastructure collision risk (i.e., biological, technical, topographic; Bevanger 1994).

Although most avian collision studies only measured covariates at the site-scale, many studies reported spatial variability in collision risk at regional scales. Bevanger and Brøseth (2000) found significant variation of ptarmigan collision risk among fence segments in different regions in Norway. Barrios and Rodríguez (2004) reported 47 avian-wind turbine collisions at 1 site, and only 2 collisions at a second wind-power facility. Patten et al. (2005) found lesser prairie-chicken fence collision mortality was higher in Oklahoma than New Mexico, and hypothesized this was likely due to larger levels of landscape-scale infrastructure fragmentation. Shaw et al. (2010) reported blue crane (*Anthropoides paradiseus*) power line collision in South Africa was influenced by region and presence of cereal grain agriculture. Despite commonly reporting regional variation in collision risk, most studies failed to measure covariates at broad spatial scales and did not explicitly consider how factors at multiple scales may affect collision (e.g., Bevanger and Brøseth 2000, Barrios and Rodríguez 2004).

Differences in broad-scale sage-grouse population densities seem the most plausible explanation for regional differences in collision risk. Landscape-scale sage-grouse lek counts observed in this study suggested broad-scale population densities may influence sage-grouse fence collision risk on southern Idaho breeding areas. Lek counts within 8 km of sampling areas in the BD and US regions were more than double those observed in the MV region. Subsequently, both probability of sage-grouse collision presence and expected collision counts were greater in the BD and US regions than the MV region. Landscape scale lek counts were relatively similar between the BD and US regions, as was expected collision risk. Regional variation of avian-infrastructure collision risk was related to local population indices for a variety of avian species in Scotland, Spain, and South Africa (Baines and Andrew 2003, Barrios and Rodríguez 2004, Shaw et al. 2010). Moreover, modeling efforts accounted for broad-scale differences in biological (e.g., vegetation characteristics, lek distribution, etc.), technical (e.g., fence length), and topographical (e.g., terrain ruggedness) features among regions, and it seems unlikely that regional differences in meteorological conditions were large enough to cause the regional variation observed. However, it is possible regional variation in sage-grouse fence collision risk was related to unmeasured broad-scale covariates.

In addition to regional variation in collision risk, I found evidence for the influence of topographic ruggedness and fence density (i.e., fence length per square km) on sage-grouse collision risk. My results suggest moderate increases in topographic ruggedness resulted in strong reductions in sage-grouse fence collision risk. I hypothesize increasing topographic variation may result in higher flight altitudes, thus reducing fence-collision risk. Direct comparisons of topographic influences on collision reported in other studies are difficult due to differences in measurement or scale. Most previous studies treated topography as a categorical variable. Bevanger (1990) categorized topography of collision and random power line sites, and reported tetraonid collision frequency highest at sites categorized as sloping, top formations, or depressions. Others reported geomorphic forms (e.g., coastline, ridges, etc.) that affected local movement corridors influenced avian-infrastructure collision risk (Bevanger 1994, Cooper and Day 1998); however, I am not aware of any studies quantifying influence of topographic variation at broad spatial scales (e.g., per square-km) on collision risk.

Similarly, I am not aware of studies quantifying the influence of infrastructure density on broad-scale collision risk. My results support the hypothesis that increasing fence density increases collision risk in sage-grouse breeding areas. Both Patten et al. (2005) and Wolfe et al. (2007) suggested landscape-scale infrastructure fragmentation likely increased lesser prairie-chicken collision mortality in Oklahoma relative to New Mexico, however, no estimates of fence or power-line densities were provided. Moreover, most studies used fences or power-line segments themselves as sampling units in analyses, obfuscating effects of infrastructure density on collision (e.g., Baines and Summers 1997, Bevanger and Brøseth 2000, Shaw et al. 2010).

Although probability of collision presence was influenced by region, topography, and fence density, collision counts appeared to also be influenced by a sites' proximity to the nearest sage-grouse lek. Leks are traditional congregation points for sage-grouse during the breeding season (Scott 1942, Patterson 1952, Gibson 1996), and lekking behavior has been hypothesized to influence infrastructure-collision risk for both birds and bats (Bevanger 1994, Baines and Summers 1997, Cryan 2008). Observations of sage-grouse colliding with fences near lekking areas were first reported in the 1940s (Scott 1942). Danvir (2002) reported finding 20 sage-grouse fence strikes along 8.3 km of fence near active leks in an unpublished report from Utah. Multiple studies have suggested infrastructure placed in close proximity to leks may pose a threat to lekking species, however, I provide the first quantifiable link between infrastructure-lek proximity and avian collision risk at multiple spatial scales.

Observed relationships between sage-grouse collision and site and broad-scale factors suggest these models may be useful predictive tools to identify high-risk areas across the landscape. Bevanger (1994) suggested information on behavior, geography and other factors be synthesized to develop infrastructure-collision predictive tools for species of concern. Many previous studies were limited in scope to only high-risk sites or worst-case-scenarios, making landscape-scale prediction difficult (Bevanger 1998). Inherent stochasticity in collision frequency and lack of fine scale space-use data has lead some authors to suggest collision is not predictable at broad spatial scales (Brown and Drewien 1995, Shaw et al.

2010). Shaw et al. (2010) tested broad scale model-predicted collision risk for blue cranes in South Africa, and reported the model was not successful at predicting high-risk sites. However, this model was constructed using expert opinion not field-collected data (Shaw et al. 2010). Spatial aggregation of collision sites reported in this and other studies suggest predictability at some spatial scale (e.g., Janss and Ferrer 2000, Baines and Andrew 2003, Shaw et al. 2010), and my models showed reasonable internally- and cross-validated prediction success despite lack of fine-scale space-use data. Moreover, limited successful prediction using previous avian-collision results may be a function of extrapolation across scales (Miller et al. 2004). Most research only quantified factors influencing collision at the site-scale, and did not attempt to identify the appropriate scale for collision-risk prediction. Since no previous work systematically studied sage-grouse fence collision risk across the landscape, my models could serve as a conceptual framework for prioritizing areas for future mitigation such as fence marking or removal. These models also represent testable hypotheses for future research to assess the accuracy of collision-risk predictions across space and time (Miller et al. 2004).

MANAGEMENT IMPLICATIONS

Evidence suggests sage-grouse fence collision in breeding areas is influenced by biological, topographic, and technical factors at multiple spatial scales. Regional variation in sage-grouse collision risk may be a function of broad-scale population densities. My data suggest that broad-scale fence collision risk is strongly reduced by even moderated increases in topographic ruggedness, and increases with increasing fence density on the landscape. Expected collision counts at broad spatial scales also appear influenced by distance to nearest lek, and decreasing distance to lek resulted in increasing expected collision counts. At the site scale, data suggest collision may be more common on fences constructed using only steel t-post and wider segment widths (> 4 m).

Management mitigation efforts such as fence marking, modification, or removal may be desirable in areas where collision risk is high and deemed unacceptable. Wolfe et al. (2007) suggested fences within 1 km of prairie grouse leks be targeted for marking efforts, however, my data suggest this may not be adequate in some areas. My data suggest management efforts should start with areas of high fence densities (> 1 km of fence per square km), and fences within approximately 2 km of leks. However, my models suggest topographic ruggedness can attenuate the influence of other predictor variables on sage-grouse fence collision risk, with reduced risk at TRI values > 5 m (i.e., average difference in elevation between pixels of 5 m), and nearly eliminated collision risk at TRI values > 10 m. Therefore, the above recommendations primarily apply to areas with relatively flat topography and mean TRI values < 10 m per square-km. Site-scale data also suggest constructing fences using wooden posts may reduce collision risk. My sampling, results, and inference are limited to sage-grouse breeding areas within approximately 2.5 km of active leks, and provide no information on sage-grouse fence-collision risk in other seasons or areas. Others have suggested fences in high use winter areas may pose significant risk to local sage-grouse populations (Danvir 2002). Therefore, future research should attempt to identify high-risk conditions on other sage-grouse seasonal ranges. Future research should also attempt to replicate this work in space and time, and independently validate my model predictions by treating them as testable hypotheses of factors influencing collision risk in other areas.

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LITERATURE CITED

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.

Anderson, W. L. 1978. Waterfowl collisions with power lines at a coal fired power plant. Wildlife Society Bulletin 6:77-83.

Avery, M. L., P. F. Springer, and J. F. Castle. 1978. The composition and seasonal variation of bird losses at a tall tower in southeastern North Dakota. American Birds 32:1114-1121.

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 941-948.

Barrios, L., and A. Rodríguez. 2004. Behavioural and environmental correlates of soaring-bird mortality at on-shore wind turbines. Journal of Applied Ecology 41:72-81.

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.

Bevanger, K. 1990. Topographic aspects of transmission wire collision hazards to game birds in central Norwegian coniferous forest. Fauna Norvegica Serie C, Cinclus 13:11-18.

Bevanger, K. 1994. Bird interactions with utility structures: collision and electrocution, causes and mitigating measures. Ibis 136:412-425.

Bevanger, K. 1995a. Tetraonid mortality caused by collisions with power lines in boreal forest habitats in central Norway. Fauna Norvegica Serie C, Cinclus 18:41-51.

Bevanger, K. 1995b. Estimates and population consequences of tetraonid mortality caused by collisions with high tension power lines in Norway. Journal of Applied Ecology 32:745-753.

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.

Bevanger, K., and H. Brøseth. 2004. Impact of power lines on bird mortality in a subalpine area. Animal Biodiversity and Conservation 27:67-77.

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.

Breiman, L. 2001. Random forests. Machine Learning 45:5-32.

Breiman, L., J. H. Friedman, R. A. Olshen, and C. G. Stone. 1984. Classification and regression trees. Wadsworth International Group, Belmont, California, USA.

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.

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.

Canfield, R. H. 1941. Application of the line intercept method in sampling range vegetation. Journal of Forestry 39:388-394.

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., K. P. Reese, and M. A. Schroeder. 2003. Monitoring of greater sage-grouse habitats and populations. College of Natural Resources Experiment Station, Contribution No. 979, 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. Unpublished Report. 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.

Cooper, B. A., and R. H. Day. 1998. Summer behavior and mortality of dark-rumped petrels and Newell’s shearwaters at power lines on Kauai. Colonial Waterbirds 21:11-19.

Cryan, P. 2008. Mating behavior as a possible cause of bat fatalities at wind turbines. Journal of Wildlife Management 72:845-849.

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.

Danvir, R. F. 2002. Sage-grouse ecology and management in northern Utah sagebrush-steppe. Deseret Land and Livestock Research Report, Woodruff, Utah, USA, 39 pp.

De’ath, G., and K. E. Fabricius. 2000. Classification and regression trees: a powerful yet simple technique for ecological data analysis. Ecology 81:3178-3192.

Doherty, K. E., D. E. Naugle, and B. L. Walker. 2010. Greater sage-grouse nesting habitat: the importance of managing at multiple spatial scales. Journal of Wildlife Management 74:1544-1553.

Efron, B., and R. J. Tibshirani. 1994. An introduction to the bootstrap. Monographs on Statistics and Applied Probability. Chapman & Hall/CRC, Boca Raton, Florida, USA.

Gibson, R. M. 1996. A re-evaluation of hotspot settlement in lekking sage grouse. Animal Behaviour 52:993-1005.

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

Hosmer, D. W., and S. Lemeshow. 2000. Applied logistic regression. Second edition. Wiley Series in Probability and Statistics. John Wiley and Sons, Hoboken, New Jersey, USA.

Janss, G. F. E. 2000. Avian mortality from power lines: a morphologic approach of a species-specific mortality. Biological Conservation 95:353-359.

Janss, G. F. E., and M. Ferrer. 2000. Common crane and great bustard collision with power lines: collision rate and risk exposure. Wildlife Society Bulletin 28:675-680.

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* 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, Volume 38, University of California Press, Berkeley, California, USA.

Kampichler, C., R. Wieland, S. Calmé, H. Weissenberger, and S. Arriaga-Weiss. 2010. Classification in conservation biology: a comparison of five machine-learning methods. Ecological Informatics 5:441-450.

Lambert, D. 1992. Zero-inflated Poisson regression, with an application to defects manufacturing. Technometrics 34:1-14.

Liaw, A., and M. Wiener. 2002. Classification and regression by randomForest. R News 2:18-22.

Malcom, J. M. 1982. Bird collisions with a power transmission line and their relation to botulism at a Montana wetland. Wildlife Society Bulletin 10:297-304.

Marmion, M., M. Luoto, R. K. Heikkinen, and W. Thuiller. 2009. The performance of state-of-the-art modelling techniques depends on geographical distribution of species.

Martin, G. R. 2011. Understanding bird collisions with man-made objects: a sensory ecology approach. Ibis 153:239-254.

Martin, G. R., and J. M. Shaw. 2010. Bird collisions with power lines: failing to see the way ahead? Biological Conservation 2695-2702.

Martin, T. G., B. A. Wintle, J. R. Rhodes, P. M. Kuhnert, S. J. Low-Choy, A. J. Tyre, and H. P. Possingham. 2005. Zero tolerance ecology: improving ecological inference by modeling the source of zero observations. Ecology Letters 8:1235-1246.

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.

Moss, R. 2001. Second extinction of capercaillie (*Tetrao urogallus*) in Scotland? Biological Conservation 101:255-257.

Moss, R., N. Picozzi, R. W. Summer, and D. Baines. 2000. Capercaillie *Tetrao urogallus* in Scotland – demography of a declining population. Ibis 142:259-267.

Patten, M. A., D. H. Wolfe, E. Shochat, and S. K. Sherrod. 2005. Habitat fragmentation, rapid evolution and population persistance. Evolutionary Ecology Research 7:235-249.

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

R Core Development Team. 2006. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Riley, S. J., D. D. DeGloria, and R. Elliot. 1999. A terrain ruggedness index that quantifies topographic heterogeneity. Intermountain Journal of Sciences 5:23:27.

Rollan, Á., J. Real, R. Bosch, A. Tintó, and A. Hernández-Matías. 2010. Modeling the risk of collision with power lines in Bonelli's Eagle *Hieraaetus fasciatus* and its conservation implications. Bird Conservation International 20:279-294.

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., 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, editors, The Birds of North America, Inc., Philadelphia, Pennsylvania, USA.

Scott, J. W. 1942. Mating behavior of the sage-grouse. Auk 59:477-498.

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., C. G. Thelander, M. L. Morrison, and L. M. Rugges. 2007. Burrowing owl mortality in the Altamont Pass wind resource area. Journal of Wildlife Management 71:1513-1524.

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.

White, G. C., and K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46 Supplement:120-138.

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(Suppl. 1):95-104.

Zeileis, A., C. Kleiber, and S. Jackman. 2008. Regression models for count data in R. Journal of Statistical Software 27:1-25.

Table 1. Summary statistics for continuous covariate data collected at random and sage-grouse collision fence points on southern Idaho rangelands during spring of 2009 and 2010.

	Collision points		Random points	
Variable	$\bar{x}$	SD	$\bar{x}$	SD
LS[a]	20.9	20.1	18.3	17.4
D2L[b]	1175.9	751.1	1320.8	585.3
Sum LC[c]	26.2	25.8	28.7	30
Nleks[d]	1.4	0.9	1.3	0.8
Lateral shrub height[e]	30.9	13.6	38.2	17.3
Longitudinal shrub height[f]	38.6	15.4	49.4	27.7
Distance to closest shrub[g]	46.4	31.4	72.4	86.9
SCC[h]	21.5	13.8	25.5	13.4
HDFCS[i]	79.9	17.2	71	20.3
HDFCSA[j]	72.2	20.9	62.3	24.3
FH[k]	109.8	16	108.5	16.1
DBP[l]	5.1	1.14	4.44	1.58
Slope[m]	1.8	1.5	2.6	2.6

[a] Observed sage-grouse count at known active lek nearest to point.

[b] Distance (m) from point to nearest known and active sage-grouse lek.

[c] Sum of observed lek counts from all known active sage-grouse leks within 2 km of point.

[d] Number of known active sage-grouse leks within 2 km of point.

[e] Mean height (cm) of closest shrub lateral to both sides of fence at point.

[f] Mean height (cm) of closest shrub in both directions of point longitudinally along fence.

[g] Mean distance (cm) to closest shrub lateral to both sides of fence at each point.

[h] Percent shrub canopy cover in cardinal directions at each point.

[i] Mean height difference (cm) between fence point and closest shrub lateral to both sides of fence.

[j] Mean height difference (cm) between fence and closest shrubs longitudinal to point.

[k] Height of fence (cm) at each point.

[l] Distance between fence posts (m) bounding segment of each point.

[m] Mean slope (%) measured across and along fence at each point.

Table 2. Top logistic regression models of probability of sage-grouse collision presence during fence collision surveys within 1x1 km clusters on sage-grouse breeding areas of southern Idaho, USA, during spring of 2009 and 2010. I ranked and compared models using Akaike's Information Criteria corrected for small sample sizes (AIC_C) and normalized Akaike model weights (w_i; Burnham and Anderson 2002). Covariates were region (region = Magic Valley, Big Desert, Upper Snake), mean terrain ruggedness in meters (TRI), length of fence (FL), proportion of area covered by vegetation > 1 m tall (VH>1.0), distance to nearest sage-grouse lek (D2L), proportion of area covered by sagebrush canopy cover class 20-30% (SCC2030), and proportion of area covered by sagebrush canopy cover class 40-50% (SCC4050) per square kilometer.

Model	K[a]	AICc	Δ AICc	w_i	-2LL[b]
$P_{(\text{Region + TRI + FL})}$	5	109.524	0.000	0.349	99.011
$P_{(\text{Region + TRI + FL + VH>1.0})}$	6	111.033	1.509	0.164	98.308
$P_{(\text{Region + TRI + FL + D2L})}$	6	111.550	2.026	0.127	98.826
$P_{(\text{Region + TRI})}$	4	111.915	2.391	0.106	103.576
$P_{(\text{Region + TRI + VH>1.0})}$	5	113.018	3.494	0.061	102.506
$P_{(\text{Region + TRI + SCC4050})}$	5	113.653	4.129	0.044	103.141
$P_{(\text{Region + TRI + SCC2030})}$	5	113.966	4.442	0.038	103.453
$P_{(\text{Region + TRI + D2L})}$	5	114.079	4.555	0.036	103.566
$P_{(\text{Region + SCC4050})}$	4	114.381	4.857	0.031	106.042
$P_{(\text{Region + FL})}$	4	115.803	6.279	0.015	107.464

[a] K = no. of model parameters.

[b] -2LL = -2 × maximized log-likelihood for model of interest.

Table 3. Top zero-inflated Poisson (ZIP) regression models of sage-grouse collision count during fence collision surveys within 1x1 km clusters on sage-grouse breeding areas of southern Idaho, USA, during spring of 2009 and 2010. I ranked and compared models using Akaike's Information Criteria corrected for small sample sizes (AIC$_C$) and normalized Akaike model weights (w_i; Burnham and Anderson 2002). Covariates were distance to nearest sage-grouse lek (D2L), proportion of area covered by sagebrush canopy cover class 10-20% (SCC1020), length of fence (FL), proportion of area covered by sagebrush canopy cover class 20-30% (SCC1020), proportion of area covered by sagebrush canopy cover class 30-40% (SCC3040), proportion of area covered by sagebrush canopy cover class 40-50% (SCC4050), mean terrain ruggedness in meters (TRI), and proportion of area covered by vegetation from 0.5-1.0 m tall (0.5<VH<1.0) per square kilometer.

Model[a]	K[b]	AICc	Δ AICc	w_i	-2LL[c]	Prediction error[d]
D2L	7	214.393	0.000	0.181	199.419	1.543
D2L + SCC1020	8	214.711	0.318	0.154	197.448	1.511
D2L + FL	8	215.860	1.467	0.087	198.597	1.542
D2L + SCC4050	8	216.292	1.899	0.070	199.029	1.604
D2L + SCC3040	8	216.374	1.981	0.067	199.110	1.615
D2L + SCC2030	8	216.528	2.136	0.062	199.265	1.540
D2L + TRI	8	216.535	2.143	0.062	199.272	1.561
D2L + 0.5<VH<1.0	8	216.618	2.225	0.059	199.355	1.546
FL + SCC1020	8	216.730	2.338	0.056	199.467	1.565
D2L + TRI + FL	9	217.956	3.563	0.030	198.363	1.576

[a] Model form is $log(\hat{\lambda}) = \hat{\beta}_0 + \hat{\beta}_1(X_1) + \ldots + \hat{\beta}_k(X_k) \mid logit(\hat{p}) = \hat{\beta}_0 + \hat{\beta}_1 \times (Region = US) + \hat{\beta}_2 \times (Region = BD) + \hat{\beta}_3 \times TRI + \hat{\beta}_4 \times FL$, where λ = expected collision count and p = binomial mixture probability.

[b] K = no. of model parameters.

[c] -2LL = -2 × maximized log-likelihood for the model of interest.

[d] Prediction error = mean root-mean-squared error calculated via leave-one-out crossvalidation. This represents the average difference between predicted and observed collision counts over each data point.

Figure 1. Site scale random forest variable importance plot for factors discriminating sage-grouse collision fence points and random fence points as a function of covariates measured during fence sampling on southern Idaho rangelands during spring of 2009 and 2010. Ten-thousand replications of the random forest algorithm were used to rank variable importance based on the normalized difference in classification accuracy for out-of-bag predictions when the variable was included as measured, and the accuracy for out-of-bag predictions when the variable was randomly permuted among the out-of-bag observations.

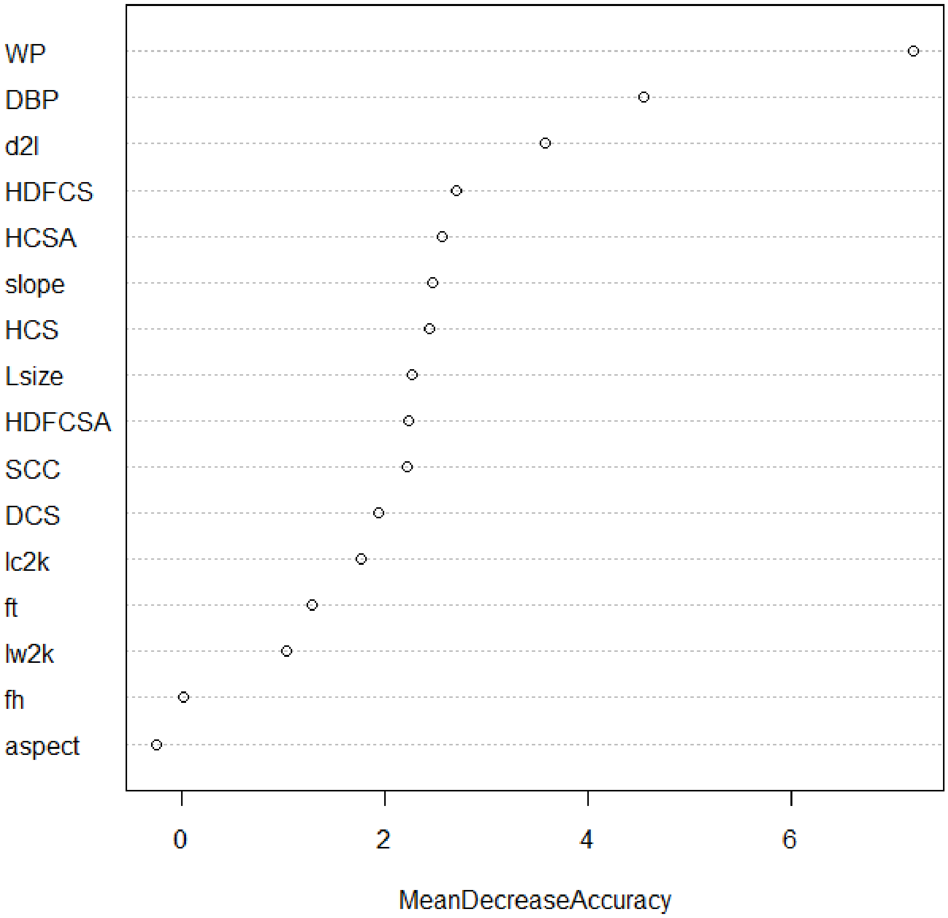
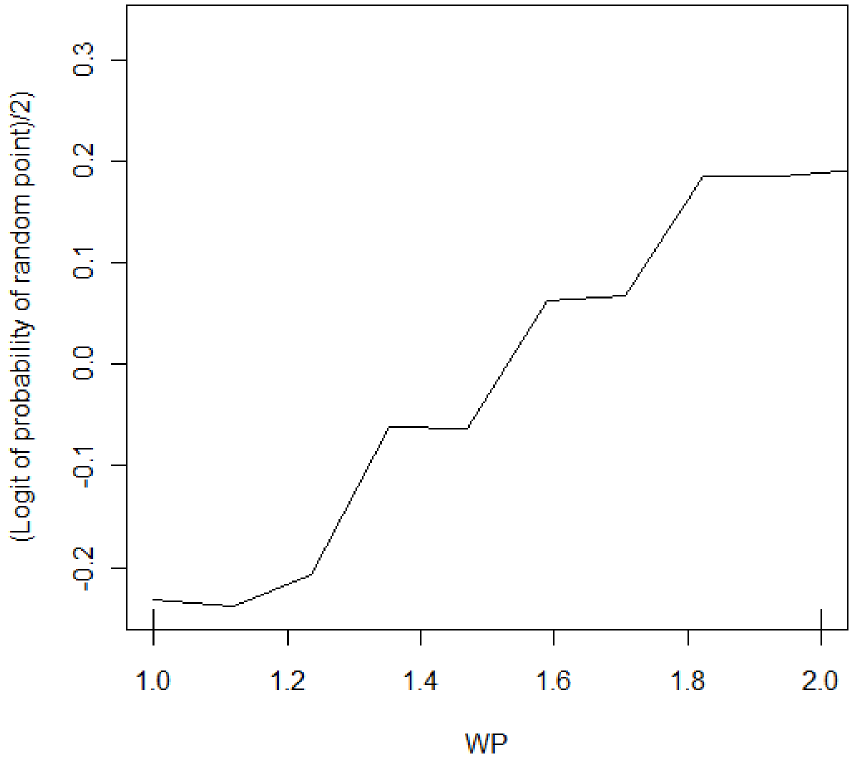
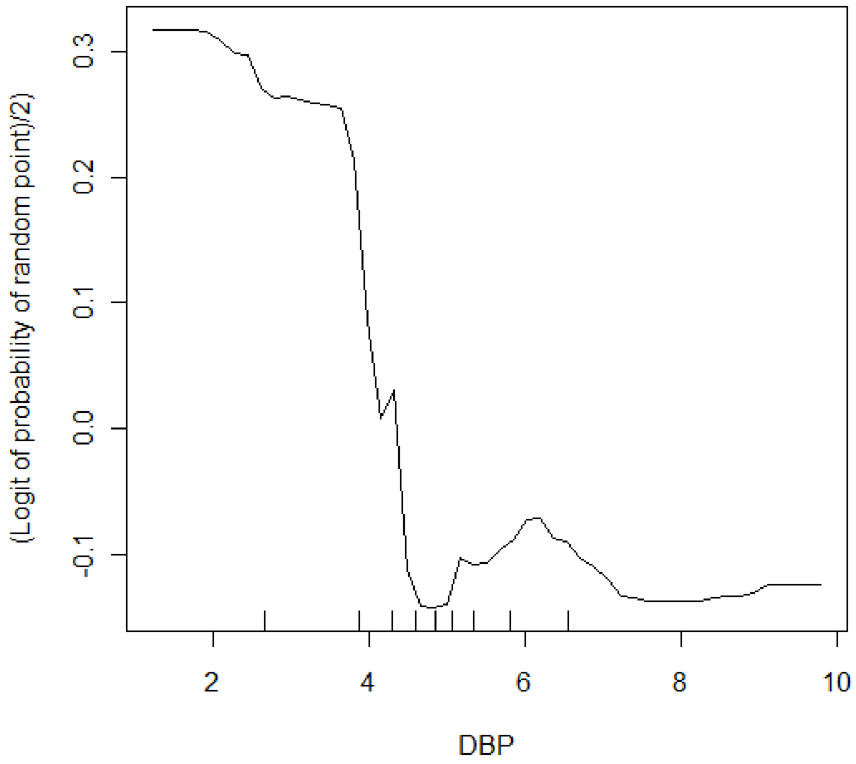


Figure 2. Partial dependence plots for classifications of sage-grouse fence collision points and random fence points (using random forests analysis) as a function of covariates measured during fence sampling on southern Idaho rangelands during spring of 2009 and 2010. Plots are dependence of probability of event on individual predictor variables after the effects of other model variables are averaged out. Partial dependence plots show influence of (a) number of wooden fence posts (WP), (b) distance (m) between fence posts at the fence segment (DBP), and (c) distance (m) to the closest sage-grouse lek (d2l) on probability of a random fence point in 10,000 random forest replications.

a)



b)


(Logit of probability of random point)/2)
0.3
0.2
0.1
0.0
-0.1
2
4
6
8
10
DBP

c)

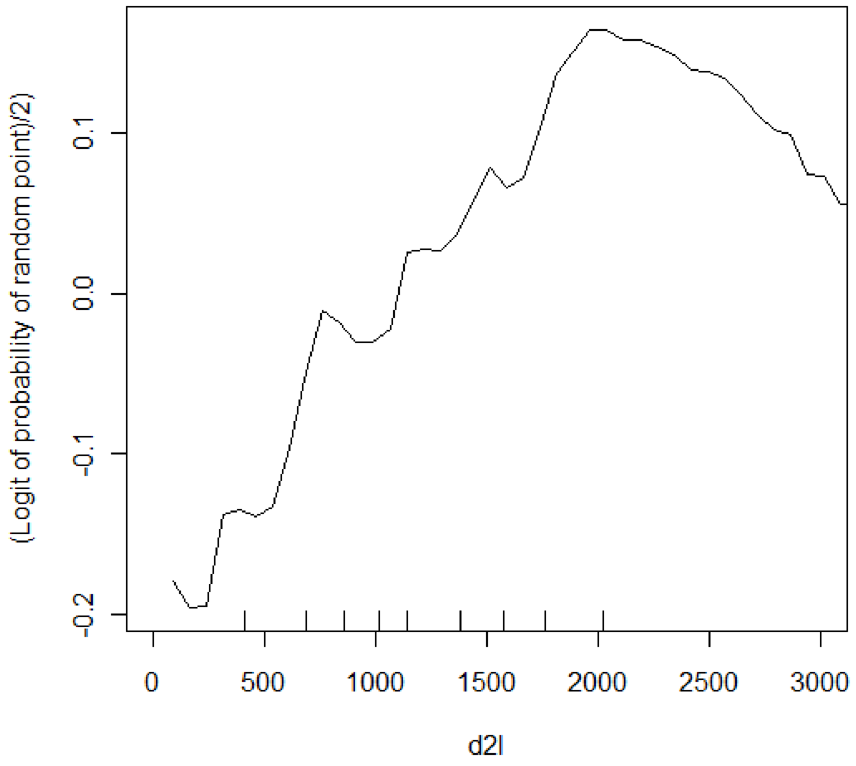
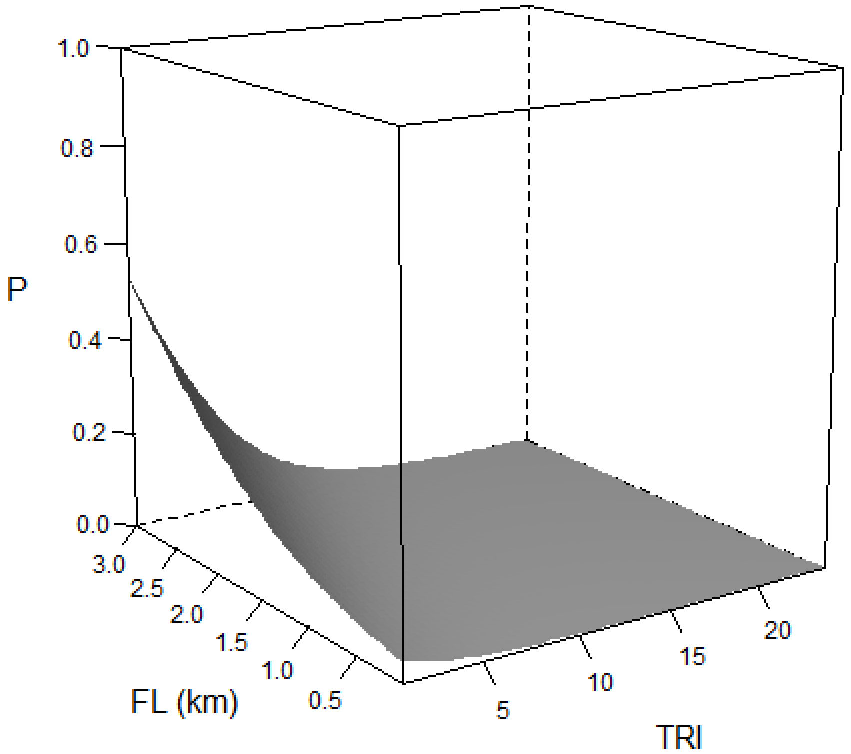
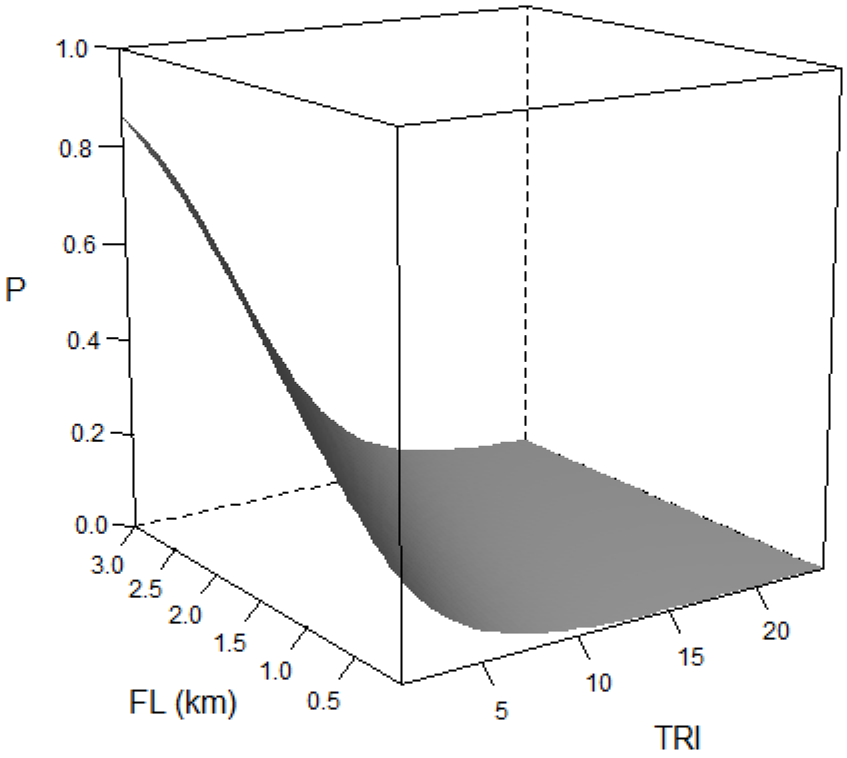

(Logit of probability of random point)/2)
-0.2
-0.1
0.0
0.1
0
500
1000
1500
2000
2500
3000
d2l

Figure 3. Plots of probability of sage-grouse collision site presence as a function of covariates from the top patch-scale logistic regression model from 1x1 km cluster sampling of fences on southern Idaho, USA, rangelands during spring of 2009 and 2010. Probability of collision site presence as a function of terrain ruggedness (TRI; measured in meters) and fence length (FL) in the (a) Magic Valley, (b) Big Desert, and (c) Upper Snake regions.

a)



b)



c)

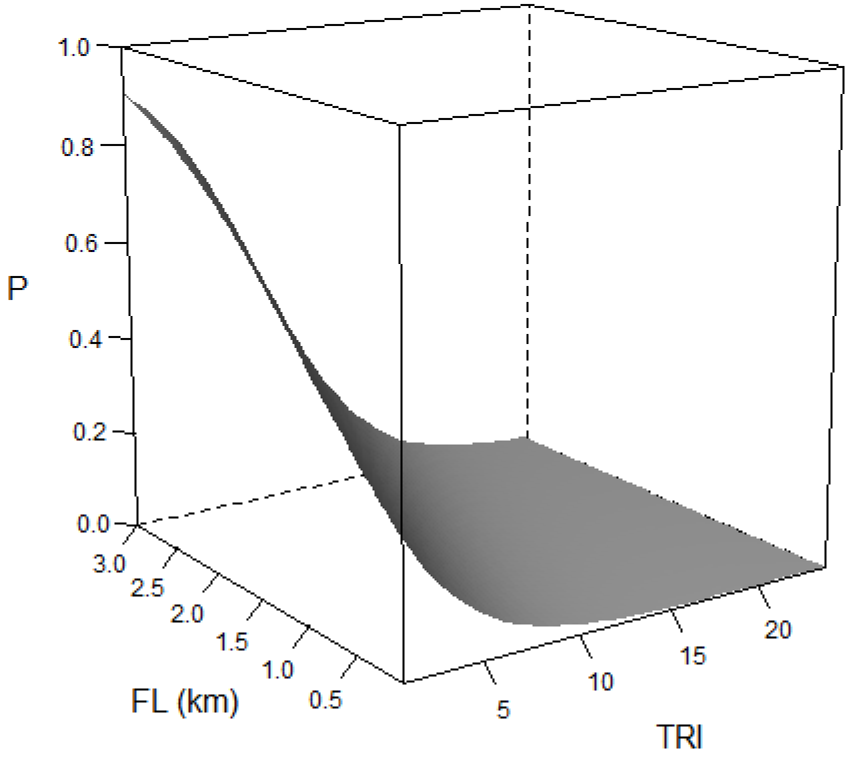
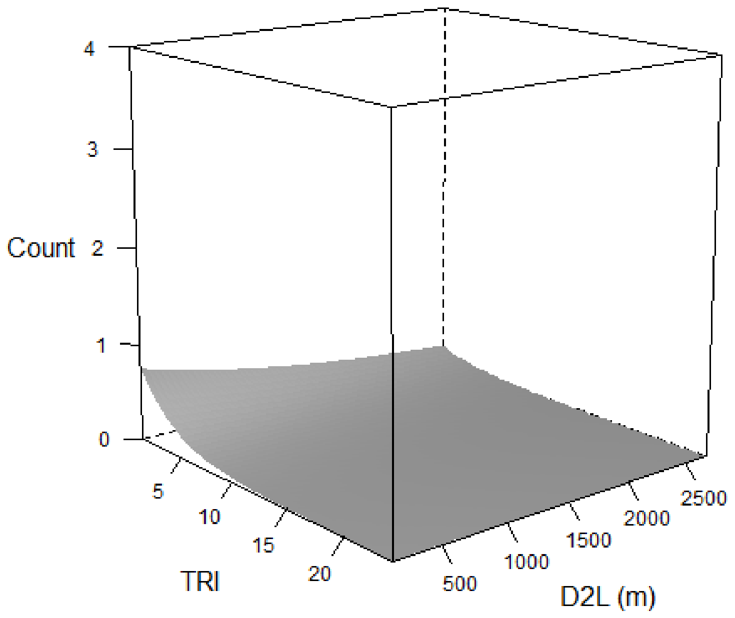
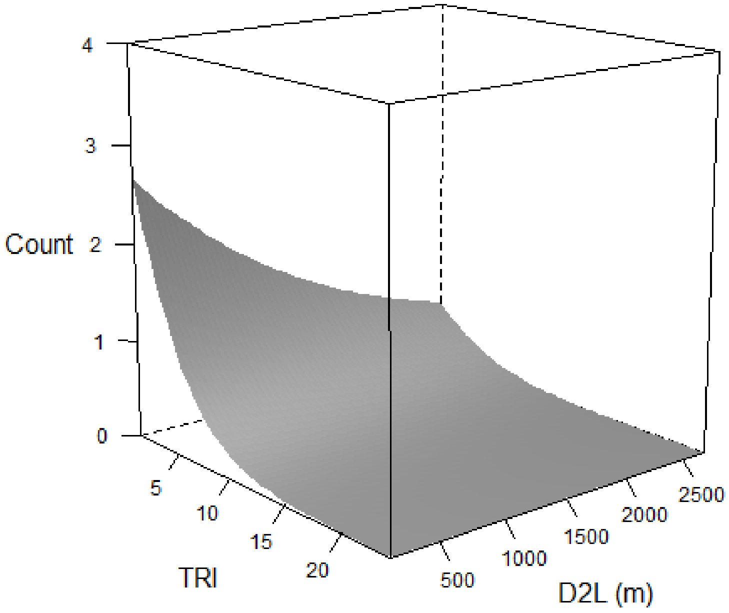


Figure 4. Plots of expected sage-grouse collision count as a function of covariates from the top zero-inflated Poisson regression model, from 1x1 km cluster sampling of fences on southern Idaho, USA, rangelands during spring of 2009 and 2010. Expected collision count per square km as a function of terrain ruggedness (TRI; measured in meters) and distance to lek (D2L), with fence length (FL) held at its mean value in the (a) Magic Valley, (b) Big Desert, and (c) Upper Snake regions. Expected collision count per square km as a function of D2L and FL with TRI held at its mean value in the (d) Magic Valley, (e) Big Desert, and (f) Upper Snake regions.

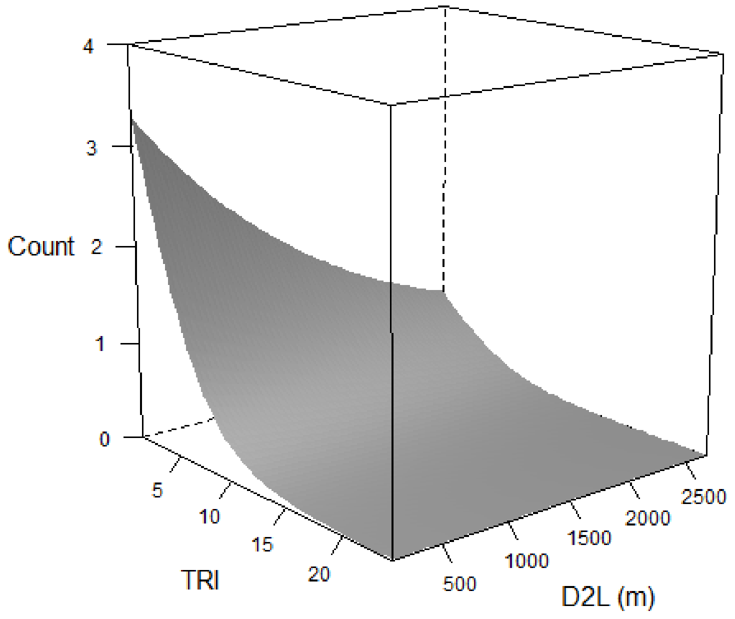
a)



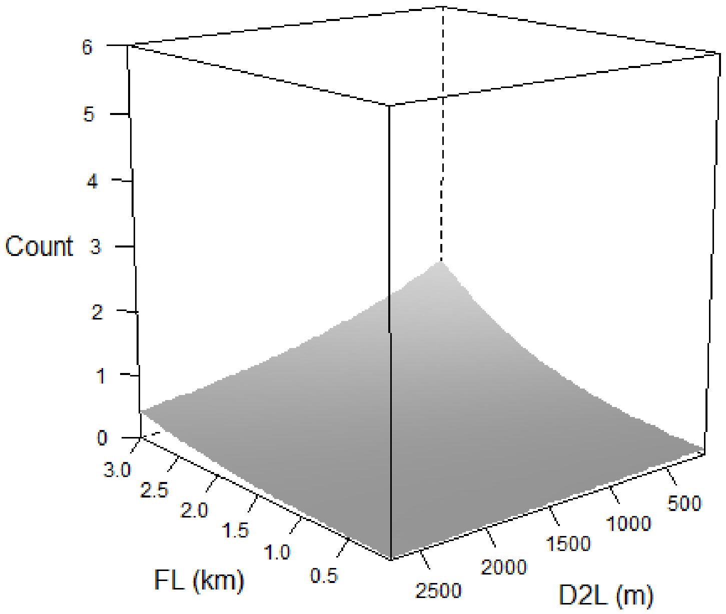
b)



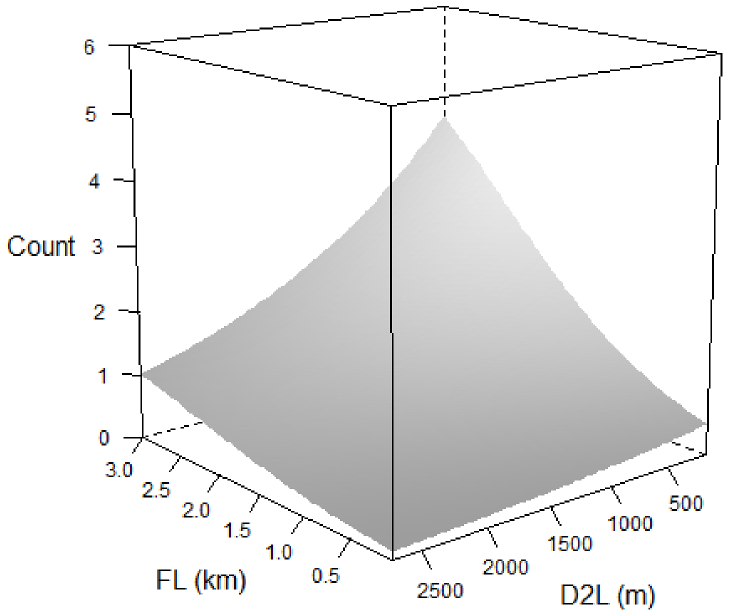
c)



d)


Count
0
1
2
3
4
5
6
FL (km)
3.0
2.5
2.0
1.5
1.0
0.5
D2L (m)
2500
2000
1500
1000
500

e)


Count
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FL (km)
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D2L (m)
2500
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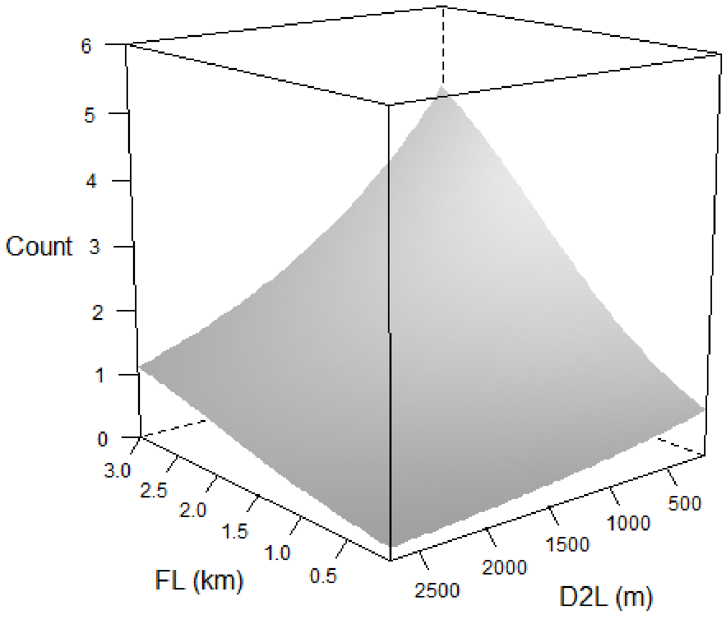
f)


Count
0
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FL (km)
3.0
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D2L (m)
2500
2000
1500
1000
500

CHAPTER 5. GREATER SAGE-GROUSE AND FENCES: AN EXPERIMENTAL TEST OF FENCE MARKING AS A MITIGATION METHOD TO REDUCE COLLISION

INTRODUCTION

Collision with elevated infrastructure has been identified as a widespread and significant mortality source for grouse species in Scotland (Catt et al. 1994, Baines and Summers 1997, Moss et al. 2000, Moss 2001, Baines and Andrew 2003), France (Miquet 1990), and Norway (Bevanger 1990, Bevanger 1995b, Bevanger and Brøseth 2000, Bevanger and Brøseth 2004). In contrast, evidence documenting collision mortality for grouse in North America is limited to studies in Oklahoma with lesser prairie chickens (*Tympanuchus pallidicinctus*; Patten et al. 2005, Wolfe et al. 2007), and studies in Idaho with greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse; Beck et al. 2006).

Although tetraonid collision in Europe is common and widespread, magnitude of collision risk and population level consequences appear to vary by species and region. For example, red grouse (*Lagopus lagopus scoticus*) collisions with fences was more common in Scotland than were capercaillie (*Tetrao urogallus*) and black grouse (*Tetrao tetrix*) collisions (Baines and Summers 1997, Baines and Andrew 2003). Baines and Summers (1997) found 281 avian fence collisions over a one year period, two-thirds of which were red grouse, while 13% each were black grouse and capercaillie. While red grouse collisions were more common in both studies, collision mortality did not appear to influence local red grouse populations (Baines and Summers 1997, Baines and Andrew 2003), whereas collision mortality was a threat to capercaillie due to small population sizes (Catt et al. 1994, Baines and Summers 1997, Moss et al. 2000), and likely contributed to long term capercaillie population declines (Moss 2001).

Bevanger and Brøseth (2000) documented 253 fence collision victims in Norway over a 4 year period, 85% of which were ptarmigan. Although ptarmigan collisions were commonly reported in Norwegian studies, capercaillie and black grouse were also frequent victims, and collision rate differences among species may have been influenced by local space use (Bevanger 1995b) and population densities (Bevanger and Brøseth 2004). Population level impacts of tetraonid collision in Norway are not well understood, however, and losses may exceed harvest mortality in some areas (Bevanger 1995b, Bevanger and Brøseth 2004). Furthermore, collision was often common in winter and early spring, suggesting victims may otherwise have survived to reproduce (Bevanger 1995b, Bevanger and Brøseth 2004). Thus, there is little reason to suspect collision mortality is compensatory in Norwegian tetraonid populations (Bevanger and Brøseth 2004).

Limited research in North America suggests grouse-infrastructure collision poses a serious threat in some areas (Patten et al. 2005, Beck et al. 2006, Wolfe et al. 2007). Wolfe et al. (2007) found 39.8% and 26.5% of all lesser prairie chicken mortality in Oklahoma and New Mexico was caused by collision with barbed-wire fences. Furthermore, the adult female component had the highest collision mortality with both fences and power lines, and mortality peaked during the breeding season (Wolfe et al. 2007). Patten et al. (2005) suggested higher mortality rates in Oklahoma may have resulted in more variable life-history strategies (i.e., larger clutches, and fewer nesting years) relative to lesser prairie chickens in

New Mexico. Further, 33% of mortality for juvenile sage-grouse on an Idaho study site was caused by collision with power lines (Beck et al. 2006), although broad scale collision risk for sage-grouse remains unknown.

Reducing collision risk through increasing infrastructure visibility via marking is a commonly suggested practice where avian collision poses a conservation concern (Bevanger 1994, Alonso et al. 1994, Baines and Andrew 2003). Most research on marking of elevated infrastructure focused on marking static ground wires of power lines (Morkill and Anderson 1991, Alonso et al. 1994, Brown and Drewien 1995). Marking methods used included white, yellow, and red PVC spirals wrapped around the wire (Alonso et al. 1994, Brown and Drewien 1995, Janss and Ferrer 1998), hanging black neoprene strips (Janss and Ferrer 1998), yellow aviation balls (Morkill and Anderson 1991, Brown and Drewien 1995), and yellow square fiberglass plates (Brown and Drewien 1995). Although most research lacked spatial replication, marking generally appears to reduce avian-collision risk for overhead ground wires (Alonso et al. 1994, Savereno et al. 1996, Janss and Ferrer 1998).

Research evaluating fence marking as a mitigation method to reduce collision risk is much less common. However, Baines and Andrew (2003) examined the effectiveness of orange plastic barrier netting at reducing grouse collision risk in Scotland. They found collision rates were reduced on treatment segments for all bird species (71% lower) and woodland grouse (84% lower), with collision rate reductions of 64%, 91%, and 49% for capercaillie, black grouse, and red grouse, respectively (Baines and Andrew 2003).

Concerns over declining populations of sage-grouse range-wide and in Idaho stimulated interest in impacts of fence collision on this species (Braun 1998, Connelly et al.

2004). That sage-grouse occasionally collide with fences has been known since at least the 1940’s (Scott 1942), however this phenomenon has never been systematically studied. Fence marking methods have been developed for reducing grouse collision risk in rangeland habitats, and anecdotal evidence suggests fence marking reduces prairie grouse collision risk (Wolfe et al. 2009). However, no published studies have experimentally evaluated the effectiveness of fence marking at reducing prairie grouse collision risk. Marking fences in sage-grouse habitats may be a desired management option, thus it is necessary to determine effectiveness of marking prior to application of a potentially expensive but untested mitigation method. Therefore, the objective of this study was to determine effectiveness of fence marking as a mitigation method to reduce collision frequency in high-risk sage-grouse breeding habitats.

STUDY AREA

I conducted this study on 8 sage-grouse breeding areas across south-central and southeast Idaho (Fig. 1). Browns Bench was located in southern Twin Falls County, and bounded by Salmon Falls Creek reservoir on the east, and China Mountain on the west, and was dominated by little sagebrush (*Artemisia arbuscula*) and black sagebrush (*A. nova*) habitats. Paddelford Flat was south of Idaho Highway 22 in Blaine County, bordered on the east by Craters of the Moon National Monument, and dominated by mixtures of three-tip (*A. tripartita*) and big sagebrush (*A. tridentata*). Big Desert sites 1 and 3 were south of Arco, Idaho, and southwest of Big Southern Butte in Butte County, and dominated by mixtures of big and three-tip sagebrush, with previously burned grasslands interspersed. Fingers Butte was southwest of Quaking Aspen Butte and east of Fingers Butte in the Big Desert south of

Arco in Butte County, Idaho, and dominated by stands of little and big sagebrush, with areas of bare ground pasture interspersed. Crooked Creek drainage was dominated by little and mixed sagebrush types, and bounded to the south by Idaho Highway 22, and to the north, east, and west by the Beaverhead Mountains and Caribou Targhee National Forest. Table Butte site was located just northwest of Table Butte in Jefferson County, and was primarily dominated by big sagebrush with interspersed areas of grasslands as a result of previous fires. Lidy site was approximately 8 km west of Dubois, Idaho, and bounded to the north by the Beaverhead Mountains and to the south by Idaho Highway 22, and dominated primarily by stands of little sagebrush. Both Lidy and Crooked Creek sites were in Clark County, Idaho. Elevations on fence marking study sites ranged from approximately 1580 m on Browns Bench to approximately 1900 m at Crooked Creek.

METHODS

Field Methods

I conducted a field experiment on 8 sites in south-central and southeast Idaho to test effectiveness of marking fences to reduce sage-grouse fence collision frequency. I found spatially aggregated collision sites (≥ 2 collision sites/km) during preliminary field sampling in spring 2009 (B. S. Stevens, University of Idaho, unpublished data), and replicated the study on these sites to address spatial variability in marking effectiveness. Sites with ≥ 2 collision sites/km in 2009 were used to define potentially high risk areas because high risk sites were necessary to obtain adequate sample sizes (i.e., adequate number of collisions), and spatial aggregation suggests areas of potential high risk. I framed a 3-km fence segment around each collision site aggregation, and used the approximate center of these points as the center of the 3 km study fence segments on each site. I subsequently divided each 3 km segment into 6, 500-m fence segments, which served as the experimental units in this study. I placed 50 m buffer segments between each 500-m experimental unit fence segment to prevent birds from adjusting flight paths laterally along the fence and artificially increasing collision rates in unmarked segments by subsequently striking fences (Baines and Andrew 2003). I randomly selected 3, 500-m fence segments at each site to serve as marked treatments, and used the remaining 3, 500-m segments at each site for unmarked controls. I constructed fence markers using vinyl siding undersill (Wolfe et al. 2009), and increased their visibility further by adding reflective metallic tape (Fig. 2). I placed markers on the top strand of barbed-wire at approximate 1 m intervals, and did not mark lower strands of wire. One-meter intervals were used to approximate the wingspan of sage-grouse. Wolfe et al. (2009) staggered markers on first and third strands of fence for lesser prairie chickens in shortgrass prairie; however, I did not feel this was necessary due to taller vegetation on Idaho rangelands. Further, fence marking would likely not be used as a mitigation strategy if it is not cost effective; therefore this strategy reduced the time needed for fence marking, number of markers and amount of materials needed to mark fences. I initiated this study and completed all fence marker construction and marking activities in February 2010.

I used a repeated measures framework for fence sampling. I sampled study fences at approximate 2-week intervals from 17 March – 26 May 2010, and all sites were sampled 5 times during the study. I conducted sampling with an observer walking each side of the fence in turn while searching for sage-grouse carcasses or feather sign on the ground and feather tufts stuck on the barbed-wire. During these surveys observers walked approximately

1-3 m on either side of the fence, and monitored the area up to approximately 15 m from the fence for carcasses or collision evidence. I defined a collision in this study as detection of a whole carcass or a feather pile (> 5 feathers) within 15 m of the fence, detection of feather tufts stuck in the barbed-wire fence, or detection of a combination of these factors. Despite this definition of a collision site, I was cautious when only feather sign was detected, and if a likely raptor plucking post was present I was conservative and did not classify these sites as collision locations. For example, raptor plucking posts were common in some areas (mostly for passerine species) and were usually located at large wooden fence-posts, with the resulting feather piles scattered from the base of the post in the prevailing wind direction. In contrast, sites deemed collision locations based solely on feather-pile evidence commonly contained large numbers of feathers scattered in the prevailing wind direction from under the fence itself, or very close to the fence. Given this definition of a fence collision the only birds not accounted for were those flying into fences and leaving no feathers either in the fence or on the ground, and no carcass, or victims whose evidence was removed prior to sampling (e.g., Smallwood 2007, Stevens et al. 2011). I used feather characteristics to identify sage-grouse collision remains (Dalke et al. 1963), and sent evidence from unknown avian species to the Feather Identification Laboratory at the Smithsonian Institution, who used whole feather characteristics (Sabo and Laybourne 1994, Woodman et al. 2005), microscopic feather characteristics (Dove and Koch 2010), and DNA barcoding (Dove et al. 2007) to identify species of individual collision victims. Lastly, I measured microsite shrub height at the location of each collision site, and at 1 m in each cardinal direction from collision evidence, to correct collision counts for the influence of vegetation height on detection probability (Stevens et al. 2011).

Traditional experimental design assumes homogenous experimental units to prevent erroneous inference in the presence of confounding variables. All fence segment experimental units in this study were not identical with respect to biological or technical factors (i.e., fence height, type, etc.); therefore we collected data on covariates potentially influencing local scale collision at each segment. To quantify fence segment attributes I used 1-in-5 systematic sampling (Schaeffer et al. 2006) on each of the 48, 500-m fence segments used in this study, where the first individual fence section (defined as length of fence between 2 posts) sampled on each 500-m segment was randomly selected from sections 1-5, and data was recorded at every 5th section thereafter until end of the 500-m fence segment. I collected covariate data at each section, including fence type (e.g., 4-strand barbed-wire, woven-wire, etc.), fence height, and number of markers in the section. Because I collected data during the breeding season, I calculated an index to local sage-grouse population abundance for each fence segment with maximum lek counts at the nearest lek, using data provided by Idaho Department of Fish and Game (hereafter IDFG) during annual lek route surveys (Jenni and Hartzler 1978, Emmons and Braun 1984, Connelly et al. 2003). Lastly, I calculated distance to nearest lek from the middle point of each fence segment in ArcGIS Geographic Information System (GIS) software using lek location data provided by IDFG.

Statistical Methods

Modeling probability of collision presence.-

I used logistic regression to model probability of sage-grouse collision site presence for a given fence-segment survey as a function of covariates using the known fate model and logit link function in Program MARK (White and Burnham 1999). I ranked and compared models using Akaike's Information Criterion corrected for small samples sizes (AIC_C) and normalized Akaike model weights (w_i) within an information-theoretic model selection framework (Burnham and Anderson 2002). I used information-theoretic model selection methods to facilitate comparison of non-nested models, and used Program MARK in this analysis to facilitate use of information theoretic methods and calculation of model weights. I treated individual surveys for each sampling round as experimental units during modeling, and ignored temporal dependence at each fence segment caused by the repeated measures design. Ignoring repeated measures dependence in logistic regression results in unbiased linear model parameter estimates, but potentially underestimated standard errors on model coefficients (Allison 2001). Therefore, I evaluated the influence of ignoring repeated measures on estimated standard errors by re-fitting the top model using generalized estimating equations for longitudinal logistic regression using SAS Version 9.2 (Diggle et al. 1994, Allison 2001; SAS, Cary, NC).

I constructed logistic regression models in this analysis using a priori hypothesized treatment effects and covariates (Table 1). Experimental design-related parameters used in modeling included a 2-level treatment effect (i.e., marked and unmarked fence), and a time effect representing each of the 5 sampling rounds. Biologically plausible random covariates used included size of the nearest lek, distance to nearest lek, and average fence height measurements for each 500-m fence segment. I compared 25 models representing combinations of the above covariates, as well as the constant intercept only or null model in this analysis (Appendix H).

I evaluated goodness-of-fit, discrimination ability, and classification success for the top logistic regression model. I used the Hosmer and Lemeshow test to evaluate goodness-of-fit for the top logistic regression model, using a *P*-value of 0.05 to test the null hypothesis that the model fits the observed data (Hosmer and Lemeshow 2000). I used area under the Receiver Operating Characteristic curve (hereafter ROC) to evaluate ability of the top logistic regression model to discriminate between fence-segment surveys with a collision present and those with no collision present (Hosmer and Lemeshow 2000). Calculation of area under the ROC curve requires pairwise comparison of all subjects where the event occurred (i.e., collision site present) with all subjects where the event did not occur (i.e., no collision site present). For each pairwise comparison the model was used to predict probability of occurrence of a collision site for each observation, and area under the ROC curve represents the proportion of pairwise comparisons where the subject that had a collision site present had a higher predicted probability than the subject without a collision site present (Hosmer and Lemeshow 2000). I conducted goodness-of-fit testing and estimated area under the ROC curve using SAS Version 9.2 (SAS, Cary, NC).

I evaluated classification success for the top logistic regression model using classification tables (Hosmer and Lemeshow 2000). In logistic regression analysis evaluation, sensitivity is defined as the probability of correctly predicting a true event when it occurs, and specificity is defined as the probability of correctly predicting absence of an event when it does not occur (Hosmer and Lemeshow 2000). It is common in logistic regression to use a prediction threshold of 0.5 (i.e., $P(y = 1) \geq 0.5$ = predicted event occurrence), however, this is only optimal when probability of event occurrence and probability of event not occurring are approximately equal (Hosmer and Lemeshow 2000). Therefore, I calculated an optimum prediction threshold for the top logistic regression model as the threshold where sensitivity and specificity were approximately equal, thereby jointly minimizing both types of prediction error (Hosmer and Lemeshow 2000). I used the optimum prediction threshold to construct classification tables used to estimate classification success of the top logistic regression model. I calculated optimum prediction thresholds and classification tables using the R statistical computing language (R Core Development Team 2006).

Modeling collision counts.-

I summarized collision count data over the lekking season for each 500-m fence segment to evaluate influence of marking treatment and covariates on collision counts. I corrected collision counts for detectability based on attributes of individual collision sites with the following model:

$$Corrected\ Count = \frac{1}{\hat{P}}$$

where $\hat{P}$ is estimated detection probability for the given collision site, estimated from the following logistic regression model (Stevens et al. 2011):

$$logit(\hat{P}) = \ln\left(\frac{\hat{P}}{1-\hat{P}}\right) = \hat{\beta}_0 + \hat{\beta}_1(X_1)$$

and

$$\hat{P} = \frac{1}{1 + e^{-\{\hat{\beta}_0 + \hat{\beta}_1(X_1)\}}}$$

where $\hat{\beta}_i$ are linear model coefficients and X_1 is the mean shrub height measured for the individual collision site under consideration. However, I assumed perfect detection for collision sites whose evidence included feathers stuck in barbed wire, because visibility of these collision sites was high and not influenced by local vegetation characteristics. Since count based modeling requires integer-valued random variables, I summed corrected counts over the lekking season for each fence segment, and rounded corrected collision counts for each segment to the nearest whole number. I did not correct counts for collision sign-survival bias because I was unable to measure sign longevity at each study site. As such, my collision counts should be considered as likely biased low due to removal of collision evidence between sampling intervals (Smallwood 2007, Stevens et al. 2011). Further, during searches feathers in the fence and feather piles were counted as collisions with no knowledge of fate of the collision victim. Therefore, my counts are of the number of collision sites present at the time of sampling, and not of collision mortalities, as I had no way to assess crippling bias caused by individual birds flying into fences and dying at a later time or in a different area (Bevanger 1999). Thus, the relationship between the collision itself and the extent of the negative effects on the individual birds was left unstudied, as this is extremely difficult to assess accurately (Bevanger 1999).

I used zero-inflated Poisson regression to model the influence of treatment effects and covariates on corrected collision counts (Lambert 1992, Martin et al. 2005). Zero-inflated Poisson models (hereafter ZIP models) are statistical mixture models, where observed counts are treated as a binomial mixture of Poisson random variables and a point mass at zero. As such, ZIP models take the following form:

$$P(Y = 0) = 1 - \hat{p} + \hat{p} \times e^{-\hat{\lambda}},$$

$$P(Y = r) = \hat{p}\frac{e^{-\hat{\lambda}}\hat{\lambda}^{r}}{r!}, \quad r = 1,2,\ldots,$$

where

$$logit(\hat{p}) = \hat{\beta}_0 + \hat{\beta}_1(X_1) + \ldots + \hat{\beta}_k(X_k)$$

$$log(\hat{\lambda}) = \hat{\beta}_0 + \hat{\beta}_1(X_1) + \ldots + \hat{\beta}_k(X_k).$$

In this analysis r represents integer valued count data, $\hat{\lambda}$ represents the mean number of collisions at a given fence segment, and $\hat{p}$ represents the binomial mixture probability (i.e., probability of the data coming from the Poisson distribution, where 1 - $\hat{p}$ is probability of data coming from inflated zero count; Lambert 1992, Martin et al. 2005). As such, ZIP models are a type of generalized linear model that facilitate modeling both expected counts and probability of an event occurring simultaneously as a function of covariates. I built and compared ZIP models using an information-theoretic model selection framework to facilitate comparison of non-nested models (Burnham and Anderson 2002), and completed this analysis using the pscl package and zeroinfl function (Zeileis et al. 2008) in the R statistical computing language (R Core Development Team 2006).

I constructed ZIP models in this analysis using a priori hypothesized treatment effects and covariates (Table 1). Covariates used in the count segment of ZIP models included a 2-level treatment effect (i.e., marked and unmarked fence), size of the nearest lek, and distance to the nearest lek for each fence segment. Due to limited samples sizes with count data pooled over time (*n* = 48), I used intercept only models for binomial mixture probabilities of the ZIP models. To evaluate evidence for additional overdispersion in the dataset I also fit the top models ($\Delta AIC_c \leq 2$) as zero-inflated negative binomial models (hereafter ZINB models) and compared them to remaining models using information-theoretic methods. Furthermore, I hypothesized 4 biologically plausible interactions (treatment × distance, treatment × lek size, lek size × distance, and treatment × lek size × distance) and added these terms to the count segment individually to the top model. I compared 10 models representing combinations of the above covariates, as well as the constant intercept only or null model in this analysis (Appendix I).

I evaluated goodness-of-fit, calculated confidence intervals on expected collision counts, and prediction success for the top ZIP model with parametric bootstrap and cross-validation procedures (Efron and Tibshirani 1994). I used 1,000 parametric bootstrap samples of the Pearson χ^2 statistic to test goodness-of-fit for the top ZIP model, using a *P*-value of 0.05 to test the null hypothesis that the model fits the observed data. I used 1,000 parametric bootstrap samples to estimate 95% confidence intervals on expected collision counts for various combinations of count model covariate values for the top ZIP model. I used leave-one-out cross-validation to estimate prediction success for ZIP models. Cross-validation procedures used re-fit each model leaving out each data point in turn and subsequently calculated the estimated collision count for the dropped point with the re-fit model. Root-mean-squared error was calculated for each data point, and mean root-mean-squared error was calculated for each model to show the average error between predicted and observed collision counts for each ZIP model. I conducted all bootstrap and cross-validation analyses using the R statistical computing language (R Core Development Team 2006).

RESULTS

I monitored 48, 500-m fence segment experimental units on 8 study areas over 5 sampling occasions during spring 2010. I found a total of 60 avian collision sites during sampling, 56 of which were sage-grouse, and total sage-grouse collision count corrected for influence of microsite shrub height on detectability was 77.9 (Table 2). I found evidence for temporal variation in collision rates within the lekking season, and spatial variation in collision frequency among study sites (Fig. 3, Table 2). My data suggest peak of collision risk from mid March - mid April, with reduced but stable collision risk through the end of the lekking season (Fig. 2). Moreover, 3 times more collision sites were found at Fingers Butte (*n* = 33) than Lidy (*n* = 11), the second highest collision count site. Within sites, sage-grouse collisions were spatially aggregated near leks; approximately 73% (*n* = 41) of sage-grouse collisions were < 500 m from a known lek, 14% (*n* = 8) were from 500-1000 m from a lek, 13% (*n* = 7) were > 1000 m and 2% (*n* = 1) were > 1500 m from a lek. Composition of evidence types found at collision sites was dominated by feather piles (72%, *n* = 42), with smaller numbers of sites having feather piles and feathers lodged in the fence (22%, *n* = 13), only feathers lodged in the fence (5%, *n* = 3), and intact carcasses (2%, *n* = 1). I was able to determine sex from approximately 43% of sage-grouse collision remains (*n* = 24), resulting in 22 male and 2 female known sex collision sites. However, identification of male sage-grouse was greatly facilitated by presence of air-sac and filoplume feathers, thus the observed sex-composition of collisions is likely male-biased.

In addition to variation among sites and within season, I found differences in number of collisions found by treatment type (Fig. 3). Un-corrected sage-grouse fence collision rates pooled across all sites and times were approximately 6 times greater in unmarked control and buffer fence segments (3.5 strikes/km; unmarked: $n = 42$, buffer: $n = 7$) than marked treatment segments (0.6 strikes/km; $n = 7$), resulting in an approximate 83% reduction in uncorrected collision rate at marked fence segments. Corrected collision rates pooled across sites and times were approximately 5.7 times greater in unmarked control segments (4.9 strikes/km; $n = 58.6$) than marked treatment segments (0.9 strikes/km, $n = 10.2$), resulting in an approximate 82% reduction in detection bias corrected collision rate at marked fence segments.

The average of estimated number of markers per treatment fence segment was 479.3 ($SD = 42.2$, $n = 24$, *range* = 370.5-578.1), and average of mean fence height per segment was 110.5 cm ($SD = 9.7$, $n = 48$). Average of estimated proportion of each fence type per 500-m segment was dominated by 4-strand ($\bar{x} = 0.58$, $SD = 0.46$, $n = 48$) and 3-strand ($\bar{x} = 0.32$, $SD = 0.45$, $n = 48$) barbed wire, with lesser amounts of 5-strand ($\bar{x} = 0.07$, $SD = 0.23$, $n = 48$), woven-wire ($\bar{x} = 0.02$, $SD = 0.12$, $n = 48$), 6-strand ($\bar{x} = 0.001$, $SD = 0.008$, $n = 48$), and 2-strand ($\bar{x} = 0.003$, $SD = 0.014$, $n = 48$) fence also present. Mean of maximum sage-grouse lek count at the nearest lek during 2010 for each fence segment was 47.1 birds ($SD = 44.3$, $n = 48$, *range* = 1-127), and mean distance from the midpoint of each fence segment to the nearest sage-grouse lek was 1364.4 m (*SD* = 1194.9, *n* = 48, *range* = 104-4650). Initial logistic regression modeling diagnostics showed the presence of outliers and overly influential data points in the model, therefore logistic regression models were fit without 3 sample-survey data points. Remaining covariate data used in logistic regression modeling had mean lek size of 46.9 (*SD* = 44.9, *n* = 237), mean distance to lek of 1361.8 m (*SD* = 1176.1, *n* = 237), and average of mean fence height per 500-m segment of 110.6 cm (*SD* = 9.5, *n* = 237).

Logistic regression modeling suggested that the probability of sage-grouse collision for a given fence-segment survey was influenced by marking treatment, lek size, distance to nearest lek, and time ($\Delta AIC_c = 0$, $w_i = 0.458$; Table 3). The top model suggested marking treatment reduced sage-grouse collision probability ($\beta = -2.89$, 95% CI = -4.23 to -1.55; Fig. 4), increasing size of nearest lek increased collision probability ($\beta = 0.03$, 95% CI = 0.01-0.04; Fig. 4), and increasing distance of fence segment from the lek reduced collision probability ($\beta = -0.0015$, 95% CI = -0.0026 to -0.0004; Fig. 3). The top model also suggested temporal variation in sage-grouse collision probability, with highest collision risk from approximately mid March – mid April, reduced collision risk from mid April – mid May, and increased collision risk again from mid-late May (Fig. 4). However, time effect parameters were estimated imprecisely (Table 4), and 95% confidence intervals overlapped zero despite a 2-fold increase in observed collision count during the second sampling round relative to the remainder of the lekking season. The top logistic model predicted collision probability was reduced by an average of 93.7% over 5 sampling rounds at marked segments with mean values of lek size and distance to lek (lek size = 46.9 birds; distance = 1361.8 m).

Despite the reduction in collision probability for marked fence segments, predicted sage-grouse collision probabilities at extreme observed values of lek size and distance to lek (i.e., lek size = 127, distance to lek = 104 m) remained high for both unmarked ($\bar{x}$ = 0.79, *range* = 0.60-0.94, *n* = 5) and marked ($\bar{x}$ = 0.25, *range* = 0.08-0.47, *n* = 5) fence segments over the 5 sampling intervals.

I found weak evidence for the influence of fence height in addition to the previously mentioned covariates on sage-grouse collision probability (ΔAIC_c = 0.348, w_i = 0.385; Table 3). The second best model suggested sage-grouse collision probability was reduced by increasing fence height, however, the regression coefficient estimate was very imprecise and the confidence interval overlapped zero (β = -3.8, 95% CI = -9.5-1.9), providing weak support for this covariate. Lastly, fitting the top model using generalized estimating equations to account for temporal dependence in sampling resulted in relatively similar confidence interval coverage and inference as standard least-squares methods (Table 4). However, generalized estimating equations suggest standard logistic regression-methods may have slightly overestimated marking treatment effects, and slightly under-estimated lek size and distance to lek parameters (Table 4).

I evaluated goodness-of-fit, discrimination ability, and classification success for the top logistic regression model in this analysis. Hosmer and Lemeshow goodness-of-fit testing failed to reject the null hypothesis that the model fits observed data (P = 0.95, χ^2 = 2.65, DF = 8). Hosmer and Lemeshow (2000) considered models with area under ROC curve values of 0.8-0.9 as having excellent discrimination ability, and models with area under ROC curve values ≥ 0.9 as having outstanding discrimination ability. Area under the ROC curve for our top logistic regression model was 0.92, suggesting the model has outstanding ability to discriminate between fence segment-surveys with sage-grouse collisions present and those without sage-grouse collisions present. Further, jointly maximizing sensitivity (0.86) and specificity (0.86) resulted in an optimum prediction threshold of 0.16, and overall classification success of 0.86 using the optimum prediction threshold suggests excellent predictive ability for the top model.

I used ZIP models to evaluate marking treatment and covariate influences on corrected collision counts summed over the lekking season for each 500-m fence segment. Marking treatment, lek size, and distance to lek influences of collision count were most supported by the data ($\Delta AIC_c = 0$, $w_i = 0.383$; Table 5), however, I also found some support for an interaction between marking treatment and distance to lek in addition to above covariates ($\Delta AIC_c = 0.365$, $w_i = 0.319$; Table 5). The top ZIP model suggested a reduction in expected collision count as a function of marking treatment ($\beta = -1.35$, 95% CI = -2.2 to -0.5; Fig. 5) and decreasing distance to lek ($\beta = -0.001$, 95% CI = -0.001 to –0.0008; Fig. 5), an increase in expected collision count as a function of increasing lek size ($\beta = 0.017$, 95% CI = 0.010-0.023; Fig. 5), and a binomial mixture probability of 0.62. Despite reduction in expected collision count for marked fence segments, extreme observations of lek size and distance to lek (i.e., lek size = 127, distance to lek = 104 m) resulted in relatively high expected sage-grouse collision count per 500-m segment over the entire lekking period for both unmarked ($E(y) = 8.33$, 95% CI = 8.14-8.51) and marked ($E(y) = 2.17$, 95% CI = 2.10-2.23) fence segments. Parametric bootstrap goodness-of-fit testing for the top model failed to reject the null hypothesis that the model fits the data ($P = 0.31$), and cross-validation procedures suggested models receiving any support from the data (i.e., $\Delta AIC_c < 3$) performed relatively similar in prediction error (*range* = 1.774-2.164, *n* = 5; Table 5). Lastly, the second best model including the treatment × distance interaction predicted a stronger marking treatment effect ($\beta = -2.07$, 95% CI = -3.3 to -0.8), and similar lek size ($\beta = 0.017$, 95% CI = 0.010-0.024) and distance to lek effects ($\beta = -0.001$, 95% CI = -0.001 to –0.0001). However, the interaction effect essentially eliminated the distance to lek effect for marked fence segments only ($\beta = 0.001$, 95% CI = -0.00005 to –0.002).

DISCUSSION

Spatiotemporal Variation in Collision

I observed spatial and temporal variation in sage-grouse fence collision risk, with spatiotemporal aggregation at the site and fence-segment level during the breeding season. While the temporal pattern of collision risk appeared consistent at sites where collision was observed, we found differences in collision probability and counts among sites and fence segments that appeared strongly influenced by lek sizes and proximity on each study area. This evidence suggests collision risk for a given fence segment across the lekking season peaked during mid March-mid April, increased with increasing lek sizes, and decreased with increasing distance from the nearest lek.

Spatiotemporal variation in risk is common in avian collision studies across a range of species and infrastructure types. Baines and Summers (1997) found a 7-fold range of red grouse fence collision rates between study sites in Scotland (0.4-2.7 birds/km/yr). Bevanger and Brøseth (2000) found significant variation in ptarmigan fence collision frequency across both fence segments and years in Norway. Brown and Drewien (1995) reported variation in power line collision among species group, season (fall vs. spring), and years in Colorado. Despite high variability in collision risk over space and time, multiple studies reported spatially aggregated collisions, often referred to as “hot spots” (Miquet 1990, Janss and Ferrer 2000, Baines and Andrew 2003, Shaw et al. 2010). Janss and Ferrer (2000) found 88% of common crane (*Grus grus*) collisions in only 37.5% of power lines under study in Spain, and Shaw et al. (2010) reported multispecies avian collision hot spots common at power line segments in South Africa. Barrios and Rodríguez (2004) indicated 15% of wind turbines responsible for 57% of avian collisions at one site in Spain. High levels of variability in avian-infrastructure collision studies has led several authors to suggest collision is unpredictable (Brown and Drewien 1995, Shaw et al. 2010), however, others have suggested it unlikely that collisions occur randomly (Bevanger 1990), and spatiotemporal aggregation in collision frequency suggests some level of predictability.

I observed spatial variability in sage-grouse collision risk for a given fence segment as a function of proximity to nearest lek. Leks are focal points for sage-grouse breeding, thus the relationship between collision risk and lek proximity is likely a function of spring space-use (Patterson 1952, Jenni and Hartzler 1978, Gibson 1996, Holloran and Anderson 2005). Scott (1942) found 4 dead male sage-grouse collision victims along a fence bisecting a lekking area in Wyoming, and observed a fifth male collide and limp away during lek observations. In an unpublished report, Danvir (2002) reported finding 20 sage-grouse collisions (7 carcasses and 13 other strikes) along 8.3 km of fence within 0.8 km of a lek. Lek mating strategy has been hypothesized to influence collision risk for both birds and bats (Bevanger 1994, Baines and Summers 1997, Cryan 2008), however, I provide the first quantifiable link between lek proximity and infrastructure collision risk. Although no studies quantified the link between lekking and collision risk, several studies have found collision risk to vary with space-use. For example, avian collision risk has been linked to presence of preferred foraging habitats (Baines and Summers 1997, Shaw et al. 2010), home range size (Rollan et al. 2010), and infrastructure proximity to territory (Smallwood et al. 2007).

Spatial variability in sage-grouse collision risk for a given fence segment was also influenced by size of the nearest lek. Lek counts are a commonly used index of local sage-grouse abundance (Jenni and Hartzler 1978, Emmons and Braun 1984, Connelly et al. 2003), suggesting collision risk increases with increasing local sage-grouse density. Similarly, Baines and Andrew (2003) found collision rate correlated with counts of displaying male black grouse in Scotland ($r = 0.60$). Anderson (1978) found collision counts at a power plant slag pit in Illinois were positively correlated with counts of mallard (*Anas platyrhynchos*) and blue winged teal (*Anas discors*) using the area. Positive correlation of collision counts with seasonal abundance indices has also been observed for common kestrels (*Falco tinnuculus*) and griffon vultures (*Gyps fulvus*) at wind turbines in Spain (Barrios and Rodríguez 2004). Increasing abundance likely increases collision risk by increasing the number of flights over a given infrastructure segment (Janss and Ferrer 2000). Although increased risk with increased local abundance is an intuitive result, daily and seasonal variation in lek attendance probability calls into question the use of lek counts as surrogates for local population sizes (Emmons and Braun 1984, Walsh et al. 2004). However, measurement error in regression predictor variables tends to attenuate their estimated effect sizes (Fox 2008); this suggests local abundance may have a stronger influence on collision risk than I measured in this study.

Further, Broms et al. (2010) found annual sage-grouse lek count indices were strongly correlated with the estimated adult male population size ($r = 0.85$), and moderately correlated with estimated size of the entire population ($r = 0.66$) over a 13 year period in Oregon.

In addition to variation related to lek distribution and count, sage-grouse fence collision risk for a given fence segment appeared to vary within the lekking season. Numerous studies reported seasonal variation in grouse collision risk (Baines and Summers 1997, Baines and Andrew 2003, Bevanger and Brøseth 2004, Patten et al. 2005). Ptarmigan fence collision in Norway peaked during winter (0.7 birds/km/month) and spring (0.6 birds/km/month; Bevanger and Brøseth 2000). Baines and Summers (1997) found species-specific seasonal variation in grouse fence collision risk in Scotland, where red grouse collision counts peaked in March (21%), black grouse collision peaked in April (24%), and capercaillie collision peaked in September (26%). Patten et al. (2005) showed a sharp peak in collision mortality during spring breeding (April-June) for radio-marked lesser prairie-chickens in Oklahoma. Radiomarked female lesser prairie-chickens in Oklahoma were disproportionately more susceptible to collision mortality than males, and possibly due to increased movement associated with multiple lek visitation during the breeding season (Wolfe et al. 2007). Interestingly, hen sage-grouse lek attendance typically peaks mid March – mid April (Eng 1963, Jenni and Hartzler 1978, Emmons and Braun 1984), which coincides with my observed peak in sage-grouse fence collision counts. However, I did not monitor lek attendance on my study areas, and lek attendance timing often varies annually (Dalke et al. 1963, Jenni and Hartzler 1978, Emmons and Braun 1984). Moreover, I was unable to determine sex for most collision victims (57%) from remaining evidence, and time effect parameters were imprecisely estimated, despite observing a collision count during the second sampling occasion that was more than double the counts observed for the remainder of the lekking season. Lastly, these data provide no evidence on temporal collision risk outside of the breeding season.

Fence Marking

In addition to variation related to lek size and proximity, observed collision counts per 500 m fence segment were strongly related to presence of fence markers. Wolfe et al. (2009) provided anecdotal evidence for reduced prairie-chicken collision risk after fence marking in Oklahoma; however, no previous studies have experimentally tested fence marker effectiveness with prairie grouse. Baines and Andrew (2003) marked deer fences in Scotland with orange barrier netting and reported approximately 84% fewer woodland grouse on treatment fence segments using uncorrected data, results remarkably similar to mine (83% reduction using raw data). Baines and Andrew (2003) documented species-specific variation in reduced collision counts on marked fence segments, with reductions of 64% and 91% for capercaillie and black grouse, respectively. However, fence marking in Scotland did not completely eliminate collision in high risk areas, which is consistent with my results despite different fence marking methodologies. Although biological, technical, and methodological differences preclude meaningful comparison of fence and power line marker effectiveness, power line marking has generally been successful at reducing avian collision counts. Power line marking has proven effective with both yellow aviation balls and spiral dampers, and observations suggest reduced collision risk was manifested through changes in flight behavior and altitude at marked sites (Morkill and Anderson 1991, Brown and Drewien 1995, Savereno et al. 1996).

Land managers have expressed concerns that reflective markers may be too conspicuous, and therefore aesthetically unpleasing for private landowners and public land users (B. Stevens, University of Idaho, personal observation). However, sage-grouse behavioral ecology and current research on avian vision suggests highly conspicuous markers may be necessary. Bevanger (1994) hypothesized lekking behavior of grouse may influence collision susceptibility due to increased activity patterns in low light conditions. Sage-grouse commonly fly at night and during low-light, early morning conditions during the lekking season (Scott 1942, Patterson 1952, Jenni and Hartzler 1978). Moreover, Scandinavian studies at high latitudes suggest grouse collision often peaks during winter when low light conditions are prevalent on the landscape (Bevanger 1995a, Bevanger and Brøseth 2004).

Current research on avian vision fields suggests many birds have forward projecting blind spots during flight, particularly if they are scanning the ground below for preferred habitats or conspecifics (Martin 2007, Martin and Osorio 2008, Martin and Shaw 2010, Martin 2011). While considerable species-specific variation in avian vision fields exists, evidence suggests birds with precocial young may have smaller amounts of binocular overlap in frontal vision, and forward vision in laterally eyed birds is processed by peripheral areas of the retina, suggesting lower visual resolution (Martin 2007, Martin and Osorio 2008, Martin 2009, Martin 2011). Visual fields have not been estimated for any grouse species, but current visual field estimates for all species with large eyes and enlarged external eyebrows have found presence of blind spot regions above the head that may project forward during flight (Martin 2007, Martin and Orazio 2008). Moreover, fundamental differences in visual perception and processing of visual information exist between birds and humans, and conspicuous or distracting markers may be necessary to catch the attention of flying birds that evolved in open habitats and whose attention may be focused on purposes other than obstacles in their flight path (Martin and Shaw 2010, Martin 2011). More research into the effectiveness of other fence-marker designs and the daily timing of sage-grouse collision risk relative to behavioral ecology and low-light conditions is necessary.

Methodological limitations in this study prevented us from correcting collision counts for sign-survival bias of collision evidence over time between sampling occasions (Smallwood 2007, Smallwood et al. 2010, Stevens et al. 2011). Evidence suggests spatial variability in survival of collision evidence over time is common (Smallwood 2007, Stevens et al. 2011), and this may have introduced bias in our modeling results. I attempted to minimize this source of bias by conducting repeat sampling with a minimum possible interval length (~ 2 weeks, Stevens et al. 2011), however, I did not have the resources to conduct carcass survival experiments on all sites or sample at a higher temporal frequency. There is no evidence to suggest survival bias should vary at small spatial scales within sites as a function of marking treatment, thus my marking assessment should remain valid in the presence of small amounts of sign-survival bias. Despite not correcting for collision sign-survival bias, my temporal sampling intensity was much greater than many published studies which sampled at ≥ 1 month intervals (e.g., Baines and Summers 1997, Janss and Ferrer 2000, Bevanger and Brøseth 2000, Baines and Andrew 2003, Smallwood et al. 2007). Also, some studies used previously published carcass removal rates to correct for this source of bias (Bevanger 1995b, Janss and Ferrer 2000, Smallwood et al. 2007), however spatiotemporal variation in sign-survival bias suggests this is highly imprecise (Smallwood 2007, Stevens et al. 2011), and resulting collision estimates are sensitive to bias estimates (Bevanger 1995b). Thus, despite correcting for influences of microsite vegetation on detection probability of collision evidence, my estimates should still be considered conservative. Clearly, more precise statistical methods are needed to estimate wildlife-infrastructure collision risk while incorporating the sources of bias in field sampling methods.

MANAGEMENT IMPLICATIONS

I provide the first experimental estimates of effectiveness of fence marking at reducing collision risk for any prairie-grouse species. Evidence suggests spatiotemporal variation in sage-grouse collision risk during the breeding season is high, and spatial variation in risk appeared to be a function of lek proximity, local abundance, and fence marking treatment. Data suggest sage-grouse collision risk may be greatest in areas with locally abundant sage-grouse populations, and for fence segments in close proximity to sage-grouse leks. My data also show a strong reduction in sage-grouse fence collision frequency with reflective markers placed in potentially high risk areas, but some level of collision risk should still be expected in extreme circumstances with fences very close to large leks. Fence removal may be a desired alternative to marking in these circumstances if collision risk is deemed unacceptable. My data also suggest previous recommendations to mark fences within 1 km of prairie-grouse leks (Wolfe et al. 2007) may be insufficient for sage-grouse in high risk areas, as we found high probability of collision and expected collision counts out to distances of approximately 2 km from the nearest known lek at large lek sizes. Future experimental research is necessary to evaluate other marker designs prior to field application if reflective markers are not a desired management option, and research should address the social acceptance of different fence marker designs. Future sage-grouse research should also evaluate sex-specific collision risk as a function of lekking behavior and low-light activity patterns. Lastly, unpublished reports have suggested fence collision risk may be high in some high-density winter areas (Danvir 2002). Therefore, more research is necessary to determine sage-grouse fence collision risk in spaces and times outside of breeding seasons and areas.

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LITERATURE CITED

Allison, P. D. 2001. Logistic regression using the SAS system: theory and application. First edition. SAS Institute, Inc., Cary, North Carolina, USA.

Alonso, J. C., J. A. Alonso, and R. Muñoz-Pulido. 1994. Mitigation of bird collisions with transmission lines through groundwire marking. Biological Conservation 67:129-134.

Anderson, W. L. 1978. Waterfowl collisions with power lines at a coal fired power plant. Wildlife Society Bulletin 6:77-83.

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 941-948.

Barrios, L., and A. Rodríguez. 2004. Behavioural and environmental correlates of soaring-bird mortality at on-shore wind turbines. Journal of Applied Ecology 41:72-81.

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.

Bevanger, K. 1990. Topographic aspects of transmission wire collision hazards to game birds in central Norwegian coniferous forest. Fauna Norvegica Serie C, Cinclus 13:11-18.

Bevanger, K. 1994. Bird interactions with utility structures: collision and electrocution, causes and mitigating measures. Ibis 136:412-425.

Bevanger, K. 1995a. Tetraonid mortality caused by collisions with power lines in boreal forest habitats in central Norway. Fauna Norvegica Serie C, Cinclus 18:41-51.

Bevanger, K. 1995b. Estimates and population consequences of tetraonid mortality caused by collisions with high tension power lines in Norway. Journal of Applied Ecology 32:745-753.

Bevanger, K. 1999. Estimating bird mortality caused by collision with power lines and electrocution: a review of methodology. Pages 29-56 *in* M. Ferrer and G. F. E. Janss, editors. Birds and Power Lines: Collision, Electrocution and Breeding. Quercus, Madrid, Spain.

Bevanger, K., and H. Brøseth. 2000. Reindeer *Rangifer tarandus* fences as a mortality factor for ptarmigan *Lagopus* spp. Wildlife Biology 6:121-127.

Bevanger, K., and H. Brøseth. 2004. Impact of power lines on bird mortality in a subalpine area. Animal Biodiversity and Conservation 27:67-77.

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.

Broms, K., J. R. Skalski, J. J. Millspaugh, C. A. Hagen, and J. H. Schulz. Using statistical population reconstruction to estimate demographic trends in small game populations. Journal of Wildlife Management 74:310-317.

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.

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.

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., K. P. Reese, and M. A. Schroeder. 2003. Monitoring of greater sage-grouse habitats and populations. College of Natural Resources Experiment Station, Contribution No. 979, 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. Unpublished Report. Cheyenne, Wyoming, USA.

Cryan, P. 2008. Mating behavior as a possible cause of bat fatalities at wind turbines. Journal of Wildlife Management 72:845-849.

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 27:811-841.

Danvir, R. F. 2002. Sage-grouse ecology and management in northern Utah sagebrush-steppe. Deseret Land and Livestock Research Report, Woodruff, Utah, USA, 39 pp.

Dove, C. J., and S. L. Koch. 2010. Microscopy of feathers: a practical guide for forensic feather identification. Journal of American Society of Trace Evidence Examiners 1:15-61.

Dove, C. J., N. C. Rotzel, M. Heacker, and L. A. Weigt. 2007. Using DNA barcodes to identify bird species involved in birdstrikes. Journal of Wildlife Management 72:1231-1236.

Diggle, P. J., K. Y. Liang, and S. L. Zeger. 1994. The analysis of longitudinal data. Oxford University Press, New York, New York, USA.

Efron, B., and R. J. Tibshirani. 1994. An introduction to the bootstrap. Monographs on Statistics and Applied Probability. Chapman & Hall/CRC, Boca Raton, Florida, 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. Observation on the breeding behavior of make sage-grouse. Journal of Wildlife Management 27:841-846.

Fox, J. 2008. Applied Regression Analysis and Generalized Linear Models. Second edition. Sage Publications, Inc. Thousand Oaks, California, USA.

Gibson, R. M. 1996. A re-evaluation of hotspot settlement in lekking sage grouse. Animal Behaviour 52:993-1005.

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

Hosmer, D. W., and S. Lemeshow. 2000. Applied logistic regression. Second edition. Wiley Series in Probability and Statistics. John Wiley and Sons, Hoboken, New Jersey, USA.

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.

Janss, G. F. E., and M. Ferrer. 1998. Rate of bird collision with power lines: effects of conductor-marking and static wire-marking. Journal of Field Ornithology 69:8-17.

Janss, G. F. E., and M. Ferrer. 2000. Common crane and great bustard collision with power lines: collision rate and risk exposure. Wildlife Society Bulletin 28:675-680.

Lambert, D. 1992. Zero-inflated Poisson regression, with an application to defects manufacturing. Technometrics 34:1-14.

Martin, G. R. 2007. Visual fields and their functions in birds. Journal of Ornithology 148(Suppl. 2):S547-S562.

Martin, G. R. 2009. What is binocular vision for in birds? A birds' eye view. Journal of Vision 9:1-19.

Martin, G. R. 2011. Understanding bird collisions with man-made objects: a sensory ecology approach. Ibis 153:239-254.

Matrin, G. R., and D. Osorio. 2008. Vision in birds. Pages 25-52 *in* A. I. Basbaum, A. Kaneko, G. M. Shepherd, and G. Westheimer, editors. The Senses: A Comprehensive Reference, Vision 1. Volume 1. Academic Press, San Diego, California, USA.

Martin, G. R., and J. M. Shaw. 2010. Bird collisions with power lines: failing to see the way ahead? Biological Conservation 2695-2702.

Martin, T. G., B. A. Wintle, J. R. Rhodes, P. M. Kuhnert, S. J. Low-Choy, A. J. Tyre, and H. P. Possingham. 2005. Zero tolerance ecology: improving ecological inference by modeling the source of zero observations. Ecology Letters 8:1235-1246.

Miquet, A. 1990. Mortality in black grouse *Tetrao tetrix* due to elevated cables. Biological Conservation 54:349-355.

Morkill, A. E., and S. H. Anderson. 1991. Effectiveness of marking powerlines to reduce sandhill crane collisions. Wildlife Society Bulletin 19:442-449.

Moss, R. 2001. Second extinction of capercaillie (*Tetrao urogallus*) in Scotland? Biological Conservation 101:255-257.

Moss, R., N. Picozzi, R. W. Summer, and D. Baines. 2000. Capercaillie *Tetrao urogallus* in Scotland – demography of a declining population. Ibis 142:259-267.

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.

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

R Core Development Team. 2006. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rollan, Á., J. Real, R. Bosch, A. Tintó, and A. Hernández-Matías. 2010. Modeling the risk of collision with power lines in Bonelli's Eagle *Hieraaetus fasciatus* and its conservation implications. Bird Conservation International 20:279-294.

Sabo, B. A., and R. C. Laybourne. 1994. Preparation of avian material recovered from pellets and as prey remains. Journal of Raptor Research 28:192-193.

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.

Scheaffer, R. L., W. Mendenhall III, and R. L. Ott. 2006. Elementary survey sampling. Sixth edition. Duxbury Advanced Series, Thompson Brooks/Cole, Belmont, California, USA.

Scott, J. W. 1942. Mating behavior of the sage-grouse. Auk 59:477-498.

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.

Smallwood, K. S., C. G. Thelander, M. L. Morrison, and L. M. Rugges. 2007. Burrowing owl mortality in the Altamont Pass wind resource area. Journal of Wildlife Management 71:1513-1524.

Smallwood, K. S., D. A. Bell, S. A. Snyder, and J. E. DiDonato. 2010. Novel scavenger removal trials increase wind turbine-caused avian fatality estimates. Journal of Wildlife Management 74:1089-1097.

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.

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.

White, G. C., and K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46 Supplement:120-138.

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 (Suppl. 1):95-104.

Wolfe, D. H., M. A. Patten, and S. K. Sherrod. 2009. Reducing grouse collision mortality by marking fences. Ecological Restoration 27:141-143.

Woodman, N., C. J. Dove, and S. C. Peurach. 2005. A curious pellet from a great horned owl (*Bubo virginianus*). Northeastern Naturalist 12:127-132.

Zeileis, A., C. Kleiber, and S. Jackman. 2008. Regression models for count data in R. Journal of Statistical Software 27:1-25.

Table 1. Parameters used, and justification for each parameter, in logistic and zero-inflated Poisson regression modeling for probability of collision and expected collision count along marked treatment and unmarked control fence segments on 8 sites in southern Idaho, USA, during spring 2010.

Parameter	Justification for hypothesized parameter
Treatment[a,b,c]	Experimental test of marking treatment
Lek size[a,b,d]	Potential density dependence in collision risk[i]
Distance[a,b,e]	Leks are focal points of spring space use[j]
Time[a,f]	Lek attendance is not constant during season[k]
Fence height[a,g]	Influence of infrastructure height on collision[l]
Treatment × distance[b,h]	Treatment effect may vary by lek proximity
Treatment × lek size[b,h]	Treatment effect may vary by lek size
Lek size × distance[b,h]	Density effect may vary by lek proximity
Treatment × lek size × distance[b,h]	Treatment effect may vary by lek size and proximity

[a] Parameter used in logistic regression modeling.

[b] Parameter used in zero-inflated Poisson regression modeling.

[c] Treatment = marked (trt = 1) or unmarked (trt = 0) fence segment.

[d] Lek size = largest count of sage-grouse lek nearest to 500-m fence segment.

[e] Distance = distance from midpoint of 500-m fence segment to nearest sage-grouse lek.

[f] Time = sampling rounds (1-5) of fence collision surveys.

[g] Fence height = mean fence height estimated for each 500-m fence segment.

[h] I added interaction terms individually to the top zero-inflated Poisson model (i.e., $\Delta AIC_c = 0$).

[i] Baines and Andrew (2003), Bevanger and Brøseth (2004).

[j] Patterson (1952), Gibson (1996), Holloran and Anderson (2005).

[k] Jenni and Hartzler (1978), Walsh et al. (2004).

[l] Bevanger (1990), Bevanger (1994).

Table 2. Total number of avian collision sites found by species, and sage-grouse collision count corrected for the influence of microsite vegetation height on detection probability, on marked treatment and unmarked control fence segments for 8 study sites sampled during experimental fence marking studies in southern Idaho, USA, during spring 2010. Avian fence collision species include: greater sage-grouse (GRSG), unknown (UNK), and Other.

	Raw counts			
Site	GRSG	UNK	Other[b]	Corrected GRSG counts
Browns Bench	1	-	-	1.4
Big Desert #1	1	-	-	1.5
Big Desert #3	8	-	-	12.1
Crooked Creek	4	-	1	5.2
Fingers Butte	33	-	-	44.6
Lidy	8	1[a]	2	11.8
Paddelford Flat	-	-	-	0
Table Butte	1	-	-	1.3
Total	56	1	3	77.9

[a] Fence collision victim identified as an unknown Galliforme, but was not identifiable to species.

[b] Other species include rough-legged hawk, horned lark, and Western meadowlark.

Table 3. Top logistic regression models of probability of sage-grouse collision presence during fence collision surveys on experimentally marked and unmarked control fence segments on 8 sage-grouse breeding areas of southern Idaho, USA, during spring 2010. I ranked and compared models using Akaike's Information Criteria corrected for small sample sizes (AIC_C) and normalized Akaike model weights (w_i; Burnham and Anderson 2002).

Model	K[a]	AICc	Δ AICc	w_i	-2LL[b]
$P_{(\text{Time+Treatment+Lek Size+Distance})}$	8	119.115	0.000	0.458	102.483
$P_{(\text{Time+Treatment+Lek Size+Distance+Height})}$	9	119.462	0.348	0.385	100.669
$P_{(\text{Treatment+Lek Size+Distance})}$	4	122.591	3.476	0.081	114.419
$P_{(\text{Treatment+Lek Size+Distance+Height})}$	5	123.018	3.903	0.065	112.758
$P_{(\text{Time+Treatment+Lek Size+Height})}$	8	127.335	8.221	0.008	110.704
$P_{(\text{Time+Treatment+Lek Size})}$	7	129.227	10.113	0.003	114.738
$P_{(\text{Treatment+Lek Size})}$	3	132.327	13.212	0.001	126.224
$P_{(\text{Time+Treatment+Distance+Height})}$	8	132.557	13.442	0.001	115.925

[a] K = no. of model parameters.

[b] -2LL = -2 × maximized log-likelihood for model of interest.

[c] Time = sampling occasions 1-5, Treatment = marked (1) or unmarked (0) fence, Lek Size = maximum count at nearest sage-grouse lek, and Distance = distance to nearest sage-grouse lek.

Table 4. Parameter estimates and 95% confidence intervals from the top logistic regression models of probability of sage-grouse collision presence during fence collision surveys on experimentally marked and unmarked control fence segments on 8 sage-grouse breeding areas of southern Idaho, USA, during spring 2010. Parameter estimates and confidence intervals are presented for logistic regression estimation using generalized estimating equations (GEE) for repeated measures using an unstructured covariance matrix, and standard logistic (Standard) regression treating individual fence-segment surveys as temporally independent sampling units.

	GEE			Standard		
Covariate	Estimate[a]	95% CI		Estimate	95% CI	
Time 1	1.110	-0.362	2.583	1.009	-0.420	2.439
Time 2	0.584	-0.115	1.283	0.547	-0.914	2.008
Time 3	-1.214	-3.058	0.630	-1.348	-3.256	0.560
Time 4	-0.934	-2.017	0.149	-1.136	-2.906	0.635
Time 5[b]	-1.071	-2.378	0.236	-1.391	-2.974	0.192
Treatment[c]	-2.689	-4.162	-1.216	-2.888	-4.229	-1.546
Lek size[d]	0.020	0.009	0.030	0.026	0.014	0.037
Distance[e]	-0.002	-0.003	-0.001	-0.001	-0.003	0.000

[a] β_i regression coefficient estimate from logistic regression modeling.

[b] Time 5 was used as the intercept term in logistic regression modeling.

[c] Treatment = marked (trt = 1) or unmarked (trt = 0) fence segment.

[d] Lek size = largest count of sage-grouse lek nearest to 500-m fence segment.

[e] Distance = distance from midpoint of 500-m fence segment to nearest sage-grouse lek.

Table 5. Top zero-inflated Poisson (ZIP) and zero-inflated Negative Binomial (ZINB) regression models of sage-grouse collision count on experimentally marked and unmarked control fence segments on 8 sage-grouse breeding areas of southern Idaho, USA, during spring 2010. I ranked and compared models using Akaike's Information Criteria corrected for small sample sizes ($\mathrm{AIC_C}$) and normalized Akaike model weights (w_i; Burnham and Anderson 2002). Model covariates are marking treatment (TRT), size of nearest lek (LSize), distance to nearest lek (Dist), and intercept only (.). All models are ZIP models unless otherwise indicated, and all models were fit with an intercept only binomial mixture probability model.

Model[a,b]	K[c]	AICc	Δ AICc	w_i	-2LL[d]	Prediction error[e]
Trt + LSize + Dist	5	111.775	0.000	0.383	100.346	1.871
Trt + LSize + Dist + Trt×Dist	6	112.140	0.365	0.319	98.092	1.774
Trt + Lsize + Dist + Trt×Lsize	6	114.324	2.549	0.107	100.276	1.898
Trt + Lsize + Dist NB[f]	6	114.397	2.622	0.103	100.348	1.863
Trt + Lsize	4	114.700	2.925	0.089	105.770	2.164
Trt	3	140.031	28.256	0.000	133.485	2.836
.	2	159.756	47.981	0.000	155.489	2.997

[a] Model form is $log(\hat{\lambda}) = \hat{\beta}_0 + \hat{\beta}_1(X_1) + \ldots + \hat{\beta}_k(X_k) \mid logit(\hat{p}) = \hat{\beta}_0$, where λ = expected collision count and p = binomial mixture probability.

[b] The following models were attempted but could not be fitted because they could not converge: (Trt + Lsize + Dist + Lsize×Dist); (Trt + Lsize + Dist + Trt×Lsize×Dist); (Trt + Lsize + Dist + Trt×Dist NB).

[c] K = no. of model parameters.

[d] -2LL = -2 × maximized log-likelihood for the model of interest.

[e] Prediction error = mean root-mean-squared error calculated via leave-one-out crossvalidation. This represents the average difference between predicted and observed collision counts for each data point.

[f] NB = zero-inflated negative binomial regression model.

Figure 1. Southern Idaho, USA, study areas, where I studied effectiveness of fence marking at reducing fence collision risk in sage-grouse breeding areas. Stars represent 8 study sites used for fence marking field experiment during spring of 2010.

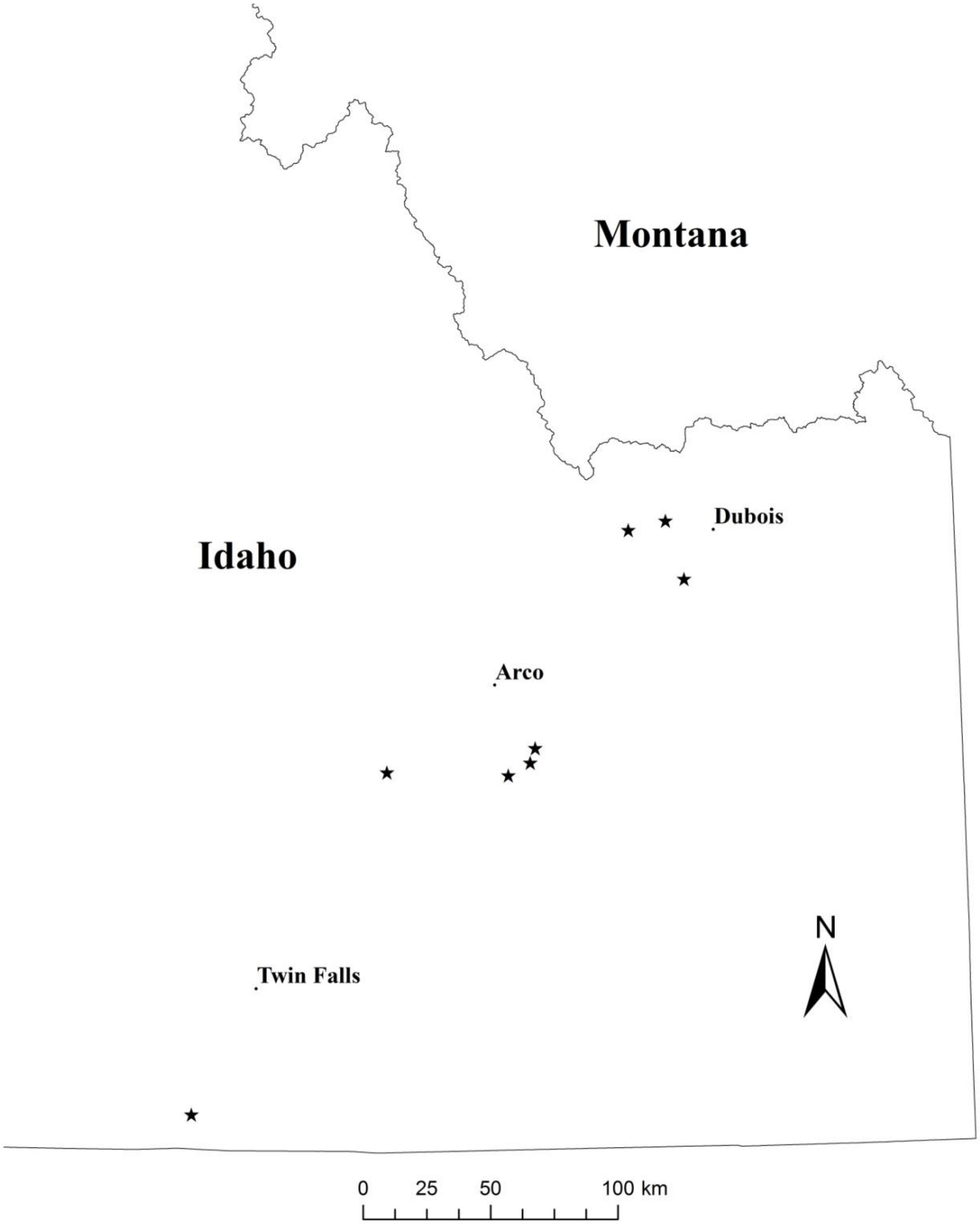
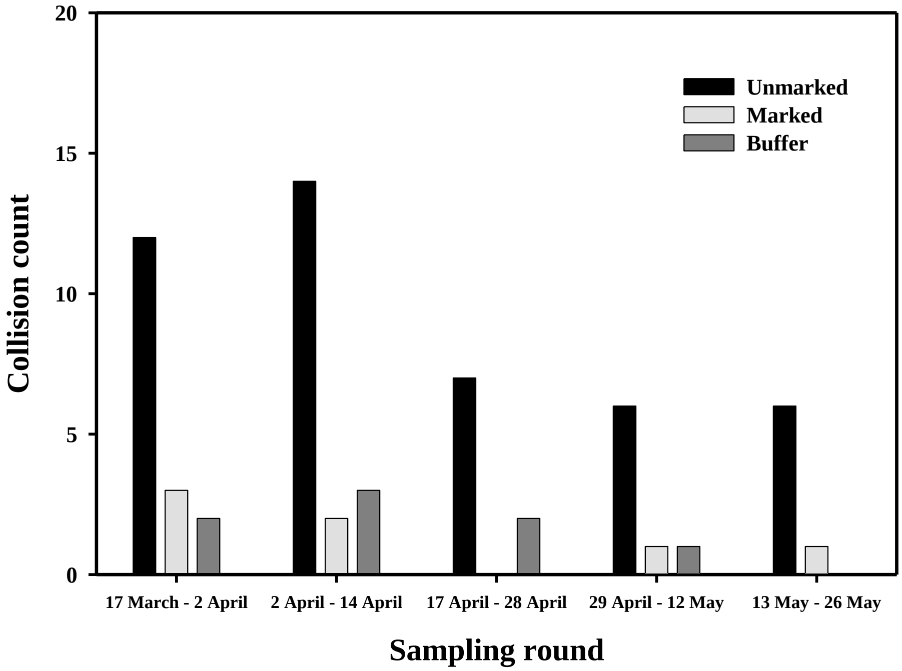


Figure 2. Reflective 8 cm vinyl siding undersill fence markers (modified from Wolfe et al. 2009) used to mark fences on 8 sites in southern Idaho during spring 2010.

Figure 3. Total number of avian collision sites located per treatment type and time period during experimental fence marking studies in southern Idaho during spring 2010. (a) Raw uncorrected counts of avian collision sites. (b) Counts of avian collision sites corrected for the influence of microsite shrub height on collision site detectability.

a)



b)

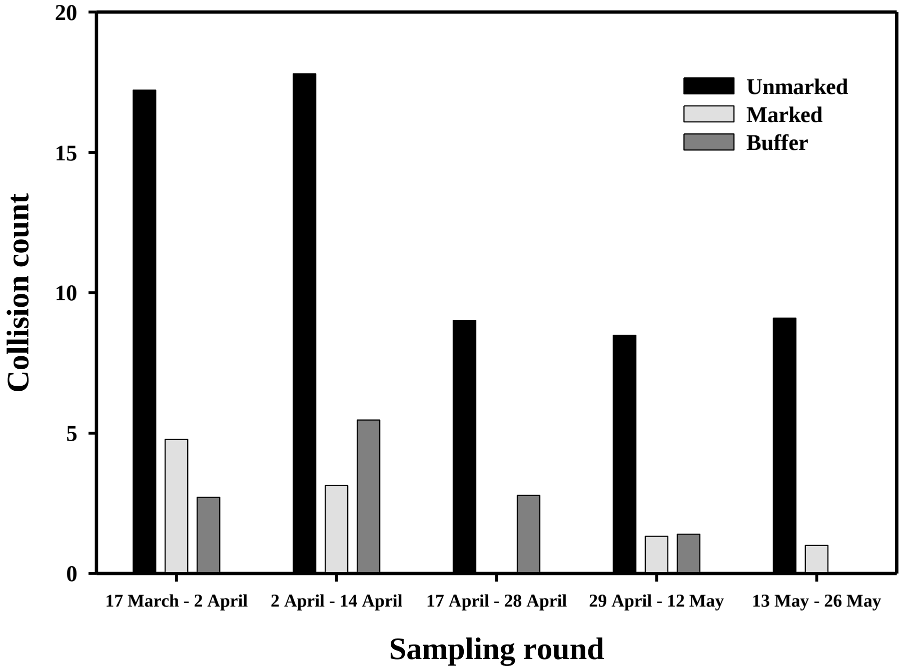
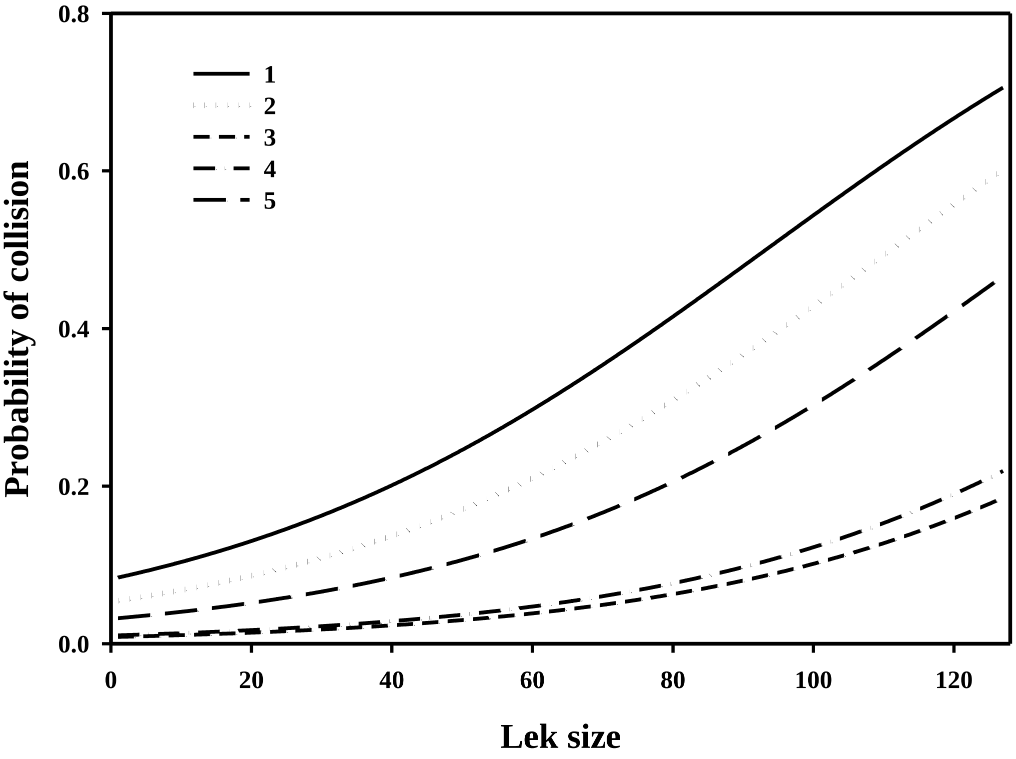
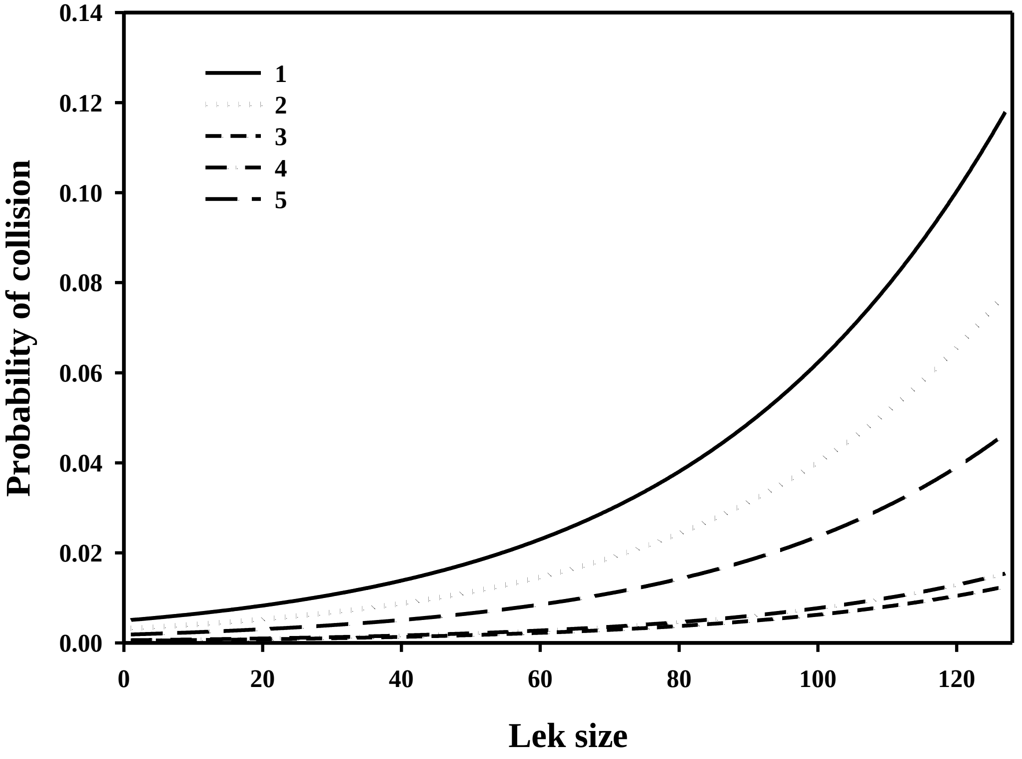

20
15
10
5
0
Collision count
Unmarked
Marked
Buffer
17 March - 2 April
2 April - 14 April
17 April - 28 April
29 April - 12 May
13 May - 26 May
Sampling round

Figure 4. Plots of probability of sage-grouse collision site present as a function of covariates from the top logistic regression model, for a given fence segment-survey on southern Idaho, USA, study sites during spring 2010. The 5 lines on each figure represent 5 sampling occasions over the sage-grouse lekking season. All plots were constructed with other covariates held at mean values (lek size = 46.9 birds, distance to lek = 1361.9 m). (a) Probability of collision site present as a function of size of nearest lek and time for unmarked fence segments. (b) Probability of collision site present as a function of size of nearest lek and time for marked fence segments. (c) Probability of collision site present as a function of distance to nearest lek and time for unmarked fence segments. (d) Probability of collision site present as a function of distance to nearest lek and time for marked fence segments.

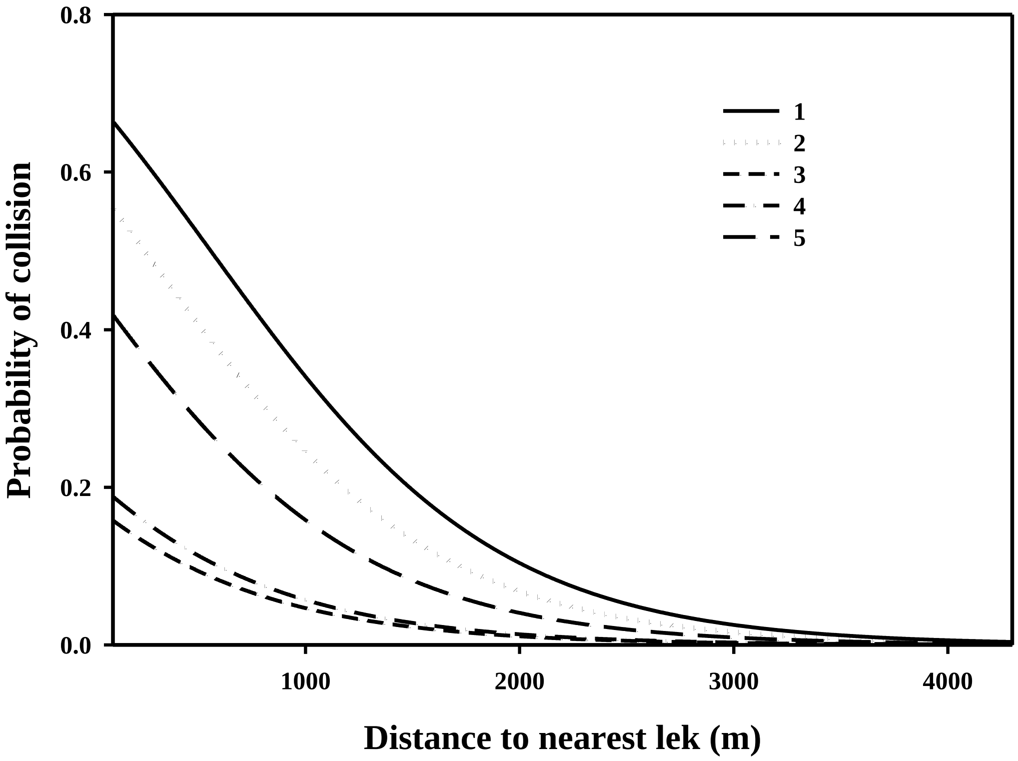
a)



b)


0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Probability of collision
0
20
40
60
80
100
120
Lek size
1
2
3
4
5

c)


0.8
0.6
0.4
0.2
0.0
Probability of collision
1000
2000
3000
4000
Distance to nearest lek (m)
1
2
3
4
5

d)

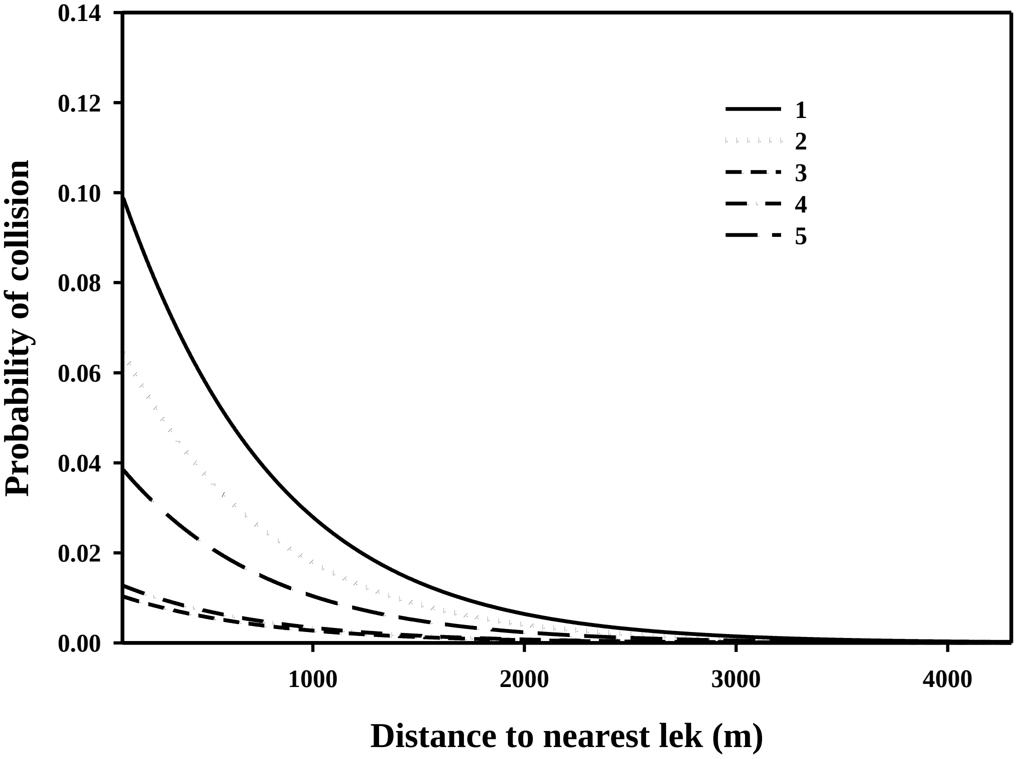
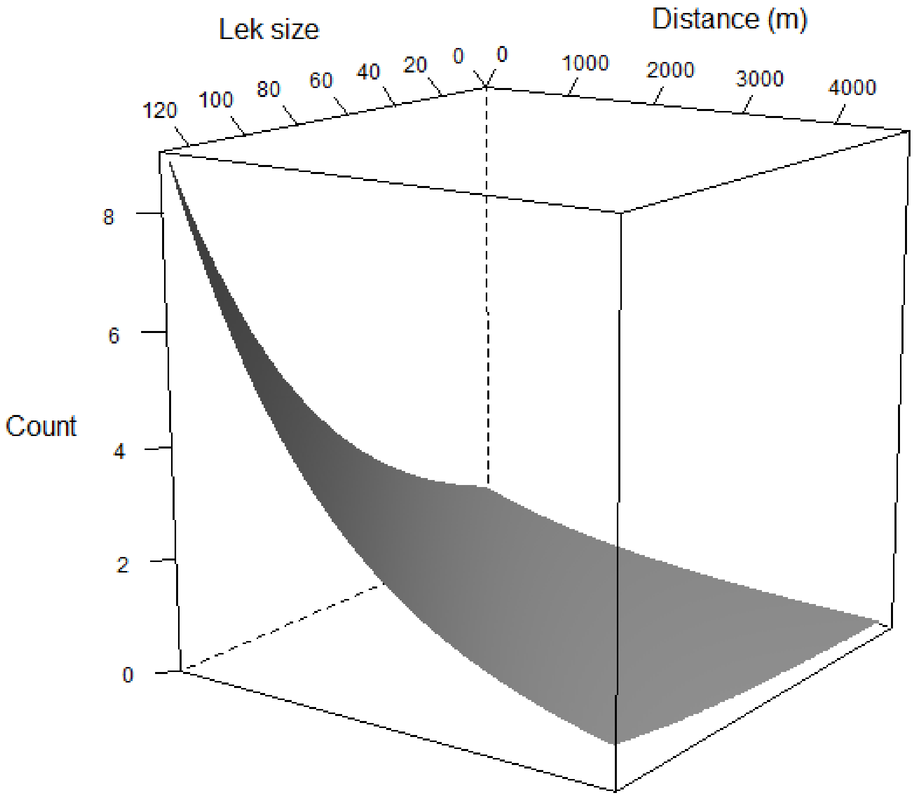
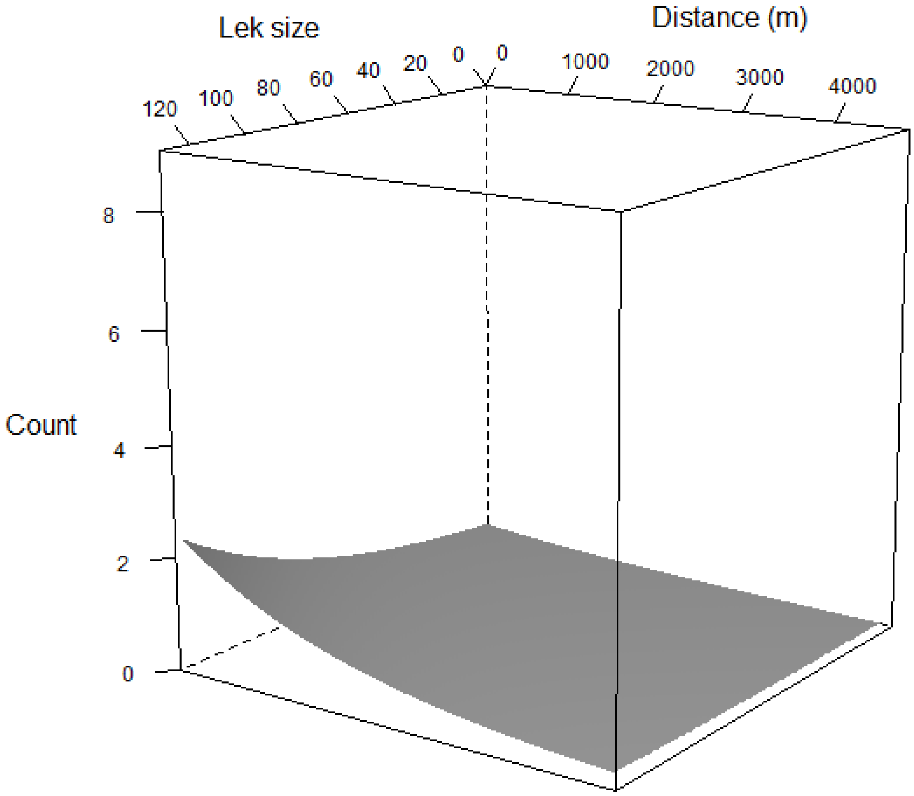

0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Probability of collision
1000
2000
3000
4000
Distance to nearest lek (m)
1
2
3
4
5

Figure 5. Plots of expected sage-grouse collision count per lekking season as a function of covariates from the top zero-inflated Poisson regression model, for a given 500-m fence segment on southern Idaho, USA, study sites during spring 2010. (a) Expected collision count as a function of size of nearest lek and distance to lek for unmarked fence segments. (b) Expected collision count as a function of size of nearest lek and distance to lek for marked fence segments.

a)



B)


Lek size
120
100
80
60
40
20
0
0
1000
2000
3000
4000
Distance (m)
Count
8
6
4
2
0

Appendix A. Models of daily survival rate (*S*) of female pheasant carcasses and their sign as a function of local-scale and microhabitat characteristics on southern Idaho sites, during spring 2009.

Model	Model description
Local-scale models	
$S_{(.)}$	*S* constant on both study areas
$S_{(\mathrm{Habitat})}$	*S* differs between areas dominated by big and little sagebrush
$S_{(\mathrm{Road})}$	*S* influenced by presence of roads
$S_{(\mathrm{Site})}$	*S* differs between BB and US study areas
$S_{(\mathrm{Dist})}$	*S* influenced by distance of evidence from the fence
$S_{(\mathrm{Habitat+Road})}$	*S* a function of both habitat type and road presence
$S_{(\mathrm{Site+Habitat})}$	*S* a function of both study area and habitat type
$S_{(\mathrm{Habitat+Dist})}$	*S* influenced by habitat and distance of evidence from fence
$S_{(\mathrm{Site+Road})}$	*S* a function of both study area and road presence
$S_{(\mathrm{Road+Dist})}$	*S* a function of road presence and distance of evidence from fence
$S_{(\mathrm{Site+Dist})}$	*S* a function of study area and distance of evidence from fence
$S_{(\mathrm{Habitat+Road+Dist})}$	*S* a function of habitat, road presence, and distance from fence
$S_{(\mathrm{Site+Habitat+Road})}$	*S* a function of study area, habitat type, and road presence
$S_{(\mathrm{Site+Habitat+Dist})}$	*S* a function of site, habitat, and distance of evidence from fence
$S_{(\mathrm{Site+Road+Dist})}$	*S* a function of site, road presence, and distance from fence
$S_{(\mathrm{Site+Habitat+Road+Dist})}$	*S* a function of site, habitat, road presence, and distance from fence
Microhabitat models	
$S_{(.)}$	*S* constant on both study areas
$S_{(\mathrm{GH})}$	*S* influenced by grass ht at evidence location
$S_{(\mathrm{VC})}$	*S* influenced by visual concealment at evidence location
$S_{(\mathrm{CC})}$	*S* influenced by % shrub canopy coverage at evidence location
$S_{(\mathrm{SH})}$	*S* influenced by shrub ht at evidence location
$S_{(\mathrm{GH+SH})}$	*S* a function of grass ht and shrub ht at evidence location
$S_{(\mathrm{GH+CC})}$	*S* a function of grass ht and % shrub canopy coverage at location
$S_{(\mathrm{GH+SH+GH} \times \mathrm{SH})}$	*S* a function of grass ht, shrub ht, and their interaction
$S_{(\mathrm{GH+CC+GH} \times \mathrm{CC})}$	*S* a function of grass ht, % shrub canopy coverage, and the interaction between grass ht and % shrub canopy coverage

$S_{(GH+CC+SH+VC)}$	*S* a function of grass ht, % shrub canopy coverage, shrub ht, and visual concealment

Appendix B. Models of female pheasant carcass detectability (p) during fence line surveys as a function of local-scale and microhabitat characteristics on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009.

Model	Model description
Local-scale models	
$p_{(.)}$	p constant
$p_{(\mathrm{Habitat})}$	p differs between big and little sagebrush
$p_{(\mathrm{Snow})}$	p influenced by snowing conditions during sampling
$p_{(\mathrm{Dist})}$	p influenced by distance of carcass from fence
$p_{(\mathrm{Observer})}$	p differs by experience level of observer
$p_{(\mathrm{Habitat+Dist})}$	p a function of habitat and distance from fence
$p_{(\mathrm{Habitat+Observer})}$	p a function of habitat type and observer experience
$p_{(\mathrm{Habitat+Snow})}$	p a function of habitat type and snowing conditions
$p_{(\mathrm{Observer+Snow})}$	p a function of observer and snowing conditions
$p_{(\mathrm{Dist+Snow})}$	p a function of distance from fence and snow
$p_{(\mathrm{Dist+Observer})}$	p a function of distance from fence and observer
$p_{(\mathrm{Habitat+Dist+Snow})}$	p a function of habitat type, distance from fence, and snowing conditions
$p_{(\mathrm{Habitat+Observer+Snow})}$	p a function of habitat type, observer experience, and snowing conditions
$p_{(\mathrm{Habitat+Dist+Observer})}$	p a function of habitat type, distance from fence, and observer experience
$p_{(\mathrm{Dist+Observer+Snow})}$	p a function of distance from fence, observer experience, and snowing conditions
$p_{(\mathrm{Habitat+Dist+Observer+Snow})}$	p a function of habitat, distance from fence, observer experience, and snowing conditions
Microhabitat models	
$p_{(.)}$	p constant
$p_{(\mathrm{GH})}$	p influenced by grass ht
$p_{(\mathrm{SH})}$	p influenced by shrub ht
$p_{(\mathrm{CC})}$	p influenced by % shrub canopy coverage

$p_{(VC)}$	p influenced by visual concealment
$p_{(GH+SH)}$	p a function of grass ht and shrub ht
$p_{(GH+CC)}$	p a function of grass ht and % canopy coverage
$p_{(GH+CC+GH \times CC)}$	p a function of grass ht, % canopy coverage, and the interaction between grass ht and % canopy coverage
$p_{(GH+SH+GH \times SH)}$	p a function of grass ht, shrub ht, and the interaction between grass ht and shrub ht

Appendix C. Length of fence sampled during avian fence collision surveys in southern Idaho, USA, during spring 2009 and 2010.

	Fence length[a]	
Site	2009	2010
East Jarbidge		
Antelope Pocket	7	7.3
Browns Bench	9.1	4.5
North Magic Valley		
North Shoshone	2	4.3
Paddleford Flats	3	1.6
Picabo Hills	5.6	4.9
Timmerman	1.3	-
Big Desert		
Big Desert #1	1.1	1.6
Big Desert #3	1	2.1
Big Desert #5	5.6	-
Fingers Butte	5.5	7.5
Upper Snake		
Crooked Creek	2.5	1
Lidy	3.1	5.3
Medicine Lodge	4.1	5.8
Plano	6.1	6.9
Red Road	5.7	6.2
Table Butte	3.3	4.2
Total	66.2	63.3

[a] Fence length was measured in km.

Appendix D. Site and broad-scale parameters used, and justification for each parameter, in CART, random forest, logistic regression, and zero-inflated Poisson regression modeling to classify collision and random fence points, model probability of collision site presence, and model expected collision counts observed during spring 2009 and 2010 in southern Idaho, USA.

Parameter	Justification for hypothesized parameter
LS[a,b,c]	Local abundance may influence collision
D2L[a,b,d]	Leks are focal points of spring space-use
Sum LC[a,b,e]	Local abundance may influence collision
Nleks[a,b,f]	Leks are focal points of spring space-use
Lateral shrub height[a,g]	Shrub height may influence concealment of fence
Longitudinal shrub height[a,h]	Shrub height may influence concealment of fence
Distance to closest shrub[a,i]	Shrubs may influence visual concealment of fence
SCC[a,j]	Shrub density may influence concealment of fence
HDFCS[a,k]	Influence of shrub height may depend of fence height
HDFCSA[a,l]	Influence of shrub height may depend of fence height
FT[a,m]	Infrastructure design may influence collision risk
FH[a,n]	Infrastructure height may influence collision risk
DBP[a,o]	Fence posts are more conspicuous than fence itself
WP[a,p]	Wooden posts are more conspicuous than steel t-posts
Slope[a,q]	Topography may influence collision risk
Aspect[a]	Topography may influence collision risk
Region[b,r]	Regional scale attributes may influence collision
Year[b]	Collision risk may vary between 2009 and 2010
FL[b,s]	Fence density may influence collision risk
VH<0.5[b,t]	Visual concealment of fence and sage-grouse space-use
0.5<VH<1.0[b,u]	Visual concealment of fence and sage-grouse space-use
VH>1.0[b,v]	Visual concealment of fence and sage-grouse space-use
SCC1020[b,w]	Shrub cover can influence sage-grouse space-use
SCC2030[b,x]	Shrub cover can influence sage-grouse space-use
SCC3040[b,y]	Shrub cover can influence sage-grouse space-use
SCC4050[b,z]	Shrub cover can influence sage-grouse space-use

VTBS[b,aa]	Sage-grouse habitats
VTLS[b,ab]	Sage-grouse habitats
TRI[b,ac]	Topography may influence collision risk

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[a] Variable used in site-scale modeling to discriminate collision and random fence points.

[b] Variable used in broad scale logistic regression and zero-inflated Poisson regression to model probability of collision site presence, and expected collision counts for 1x1 km cluster sampling units.

[c] Observed sage-grouse count at lek nearest to point (site-scale), and observed sage-grouse count at lek nearest to 1x1 km cluster centroids (broad-scale).

[d] Distance from point to nearest sage-grouse lek (site-scale), and distance to nearest sage-grouse lek from 1x1 km cluster centroids (broad-scale).

[e] Sum of observed lek counts from all sage-grouse leks within 2 km of point (site-scale), and sum of observed lek counts from all sage-grouse leks within 2 km of 1x1 km cluster centroids (broad-scale).

[f] Number of sage-grouse leks within 2 km of point (site-scale), and number of sage-grouse leks within 2 km of 1x1 km cluster centroids (broad-scale).

[g] Mean height of closest shrub lateral to both sides of fence at each point.

[h] Mean height of closest shrub in both directions of point longitudinally along fence.

[i] Mean distance to closest shrub lateral to both sides of fence at each point.

[j] Percent shrub canopy cover in cardinal directions at each point.

[k] Mean height difference between fence point and closest shrub lateral to both sides of fence.

[l] Mean height difference between fence point and closest shrub longitudinal to point.

[m] Type of fence (i.e., 3-strand wire, 4-strand wire, etc.).

[n] Height of fence at each point.

[o] Distance between fence posts bounding segment of each point.

[p] Indicator variable for presence of wooden fence posts bounding segment of each point.

[q] Mean slope measured across and along fence at each point.

[r] Region = Magic Valley (MV), Big Desert (BD), and Upper Snake (US).

[s] Fence density or length of fence per 1x1 km cell.

[t] Arcsine-square-root transformed proportion of 1x1 km cell covered by vegetation < 0.5 m tall.

[u] Arcsine-square-root transformed proportion of 1x1 km cell covered by vegetation > 0.5 but < 1.0 m tall.

[v] Arcsine-square-root transformed proportion of 1x1 km cell covered by vegetation > 1.0 m tall.

[w] Arcsine-square-root transformed proportion of 1x1 km cell covered by 10-20% sagebrush cover.

[x] Arcsine-square-root transformed proportion of 1x1 km cell covered by 20-30% sagebrush cover.

[y] Arcsine-square-root transformed proportion of 1x1 km cell covered by 30-40% sagebrush cover.

[z] Arcsine-square-root transformed proportion of 1x1 km cell covered by 40-50% sagebrush cover.

[aa] Arcsine-square-root transformed proportion of 1x1 km cell covered by big sagebrush.

[ab] Arcsine-square-root transformed proportion of 1x1 km cell covered by low sagebrush..

[ac] Mean terrain ruggedness index calculated for each 1x1 km cell.

Appendix E. Correlation of broad-scale predictor variables used in logistic and zero-inflated Poisson regression modeling. Correlated variables are significantly correlated with the Variable column.

Variable	Correlated variable
FL	none
D2L	Nleks, Sum LC
Nleks	D2L, Sum LC, VH<0.05, 0.5<VH<1.0, VH>1.0, SCC4050, VTBS, VTLS
Sum LC	D2L, Nleks, LC, SCC1020, VTLS
LC	Sum LC, VH<0.5, 0.5<VH<1.0, SCC2030, TRI
VH<0.5	Nleks, LC, 0.5<VH<1.0, VH>1.0, SCC2030, SCC3040, SCC4050, VTBS, VTLS, TRI
0.5<VH<1	Nleks, LC, VH<0.5, SCC2030, SCC3040, SCC4050, VTBS, VTLS, TRI
VH>1	Nleks, VH<0.5, VTBS, VTLS
SCC1020	Sum LC, SCC2030, SCC3040
SCC2030	VH<0.5, 0.5<VH<1.0, SCC1020, SCC3040, SCC4050, VTBS, TRI
SCC3040	VH<0.5, 0.5<VH<1.0, SCC1020, SCC2030, SCC4050, VTBS, VTLS, TRI
SCC4050	VTBS, VTLS, TRI, VH<0.5, 0.5<VH<1.0, SCC2030, SCC3040
VTBS	Nleks, VTLS, TRI, VH<0.5, 0.5<VH<1.0, VH>1.0, SCC2030, SCC3040, SCC4050
VTLS	Nleks, Sum LC, VH<0.5, 0.5<VH<1.0, VH>1.0, SCC3040, SCC4050, VTBS
TRI	LC, VH<0.5, 0.5<VH<1.0, SCC2030, SCC3040, SCC4050, VTBS

Appendix F. Logistic regression models of collision probability during collision surveys of fence segments on 16 sage-grouse breeding areas during spring 2009 and 2010 in southern Idaho, USA.

Model	Model description
Initial models	
$P_{(\mathrm{D2L})}$	P influenced by distance to lek
$P_{(\mathrm{Nleks})}$	P influenced by number of leks within 2 km
$P_{(\mathrm{LC})}$	P influenced by lek count at nearest lek
$P_{(\mathrm{Sum\ LC})}$	P influenced by sum of lek counts within 2 km
$P_{(\mathrm{D2L\ +\ LC})}$	P influenced by distance to lek and lek count at nearest lek
$P_{(\mathrm{Nleks\ +\ LC})}$	P influenced by number of leks within 2 km and lek count
$P_{(\mathrm{FL})}$	P a function of fence length per 1x1 km cell
$P_{(\mathrm{TRI})}$	P a function of terrain ruggedness
$P_{(\mathrm{FL\ +\ TRI})}$	P a function of fence length and terrain ruggedness
$P_{(\mathrm{VH<0.5})}$	P a function of cover of vegetation < 0.5 m tall
$P_{(\mathrm{VH>0.5<1.0})}$	P a function of cover of vegetation > 0.5 but < 1.0 m tall
$P_{(\mathrm{VH>1.0})}$	P a function of cover of vegetation > 1.0 m tall
$P_{(\mathrm{SCC1020})}$	P a function of cover of 10-20% sagebrush canopy cover
$P_{(\mathrm{SCC2030})}$	P a function of cover of 20-30% sagebrush canopy cover
$P_{(\mathrm{SCC3040})}$	P a function of cover of 30-40% sagebrush canopy cover
$P_{(\mathrm{SCC4050})}$	P a function of cover of 40-50% sagebrush canopy cover
$P_{(\mathrm{VTBS})}$	P a function of cover of vegetation type big sagebrush
$P_{(\mathrm{VTLS})}$	P a function of cover of vegetation type low sagebrush
$P_{(\mathrm{Year})}$	P varies by year
$P_{(\mathrm{Region})}$	P varies by region
$P_{(\mathrm{Region\ +\ Year})}$	P a function of region and year
$P_{(\mathrm{Region\ +\ TRI})}$	P a function of region and terrain ruggedness
$P_{(\mathrm{Region\ +\ TRI\ +\ FL})}$	P a function of region, terrain ruggedness, and fence length
Final model suite	
$P_{(\mathrm{TRI})}$	P a function of terrain ruggedness
$P_{(\mathrm{Region})}$	P varies by region
$P_{(\mathrm{Region\ +\ TRI})}$	P a function of region and terrain ruggedness

$P_{(\text{FL + TRI})}$	P a function of fence length and terrain ruggedness
$P_{(\text{Region + FL})}$	P a function of region and fence length
$P_{(\text{TRI + D2L})}$	P a function of terrain ruggedness and distance to lek
$P_{(\text{Region + LC})}$	P a function of region and lek count
$P_{(\text{Region + D2L})}$	P a function of region and distance to lek
$P_{(\text{Region + VH>1.0})}$	P a function of region and cover of vegetation > 1.0 m tall
$P_{(\text{TRI + VH>1.0})}$	P a function of terrain ruggedness and vegetation cover > 1.0 m
$P_{(\text{Region + SCC2030})}$	P a function region and cover of 20-30% sagebrush canopy cover
$P_{(\text{Region + SCC4050})}$	P a function region and cover of 40-50% sagebrush canopy cover
$P_{(\text{Region + TRI + FL})}$	P a function of region, terrain ruggedness, and fence length
$P_{(\text{Region + TRI + VH>1.0})}$	P a function of region, terrain ruggedness, and vegetation > 1.0 m
$P_{(\text{Region + TRI + SCC4050})}$	P a function region, terrain ruggedness, and 40-50% canopy cover
$P_{(\text{Region + TRI + SCC2030})}$	P a function region, terrain ruggedness, and 20-30% canopy cover
$P_{(\text{Region + TRI + D2L})}$	P a function of region, terrain ruggedness, and lek distance
$P_{(\text{TRI + FL + VH>1.0})}$	P a function of ruggedness, fence length, and vegetation > 1.0 m
$P_{(\text{TRI + FL + D2L})}$	P a function of ruggedness, fence length, and lek distance
$P_{(\text{Region + Nleks + LC})}$	P influenced by region, number of leks within 2 km, and lek count
$P_{(\text{Region + D2L + LC})}$	P influenced by region, lek distance, and lek count
$P_{(\text{Region + TRI + FL + VH>1.0})}$	P a function of region, ruggedness, fence length, and vegetation cover > 1.0 m
$P_{(\text{Region + TRI + FL + D2L})}$	P a function of region, ruggedness, fence length, and lek distance

Appendix G. Zero-inflated Poisson regression models of expected collision counts during collision surveys of fence on 16 sage-grouse breeding areas during spring 2009 and 2010 in southern Idaho, USA

Model	Model description
Initial models	
D2L	Collision influenced by distance to lek
Nleks	Collision influenced by number of leks within 2 km
LC	Collision influenced by lek count at nearest lek
Sum LC	Collision influenced by sum of lek counts within 2 km
D2L + LC	Collision influenced by distance to lek and lek count
Nleks + LC	Collision influenced by number of leks within 2 km and lek count
FL	Collision a function of fence length per 1x1 km cell
TRI	Collision a function of terrain ruggedness
FL + TRI	Collision a function of fence length and terrain ruggedness
VH<0.5	Collision a function of cover of vegetation < 0.5 m tall
VH>0.5<1.0	Collision a function of vegetation > 0.5 but < 1.0 m tall
VH>1.0	Collision a function of cover of vegetation > 1.0 m tall
SCC1020	Collision a function of 10-20% sagebrush canopy cover
SCC2030	Collision a function of 20-30% sagebrush canopy cover
SCC3040	Collision a function of 30-40% sagebrush canopy cover
SCC4050	Collision a function of 40-50% sagebrush canopy cover
VTBS	Collision a function of vegetation type big sagebrush
VTLS	Collision a function of vegetation type low sagebrush
Year	Collision varies by year
Region	Collision varies by region
Region + TRI	Collision a function of region and terrain ruggedness
Region + FL	Collision a function of region and fence length
Region + TRI + FL	Collision a function of region, terrain ruggedness, and fence length

Final model suite

D2L	Collision influenced by distance to lek
FL	Collision a function of fence length per 1x1 km cell
TRI	Collision a function of terrain ruggedness
D2L + TRI	Collision a function of lek distance and terrain ruggedness
D2L + FL	Collision a function of lek distance and fence length
D2L + SCC1020	Collision a function of lek distance and 10-20% sagebrush canopy cover
D2L + SCC2030	Collision a function of lek distance and 20-30% sagebrush canopy cover
D2L + SCC3040	Collision a function of lek distance and 30-40% sagebrush canopy cover
D2L + SCC4050	Collision a function of lek distance and 40-50% sagebrush canopy cover
D2L + VH>0.5<1.0	Collision a function of lek distance and vegetation > 0.5 but < 1.0 m tall
FL + SCC1020	Collision a function of fence length and 10-20% sagebrush canopy cover
FL + SCC2030	Collision a function of fence length and 20-30% sagebrush canopy cover
FL + SCC3040	Collision a function of fence length and 30-40% sagebrush canopy cover
FL + SCC4050	Collision a function of fence length and 40-50% sagebrush canopy cover
FL + VH>0.5<1.0	Collision a function of fence length and vegetation > 0.5 but < 1.0 m tall
TRI + SCC1020	Collision a function of ruggedness and 10-20% sagebrush canopy cover
TRI + SCC2030	Collision a function of ruggedness and 20-30% sagebrush canopy cover

TRI + SCC3040	Collision a function of ruggedness and 30-40% sagebrush canopy cover
TRI + SCC4050	Collision a function of ruggedness and 40-50% sagebrush canopy cover
TRI + VH>0.5<1.0	Collision a function of ruggedness and vegetation > 0.5 but < 1.0 m tall
D2L + TRI + FL	Collision a function of lek distance, ruggedness, and fence length

Appendix H. Models of collision probability (*P*) during fence collision surveys on experimentally marked and unmarked control fence segments on 8 sage-grouse breeding areas of southern Idaho, USA, during spring 2010.

Model	Model description
$P_{(.)}$	*P* constant
$P_{(\text{Treatment})}$	*P* differs between marked and unmarked fence segments
$P_{(\text{Distance})}$	*P* influenced by distance to nearest lek
$P_{(\text{Lek Size})}$	*P* influenced by size of nearest lek
$P_{(\text{Time})}$	*P* influenced by time of sampling during lekking season
$P_{(\text{Height})}$	*P* influenced by mean fence height
$P_{(\text{Time+Treatment})}$	*P* a function of both time of sampling and marking
$P_{(\text{Time+Lek Size})}$	*P* a function of both time of sampling and lek size
$P_{(\text{Time+Distance})}$	*P* a function of both time of sampling and lek distance
$P_{(\text{Lek Size+Distance})}$	*P* a function of both time of lek size and distance to lek
$P_{(\text{Lek Size+Height})}$	*P* a function of both time of lek size and fence height
$P_{(\text{Treatment+Height})}$	*P* a function of marking treatment and fence height
$P_{(\text{Treatment+Lek Size})}$	*P* a function of marking treatment and lek size
$P_{(\text{Time+Distance+Height})}$	*P* a function of time, distance to lek, and fence height
$P_{(\text{Time+Lek Size+Distance})}$	*P* a function of time, lek size, and distance to lek
$P_{(\text{Time+Treatment+Distance})}$	*P* a function of time, marking, and lek distance
$P_{(\text{Time+Treatment+Lek Size})}$	*P* a function of time, marking treatment, and lek size
$P_{(\text{Treatment+Distance+Height})}$	*P* a function of marking, lek distance, and fence height
$P_{(\text{Treatment+Lek Size +Distance})}$	*P* a function of marking, lek size, and distance to lek
$P_{(\text{Time+Lek Size+Distance+Height})}$	*P* influenced by time, lek size, lek distance, and height
$P_{(\text{Time+Treatment+Distance+Height})}$	*P* a function of time, marking, lek distance, and height
$P_{(\text{Time+Treatment+Lek Size+Height})}$	*P* a function of time, marking, lek size, and fence height
$P_{(\text{Treatment+Lek Size+Distance+Height})}$	*P* varies by marking, lek size, lek distance, and height
$P_{(\text{Time+Treatment+Lek Size+Distance})}$	*P* a function of time, marking, lek size, and lek distance
$P_{(\text{Time+Treatment+Lek Size+Distance+Height})}$	*P* varies by time, marking, lek size, lek distance, height

Appendix I. Zero-inflated Poisson and zero-inflated Negative Binomial (NB) regression models of expected collision count on experimentally marked and unmarked control fence segments on 8 sage-grouse breeding areas of southern Idaho, USA, during spring 2010. All models are zero-inflated Poisson unless otherwise indicated.

Model	Model description
(.)	Collision constant
(Trt)	Collision influenced by marking treatment
(Trt + Lsize)	Collision influenced by marking nd lek size
(Trt + Lsize + Dist--NB)	Collision influenced by marking, lek size, and distance to lek, NB model
(Trt + Lsize + Dist)	Collision influenced by marking, lek size, and lek distance
(Trt + Lsize + Dist + Trt × Lsize)	Collision influenced by marking, lek size, distance to lek, and marking by lek size interaction
(Trt + Lsize + Dist + Trt × Dist)	Collision influenced by marking, lek size, distance to lek, and marking by distance to lek interaction
(Trt + Lsize + Dist + Lsize × Dist)	Collision influenced by marking, lek size, distance to lek, and lek size by distance interaction
(Trt + Lsize + Dist + Trt × Dist--NB)	Collision influenced by marking, lek size, distance, and marking by distance interaction, NB model
(Trt + Lsize + Dist + Trt × Dist)	Collision influenced by marking, lek size, distance to lek, and marking by distance interaction
(Trt + Lsize + Dist + Trt × Lsize × Dist)	Collision influenced by marking, lek size, distance, and treatment by lek size by distance interaction

Appendix J. University of Idaho Animal Care and Use Committee Approval Verification.

University of Idaho
Animal Care and Use Committee

Date: Monday, December 22, 2008
To: Kerry Reese
From: University of Idaho
Re: Protocol 2009-21
Factors Influencing Detectability and Scavenging Associated with Fence Collisions in Sagebrush Steppe

Your animal care and use protocol for the project shown above was reviewed by the University of Idaho on Monday, December 22, 2008.
This protocol was originally submitted for review on: Wednesday, December 17, 2008
The original approval date for this protocol is: Monday, December 22, 2008
This approval will remain in affect until: Tuesday, December 22, 2009
The protocol may be continued by annual updates until: Thursday, December 22, 2011

Federal laws and guidelines require that institutional animal care and use committees review ongoing projects annually. For the first two years after initial approval of the protocol you will be asked to submit an annual update form describing any changes in procedures or personnel. The committee may, at its discretion, extend approval for the project in yearly increments until the third anniversary of the original approval of the project. At that time, the protocol must be replaced by an entirely new submission.

Brad Williams, DVM
Campus Veterinarian
University of Idaho
208-885-8958