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Original Article

Roads elicit negative movement and habitat-selection responses by wolverines (*Gulo gulo luscus*)

0.66 mi²
Robust

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The fine-scale behavior of wildlife when crossing roads and interacting with traffic is likely to mirror natural responses to predation risk including not responding, pausing, avoiding, or increasing speed during crossing. We generated coarse-scale behavioral predictions based on these expectations that could be assessed with GPS radiotelemetry. We evaluated our predictions using an integrated step-selection analysis of wolverine (*Gulo gulo luscus*) space use in relation to spatially and temporally dynamic vehicle traffic on industrial roads in northern Alberta. We compared support for alternative models of road avoidance, increased speed near roads, and road avoidance and increased speed near roads. We predicted that wolverines would avoid roads and increase their speed near roads and that these behaviors would increase with traffic volume. We found that vehicle traffic was relatively low (0–30 vehicles/12 h) but important for explaining wolverine space use. Top winter and summer models indicated that wolverines avoided and increased speed near roads. Wolverine movement, but not avoidance, increased with traffic volume. We suggest that movement is a fine-scaled response that is more responsive to vehicle traffic than habitat selection. We show that roads, regardless of traffic volume, reduce the quality of wolverine habitats and that higher-traffic roads might be most deleterious. We suggest that wildlife behavior near roads should be viewed as a continuum and that accurate modeling of behavior when near roads requires quantification of both movement and habitat selection. Mitigating the effects of roads on wolverines would require clustering roads, road closures, or access management.

Key words: **boreal forest, *Gulo gulo*, habitat selection, movement, oil extraction, roads, step selection, traffic volume, wolverine.**

INTRODUCTION

The growth of human populations and our demand for natural resources has caused the development of roads in wild areas throughout the world (Hansen et al. 2013). Road networks have expanded and densified for the extraction of oil, gas, mining, and timber resources resulting in forest fragmentation and reduction in core areas (Schnieder 2002; Pickell et al. 2014; Pickell et al. 2016). The change in the characteristics of forested landscapes has had cascading effects on the behavior and abundance of many wildlife species (Rytwinski and Fahrig 2012; Latham and Boutin 2015).

Roads are an interface between humans and wildlife. Roads can have positive effects on wildlife by providing protection from predators (Berger 2007), enhanced movement (Whittington et al. 2011), or foraging opportunities (Scrafford et al. 2017b). Roads also can have negative effects on wildlife (Forman and Alexander 1998; Rytwinski and Fahrig 2012) including vehicle-caused mortality (Da Rosa and Bager 2013; McClintock et al. 2015; Niemi et al. 2017) or displacement (Shannon et al. 2014; Abrahms et al. 2016). As per the risk-disturbance hypothesis, displacement of wildlife by roads is because wildlife perceives roads, and associated human activity, as a predation risk (Frid and Dill 2002). Road avoidance can result in a reduction of suitable habitats for wildlife (Beyer et al. 2016; D'Amico et al. 2016; Kite et al. 2016).

Road effects on wildlife are often discerned without data on traffic volume (e.g. Krebs et al. 2007; Roever et al. 2010; Beyer et al. 2016). However, this approach neglects the varied responses of wildlife to the magnitude of human use of roads. The risk-disturbance hypothesis predicts that wildlife perception of risk increases with human activity. A high frequency of vehicles should therefore be viewed by wildlife as a large-predator group that is a threat to their security (Frid and Dill 2002; Jacobson et al. 2016). For example, grizzly bears (*Ursus arctos*) avoided high-traffic roads by crossing roads at night when traffic volume was low (Northrup et al. 2012). Similarly, squirrels (*Tamiasciurus hudsonicus grahamensis* and *Sciurus aberti*) reduced their movement across roads as traffic volume

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NEST SITE CHARACTERISTICS AND BREEDING BIOLOGY OF FLAMMULATED OWLS (*Otus flammeolus*) IN MISSOULA VALLEY

By

MATHEW THOMAS SEIDENSTICKER

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Thesis

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INTRODUCTION

In the western United States the Flammulated Owl (*Otus flammeolus = Psiloscops flammeolus)* is considered a Neotropical migrant and locally common breeder (AOU 1998). In Montana, the Flammulated Owl occurs in the western third of the state (Lenard and others 2003). Throughout its range, the Flammulated Owl primarily breeds in montane coniferous forests at moderate to high elevations, and shows a preference for mature- or old growth yellow pine forests (*Pinus ponderosa, Pinus jeffreyi*) mixed with other conifers, such as Douglas-fir (*Pseudotsuga menziesii*) and true firs (*Abies* spp.) (McCallum 1994a).

Flammulated Owls also breed in lower elevation yellow pine forest mixed with pinyon pine (*Pinus edulis*), juniper (*Juniperus* spp.), quaking aspen (*Populus tremuloides*), and cottonwood *(Populus* spp.) (Bent 1938, Marshall 1939, Johnson and Russell 1962, Reynolds and Linkhart 1987b, McCallum and Gehlbach 1988, Bull and others 1990, McCallum 1994a, Groves and others 1997, Arsenault 2004) and habitat without yellow pine such as, Douglas-fir forests (Powers and others 1996), quaking aspen stands (Marti 1997), white fir (*Abies concolor*), subalpine fir (*A. lasiocarpa*), limber pine (*Pinus flexilis*) (Dunham and others 1996), and possibly black oak forests (*Quercus* spp.) (Winter 1974, Marcot and Hill 1980). Howie and Ritcey (1987) and Christie and van Woudenberg (1997) reported Flammulated Owls to avoid pure ponderosa pine stands, opting instead for dry Douglas-fir forests with open aspect, including selectively logged forests. Flammulated Owls may also use small forest patches (Dunham and others 1996).

In Montana, Flammulated Owls were reported to occur in mature and old-growth xeric ponderosa pine/Douglas-fir stands (Holt and Hillis 1987), and in landscapes with higher proportions of ponderosa pine/Douglas-fir forest with low to moderate canopy closure (Wright and others 1997). Generally, the most common features that describe Flammulated Owl breeding habitat throughout its range in the western United States are: a cold temperate and semiarid climate, a high abundance or diversity of nocturnal arthropod (mostly insect) prey, open physiognomy, dense foliage for roosting, and trees providing nest cavities (McCallum 1994b).

Mature- to old-growth ponderosa pine/Douglas-fir forests have been heavily affected by wildfire suppression and logging practices. For nearly 100 years, removal of over-story ponderosa pine and wildfire suppression has changed pre-settlement forest physiognomy throughout the Intermountain West. These practices have led to replacement of most old-growth ponderosa pine forests by younger forest with a greater proportion of Douglas-fir, fewer large trees, and reduced open perennial grass/shrub understory (Habeck 1990). Indeed, across western United States, finding old ponderosa pine forests that have not been influenced by fire suppression, grazing, timber harvest, or other anthropogenic influences is uncommon (Morgan 1994). Ultimately, these influences could have affected habitat suitability for breeding Flammulated Owl populations.

As a result of these anthropogenic changes, lack of long-term monitoring and breeding season studies, and detailed habitat needs, the Flammulated Owl was listed as a sensitive species by the United States Forest Service Region 1 (USFS 2004). This listing provides special emphasis in planning and management activities to ensure its conservation (USFS 2004). Additionally, the United States Bureau of Land Management considers the Flammulated Owl sensitive, and imperiled in at least part of its range, including Montana (BLM 2001). In Montana, Flammulated Owls are listed as Tier 1 Species of Concern, in greatest need of conservation, and requiring adequate, systematic monitoring (MFWP 2005). Despite these mandates, few efforts have been undertaken to understand the breeding biology of Flammulated Owls at the northeastern edge of their breeding range in western Montana.

BREEDING STATUS OF FLAMMULATED OWLS IN MONTANA

To ascertain breeding status of Flammulated Owls in Montana, I examined and evaluated historical and current records. My intent was to: (1) provide a general overview of Flammulated Owl distribution in Montana; (2) review the current accepted breeding records for Montana and clarify criteria for what constitutes proof of breeding; and (3) underscore the need for long-term breeding research in Montana.

To evaluate breeding records and distribution, I compiled historical and current records of Flammulated Owls in Montana by reviewing scientific literature, Montana Natural Heritage Program (MTNHP) records, University of Montana Avian Science Center (UMASC) data, and Owl Research Institute (ORI) records. I based my decisions on my more than 10 years experience researching owls in Montana and current Montana breeding bird criteria (Lenard and others 2003).

Lenard and others (2003) listed the following as direct evidence of eggs or young: (1) an occupied nest (adults entering or leaving a nest site in circumstances indicating an occupied nest, includes high nests or nest holes, the contents of which cannot be seen) or adults incubating or brooding; (2) recently fledged young (of altricial species) incapable of sustained flight, or downy young (of precocial species) restricted to the area by dependence on adults or limited mobility; (3) adults attending young – adult carrying food or fecal sac for young, or feeding recently fledged young; (4) a used nest or eggshell found (identification must be convincing for such records to be accepted); and (5) a nest with egg(s) that can be clearly identified.

Lenard and others (2003) explicitly stated that breeding is not assumed by the presence of adults or from behavior. Although the five criteria – particularly number 1 – give some credence pine/Douglas-fir forests. However, longitudinal surveys in other forest types have not been conducted. Although surveys have been conducted in the dry forest mountain ranges in eastern Montana, no Flammulated Owls have been detected (Cilimburg 2006).

Need for Long Term Breeding Research

The surveys conducted from 1994-1996 (Wright and others 1997) and in 2005 and 2008 (Cilimburg 2006, Smucker and others 2008) have done much to increase our understanding of distribution and landscape-level habitat associations of Flammulated Owls in Montana. In addition, a thorough and effective survey protocol has been developed (Cilimburg 2006, Smucker and others 2008). However, observations of vocalizing males may indicate breeding, but may also simply indicate transitory presence. It is well known that many species of migratory birds, including owls, will vocalize during migration, and may be there one day and not the next. Indeed, Flammulated Owls vocalize and defend territories even when apparently not breeding (Reynolds and Linkhart 1987b) – a suggestion that singing does not always indicate nesting. Thus it is important to extend efforts throughout the nesting period (May-Sept) to include; detection surveys, nest searches and nest monitoring.

The limited direct evidence of breeding to date in Montana underscores the need for rigorous long-term breeding season research. This research is urgently needed in order to develop complete, scientifically robust habitat and population models for Flammulated Owls. Federal and state agencies need to provide such models for management, and consequently conservation of listed species. To address the need for information about nest site characteristics and breeding habitat I attempted to locate Flammulated Owl nests in the Missoula Valley from 2008-2010. My project goals were to (1) locate FLOW nests; (2) quantify nest site characteristics, and (3) describe breeding habitat around nest trees.

METHODS

STUDY AREAS

My study was located in the Lolo National Forest, within a 30-mile radius of Missoula, Montana (Figure 2). Survey areas are characterized by steep, mountainous ridges containing habitat ideal for Flammulated Owls (hereafter FLOWs): open stands of ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*), and mixed shrub understory. I did not randomly select survey areas. Instead, I focused on areas of previous FLOW detection, including 4 survey routes established by University of Montana Avian Science Center in 2005 and areas where FLOWs were detected by the Owl Research Institute in the 1980s, particularly in the Marshall Canyon watershed. These areas included Crazy Canyon in Pattee Canyon Recreation Area, parts of Rattlesnake National Recreation Area, Blue Mountain, Woods Gulch, and Marshall Canyon. Forest type in all these areas is dominated by xeric ponderosa pine/Douglas-fir forests. I did not survey in other forest types.

FIELD SEASON 2009

May to mid-June 2009. The first 6 weeks of my field season entailed surveys to detect FLOWs. Surveys were conducted using methods described by Takats and others (2001) for nocturnal owls with the following modifications. Survey routes usually consisted of 10 survey points spaced 400 m apart. Starting points for routes already established were predetermined. I determined starting points for new survey routes arbitarily. Usually, I began surveys no less than 400 m but sometimes up to 1600m from trailheads depending on habitat. For example, if 400 m from a trailhead was located in a meadow I opted to start the survey at the interface of the meadow and trees. I alternated the direction that I conducted surveys along routes. If I conducted surveys at points 1-10 one night, I then surveyed points 10-1 the next time I surveyed the route. Surveys began at the end of civil twilight and typically continued to 1 a.m. Surveys points consisted of

The distribution of DBH of cavity bearing trees appears rather symmetric (Figure 10). The modal interval was the 36-45 cm DBH class. Twenty five (45%) of the cavity bearing trees we located were in the 36-55 cm DBH classes. Sample median DBH was 54.6 cm and range 92.7 cm (116.8 – 24.1 cm). Grouped median DBH was 53.0 cm.

CAVITY CHARACTERISTICS IN TERRITORIES

A narrow majority (52%) of the 131 cavities that I located were usable for FLOWs. Forty percent (61 of 115) of the cavities that I located were bearing S-E on trees. The distribution of height of cavities appears skewed right with an outlier in the 26-30 m height class (Figure 11). The modal interval was the 6-10 m height class, comprising over half (55%) of the cavities we located. Sample median was 8.45 m and range 22 m (26.3 – 4.3 m). Grouped median was 9.09 m. Sample median of height of nest cavities was 8.2 m and range 7.7 m (n = 3).

DISCUSSION

OVERVIEW OF FLAMMULATED OWL BREEDING RESEARCH

The breeding biology of Flammulated Owls has been rather well studied at the core of their range in New Mexico, Colorado, Utah, Oregon, and Idaho. Most of the studies have demonstrated an affinity to xeric ponderosa pine/Douglas-fir forests containing large diameter snags for breeding, although several studies have indicated that Flammulated Owls breed in other forest types such as Douglas-fir forests and aspen stands (Reynolds and Linkhart 1987b, Powers and others 1996, Marti 1997).

Several studies have investigated nest site preferences of Flammulated Owls and collected data comparable to our preliminary data. In New Mexico, McCallum and Gehlbach (1988) reported mean and standard deviation of nest tree DBH was 46.2 ± 10.7 cm (n = 17). Mean and standard deviation of cavity height, entrance diameter, depth, and internal diameter at nest sites was 4.9 ± 1.6 m, 5.9 ± 0.9 cm, 21.2 ± 5.2 cm, and 13.5 ± 2.8 cm, respectively. Arsenault (2004) reported mean and standard deviation of entrance length and width, cavity depth, and internal diameter 5.64 ± 0.9 and 5.68 ± 1.1 cm, 25.21 ± 9.2 cm, and 14.6 ± 4.1 cm, respectively in New Mexico (n = 34). Both of these studies reported cavity dimensions smaller than cavity dimensions in our study. Arsenault (2004) did not report height of nest cavities but did report a majority (18) of nests in live trees, mainly Gambel oak. Also, Arsenault (2004) concluded that Flammulated Owls nested in more Northern Flicker cavities than expected based on availability. However, Arsenault (2004) reported that most Acorn Woodpecker (*Melanerpes formicivorus*) nests were in live trees and Northern Flicker nests most often in dead trees. Given the fact that Arsenault (2004) found 18 Flammulated Owl nests in live trees may indicate that these owls largely depend on cavities excavated by Acorn Woodpeckers for nesting in New Mexico.

The assemblage of primary excavator species may differ throughout the range of Flammulated Owls. Depending on the assemblage this may result in a large variation in the dimensions of available cavities used for nesting by these owls. For example, Bull and others (1990) reported mean and standard deviation of nest tree DBH 72 ± 14.4 cm and nest cavity ~~height 12 ± 4.7 m for Flammulated Owl nests in Oregon (n = 33), seventy percent of which were~~ in ponderosa pine trees. No information about nest cavity dimensions was reported by Bull and others (1990). However, they did report most Flammulated Owl nests in cavities excavated by Pileated Woodpeckers in dead trees. Nest sites were located at sites dominated by ponderosa pine as the overstory tree species, similar to breeding habitat associations reported for Flammulated Owls in Colorado (Reynolds and Linkhart 1992). Bull and others (1990) concluded that relative to availability, Flammulated Owls used a higher percentage of Pileated Woodpecker cavities than expected.

In Idaho, Powers and others (1996) studied 24 Flammulated Owl nests in habitat without ponderosa pine but largely dominated by Douglas-fir with isolated pockets of trembling aspen. They reported mean and standard deviation for nest tree DBH, nest cavity height, and entrance diameter 49.9 ± 18.9 cm, 5.1 ±0.6 m, and 6.8 ± 1.3 cm, respectively ($n = 13$). They also reported most (20) of the nests were in dead trees of which 13 (54%) were broken-top Douglas-fir snags. In my study I found one nest in a 28 cm DBH dead, broken-top Douglas-fir snag. Eleven of the nests Powers and others (1996) found were in trembling aspen, seven in dead trees and four in live trees. Flammulated Owls are also known to nest in aspen in Colorado (Reynolds and Linkhart 1987b). Although our sample size ($n = 3$) is very limited our median nest tree DBH, nest cavity height and internal dimensions were within the range reported by other Flammulated Owl investigators. Bull et al. (1990) reported similar although slightly larger dimensions in ponderosa pine tree nests, while other studies (McCallum and Gehlbach 1988, Powers 1996, Arsenault 2004) have reported smaller values for these dimensions in Gambel oak and Douglas-fir tree nests.

~~The New Mexico and Idaho studies were in areas where primary cavity excavators were~~ Northern Flickers. In Oregon, Pileated Woodpeckers were the primary excavators, similar to our study area. Both species are present in our study area and differentiating Pileated Woodpecker and Northern Flicker nest cavities can be difficult (Arsenault 1999). The mean entrance diameter of the nest cavities reported from Powers and others (1996) study (6.8 cm, SD = 1.3 cm) is similar to the mean length and width entrance dimensions for Flammulated Owl nests in Northern Flicker cavities reprorted by McCallum and Gehlbach (1988) and Arsenault (2004) from New Mexico. This may indicate that the dimensions of Northern Flicker nest cavities used by Flammulated Owls are similar throughout the owls' range in the intermountain western United States. However, it should be noted that Powers and others (1996) did not identify primary excavator species but only stated which primary excavator species were present in their study area.

Although my study does not provide a quantitative method for differentiating the nest sites of primary excavator species we observed few Northern Flickers and found several Pileated Woodpecker nests in our study area. Intuitively this leads to thinking that Pileated Woodpeckers were the most likely primary excavators of Flammulated Owl nest cavities that I found. The dimensions of the nest cavities in my study are much larger than the dimensions of Northern Flicker nest cavities used by Flammulated Owls reported by McCallum and Gehlbach (1988), Powers and other (1996), and Arsenault (2004), but similar to Bull and others (1990) in Pileated Woodpecker excavated cavities. However, a much larger sample size is needed before inferences can be made about Flammulated Owl nest characteristics in Montana. Further, the interaction between excavator species and nest tree species complicates the identification of common Flammulated Owl nest cavity physical characteristics throughout its entire range. Median clutch (2 eggs) and brood sizes (2 young) in my study are similar to sizes reported by other investigators (Reynolds and Linkhart 1987b, Powers and others 1996, Linkhart and Reynolds 2006).

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Interestingly, Nest #4 that I located may be the latest occurrence of FLOW egg laying ever reported. Egg laying at this nest occurred between 02 – 05 July. To our knowledge, the closest report is that of a 29 June egg laying at a nest in Idaho (Barnes and Belthoff 2008), some 200 miles and nearly 2 degrees latidute farther south than my study area. A review of the literature indicates that FLOWs initiate breeding earlier in the southwestern U.S. (New Mexico), later in Colorado, and still later in Idaho. This variance in phenology makes intuitive sense, with colder, wetter conditions persisting longer at more northern latitudes of the FLOW's breeding range. Further seasons of research will clarify how common a July egg-laying date is, and whether mean egg-laying dates in Montana follow this phenological gradient.

In a review of Flammulated Owl biology, management, and conservation McCallum (1994b) provided good discussions of habitat preference, fitness functions, and the concept of source and sink populations. However, Holt (1995) concluded that given the scarcity of information identified in McCallum (1994b) not enough demographic data were available to build source and sink models for Flammulated Owls. Since that time, only the long-term study (23 years) by Linkhart and Reynolds (2006, 2007) provide enough demographic data to model Flammulated Owls in Colorado.

Occupancy studies have increased our knowledge of Flammulated Owl distribution and landscape level habitat associations in Montana. However, if occupancy does not imply breeding, then inferring breeding habitat associations with occupancy surveys may be limited because actual breeding is not taking place in the habitat occupied. Further, survey protocols using playback methods may draw owls away from core territories and habitat recorded around the point of detection may be important to predict occupancy but not necessarily breeding. For example, we captured and attached a radio-transmitter to a breeding female over 300 m from her nest site. Thus, it is important to continue efforts throughout the nesting period (May to Sept) to include detection surveys, nest searches and nest monitoring.

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Montana remains one of 4 states (Montana, Wyoming, Washington, and California) within the 11 western states with no detailed breeding study of Flammulated Owls. In order to

Using a peeper camera is very effective for determining egg laying/clutch size and monitoring development of FLOW young. We recommend the use of such an instrument for any project conducting breeding research on FLOWs. It is also very useful for determining whether cavities are useable or not.

Using mist nest and audio lures to capture FLOWs seems to be effective and has been the most commonly used technique reported in the literature. However, most authors do not report the height at which nets should be placed. In my trapping attempts I observed that most often FLOWs tend to perch and fly 15 feet or more above the ground. As a result, I modified my trapping set to allow nets be raised so that the top of nets were at 15 feet. I suggest that researchers attempting to capture FLOWs place nets as high as possible to increase chances of success.

Our research attempted to address the need for breeding information about Flammulated Owls in Montana but much more needs to be done. Below we offer suggestions for future research and management.

SUGGESTIONS FOR FUTURE RESEARCH AND MANAGEMENT

Research

We strongly recommend that Flammulated Owl research extend beyond detection and occurrence to continue from detection through nesting and fledging (May-Sept.). We realize that replication of routes and repeated observations can lend strong inference towards which habitats singing males are using. However, as mentioned previously – singing does not always indicate nesting. Indeed, in three years of research in our study area we did not find nests one year. Whether this resulted from our failure to find them or the owls we monitored did not breed is not known, but may indicate that breeding of Flammulated Owls in local areas in Montana is not consistent from year to year. Essentially, we have no idea if Flammulated Owls are breeding in a way to sustain viable populations in Montana, important information that may affect management strategies if a sink population exists. Further, the outcomes of demographic modeling may be misleading if no annual variation in productivity is assumed. No doubt substantial commitment is needed to comprehensively assess breeding status, nest site selection, habitat requirements, and general breeding biology in Montana. This will require much in terms of money, resources, and personnel. However, as much as possible within these constraints managers should emphasize demographic variables if concerned with habitat quality (Johnson 2007).

Presence-absence data are useful for managing wildlife in many contexts, including identifying habitats that are of high value to specific species of conservation concern (MacKenzie 2005). For example, occupancy studies conducted by researchers in the mid-1990s and Avian Science Center at the University of Montana have provided a solid understanding of distribution and landscape level habitat associations in Montana. These studies have also demonstrated that singing Flammulated Owls can be quite numerous in local areas during the breeding season and occupying specific habitat (i.e. dry montane forests, open canopy and understory). In a review of density as an indicator of habitat quality and breeding Bock and Jones ~~(2004) concluded that of 109 published cases on 67 species across North America and Europe~~ that, in most cases, density will be a reliable indicator of breeding habitat quality. Intuitively, this implies that areas where there is lots of singing there is lots of breeding. However, it is known that Flammulated Owls will sing and defend territories if not breeding. Interestingly, Bock and Jones (2004) also reported a negative association between density and per capita reproduction in highly territorial species. However, it should be noted that most of the studies reviewed by Bock and Jones (2004) were conducted on non-strigiform species, mainly passerine birds. Nonetheless, in the end habitat characteristics associated with breeding must form the basis of modeling habitat for Flammulated Owls (van Woudenberg and Christie 2000). If it cannot we offer the following suggestions.

Researchers may be able to obtain data about breeding habitat by conducting properly designed presence-absence surveys, collecting habitat information, and then inferring that density of singing males indicates successful breeding if they had the evidence. A well designed long term presence-absence study coupled with efforts to locate nesting individuals at points of repeated observations may provide such evidence.

Future research should also examine Flammulated Owl habitat use in other forest types. Flammulated Owls are known to nest in forest types other than ponderosa pine (Christie and van Woudenberg 1997, Dunham and others 1996, Howie and Ritcey 1987, Marti 1997, Powers and others 1996) and this could have important conservation implications if determined to be the case in Montana. It also presents an opportunity to compare productivity between different habitat types.

Management

If models with little supporting demographic data are used to manage habitat for Flammulated Owls, a significant commitment to monitoring is required to determine the viability of this approach. Models based on short-term studies may capture relationships between animals and habitat during the course of investigation but may be ineffective in accurately predicting these relationships over time (see Rotenberry and Wiens 2009). Researchers and wildlife managers must have longitudinal data that includes breeding data to make strong inferences that decipher these changes. Further, models without these data could be honestly challenged in court if agency actions concerning forest management and Flammulated Owls in Montana are litigated. For example, will ground-truthing and monitoring be conducted to determine if habitat models are biologically relevant? Federal and state agencies should provide contingency plans to assess models. Models should also allow for catastrophic events. Indeed, wildfire is an annual occurrence in forests of the western United States. Fire should be a model parameter as should potential effects of global warming on Flammulated Owl breeding ecology.

Snag Management—Flammulated Owls a e obligate secondary cavity nesters and the presence of suitable cavities is a prerequisite for successful nesting. A large majority of Flammulated Owl nests have been reported in large, dead trees (dbh > 40 cm), especially cavities excavated by Northern Flickers (*Colaptes auratus*), and Pileated Woodpeckers (*Dryocopus pileatus*) (McCallum 1994a). We have yet to adequately describe characteristics of nest trees (snags) used by Flammulated Owls in Montana. Not all snags are created equally. They vary in height, dbh, age, and many other variables. For example, studies by D. Holt (unpubl.data) have shown Northern Saw-whet and Northern Pygmy-owls have different requirements for nest trees, and that leaving the "classic" old snag with woodpecker holes may not provide nests for both species. Additionally, the interior of all cavities are also not created equally. Evaluation of the interior of snags revealed that although the entrance hole looked appealing, the interior can vary from structurally sound to unusable (D. Holt, unpubl. data). Thus, snag retention policies for secondary obligate cavity nesters may need revaluation. 16

Long-term, rigorous study of breeding biology and habitat is urgently needed in order to develop scientifically robust and reliable population models for Flammulated Owls. This will require surveying to locate owls and intensive nest searching and monitoring. A well thought out presence-absence survey coupled with nest searches at points of repeated observations may lend

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Wolverine behavior varies spatially with anthropogenic footprint: implications for conservation and inferences about declines

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Abstract

Understanding a species' behavioral response to rapid environmental change is an ongoing challenge in modern conservation. Anthropogenic landscape modification, or "human footprint," is well documented as a central cause of large mammal decline and range contractions where the proximal mechanisms of decline are often contentious. Direct mortality is an obvious cause; alternatively, human-modified landscapes perceived as unsuitable by some species may contribute to shifts in space use through preferential habitat selection. A useful approach to tease these effects apart is to determine whether behaviors potentially associated with risk vary with human footprint. We hypothesized wolverine (*Gulo gulo*) behaviors vary with different degrees of human footprint. We quantified metrics of behavior, which we assumed to indicate risk perception, from photographic images from a large existing camera-trapping dataset collected to understand wolverine distribution in the Rocky Mountains of Alberta, Canada. We systematically deployed 164 camera sites across three study areas covering approximately 24,000 km^2, sampled monthly between December and April (2007–2013). Wolverine behavior varied markedly across the study areas. Variation in behavior decreased with increasing human footprint. Increasing human footprint may constrain potential variation in behavior, through either restricting behavioral plasticity or individual variation in areas of high human impact. We hypothesize that behavioral constraints may indicate an increase in perceived risk in human-modified landscapes. Although survival is obviously a key contributor to species population decline and range loss, behavior may also make a significant contribution.

WINTER ECOLOGY OF MOOSE ON THE

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by

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ABSTRACT

Shiras moose (*Alces alces shirasi*) invaded the Northern Yellowstone Winter Range (NYWR) in the1800's. Management of this species has been handicapped by limited population data and reliance on habitat use paradigms from other areas in North America. For example, although moose in most regions forage in recently disturbed forests with abundant deciduous shrubs, I did not find this to be true on the NYWR. I used multiple methods to determine population status, including aerial surveys, horseback surveys, road surveys, and ground counts in willow habitats. Moose were most easily seen when they foraged in willow (*Salix* spp.) stands in November–December and May–June. Attempts to count moose at other times resulted in marked under-sampling. Fires in 1988 removed mature forests at a landscape level resulting in a loss of preferred winter habitat and a sustained population reduction. I also investigated moose winter foraging activities among burned (1988), logged, forested, and forest edge environments. Logged areas and recent burns were rarely used, but forest interiors were important habitat. The edge effect did not benefit moose in winter. NYWR moose coped with winter by seeking concentrations of food to maximize feeding and minimize movement. They optimized foraging by using 2 taxa that grow in patches, willow and subalpine fir (*Abies lasiocarpa*). Moose fed in willow until snow forced them to nearby forested hillsides where they browsed subalpine fir <5 m in height. Subalpine fir, a shade tolerant tree, grows more rapidly under a forest canopy. It can reproduce through layering, creating an expanding patch of small trees around the parent. Patches are largest and most numerous in the oldest lodgepole pine (*Pinus contorta*) forests. Moose used other characteristics of lodgepole forests to reduce energy output by finding routes with less than average snow depth to reach subalpine fir patches. Moose also browsed gooseberry (*Ribes* spp.) and buffaloberry (*Shepherdia canadensis*), nutrient rich shrubs that are scattered and energetically expensive to obtain. Exclosure studies showed willow was suppressed before 1988, and fires and drought increased negative effects. Higher willow utilization rates in 1989 probably reflected a response by moose to a loss of winter habitat due to the 1988 fires, a foraging strategy that may have contributed to observed willow mortality. From 1989 to 1997, some willow showed signs of recovery. Replacement of older forests where moose foraged before 1988 could take several hundred years. Maintaining remaining populations should concentrate on minimizing loss of mature forests and adjacent willow and careful hunting management.

CHAPTER 4

MOOSE COVER TYPE USE ON THE NORTHERN YELLOWSTONE WINTER RANGE

Introduction

Moose are considered "selective generalists" because they can occupy a wide variety of habitats. They are also capable of using habitat components in higher proportion than they occur and selecting seasonally advantageous areas within those environments (Neu et al. 1974, Peek 1997). The selection of cover types by moose reflects the location of resources needed to meet habitat requirements (Neu et al. 1974, Van Horne 1983, Hobbs and Hanley 1990, Peek 1997).

Cover type use patterns have implications beyond forage availability for moose. The quality of thermal and hiding cover and differences in snow conditions also influence moose habitat use (Kelsall and Telfer 1974, Coady 1982, Telfer 1984, Renecker and Schwartz 1997, Schwartz and Renecker 1997, Peek 1997). Requirements and availability of these resources may vary with environmental changes (Geist 1971, 1974, Neu et al. 1974, Kelsall et al. 1977, Oldemeyer et al. 1977). In managed forests, cover type is routinely manipulated through a variety of silvicultural practices that affect moose winter survival strategies. In addition, moose habitat in boreal forests is often substantially altered by fire requiring moose to adapt to a radically different environment (Spencer and Chatelain 1953, Spencer and Hakala 1964, Peek et al. 1976, Coady 1982, Risenhoover and Maass 1987, Thompson and Stewart 1997). Peek (1997) concluded that general descriptions are available for moose habitat use patterns in most regions but also indicated the need to determine habitat components preferred by moose in local situations, especially where significant habitat alterations have occurred.

Recent landscape level habitat alterations, primarily the 1988 Yellowstone fires, have changed about 30% of the NYWR from mature conifer forests (143,856 ha pre-fire) to early seral stages, dramatically affecting proportions of successional communities available to moose. Romme and Despain (1989) postulated that the GYA experienced a similar major fire event or series of events about 1700 followed by a period that lacked comparable disturbances. In 1988 when the fire cycle apparently repeated, more >300-year-old lodgepole pine forests existed than at any time in the past 3 centuries (Romme and Despain 1989). The 1988 Yellowstone fires affected 35% of my study area, including 29% of mature forests. In addition, although GNF timber harvesting over the past 50 years has involved <1.0% (885 ha) of the mature forests in the study area, it has added cumulatively to fire effects by increasing the proportion of young forests in moose winter range. CEA

Across most of North America, fire and logging in mature forests are considered beneficial for moose winter range because removal of conifer competition can stimulate growth of post-disturbance shrub fields providing concentrations of deciduous moose browse and efficient foraging opportunities (Peterson 1955, Geist 1971, Kelsall and Telfer 1974, Coady 1982). However, the effects of disturbances on NYWR moose cover type use patterns have not been investigated, and others have speculated that moose habitat relationships in this area may be anomalous. Loope and Gruell (1973) hypothesized that GYA moose do not benefit from early seral stages but rather are associated with mature conifer forests where double canopies ameliorate snow conditions and provide browse. They forwarded the idea that GYA moose numbers have increased due to a general lack of stand replacing fires over several centuries. They believe subalpine fir, a shade-tolerant tree that reaches its greatest densities in the oldest forests (see Chapter 6), is the primary food item of GYA moose. In support of this theory, McDowell and Moy (1942) and Peek (1974a) speculated that NYWR moose cover type use involves an early winter association with permanent streamside willow stands until snow accumulation forces a mid-winter move to adjacent forested mountainsides where dense cover and abundant conifer browse provide for habitat needs.

Because a greater variety of cover types may be available post-disturbance in boreal forest landscapes for moose to select from, moose habitat use patterns following fires or logging can help identify critical habitat components; i.e. post-disturbance landscapes may have a greater mix of forest types and ages than more uniform undisturbed environments allowing for an assessment of moose proportional use of available successional stages (Neu et. al 1974). I used the 1988 Yellowstone fires as an opportunity to examine the theories of Loope and Gruell (1973), McDowell and Moy (1942), and Peek (1974a), and contrast NYWR moose cover type use with other North American populations, the preponderance of which are associated with post-disturbance shrub fields. Results of this investigation have implications for management of moose populations and remaining NYWR mature forest habitats.

Peek (1997) maintained that a significant shortcoming in moose research and management has been the inability to determine accurately the limits of adaptability of moose to habitat alteration. To rectify this he recommended investigating patterns of moose habitat use illustrating variation among seasons, years, and areas, and determining reasons for variation. Generally, habitat use patterns for moose are obtained by comparing frequency distributions of tracks, pellet groups, or radiolocations within each habitat parameter of interest with the availability of that parameter to the population, the assumption being that that there is a positive correlation between moose abundance and habitat quality (Neu et al. 1974, Van Horne 1983, Bookhout 1994, Peek 1997). Although commonly accepted, the techniques used to collect moose habitat

Studies indicate that moose habitat is often enhanced post-disturbance by a temporary increase in available forage. Parker and Morton (1978) demonstrated that woody biomass peaked 8 years following a clearcut in Newfoundland. On the Kenai Peninsula, Alaska, important browse species peaked 15 (Spencer and Hakala 1964) and 10 years (Oldemeyer and Regelin 1987) post-fire. Based on fluctuations in browse biomass and moose populations in boreal forests, Kelsall et al. (1977) concluded that the optimum successional stages for moose were 11 to 30 years after burning. Thompson and Stewart (1997) reported that moose need young successional stands in boreal forests but that their value is transient, lasting less than 30 years. Peek et al. (1976) found that the effects of logging in fire-prone Minnesota forests were similar to burning, comparable vegetation mosaics and increases in forage and moose populations resulted.

North American moose are also occasionally associated with habitats where fire is infrequent. High-density moose populations can exist where fire is either not a major influence or is detrimental to habitat availability (Peek 1997). Examples include river delta systems in Alaska (Doerr 1983, MacCraken 1992), habitats at or above timberline in central Alaska (Miquelle and Van Ballenberghe 1989, Risenhoover 1989), and mesic maritime forests of eastern Canada (Telfer 1984). In central Idaho, moose inhabit areas of densely forested steep terrain with limited riparian vegetation that burn infrequently (Peek et al. 1987). These forests have double canopies with Pacific yew (*Taxus brevifolia*) as a subcanopy beneath mixed conifers that are >150 years old. The yew provides food and cover and the double canopy intercepts snowfall. In other areas of Idaho where yew is not abundant, spruce-fir forests provide similar habitat with a mixture of seral and climax species including subalpine fir, Engelmann spruce, western larch (*Larix occidentalis*), lodgepole pine, and Douglas fir (Peek et al. 1987, Peek 1997).

Where moose are associated with mature boreal forests with intact canopies, as opposed to recently disturbed areas, juxtaposition of food and shelter during late winter may be essential for their survival. This spatial relationship between cover and food becomes limiting because snow makes low growing forage unavailable (Coady 1974, Telfer 1978, Welsh et al. 1980, Thompson and Vukelich 1981, Eastman and Ritcey 1987, Peek et al. 1987) (see Chapter 5). In Alberta, Stelfox (1981) found that regrowth of adequate winter shelter did not begin until 25 years after logging, and moose did not use the disturbed area in winter before that time (Stelfox et al. 1976). Welsh et al. (1980) showed that moose in boreal Ontario migrated from previously occupied habitat that was recently clearcut to mature forests as much as 50 miles (80.5 km) away. In response to progressively deeper snows and less accessible food in open areas, a mid-winter shift into closed canopy habitats has been documented in Alberta (Rolley and Keith 1980), British Columbia (Eastman 1974), Wyoming (Houston 1968), Montana (Stevens 1970), eastern Minnesota (Peek et al. 1976), western Minnesota (Phillips et al. 1973), Ontario (Thompson and Vukelich 1981), and Nova Scotia (Telfer 1970).

digestible food in patches that permit high rates of intake (Saether 1990, Saether and Anderson 1990). Moose could not survive only on sparse high-quality food because too much time would be spent in searching rather than harvesting. Thus the operating niche of moose is defined in terms of forage and nutrient distribution (Regelin et al. 1985, Renecker and Schwartz 1997).

Assessing seasonal diet composition changes for moose is difficult because not only must the food items be identified, but their amount, availability, current nutritional value, and risk of acquisition should also be determined (Schwartz and Renecker 1997). To meet winter food needs, NYWR moose used season-specific foraging patterns in response to some combination of these determinants. Well-developed pre-fire methods for finding food were interrupted by the 1988 fires that significantly altered habitat and required new foraging strategies.

The 5 principal browse species used by NYWR moose occur in the diets of other North American moose (Renecker and Schwartz 1997), but the combination and relative proportions of plant species may be unique to this area. Subalpine fir, the principal food for NYWR moose (especially during mid- and late winter), has not been widely reported in other food habits studies, but it is browsed by moose in British Columbia (Eastman and Ritcey 1987), Alberta (Cowan et al. 1950), northwest Wyoming (Harry 1957, Houston 1968), and southwest Montana (Knowlton 1960, Smith 1962, Dorn 1970, Stevens 1970). Willow is a preferred moose food and is used whenever and wherever it is available (Renecker and Schwartz 1997), including the subalpine and subarctic regions of Alaska (Risenhoover 1989) and throughout the montane regions of North America (Peek 1974a and b). Lodgepole pine has been reported in the diet of moose in British Columbia (Eastman and Ritcey 1987), Washington (Poelker 1972), southwest Montana (Smith 1962), and central Canada (Barrett 1972, Risenhoover 1987). Documentation of winter buffaloberry use is limited to Manitoba (Trottier et al. 1983) and Alberta (Renecker 1987, Cairns 1976), although McDowell and Moy (1942) observed summer use in the study area. Browsing on various species of gooseberry is reported for Manitoba (Trottier et al. 1983), Alberta (Renecker 1987), southwest Montana (Knowlton 1960, Stevens 1970), northwest Wyoming (Houston 1968), and Alaska (Hosley 1949).

Moose density is often positively associated with food abundance, which varies with seral age of forests (Schwartz and Franzmann 1989). Biomass of desirable forage species and moose numbers usually increases within several years after a disturbance and then declines (Parker and Morton 1978, Eastman and Ritcey 1987). However, NYWR moose showed a more consistent association with mature coniferous forests than with early seral communities. Almost 10 times more browsing occurred in late as opposed to early forest successional stages.

As McDowell and Moy (1942) and Peek (1974a) theorized, NYWR moose begin winter in streamside willow stands before a mid-winter shift to adjacent mature forests. In the early winter, NYWR moose consumed quantities of willow in widely scattered but concentrated areas where it was dominant or a co-dominant with young to mature conifers. They also sought nutrient-rich gooseberry in mature spruce-fir forests and subalpine fir in this and other old conifer types. As winter progressed, subalpine fir became the dominant food while use of willow decreased, use of lodgepole pine and buffaloberry increased, and gooseberry became only a minor food item. Increased use of lodgepole pine probably reflected more difficult late winter foraging conditions and the need for moose to make up for diet shortfalls. With its high winter-long content of the macronutrients calcium, magnesium, sodium, and phosporous, gooseberry is probably nutritionally valuable to moose during all winter periods. However, it is low growing and unavailable under snow after early winter (see Chapter 5). Buffaloberry, comparatively unused in early winter, probably increased in importance because of its high late-winter crude protein content. Although among and within species variation in the nutritional quality of browse plants was apparently important determinants for NYWR moose food habitats, changes in crude protein, mactronutrients, and ether extract content were not statistically correlated with seasonal diet shifts, indicating that other factors, such as snow conditions (see Chapter 5) and plant densities, may have been more important in regulating what moose ate.

Loope and Gruell (1973) were also correct in their speculation. Most mid- and late winter foraging was done in mature lodgepole pine forests where moose browsed subalpine fir. As a shade tolerant species and part of the lower layer in double canopy forests, nearly all browsed subalpine fir were <5 m tall and most were < 1m tall. Subalpine fir was used in proportions greater than availability in all ages and types of forests, while other conifer species were used less. Subalpine fir was an indicator of foraging conditions. Browsing intensity on subalpine fir increased during situations with more difficult foraging conditions due to deeper snow and loss of cover and foraging habitat, including late winter, after the 1988 fires, and in the areas where fire impacts were the greatest.

Moose response to the 1988 fires included changes in diet and foraging locations. Post-fire, moose abandoned burned coniferous forests in favor of nearby willow areas less affected by fire resulting in more intense early winter browsing on willow. However, use of willow stands declined in mid- and late winter to below pre-fire levels. Because these areas are spatially limited, available forage was probably exhausted in early winter with the additional browsing pressure. For example, in the SC study unit where fire effects on mature lodgepole pine and spruce-fir forests were greatest (see Chapter 2), increases in willow and lodgepole pine browsing and declines in subalpine fir browsing were also the greatest. In addition to heavy during more difficult conditions, may represent a simple attempt to maintain rumen fill. Use of lodgepole pine appears to be directly related to the magnitude of fire effects and loss of preferred foraging areas.

The reliance of NYWR moose on mature coniferous forests for winter foods makes them susceptible to natural or human-induced disturbance. Loss of these areas during the 1988 fires required moose to adopt less energy efficient foraging strategies. Where unburned woody vegetation becomes sparse or widely dispersed as occurred in 1988, moose are unlikely to prosper (Renecker and Schwartz 1997). Moose shifting to less nutritionally valuable forage and spending more time foraging illustrated this. Food shortages may compound problems for remaining moose because limited food supplies may be over-utilized. Riparian habitats with live woody vegetation tend to concentrate moose when other foraging areas are unproductive. The resulting browsing pressure may deplete food in the short term and long-term ecological changes may also ensue (Oldemeyer 1983). Because of the importance of mature conifer forests with shade-tolerant subalpine fir understories, fires had negative consequences for NYWR moose forage availability that may take centuries to erase (Loope and Gruell 1973).

Moose have evolved to survive in harsh northern climates where winter conditions require precise foraging strategies, including the use of patches of preferred food to minimize travel. NYWR moose displayed habitat use patterns commensurate with other North American moose populations by browsing rare high quality foods when available but relying on biomass concentrations of woody plants to balance energy budgets. However, the characteristics unique to NYWR moose diet and foraging patterns reveal strategies for survival specific to this environment. Unlike most populations, NYWR moose benefited from favorable foraging opportunities in mature forests rather than post-fire shrubfields (Loope and Gruell 1973).

OLD-GROWTH HABITATS AND ASSOCIATED WILDLIFE SPECIES IN THE NORTHERN ROCKY MOUNTAINS

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INVENTORY AND STAND SELECTION

INTRODUCTION

Old-growth forests are an important component of biological diversity. Old-growth stands are characterized by an increasing variety in sizes and species composition of trees, and more diversity of functions and interactions as compared with earlier successional stages. A complex, multistoried structure and a mosaic of both early and late successional stages often are important attributes (Bormann and Likens 1979). Old-growth forests are not the same as successional climax (habitat type), often being dominated by seral tree species.

The greater vertical and horizontal diversity found within an old-growth stand allows for niche specialization by wildlife. Although the individual wildlife species occurring may not be unique to old-growth stands, the assemblage of wildlife species and the complexity of interactions between them are different than in earlier successional stages.

OLD-GROWTH FORESTS: CONDITIONS IN THE NORTHERN ROCKIES

The Northern Rocky Mountains exhibit very diverse vegetation patterns. In addition to wide gradients in climate and elevation, wildfire has played a major role in the evolution of forest ecosystems in the northern Rockies (Habeck 1987). At the time of European settlement, fire-generated or fire-perpetuated forest types dominated much of the region.

Northern Idaho and northwestern Montana, which experience an inland maritime climate, support near-climax old-growth forests dominated by western redcedar and western hemlock, as well as subclimax old-growth stands dominated by western larch and western white pine. These sites historically burned infrequently, at 100 to 200-year intervals, often by high-intensity stand replacing fires (Habeck 1985).

In west-central Montana, old-growth forests at lower elevations were typically maintained as an open-canopied savanna by frequent (5 to 20-year interval), low intensity ground fires (Arno 1980). These stands were dominated by western larch and ponderosa pine, or by Douglas-fir on moister sites. Recent fire suppression has favored shade-tolerant tree species such as Douglas-fir. At higher elevations, old-growth forests are dominated by true firs and burn less frequently.

In central and eastern Montana, lodgepole pine dominates large acreages. Lodgepole pine stands typically experience frequent, stand-replacing disturbances from fire or insect outbreaks. Along the eastern ecotone between forest and grassland, ponderosa pine and limber pine stands experienced frequent low-intensity fires (Habeck 1988).

DEFINITIONS OF OLD-GROWTH

Attributes of old-growth stands include large-diameter overstory trees, presence of large standing dead and defective trees, presence of down logs, development of a deep duff layer, and formation of canopy gaps and several canopy layers. Several of these attributes may be less apparent in types experiencing frequent fire.

Long periods of time are required to develop old-growth conditions within a stand. However, stand age may not accurately predict the onset of old-growth conditions, because tree species, site conditions, fire or other disturbance, variations in weather, and other factors interact to affect the rate of succession within a stand.

Structural attributes have generally been used as descriptors of old-growth stands. Pfister (1987) attempted to identify existing old-growth stands on the Nez Perce and Kootenai National Forests, using overstory tree size, density of snags, density of down material, canopy closure, and canopy layering. Some of the attributes, particularly snags and down logs, were inadequately sampled. Using stand exam data, Pfister found that few stands met all of the pre-determined criteria for old-growth stands.

The national definition developed by the USDA Forest Service describes old growth conceptually, in terms of plant succession and general characteristics. Ecological classifications are to be developed to define the old-growth community type(s) for each major forest type (or forest type group). Definitions for old-growth communities will be based on vegetation structural characteristics which are easily measured.

OLD-GROWTH INVENTORY

Forest-Wide Inventory. Forest-wide estimates are needed of the relative abundance, patch sizes, and spatial distribution of old-growth habitats by forest type. As a first approximation, *candidate* old-growth habitats should be identified, by stand or groups of stands.

Timber stands are delineated on the basis of predominant overstory species, tree size, and tree density. Contiguous old-growth habitat may be composed of more than one stand.

Data sources for identification of candidate stands could include aerial photographs, satellite imagery, timber stand exam data, Ecodata plot data, and other inventories. Detailed information, particularly on dead trees and down logs, may not be available for each stand. For this reason, additional field analysis of candidate stands may be needed.

Candidate old-growth habitat maps should be updated periodically, to reflect actual old-growth habitat delineations as site-specific inventory information becomes available.

To develop an ecological classification for old-growth community types, data on dominant and indicator species must be collected. The Ocular Macroplot or the Cover Microplot sampling methods (FSH 2090.11) would be suitable for this purpose. The optional Density sampling method can be used to estimate the density of components such as snags and down woody material.

Site-specific Stand Analysis. Stand exam and Ecodata plot sampling methods are recommended to describe vegetation composition, tree characteristics, and structure within a stand. Additional sample plots are needed for components that occur in low densities, such as snags and down woody material.

A general rating of the ability of a given stand to provide old-growth habitat conditions can be made. If the old-growth criteria are used collectively as an "in/out" screen, some stands may be excluded which do in fact have value. Thus, a relative rating system should be used.

Old growth "Scorecards" have been used successfully in various areas. Points are scored for each parameter (such as overstory tree size, number of trees per acre, canopy cover, canopy structure, density of snags and down logs, and decadence) and summed for the stand.

Example scorecards, which are based on judgement rather than data, are shown in Exhibit 1. Scorecards should be refined to reflect local habitat types and conditions. To interpret the meaning of the scoring, comparisons could be made with reference stands (those sampled to develop old-growth community classification).

WILDLIFE SPECIES ASSOCIATED WITH OLD-GROWTH HABITATS

Meslow and Wight (1975) found that 69 percent of the birds occurring in Douglas-fir forest types of the Pacific Northwest used old-growth stands. Three species were listed as nesting primarily within old-growth forests: spotted owl, northern goshawk, and Vaux's swift. Bull (1978) identified six species which are primarily associated with old-growth forests in Oregon: great gray owl, barred owl, flammulated owl, white-headed woodpecker, northern three-toed woodpecker, and Townsend's warbler. The marten is associated with mature and old-growth spruce-fir forests in Idaho (Koehler and Hornocker 1977).

In northwestern Montana, McClelland (1977) described a general trend of increased species richness in cavity-nesting birds from young to old-growth stands of larch and Douglas-fir. Old growth was particularly important in providing an adequate number of suitable nesting trees for cavity-nesters. He also noted the association of open-nesters such as the pine grosbeak, Townsend's warbler, varied thrush, black-headed grosbeak, and goshawk with old-growth forests, as well as its value as big game thermal cover.

44%

Based on a literature review, about 40 percent of the 373 wildlife species occuring in the Northern Region were thought to use old-growth forest for feeding and/or reproduction (Harger 1978). Of these, 33 species were thought to be closely associated with old-growth habitats (Table 1).

33 sps

The following describes the importance to wildlife of various components of old-growth habitats.

Overstory Trees. Large trees are needed to provide suitable nesting sites for large birds. Bark crevices in older trees provide important foraging sites. Large-canopied trees can modify microclimate by providing shade, capturing moisture, and moderating winds.

Dead and Defective Trees. Dead trees (snags) and defective trees (partially dead, spike top, broken top) provide nesting and roosting sites for cavity-users. Snags host invertebrates which are an important food source for woodpeckers.

Down Woody Material. Downfall supports insects and other invertebrates, provides habitat for fungi and saprophytic plants, provides cover and den sites for wildlife, and creates debris dams in streams. Dead woody material holds moisture and contributes to nutrient cycling. In areas with high frequency of fire, this component will be less abundant.

Tree Canopy. A relatively closed canopy, often with two or more layers, creates a more moderate microclimate. Vertical diversity provides a variety of substrates for feeding or nesting, and supports development of forest components such as arboreal lichens.

Decadence. Presence of heart rot, mistletoe, dead or broken tree tops, diseased trees, and saprophytic plants create a variety of microsites and food sources for wildlife.

MANAGEMENT INDICATOR SPECIES

The National Forest Management Act and its implementing regulations require that fish and wildlife habitats be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area. In order to estimate the effects of each Forest Plan alternative on fish and wildlife populations, Management Indicator Species (MIS) are to be selected.

33

Table 1. Species in the Northern Region (north Idaho, Montana, North Dakota, and South Dakota) thought to prefer old-growth components for breeding or feeding (from Harger 1978).

21733

CN 18
log 7

WILDLIFE SPECIES	STRUCTURAL STAGES Breed	Feed	OPTIMUM COVER TYPE
Great Blue Heron	5-6	1	ASP, COT
log Northern Goshawk	5-6	3-6	ASP, DF, PP, LPP
log Great Gray Owl	6	1	LPP, SAF, SP
Flammulated Owl CN	5-6	1-2	PP, DF, GF, LPP
Pygmy Owl CN	5-6	1-6	PP, DF, GF, WL
Saw-whet Owl CN	5-6	1-6	PP, DF, GF, LPP, SAF, SP
Boreal Owl CN	5-6	1-6	PP, ASP, SP, SAF, DF
Barred Owl CN	5-6	3-6	COT, DF, GF
Vaux's Swift	5-6	1-6	ASP, COT, DF, WRC
White-headed Woodpecker CN	4-6	4-6	PP, WWP
Pileated Woodpecker CN	4-6	3-6	COT, PP, WL, GF
Three-Toed Woodpecker CN	4-6	3-6	SAF, WBP
Black-backed Woodpecker CN	4-6	3-6	PP, DF, WWP
Red-naped Sapsucker CN	4-6	1-6	BGA, COT
Williamson's Sapsucker CN	4-6	2-6	PP, WL, GF, SAF, SP
White-breasted Nuthatch CN	4-6	3-6	PP, DF
Red-breasted Nuthatch CN	4-6	3-6	PP, ASP, LPP, SAF, SP
Pygmy Nuthatch CN	4-6	3-6	PP, DF
Brown Creeper CN	4-6	3-6	DF, LPP, SP, SAF
Great Crested Flycatcher	4-6	2-4	BGA, COT
Hermit Thrush	5-6	2-6	GF, WH, WRC, SAF, SP
Varied Thrush	5-6	2-6	WH, WRC, SP
Townsend's Warbler	4-6	2-6	ASP, COT, PP, DF
Silver-haired Bat CN	3-6	1-3	PP, DF, WRC, SP
Long-eared Bat CN	4-6	1-6	PP, DF, GF
Long-legged Myotis	4-6	1-6	LPP, SAF, SP
California Myotis	4-6	1-6	BGA, COT, PP, JPU
Fisher	5-6	3-6	ASP, COT, PP, DF, WWP
log Marten	5-6	2-6	LPP, SAF, SP, WRC
Wolverine	3-6	1-6	SAF, WBP
Woodland Caribou	5-6	5-6	SAF, SP, WH, WBP
log Boreal Red-back Vole	4-6	4-6	SAF
Northern Flying Squirrel CN	5-6	5-6	ASP, COT, DF, WWP, SAF

Hutto

Structural Stages: 1=Grass/Forb, 2=Shrub/Seedling, 3=Pole Trees, 4=Young Trees, 5=Mature Trees, 6=Old-Growth

Cover Types: ASP=Aspen, BGA=Birch/Green Ash, COT=Cottonwood, DF=Douglas-fir, GF=Grand Fir, JPU=Juniper, LPP=Lodgepole Pine, LP=Limber Pine, PP=Ponderosa Pine, SAF=Subalpine Fir, SP=Spruce, WH=Western Hemlock, WL=Western Larch, WRC=Western Redcedar, WWP=Western White Pine, WBP=Whitebark Pine

Several categories of species are to be represented where appropriate: Endangered and threatened plant and animal species; species with special habitat needs that may be influenced significantly by planned management programs; species commonly hunted, fished or trapped; non-game species of special interest; and plant or animal species selected because their population changes are believed to indicate the effects of management activities on other species of selected major biological communities (36 CFR 219.19). The first four categories of MIS include "featured species", wherein management activities are directed to providing specific habitat components to meet management objectives for the individual species. The latter category is intended to include ecological indicators, which are selected to represent a larger community of species.

An ecological indicator will ensure the welfare of only those species whose niches are entirely included within its habitat niche and geographic range. To be effective, then, an ecological indicator should have a large home range and be closely associated with a specific habitat.

The goshawk, pileated woodpecker, and marten were identified as MIS by most Forests in the Northern Region to be used as ecological indicators for old-growth components or old-growth habitats. Each of these species have habitat requirements related to stand structure or components which are more likely to be found in old-growth habitats. Their population densities generally are higher in old-growth than in younger stands. In a California study, both marten and pileated woodpecker were found to be sensitive to habitat fragmentation (Rosenberg and Raphael 1986). Each of these species utilize a relatively large home range, which is thought to include the home ranges of other old-growth-associated species.

The habitat requirements of pileated woodpecker, goshawk, and marten, and models for assessing habitat suitability for each are detailed in subsequent sections. Although Schroeder (1987) identified several problems in HSI modeling, they are the most expedient method for evaluating impacts of habitat change.

The requirements of these species in terms of the size and spacing of habitat patches can be used to determine the distribution of old-growth habitats across the landscape.

OLD-GROWTH STAND SELECTION

Landscape Ecology Theory. Forman and Godron (1981) defined landscape patches as communities surrounded by a matrix with a dissimilar community structure or composition. They hypothesized that species diversity within a patch is a function of habitat diversity, disturbance, area, age, surrounding heterogeneity, isolation, and boundary discreteness.

Old-growth habitats offer comparatively high within-stand structural diversity. As forest management progresses through time, old-growth forests will be represented as remnant patches embedded in a younger-aged matrix.

Patch size correlates strongly with the numbers of species and individuals that can be supported and with rates of extinction and recolonization (May 1975, Simberloff and Aberle 1982). Patch size is particularly significant for large, wide-ranging species (Noss and Harris 1986). As acreage decreases, habitat patches become unsuitable for wildlife species with large home ranges. Of 48 old-growth-associated species occurring in the Northern Region, about 60 percent are thought to require stands larger than 80 acres (Harger 1978).

Patch size is modified by its shape due to edge effect. Changes in temperature, humidity, light, and wind, which influence plant species composition, can occur in a band along the ecotone (edge) (Greene 1988). In Douglas-fir old-growth forest, a circular-shaped stand smaller than 25 acres is effectively all "edge" (influenced by the ecotone); within a stand of 80 acres, half of the patch provides "interior" conditions (Lemkuhl and Ruggerio in prep.). In northern California, Rosenberg and Raphael (1986) found that old-growth Douglas-fir stands of less than 50 acres tended to lack the full complement of vertebrate wildlife, and suggested that isolated stands should exceed 125 acres in size to be effective. Wilcove et al. (1986) estimated that habitat islands should exceed 250 acres to provide for birds inhabiting forest interior.

The abruptness of the edge, or edge contrast, affects stand insularity. Edges between similar cover types and successional stages cause less isolation than do edges between very dissimilar stands. A patch totally surrounded by dissimilar habitat becomes more similar to a true island, with less movement of species between patch and matrix (Forman and Godron 1981).

Selection of Stands to Manage for Old-Growth Conditions. The recommended scale for initial analysis of the quality and distribution of existing old-growth habitats is the watershed scale (typically about 10,000 acres in size). Individual stands or groups of contiguous stands can be evaluated using a scorecard method. The habitat requirements and dispersal capabilities of old-growth Management Indicator Species should be used to determine the size, shape and spacing of old-growth patches across the forest landscape.

If a particular analysis unit does not contain existing stands of high quality, large enough size, or sufficient overall acreage, the best available (nearest to old-growth condition) stands should be selected. Alternatively, old-growth stands in adjacent watersheds could be selected, provided the spacing of old-growth patches does not exceed the mobility of the MIS.

Roads are generally undesirable within an old-growth habitat patch. The road corridor fragments the habitat by creating edge, and access may result in loss of snags to woodcutting.

Landres et al. (1988) point out that selecting old-growth stands based on within-patch habitat conditions required by the MIS, and then predicting or monitoring habitat conditions for those MIS, is circular reasoning. Because old-growth-associated MIS are intended to represent a community of wildlife species, stand selection and management should not be directed only towards providing the minimum characteristics required by the individual species. Management objectives and actions in the selected old-growth patches should be directed towards protecting or enhancing the integrity and longevity of old-growth conditions.

In devising a conservation strategy, Forman and Godron (1981:738) emphasize the importance of recognizing that "no patch stands alone", but is influenced by surrounding patches. Harris (1984) recommended surrounding old-growth habitat islands with a long-rotation management area, and interconnecting these with riparian corridors and other linkages. Similarly, Noss and Harris (1986) suggest that a regional landscape level be used, wherein high-quality nodes, such as an old-growth patch, would be integrated within interconnected "multiple-use modules", where management activities are scheduled to maintain the integrity of the nodes.

Natural disturbance regimes such as fire often occur on a spatial scale larger than a landscape patch (Urban et al. 1987). Providing for well-distributed habitat patches with interconnections between patches thus are necessary to maintain species diversity over the long term.

Pileated Woodpecker Habitat Relationships

William C. Aney and B. Riley McClelland

Revised by Nancy Warren, Karen Wilson, Mike Hillis, Bob Summerfield,
Carl Frounfelker, Tom Wittinger, and Wally Murphy

INTRODUCTION

The pileated woodpecker requires tall, large-diameter dead or live defective trees within forest stands for nesting. It is particularly characteristic of old-growth stands of ponderosa pine, western larch, and black cottonwood stands.

Most of the Forests in the Northern Region have selected the pileated woodpecker as a Management Indicator Species for old-growth habitats. The following describes habitat requirements of the pileated woodpecker, a model to evaluate habitat suitability, and recommended methods for inventory and monitoring of habitats and populations.

ECOLOGY OF THE PILEATED WOODPECKER

Distribution. The pileated woodpecker (*Dryocopus pileatus*) is a year-round resident of forested areas in the northern Rockies. This large woodpecker is absent from the central and southern Rockies (Bent 1964), probably due to the absence of dense highly productive forests in those regions (Bock and Lepthien 1975).

Reproduction. Courtship usually begins in mid-March. Both drumming and vocalizations are used by the male to attract a mate. Pileateds regularly begin breeding as one-year olds, and virtually all birds breed every year (Bull 1987).

One or more new cavities are excavated during courtship each year, within the heartwood of a tree. The distance between successive nests used by the same pair averaged 0.3 mile (0 - 1 mile) in Oregon (Bull 1987).

Clutch size is typically four. The incubation period, usually occurring in May, is about 18 days. The nestlings fledge about one month after hatching. Usually 2-3 young fledge from a nest.

Birds that were banded as nestlings and later nested withing the same study area in Oregon (Bull 1987) dispersed an average of two miles (0 - 5.2 miles).

Nesting Habitat. Typical pileated nest stand conditions can best be described as old-growth stands with a decadent overstory of western larch, ponderosa pine, black cottonwood, or aspen. Stands of 50 to 100 contiguous acres, generally below 5000 feet in elevation, with basal areas of 100-125 square feet per acre and a relatively closed canopy, were used for nesting (McClelland 1977, 1979a, Bull 1980). The grand fir cover type was preferred in Oregon and western larch/Douglas-fir cover types are preferred in Montana (Bull 1987, McClelland et al. 1979).

Nests in the northern Rocky Mountains most commonly occur in dead ponderosa pine, dead black cottonwood, or live or dead western larch trees. Nest trees average nearly 30 inches in diameter, and are more than 91 feet tall in Montana (Aney and McClelland 1985). Minimum nest tree diameter is 20 inches dbh. Nest cavities usually are located more than 30 feet above the ground, at a level within the canopy of the surrounding forest (McClelland 1979a, Bull 1980).

Heart rot appears to be an important feature of suitable nest trees. Decay fungi (*Laricifomes laracis* or *Phellinus pini*) enter the heartwood of living or dead larch trees through a broken top, fire scar, or other wound; decay gradually softens the heartwood while leaving a shell of sound sapwood. Conner et al. (1976), McClelland (1977, 1979a), Mannan et al. (1980), and Harris (1983) all reported a high incidence of broken tops in nest trees.

Pileated woodpeckers are able to excavate cavities in sound wood, however. Bull (1975, 1980) reported that the majority of nest trees in her northeastern Oregon study area, where ponderosa pine was the most common nest tree, showed no evidence of decay at the nest site. This apparently is related to characteristics of the dominant nest tree species of the two regions. Ponderosa pine was the most common nest tree in northeastern Oregon; western larch was predominant in the Montana study. Sound ponderosa pine, with a specific gravity of about 0.37 g/ml, is considerably softer than sound western larch (specific gravity of 0.48-0.52 g/ml). Decayed western larch heartwood, however, is softer than sound pine wood (0.27 g/ml) (McClelland 1977, Harris 1983).

In the Northern Region, either live or dead larch trees can be suitable nest trees, if heart rot is present. Ponderosa pine trees need not have decay at the nest site, but the excavated portion of the bole must be dead. Nest excavation in black cottonwood occurs almost always in dead portions of the bole, but the presence of heart rot is not essential (McClelland 1977). Aspen can provide suitable nest sites if trees are large enough.

Food Habits. Pileated woodpeckers feed principally on carpenter ants (*Camponotus herculaneus* and *C. pennsylvanicus* , excavating deep into ant colonies in dead and decaying wood (Sutton 1930, Bent 1964, Hoyt 1957, McClelland 1977, Bull 1980, Miller and Miller 1980). Beal (1911) examined the stomach of one pileated woodpecker and found the remains of more than 2600 ants. Estimates of the proportion of ants in the diet range from 40-60% (Koehler 1981) to 95% (Bull 1987).

Other insects reported in the diet include termites (Isoptera) and beetles (Coleoptera), which are obtained by bark gleaning, scaling, or excavating. In winter, when few insects are available on the boles of trees, pileateds feed principally by excavating.

Fruits and berries form a secondary component of the diet of pileated woodpeckers in the late summer and fall.

Feeding Habitat. Because carpenter ants make up the bulk of the pileated woodpecker's diet, suitable ant habitat is selected for foraging by pileated woodpeckers. These ant colonies occur most often in large snags with advanced decay, the moist decaying butts of live trees, logs greater than 10 in. in diameter, and natural or cut stumps (Cline 1977, Conner et al. 1975, Bull 1980, Bull and Meslow 1977, McClelland 1977, 1979a). Furniss and Carolin (1977) noted that carpenter ants expecially favor fire-scarred and butt-rotted western redcedar trees, and will make extensive use of Douglas-fir, pines, and true firs.

In Montana, carpenter ants were found to select stands of high canopy cover, but ant densities declined as canopy cover increases past 90% (Youngs and Campbell n.d.). Similarly, stands with basal area in the range of 100 sq.ft. per acre were favored over more densely stocked stands (>150 sq.ft. per acre). *C.modoc* densities were positively correlated with dead wood volume in snags and stumps.

McClelland (1977) suggested that availability of food in winter may act as an ecological "bottleneck", limiting northern Rocky Mountain woodpecker populations. Snowpack makes logs and low stumps unavailable to feeding pileated woodpeckers.

McClelland (1979a) noted that pileated woodpeckers usually avoid open areas for feeding, preferring forests with a significant old-growth component and high basal area. Shelterwood cuts and small group selection cuts are suitable, but not preferred, feeding areas. Bull and Meslow (1977) classified preferred feeding habitats as having high densities of snags and logs, dense canopies, and tall ground cover, with more than 10% of the ground area covered by logs. Kilgore (1971) found that reduction of dead and down woody material by fire caused a decline in the use of a giant sequoia forest by pileateds for feeding.

In the northern Rockies, the density of snags and stumps at pileated feeding sites (not throughout the feeding range) averaged 7 per acre (Aney and McClelland 1985). At least 500 acres of suitable feeding habitat is needed within the home range of a pair (McClelland 1979a).

Home Range Size. Nesting pairs of pileated woodpeckers in the northern Rockies often cover 500-1000 ac. in their daily feeding activities (McClelland 1979a). In high-quality habitat in the northern Rockies, densities of 1 pair per 500 acres are not uncommon (McClelland pers. comm.). Bull (1987) estimated a density of one pair per 480 acres in northeastern Oregon, while Mellen (1987) reported an average home range size of 1200 acres for pairs in western Oregon.

Dispersal Distance.

Bull (1987) recorded juvenile dispersal distances of banded nestlings averaging 2 miles. The longest dispersal was 5.2 miles, although areas outside the study area were not searched, eliminating the chance of finding those that dispersed farther than 9.6 miles.

When adult birds lose their mates, they remain on territory. Juvenile dispersers are essential to mate with unpaired territorial adults and to recolonize unoccupied habitats (Bull pers. comm.).

REVISED FOREST PLAN
for the
TARGHEE NATIONAL FOREST
Intermountain Region R-4
April 1997

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This Revised Forest Plan was prepared according to Secretary of Agriculture regulations (36 CFR 219), which are based on the Forest and Rangeland Renewable Resources Planning Act (RPA) as amended by the National Forest Management Act of 1976 (NFMA). This Revised Forest Plan was developed in accordance with regulations (40 CFR 1500) for implementing the National Environmental Policy Act (NEPA). A detailed Environmental Impact Statement (EIS) has been prepared as required by NEPA and 36 CFR 219.

If any particular provision of this Revised Plan, or the application of the action to any person or circumstances, is found to be invalid, the remainder of the proposed action and the application of that provision to other persons or circumstances shall not be affected.

3. The following conditions and criteria will apply in determining the problem status of wolves. (USDI Fish and Wildlife Service 1994 a and b) (S)

A. Wounded livestock or some remains of a livestock carcass must be present with clear evidence that wolves were responsible for the damage and there must be a reason to believe that additional losses would occur if the problem wolf or wolves were not controlled. Such evidence is essential since wolves may simply feed on carrion they have found while not being responsible for the kill.

B. Artificial or intentional feeding of wolves must not have occurred. Livestock carcasses not properly disposed of in an area where depredations have occurred will be considered attractants. Removal or resolution of such attractants must accompany any control action. Livestock carrion or carcasses not being used as bait in an authorized control action (by agencies) must be removed, burned, treated with an acceptable chemical repellent, or otherwise rendered such that the carcass(es) will not attract wolves using methods approved by the District Ranger.

C. Animal husbandry practices previously identified in existing approved Allotment Management Plans and annual operating plans for allotments must have been followed.

4. If additional livestock depredations are likely, proper animal husbandry practices are employed (proper disposal of livestock carcasses, etc.), artificial feeding does not take place, and AMPs are followed, the Forest may implement procedures to harass, capture, move, or kill wolves that attacked livestock (defined as cattle, sheep, horses, or mules only) on National Forest land. (G) Prior to the establishment of six breeding pairs, depredating females and their pups will be captured and released at or near the site of capture, one time prior to October 1. If depredations continue, or if six packs are present, females and their pups will be removed. (USDI Fish and Wildlife Service 1994 a and b) (S)

Goal - Peregrine Falcon Habitat

Plan project activities to avoid adverse impacts to falcons and their habitats.

Standards and Guidelines - Peregrine Falcon Habitat

1. For proposed projects within two miles of known falcon nests consider such items as: 1) human activities (aircraft, ground and water transportation, high noise levels, and permanent facilities) which could cause disturbance to nesting pairs and young during the nesting period March 15 to July 31; 2) activities or habitat alterations which could adversely affect prey availability. (G)

2. Within 15 miles of all known nest sites, prohibit all use of herbicides and pesticides which cause egg shell thinning as determined by risk assessment (USDA-Forest Service, September 1992). (S)

3. Restrict climbing and other human disturbances from March 15 through July 31 to avoid adverse impacts at known falcon nest sites. (S).

Objective - Wolverine Habitat

Within two years of the ROD complete a GIS inventory to identify potential wolverine natal den sites. Within 4 years of the ROD, survey all potential wolverine natal den sites to document wolverine presence.

Goal - Goshawk Habitat

Provide suitable habitat conditions for known active and historic goshawk nesting territories.

Standard and Guideline - Goshawk Habitat

Management standards and guidelines for all forest types within active and historic goshawk nesting territories follow:

Attribute	Nest Area	Post-Fledging Family Area	Foraging Area
Number of areas (S)	1	1	1
Size of each area (acres) (S)	>= 200 acres	>= 400	>= 5,400
Size-Class Distribution for forested acres (%): (G)			
nonstocked/seedling	0	<= 20	<= 20
sapling	0	<= 20	<= 20
pole	0	<= 20	<= 20
mature/old growth 1/ 2160ac	100	>= 40	>= 40
Rotation age (years) (G)	--	60 to 240	60 to 240
Maximum created opening (acres) (G)	0	<= 40	<= 40
Snags and Reserve Trees 2/ (G)	>= 60% unless specified higher in prescription	>= 60% unless specified higher in prescription	>= 60% unless specified higher in prescription
Downed logs (average/acre) (G)	Forestwide S&Gs	Forestwide S&Gs	Forestwide S&Gs
Management Season (S)	Oct-Feb	Oct-Feb	Year-long
Thinning (G)	Non-uniform 3/	Non-uniform	by silvicultural prescription
Open Road Density4/ (G)	No new system roads	No new system roads	<= Management Rx Density

1/ Mature and old growth canopy closure for nest sites and post-fledging family areas should range between 75-100 percent. (G)
2/ Refer to previous section on snag/cavity nesting habitat for explanation of biological potential.
3/ Maximize diversity of structure.
4/ Open roads in goshawk territories will be given priority for closure to meet management prescription road density standards. First priority will be to close roads in nest areas; second priority in post-fledging family areas; third priority in foraging areas. Where possible, open road density should be zero in the nest areas and the post-fledging family areas.

2160

Pole = 3-7, 8" DF

Standard and Guideline - Flammulated Owl Habitat

Do not allow timber or firewood harvest activities within a 30-acre area around all known flammulated owl active and historic nest sites. (S)

Standards and Guidelines - Boreal Owl Habitat

1. Do not allow timber or firewood harvest activities within a 30-acre area around all known boreal owl active and historic nest sites. (S)

2. Maintain over 40 percent of the forested acres in late seral age classes within a 3,600-acre area around all known boreal owl nest sites. (G)

Standards and Guidelines - Great Gray Owl Habitat

1. Do not allow timber or firewood harvest activities within a 20-acre area around all known great gray owl active and historic nest sites. Vegetation manipulation does not include tree planting. (S)

2. Maintain over 40 percent of the forested acres in late seral age classes within a 1,600-acre area around all known great gray owl nest sites. (S)

3. Restrict the use of strychnine poison to control pocket gophers within a 1/2-mile buffer around all known active great gray owl nest sites. (G)

Goals - Trumpeter Swan Habitat

1. Maintain habitat to support ten breeding pairs or more on the Forest.

2. Protect emergent vegetation along shorelines. Maintain riparian vegetation in desired vegetative condition.

Standards and Guidelines - Trumpeter Swan Habitat

1. Maintain suitable trumpeter swan nesting habitat conditions including (but not limited to) the following lakes and ponds: Boundary Pond, Swan Lake, Lily Pond, Hatchery Butte, Railroad Pond, Mesa Marsh, Bear Lake, Upper Goose Lake, Long Meadows, Thompson Hole, Twin Lakes, Chain Lakes, Widgit Lake, Rock Lake, Indian Lake, Putney Meadows, Unnamed Pond (Sec. 19, T9N, R46E). (S)

2. Change livestock grazing through management or fencing when grazing is adversely affecting trumpeter swan use or productivity. (G)

3. No vegetation management will occur within 300 feet of the lake or pond shoreline unless necessary to improve riparian habitat conditions favorable for trumpeter swans. Management may occur after the swans have left the lake or pond. (S)

4. Maintain constant water levels; allow no drawdowns from May 1 to September 30 when not in conflict with preexisting water rights. (G)

5. Do not take any recreation management actions that would encourage dispersed recreation activity at these lakes and ponds. Close these areas to recreation activity if this activity is adversely affecting trumpeter swan use or productivity. (G)

6. Implement habitat improvement projects at these lakes and ponds, such as dredging to maintain proper water depths and aquatic vegetation control. (G)

Goal - Spotted Frog Habitat

Maintain riparian vegetation in desired vegetation condition.

Goals - Common Loon Habitat

1. Evaluate the potential to provide and maintain suitable breeding habitat for common loons at these sites: Indian Lake, Thompson Hole, Bergman Reservoir, Junco lake, Fish Lake, Loon Lake, Moose Lake, unnamed pond (Sec. 9, T47N, R118W).

PA 37,320 - 2
Wild = 12,000 - 28
25,320 = PA Addl.
39.6 sq mi.

USDA

United States
Department of
Agriculture

Forest Service

April 2018

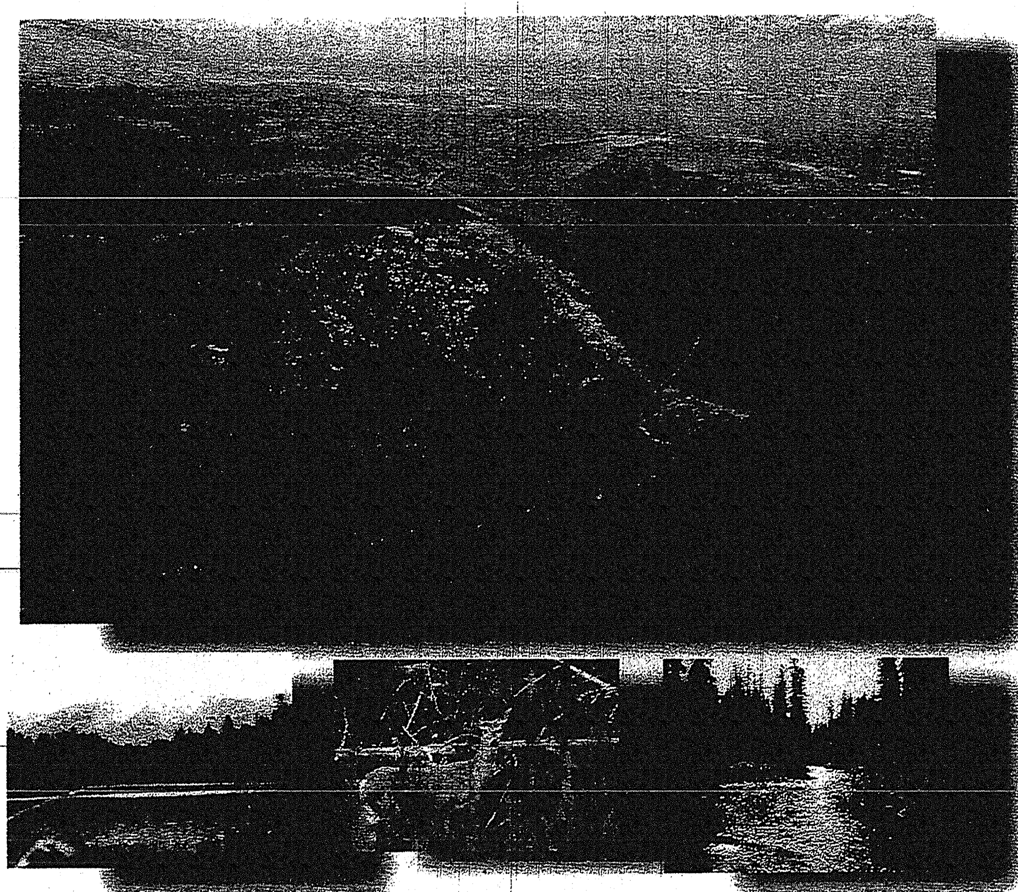
Glacier Loon Fuels Reduction and Forest Health Project

Draft Decision Notice

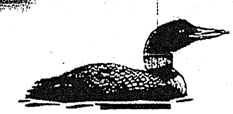
timeline
- 5 yrs after log - 30

Swan Lake Ranger District, Flathead National Forest
Missoula County, Montana

timeline 8 yrs, 5-18



monitor 0-29



OLD GROWTH ASSOCIATED WILDLIFE

INTRODUCTION

Old growth is defined in Amendment 21 of the Forest Plan as "a community of forest vegetation that has reached a late stage of plant succession." The generic description is as follows:

- The age of the dominant cohort of trees is significantly older than the average time interval between natural disturbances (interval will vary depending upon forest cover type and habitat type);
- Forest composition and structure are different from younger stands;
- Rates of change in composition and structure of the stand are slow relative to younger forests;
- There is a significant showing of decadence (wide range of defect and breakage in both live and dead trees).

In The Dictionary of Forestry (Helms 1998), old growth forests are described as having:

- Large trees for the species and site;
- Accumulations of large dead standing and fallen trees;
- Decay or breakage of tree tops, boles, or roots;
- Multiple canopy layers;
- A wide variation in tree size and spacing; and
- Canopy gaps and understory patchiness.

The characteristics of old growth forest described above provide habitat for many plant and animal species. Old growth forests are an important component of biological diversity. For the purpose of this discussion, old growth associated species includes any wildlife species that use the various attributes of old growth forests for some or all of their ecological needs. These needs could include nesting, denning, security, or foraging habitat. For some species, closed canopy old growth provides snow capture and reduces snow depths, insulates the animals from cold winds, and provides protection from predators. Some species, such as the fisher, are strongly tied to canopy cover and mature forest structure for the majority of their habitat needs. More open canopies, or open understories, provide foraging opportunities for prey and predator species alike. Wildlife may use interior old growth habitat as shelter from sun, heat, dryness, or wind and old growth cover may provide protection from predators. Some old growth associated wildlife species need only a portion of their home range to be in old growth; examples include the Canada lynx, northern goshawk, and American marten. Other species such as southern red-backed voles, chestnut-backed chickadee, Swainson's thrush, and northern flying squirrels, have relatively small home range sizes (less than 100 acres), with the necessary proportion of this home range being in old growth unknown.

The following table displays 31 old growth associated species that may be found in the Swan Valley, along with their associations with various old growth habitat characteristics (USDA 1999b).

TABLE 3-87 HABITAT REQUIREMENTS OF OLD GROWTH ASSOCIATED WILDLIFE SPECIES. (BASED ON WARREN 1998 AND LRMP AMENDMENT 21 FEIS).

SPECIES	COVER TYPE IN AFFECTED AREA	CANOPY	EDGE	LARGER PATCHES	SNAG	DOWN LOG	OCCURRENCE
American Marten	Mixed mesic, lodgepole, spruce/fir forests	Closed	-	+	X	X	Known current
Bald Eagle (S)	Mixed mesic forests, near large lake or river	Open		+	X		Known current
Black-backed Woodpecker (S)	Lower Montane & Montane; post-fire or insect-epidemic forests	Open			X		Known current
Boreal Owl	Mixed mesic and spruce/fir forest mosaic	Closed			X	X	Known current
Brown Creeper	Mixed mesic, lodgepole, and spruce/fir forests	Closed	-		X		Known current
Canada Lynx (T1)	Mixed mesic, lodgepole, and spruce/fir forests; gentle terrain		+2	+	X3	X	Known current
Chestnut-Backed Chickadee	Mixed mesic and spruce/fir forests, especially cedar-hemlock	Closed	-4		X		Known current
Fisher (S5)	Mixed mesic and lodgepole forests	Closed				X	Known current
Flammulated Owl (S, F6)	Lower Montane and Montane, single-story	Open			X		Known current
Golden-crowned Kinglet	Mixed mesic, lodgepole, and spruce/fir forests	Closed		+	X		Known current
Hairy Woodpecker	Mixed mesic, lodgepole, and spruce/fir forests	Open			X	X	Known current
Hammond's Flycatcher (F)	Mixed mesic and spruce/fir forests	Closed					Known current
Harlequin Duck (S)	Swift mountain streams, riparian old growth (weak association)	Open				X	Known current
Hermit Thrush	Dry mixed mesic and spruce/fir forests	Open		+			Known current
Lewis' Woodpecker	Lower Montane ponderosa pine and old burns	Open			X		Known current
Northern Flying Squirrel	Mixed mesic and lodgepole forests			+	X	X	Known current
Northern Goshawk	Single or multistory old growth; clear forest floor	Closed		+	X		Known current
Pileated Woodpecker	Mixed mesic forests	Closed		+	X	X	Known current
Pine Grosbeak	Mixed mesic, lodgepole, and spruce/fir forests						Known current
Pygmy Nuthatch	Large single-story ponderosa pine and mixed mesic forests	Open			X		Known current
Red-Breasted Nuthatch	Mixed mesic, lodgepole, and spruce/fir; relatively dry	Open		+	X		Known current
Silver-haired Bat	Mixed mesic and lodgepole forests; caves and snags				X		Suspected
Southern Red-backed Vole	Mixed mesic, lodgepole, and spruce-fir forest				X	X	Known current
Swainson's Thrush (F)	Mixed mesic and lodgepole forest with shrub understory			+			Known current
Tailed Frog	Cold, high gradient headwater streams					X	Known current

Table 3- 87 Habitat Requirements of Old Growth Associated Wildlife Species. (Based on Warren 1998 and LRMP Amendment 21 FEIS).

Species	Cover Type in Affected Area	Canopy	Edge	Larger Patches	Snag	Down Log	Occurrence
Three-toed Woodpecker	Mixed mesic, lodgepole, and spruce/fir forests; post-fire				X		Known current
Townsend's Warbler	Mixed mesic and lodgepole forest; dense understory	Closed	-	+			Known current
Varied Thrush	Mixed mesic and spruce/fir forests, especially cedar-hemlock	Closed		+			Known current
Vaux's Swift (F)	Mixed mesic and spruce/fir forests; large hollow snags				X		Known current
White-breasted Nuthatch	Large single-story ponderosa pine	Open			X		Known current
Winter Wren	Mixed mesic and spruce/fir forests, especially cedar-hemlock		-	+	X		Known current

[1] T = Threatened
[2] + = Positive correlation (where known)
[3] X = Important Habitat Component
[4] - = Negative correlation (where known)
[5] S = Sensitive
[6] F = Forest-dwelling Neotropical migrant with apparently declining populations

Analysis Area

Spatial Bounds

The effects analysis area for direct, indirect, and cumulative effects to old growth associated wildlife species is the Glacier Loon Project Area (37,320 acres). This area is large enough to include the home ranges of old growth associated species, and is representative of the effects of fire, natural tree mortality, timber harvest, and road management across the landscape. At the same time, this analysis area is small enough to not obscure the effects of the alternatives. A multi-scale assessment has also been conducted to address habitat diversity concerns.

Temporal Bounds

The length of time for effects from the proposed fuels reduction and forest health treatments is approximately 1 to 5 years. This is based on the probable contract length for the proposed project, and the timeframes for related activities.

Data Sources, Methods, and Assumptions Used

Data used included stand exams, field surveys of snags and down woody logs, old growth surveys, project area field visits, research literature, and GIS and dataset information for features, such as general forest attributes, habitat type, and forest type.

Measurement Indicators

The effects analysis will focus on:

1. Effects to old growth habitat, and
2. Potential effects to old growth associated wildlife species.

April 2018

SNAG AND DOWN WOODY ASSOCIATED SPECIES

INTRODUCTION

Snags, broken-topped live trees, down logs, and other woody material are required by a wide variety of species for nesting, denning, roosting, perching, feeding, and cover (Bull et al. 1997). Snags and down, dead, material are also used for communication purposes:

- Singing, (songbirds),
- Drumming (grouse and woodpeckers),
- Calling (squirrels, jays, birds of prey), and
- Sight recognition posts.

Small mammals and birds use standing and down dead material for food storage and for hunting. Down logs and stumps are important for travel, both below the snow in the winter, and as travel cover throughout the year. It is estimated that about one third of the bird and one third of the mammal species that live in the forests of the Rocky Mountains use snags for nesting or denning, foraging, roosting, cover, communication, or perching. On the Flathead National Forest, at least 42 species of birds and 10 species of mammals are dependent on dead wood habitat for nesting, feeding, or shelter (USDA 1999b). The more mobile species that depend on dead wood habitat include black bears, Canada lynx, wolverines, marten, fisher, bats, woodpeckers, and small owls. Less mobile species that depend on dead wood include snowshoe hares (the primary prey of Canada lynx), red-backed voles (the primary prey of marten, fisher, boreal owl, and several other species), shrews, bryophytes, lichen, fungi, and protozoa. As down woody material further decays, it plays an important role in nutrient cycling, soil fertility, and erosion control.

Snags and their management have become a major conservation issue in managed forests across the western United States. Biologists have recognized for a long time that snags and down woody material provide important wildlife habitat, but only in the last decade or so have managers begun to understand that not only is tree decay an important ecological process that affects wildlife habitat (Bull et al. 1997), but snags and dead wood are an essential, important part of the larger ecosystem. The number, species, size, and distribution of available snags strongly affect snag-dependent wildlife. An insufficient number of suitable snags may limit or eliminate populations of cavity-using species (Saab et al. 1998; Thomas et al. 1979).

Although various sizes of snags and down woody are used, larger birds and mammals require larger dead trees. The larger-diameter down trees provide stable and lasting structure and offer better protection from weather extremes (Bull 2002). Longer down woody pieces provide better runways, shelter, and under-snow access.

Several wildlife species that use snag and down woody habitats on the Flathead National Forest are USFS Region One Sensitive Species, including the bald eagle, black-backed woodpecker, fisher, flammulated owl, Townsend's big-eared bat, and wolverine. One of the TES Species on the Flathead National Forest, the Canada lynx, has a strong habitat association with down woody material (denning).

Snags are essential habitat for at least 42 species of birds and 10 species of mammals in Montana. Table 3- 88 displays specific habitat relationships and Montana NHP rankings for wildlife species in Montana associated with snag, "defective" live tree, or down woody habitat.

TABLE 3- 89 SPECIES THAT USE SNAGS, "DEFECTIVE" LIVE TREES, AND/OR DOWNED LOGS.

SPECIES	GLOBAL & STATE RANKS (MTNHP 2009) *	SNAG DBH (INCHES)	SNAG HEIGHT (FEET)	DOWNED LOGS?	OCCURRENCE
American Kestrel (N)	G5, S5B	17	20		Known current
Bald Eagle (S)	G5, S3, SOC	25	40		Known current
Barred Owl (former MIS)	G5, S4	25	30		Known current
Barrow's Goldeneye	G5, S4, potential SOC	25	10		Known current
Big Brown Bat	G5, S4	17	20		Known current
Black-backed Woodpecker (S)	G5,S3 (SOC)	17	10		Known current
Black-capped Chickadee	G5, S5	9	10		Known current
Bobcat	G5, S5	-	-	yes	Known current
Boreal Chickadee	G5,S3 (SOC)	9	10		Known current
Boreal Owl (former S)	G5, S4	17	10		Known current
Brown Creeper	G5, S3, SOC	15	20		Known current
Bufflehead	G5, S5B	17	10		Known current
Canada lynx (T)	G5, S3, SOC	-	-	yes	Known current
Chestnut-backed Chickadee	G5, S4	9	10		Known current
Common Goldeneye	G5, S5	25	10		Known current
Common Merganser	G5, S5B	17	10		Known current
Dark-eyed junco	G5, S5B	-	-	yes	Known current
Downy Woodpecker	G5, S5	11	10		Known current
Fisher (S)	G5, S3, SOC	25	30	yes	Known current
Flammulated Owl (S, N)	G4, S3B, SOC	17	10		Known current
Great Horned Owl	G5, S5	25	30		Known current
Hairy Woodpecker	G5, S5	17	20		Known current
Harlequin Duck (S)	G4, S2B, SOC	-	-	yes	Known current
Hooded Merganser	G5, S4, potential SOC	17	10		Known current
House Finch	G5, S5	15	10		Known current
House Sparrow	G5, undesired species	15	20		Known current
House Wren (N)	G5, S5B	15	10		Known current
Lewis' Woodpecker	G4, S2B (SOC)	17	30		Known current
~~Little Brown Myotis~~	~~G5, S4~~	~~17~~	~~10~~		Known current
Long-eared Myotis	G5, S4	17	10		Known current
Long-legged Myotis	G5, S4	17	10		Known current
Long-tailed Weasel	G5, S5	-	-	yes	Known current
Marten (former MIS)	G5, S4	17	20	yes	Known current
Mountain Bluebird	G5, S5B	15	10		Known current
Mountain Chickadee	G5, S5	9	10	yes	Known current
Northern Alligator Lizard	G5,S3 (SOC)	-	-	yes	Known current
Northern Flicker	G5, S5	17	10		Known current
Northern Flying Squirrel	G5, S4	17	20		Known current
Northern Goshawk (former S)	G5,S3 (SOC)	-	-	yes	Known current
Northern Hawk Owl	G5, S4, potential SOC	25	10		Known current
Northern River Otter	G5, S4	-	-	yes	Known current
Northern Waterthrush (N)	G5, S5B	-	-	yes	Known current
Osprey	G5, S5B	17	40		Known current

TABLE 3- 89 SPECIES THAT USE SNAGS, "DEFECTIVE" LIVE TREES, AND/OR DOWNED LOGS.

SPECIES	GLOBAL & STATE RANKS (MTNHP 2009) *	SNAG DBH (INCHES)	SNAG HEIGHT (FEET)	DOWNED LOGS?	OCCURRENCE
Painted Turtle	G5, S4	-	-	yes	Known current
Pileated Woodpecker (former MIS)	G5,S3 (SOC)	25	60		Known current
Pygmy Nuthatch	G5, S4	17	30		Known current
Pygmy Owl	G5, S4	17	30		Known current
Raccoon	G5, S5	25	10		Known current
Red-breasted Nuthatch	G5, S5	17	20		Known current
Red-naped Sapsucker (N)	G5, S4B	17	20		Known current
Rubber Boa	G5, S4	-	-	yes	Known current
Ruffed Grouse	G5, S4	-	-	yes	Known current
Saw-whet Owl	G5, S4	17	20		Known current
Silver-haired Bat	G5, S4, potential SOC	17	20		Known current
Southern Red-backed Vole	G5, S4	-	-	yes	Known current
Spruce Grouse	G5, S4	-	-	yes	Known current
Striped Skunk	G5, S5	-	-	yes	Known current
Swainson's Thrush (N)	G5, S5B	-	-	yes	Known current
Tailed Frog	G5, S4	-	-	yes	Known current
Three-toed Woodpecker	G5, S4	17	20		Known current
Tree Swallow (N)	G5, S5B	15	20		Known current
Vaux's Swift (N)	G5, S4B	25	40		Known current
Violet-Green Swallow	G5, S5B	15	20		Known current
Western Bluebird	G5, S4B	15	10		Known current
Western Jumping Mouse	G5, S4	-	-	yes	Known current
Western Screech Owl	G5, S3, potential SOC	17	20		Known current
Western (Townsend's) Big-eared Bat (S)	G4, S2 (SOC)	?	?		Known current
White-breasted Nuthatch	G5, S4	17	20		Known current
Williamson's Sapsucker (N)	G5, S4B	17	20		Known current
Wilson's Warbler (N)	G5, S5B	-	-	yes	Known current
Wolverine (S)	G4, S3 (SOC)	-	-	yes	Known current
Wood Duck	G5, S5B	25	10		Known current
Yuma Myotis	G5, S3, potential SOC	17	10		Known current

T=Threatened; S=Sensitive Species; N-Neotropical migratory bird; Natural Heritage Program Rank: G=species range-wide (global); S=state wide; 2=At risk because of very limited and/or declining numbers, range, and/or habitat, making it vulnerable to global extinction or extirpation in the state. 3=Potentially at risk because of limited and/or declining numbers, range, and/or habitat, even though it may be abundant in some areas. 4=Uncommon but not rare (although it may be rare in parts of its range), and usually widespread. Apparently not vulnerable in most of its range, but possibly cause for long-term concern. 5=Common, widespread, and abundant (although it may be rare in parts of its range). Not vulnerable in most of its range. B=State rank modifier indicating breeding for a migratory species. SOC=Montana Species of Concern.

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ANALYSIS AREA

SPATIAL BOUNDS

The Glacier Loon Project Area was considered for the evaluation of direct and indirect effects on snag and down woody associated species. This approximately 37,320-acre area is large enough to include the home ranges of several individuals or pairs of a species, and is representative of the effects of fire, natural tree mortality, timber harvest, and road management across the landscape. The actions proposed in the alternatives that could directly or indirectly affect snag or down woody associated wildlife species are contained within this area. The Upper Swan Valley was considered in the cumulative effects analysis. A multi-scale assessment was also conducted to address habitat diversity concerns for dead tree dependent species (USDA 2006).

TEMPORAL BOUNDS

The length of time for effects from the proposed fuels and forest health treatments is approximately 1 to 5 years. This is based on the probable contract length for the proposed project, and the timeframes for related activities.

DATA SOURCES, METHODS, AND ASSUMPTIONS USED

Data used included project area field visits, research literature, and GIS and dataset information for features, such as general forest attributes, habitat type, and forest type.

MEASUREMENT INDICATORS

The effects analysis will focus on:

1. Effects to snag and down woody habitat, and
2. Potential effects to snag/down woody associated wildlife species.

AFFECTED ENVIRONMENT

HISTORICAL CONDITION

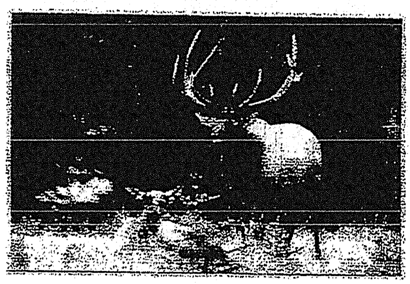
Forest ecosystems in the western United States have adapted in response to disturbances such as wildfire, insects, disease, and windstorms. Snags and down woody material have always occurred on the landscape, a direct result of these disturbance factors, either on a large scale, or on a very small scale, as individual trees grow old and die. Ritter and others have described snag populations as occurring in either "pulses" of snags following a large disturbance event, or as "continuous" populations of scattered individuals (Ritter et al. 2000).

Historically, in the Swan Valley, snag habitat and down woody material, though always present in varying amounts, experienced greater "pulses" across the landscape and in localized areas as a result of natural disturbances. Warmer and drier areas historically underwent more frequent, lower-intensity fires, and typically supported fewer snags and large down logs than cooler and moister environments, where the stands reached climax conditions before experiencing stand-replacing fire.

EXISTING CONDITION

The Northern Region of the Forest Service estimated snag densities for Western Montana Forests by

September 27, 2013



U.S. Forest Service and Montana Department of Fish Wildlife and Parks Collaborative Overview and Recommendations for Elk Habitat Management on the Custer, Gallatin, Helena, and Lewis and Clark National Forests

Abstract:

A group of wildlife biologists from the Forest Service (FS) and Montana Department Fish, Wildlife, and Parks (MDFWP) have compiled recommendations, along with a discussion of their conversations and the relevant literature, for elk habitat management. These are recommendations only, and the group is not making decisions or setting policy for either agency in this effort. While we focus on elk habitat considerations in this effort, we do not advocate for single species management. We advocate for ecologically appropriate habitat management under an umbrella of landscape scale ecosystem management, which focuses on providing a range of habitats to support all fauna native to the landscape, including elk. These recommendations apply only on the Custer, Helena, Lewis and Clark, and Gallatin National Forests.

The recommendations are based on the most current available information and the collective experiences of these biologists. They considered contemporary issues and circumstances such as increases in recreation of all types on these National Forests, changes in the numbers and distribution of elk (including the use of private lands where hunting is limited or not allowed), the restoration of large predators, the current mountain pine beetle epidemic, and small and large fires on the Custer, Helena, Lewis and Clark, and Gallatin National Forests in the Northern Region of the Forest Service. The shared goal of the two agencies is to provide for elk and other big game on National Forest System (NFS) lands throughout the year, recognizing that overall agency missions differ. That is, within the multiple use mandate of the Forest Service, management for elk will be one of many considerations on NFS lands. The overview and recommendations address an appropriate elk analysis unit, management of cover and recreation on winter ranges, security during the archery and rifle hunting seasons, motorized route management relative to habitat effectiveness, cover on spring-summer-fall ranges, cover patch size, forage, calving areas, and migration corridors. The recommendations conclude with a short discussion of considerations for bighorn sheep.

requirements for a particular species at the appropriate geographic scale does not result in the species being present for reasons outside of the FS control (e.g. not habitat related).

The species specific component (fine filter) of this approach focuses on the critical habitat elements and limiting factors for a particular species, and may be focused more on human related disturbances rather than habitat. An example of a fine filter consideration is limiting human disturbances on big game winter range to prevent displacement or harassment.

With an effective ecosystem/coarse filter component in place, the more costly and information intensive species specific/fine filter approach can be focused on the few species whose requirements and circumstances may not be fully addressed by the ecosystem/coarse filter approach.

Habitat varies in space and time given the dynamic nature of ecosystems. This may be referred to as natural range of variation (NRV); FS 2012 Planning Rule (USDA-FS 2013). NRV is defined as the spatial and temporal variation in ecosystem characteristics under historic disturbance regimes during a reference period. The reference period considered should be sufficiently long to include the full range of variation produced by dominant natural disturbance regimes, often several centuries, for such disturbances as fire and flooding and should also include short-term variation and cycles in climate. Natural range of variation (NRV) is a term used synonymously with historic range of variation or range of natural variation. The NRV is a tool for assessing ecological integrity, and does not necessarily constitute a management target or desired condition. The NRV can help identify key structural, functional, compositional, and connectivity characteristics, for which plan components may be important for either maintenance or restoration of such ecological condition. Addressing departures from NRV (e.g. conducting treatments to increase aspen habitat if aspen habitat is underrepresented on the landscape), generally contributes to providing habitat conditions across a landscape that are representative of those which produced and sustained the fauna native to that landscape. Native species (including elk) evolved with, adapted to, and contributed to, natural disturbance patterns and the resulting mosaic of habitat conditions. The diverse needs of all species across a landscape require the consideration of habitat requirements over broad geographic areas and extended time periods.

Habitat restoration opportunities may exist where conditions have departed from those that provided for the species that are native and adapted to that landscape. Restoring the range of ecological conditions that sustained native species in the past will best conserve biological diversity and will maintain habitat for a broad range of species.

3. *Winter Range Recommendation*

Human disturbance on winter ranges should be minimized to prevent displacement or harassment of elk on winter range. In addition, the amount and types of available cover and forage should accommodate the needs of elk (within the bounds described in the ecological context section above) and take into consideration the limiting factor for elk on winter range.

At the project level, the various canopy cover classes of conifer cover on winter range will need to be examined relative to their potential benefit to elk.

Discussion:

The desired condition for winter range is to maintain elk use of NFS lands during the winter and early spring (approximately December 1 to May 15, but should be flexible to account for site specific conditions), in keeping with MDFWP 2005 Elk Plan and FS Forest Plan direction for multiple resources.

The agency participants agreed that winter range would either be updated at the project level by MDFWP for an elk herd or default to the delineations currently mapped by MDFWP and available on their website. MDFWP currently does not delineate or map 'critical/crucial' winter range separate from other winter range. It was recognized that in many areas very little winter range is found on NFS lands due to their relatively higher elevations and accompanying higher snow accumulations. Given that consideration, where NFS lands do provide winter habitat for elk, the agency participants defined a desired condition of providing maximum protection, by maintaining adequate forage and cover and by restricting travel for the objective of maximizing effective use of winter ranges on NFS lands by elk. Consistent with this idea, (but not generally considered by the FS in travel management), it was recognized that in a given local situation, special area closures may be warranted for motorized and non-motorized uses (precluding those activities such as snowshoeing or skiing), or at least a consideration of prohibiting cross-country or dispersed travel on winter range by designating travel corridors for both motorized and non-motorized recreation.

Both motorized and non-motorized routes should be examined for the potential to disturb and/or displace elk. Elk may be able to habituate to predictable, limited human travel when it is limited to a designated route (with associated area closures) through winter range, although there may still be physiological effects (Canfield et al. 1999; Cassier and Ables 1990). The location of routes should be examined as well. Relocation of routes may reduce impacts on elk, for example, by relocating an over-the snow motorized route away from key areas used by elk in severe winter conditions.

The literature suggests that forested cover on winter range may have multiple functions including snow interception, thermal modulation, wind buffering, and to provide areas that "hide" animals and provide security in the face of disturbance. The first two functions are provided by relatively mature forest stands with relatively high canopy cover (40-100%) (Carter 1984, Eon 2004). Thomas et al. (1979) suggested that thermal cover is coniferous trees at least 40' in height and 70% crown closure, but these numbers have not been validated experimentally (Skovlin et al. 2002).

Lyon et al. (1985) reported that east of the Continental Divide elk forage in grasslands during the winter, but seek cover in adjacent timber stands. The research summarized that logging in these forested stands will normally be detrimental to elk. West of the Continental Divide, Thompson et al. (2005) documented the importance of forage availability in the coniferous stands when snow conditions in the grasslands become crusty. The research recommended maintaining connected patches of denser canopies for snow interception, and to maintain some more open canopy conditions that result in "forested forage". Skovlin et al. (2002) also report that different stand conditions provide varying degrees of thermal protection and cover from snow. Mysterud and Ostbye (1999), in their

NC + Cover

discussion of all the important functions of cover for ungulates, distinguish between canopy cover, which shelters an animal from above, and ground cover, which is what hides an animal viewed from ground positions, which are both potentially important functions of cover on winter range. All that being said, elk populations can prosper in areas lacking a coniferous forest structure (Skovlin et al. 2002). In those situations, elk may use topography and aspect to regulate temperature (Beall 1974).

Cook et al. (1998) concluded that elk benefited more from forage than from cover in terms of overall fat metabolism through the winter. They concluded that thermal cover may be important under certain conditions, but its value is relational to other habitat attributes that contribute to the productivity of elk herds. The agency participants had an opportunity to discuss this research with staff from the Starkey Research Station. We concluded that the conditions under which the study was conducted, including the climate of the study area and the use of penned and fed elk, may not be applicable to winter conditions for free ranging wild elk on the four forests addressed in this document. — thermal

The agency participants ultimately agreed that elk may use cover in the winter for a variety of reasons that may include thermoregulation, but that forest cover may also be important in keeping forage available to elk in some situations (crusty, icy conditions, or deep snow for example), or to buffer elk from potential disturbances on winter ranges. MDFWP representatives pointed out that from an energy expenditure reduction standpoint, big trees and multi-layer canopies may provide benefits not provided by small trees or single layer forest canopies. The participants concluded that coniferous cover should generally be maintained on elk winter ranges within the capability of the landscape (see ecological context section). The agency participants also discussed the time it takes for conifer cover to become multistoried or dense-canopied, which is the kind of structure that provides snow interception following regeneration of forest cover (e.g. burning or timber harvesting) on winter range. Although the screening function may be recovered in about 15-20 years, the structural conditions that provide for snow interception may not recover for 60 years or more. Recovery — TC

Fire suppression for the last several decades has changed the way fires burn. Fire exclusion has resulted in a more homogeneous landscape with an increased potential for larger stand-replacing fires (Arno and Bunnell 2002, Hessberg et al. 1999, Quigley et al. 1996). This is especially applicable in warm, dry low-elevation forests (the types of forest that provide elk winter range) where frequent low-intensity fires were the norm. Many of these forests are now unnaturally dense. Furthermore, as more homes are built in the wildland urban interface, there is a greater social demand for fire suppression at the same time that the risk of fire has increased. While maintaining multi-layered canopies may be beneficial for elk on winter range, the group recognized that treatment of hazardous fuels in areas where winter range overlaps with a wildland urban interface may trump the needs of elk. — outdate

After the 2000 wildfire season, which burned a substantial amount of NFS lands (as well as other lands), the Forest Service re-examined wildland fire policies. As a result, several documents were developed that set goals for wildland fire policies. Congress subsequently passed the Healthy Forests Restoration Act in 2003 which further emphasized the need to manage hazardous fuels. In conclusion, forested cover is an important consideration on elk winter range; however, the national attention and emphasis on reducing hazardous fuels to protect communities, watersheds, and other at-risk values may result in management activities that affect winter cover. This effect could be mitigated if ungulate cover needs are considered in the design of hazardous fuels projects. The agency participants

WR mngt!

unnaturally dense!

criteria

motorized use during the archery season. MDFWP's elk security concerns depending upon the individual situation relate to: 1) bull elk survival during both the archery and rifle season in some cases; 2) elk distribution during the hunting season (i.e. are elk being pushed into marginal habitats that may negatively influence survival and reproduction); and 3) availability of elk, particularly antlerless elk, on NFS land (i.e. potential displacement to private land or checker-boarded areas of NFS ownership where access to NFS land is limited or unavailable, or into only generally inaccessible areas on NFS land) during the fall hunting season. The last concern is particularly important prior to the opening of the general rifle season when the overwhelming majority of antlerless elk harvest for population control occurs.

As such, fall elk security levels should be assessed either just using the September 1 starting date or both the September 1 and October 15 hunting season related starting dates, rather than just running an analysis using October 15 as the starting date, as has often just been done in the past.

The concept of security areas is embodied by the "Hillis paradigm", a paper compiled by Hillis et al. in 1991 as part of an elk vulnerability symposium proceeding. Hillis et al. recommended identifying security areas within the hunting season home range. In practice on these four National Forests, elk have the potential, depending on weather and other conditions, to use the entire breadth of elevations within their home range during the big game archery and general rifle hunting seasons. Therefore, it is not necessary or possible to identify a consistently "separate" fall use area within an EAU.

Hillis et al. recommended identifying areas at least ½ mile from an open motorized route and at least 250 acres in size as the minimums for providing adequate security. The authors cautioned that in some cases, distance from open routes and the size of security area blocks must be increased (e.g. 350 acres >=1 mile from an open motorized route) to provide adequate security. Hillis et al. recommended that at least 30% of a valid analysis unit be comprised of security areas.

Hillis 'cover'

Although Hillis et al. (1991) define security as "non-linear blocks of hiding cover", they also suggest that effective security areas may consist of several different cover –types if the block is relatively un-fragmented. Relative to the composition of the security areas, there was discussion about whether Hillis et al. (western Montana) was applicable to these four National Forests.

The agency participants spent a fair amount of time discussing the definition, distribution and vegetative composition of security areas. MDFWP biologists felt that security areas are not meaningful to elk, if by virtue of the physical conditions of the landscape (rocks, cliffs, steep areas) elk would not be expected to use them to meet their biological needs. To be most effective, security areas should be well-distributed cross the breadth (geographic and elevations) of elk habitat within the EAU. This is especially relevant where a portion of the home range is relatively inaccessible such as represented by sanctioned wilderness areas.

The studies considered by Hillis et al. were done in areas where forests of various ages were continuous. In their discussion of security areas, Christensen et al. (1993) speak to the significance of cover in this equation and note that where cover is ubiquitous, security can be controlled by road management alone. They recommend that detailed analyses of hiding and thermal habitat components are not essential relative to security areas and that a consideration of coniferous cover is important for more than just elk vulnerability and therefore should be assessed at a landscape level to develop long-term perspectives to address the condition, quantity, location, and configuration of cover (see ecological context section and spring-summer-fall cover section).

In contrast to the Hillis et al. study areas, the landscapes on these four National Forests include many areas of more open habitat or where forests and grasslands are interspersed in a mosaic. A break out group of agency participants discussed the cover and security area relationship and concluded that consideration of the quantity and quality of forested cover at the scale of the non-winter portion of the elk analysis unit would be better than defining security areas as "blocks of hiding cover". This also allows for the situation where areas naturally include a mosaic of forest and/or open habitats, but which operationally are secure. In addition, recent analyses of elk habitat selection during the hunting season in Montana (Proffitt et al. 2012 and personal communication with Kelly Proffitt, MDFWP) did not show a significant selection for security areas comprised of total conifer cover relative to security areas just defined by size and distance from a road. In addition, Proffitt's analysis showed that security areas as a variable in habitat selection during the hunting season are strongly related to the motorized route variable.

Hillis et al. only speak to "open roads" and "closed roads". They suggest that hunting pressure is concentrated along open roads, but that (hypothetically) closed roads located within security areas may increase elk vulnerability by providing walking and shooting lanes. Unsworth and Kuck (1991) note that road closures may have varied effects on animal distribution and hunter use and success. They cite to several studies where road closures allowed elk to remain in more preferred sites for longer periods of time (Irwin and Peek 1979, Marcum 1975). Basile and Lonner (1979) reported that when vehicular travel was restricted, hunters spent more time walking, saw more elk, and had greater success and reported having a higher quality hunting experience. Based on these studies and the recent review from McCorquodale (2013) on elk and roads, the agency participants agreed that Hillis et al. recommendation to "minimize" closed roads within security areas was not necessary.

The agency participants noted that there has been a dramatic increase in all-terrain type vehicles, and therefore agreed that all FS system routes that receive any kind of motorized use by the public should be considered for their impact to elk not just open 'roads'. A review of the scientific literature regarding elk, roads and traffic (McCorquodale 2013) provides strong evidence that elk use declines as traffic volume increases (Johnson et al. 2000, Edge and Marcum 1991, Rumble et al. 2005, Stubblefield et al. 2006, Montgomery et al. 2013). Johnson et al. found that elk avoided roads that had 2-4 vehicles per 12 hours or higher.

Given this threshold for avoidance, agency participants agreed that closed routes or low intensity, occasional private or administrative travel and management activity on routes closed to the public could be reasonably excluded in identifying security areas. However, consistent, frequently-used non-public routes or temporary roads would detract from security areas, if such roads are used during hunting season(s).

The agency participants recognized that there were a lot of different variables that must be considered when applying the parameters from the Hillis et al. paper, and as the paper suggests, the numerical recommended guidelines may not be sufficient in all cases and that "**strict adherence to the guidelines should be avoided**".

One of the driving factors in the security area discussion was to try to reduce or eliminate elk displacement from public land prior to normal migration events. The objective of maintaining or enhancing elk presence on NFS lands so that elk are available to the hunting public on public land should be considered in the determination of the specific parameters (block size, distance from motorized routes) used to identify security areas, in the percentage of the EAU dedicated to security areas, and in discussions involving hunting season related seasonal road closures. Additionally, it was recognized that elk may use habitat differently on private lands than on public lands. For example, the importance of security areas may be less on private land where hunting pressure and recreational use is lower, than on public land where hunting pressure and recreational use is generally higher. The agency participants recognized that increasing elk security alone on NFS land may not be enough to maintain elk use of NFS land.

The agency participants felt strongly that posted private land was not the desired way to provide security for elk that also use FS lands, as that situation results in lost opportunity for the general public and decreased or lost management effectiveness relative to population control. In addition, when elk are displaced to private land, it negates any intentional elk –related management decisions by the FS. The agency participants recognized that although it isn't desirable for posted private land to act as security for elk using public lands, elk may choose that form of security. There was discussion regarding whether or not elk would re-occupy public land after displacement to private lands where hunting is not allowed regardless of potential management actions by the FS and/or MDFWP. We concluded that increasing security and habitat quality are important and necessary considerations, but that some elk herds may, through learned behavior, continue to use private lands regardless of conditions on public lands. This would need to be addressed outside of habitat management and hunting seasons on NFS lands. For example, there may be access incentives available for private landowners to consider.

The agency participants recognized that given high enough levels of hunter numbers and hunting pressure, security areas may not be effective, resulting in elk being displaced to other areas including private land (Vales et al. 1991; Lyon and Christensen 2002). It may be determined through the course of discussions between the two agencies that non-habitat related management actions outside of USFS control (such as a hunting season related change) may be necessary to address elk distribution and displacement issues, if habitat management actions alone are insufficient, with the recognition that any hunting season related change would have to go through MDFWP's public Commission process. Given that both land use (i.e. travel plans) and hunting seasons can represent significant public values and expectations, it is recognized that changes to either can be procedurally difficult and contentious. It is also recognized that each has its own process and that elk distribution issues need to be responsibly addressed by both. Working together, both agencies would need to assess the relative management gains represented by different FS and MDFWP management options and not unilaterally rely upon the other's actions or dismiss the potential for making changes that would increase the potential to maintain elk on NFS lands.

The group acknowledged that the FS multiple use mandates may result in lower than desired elk security levels in some areas, particularly during the archery season. Where travel management decisions have already been made, and consideration was given to elk security in the overall decision, elk security analyses at the project level should (1) focus on any additional temporary motorized routes and their potential to reduce elk security and (2) include potential mitigation opportunities to enhance elk security including but not limited to date changes to existing open motorized routes. The potential to revisit existing travel plans in order to improve elk security where needed should remain an option.

5. Habitat Effectiveness Recommendation

Habitat effectiveness is defined as the percentage of available habitat that is usable by elk outside the hunting season (Lyon and Christensen 1992). For areas intended to benefit elk summer range and retain high elk use, habitat effectiveness related to motorized routes should be 70% or greater. For areas where elk are one of the primary resource considerations habitat effectiveness should be 50% or greater. In situations involving elk summer range where habitat effectiveness is less than 50% elk use potential (which equates to an open motorized route density of about 2 miles/square mile (Christensen et al.1993), these areas must be recognized as making only minor contributions to elk management goals (Christensen et al. 1993). We recognize that due to the USFS's multiple use mandate that there will be areas that are managed for habitat effectiveness of less than 50% elk use potential. At the project level an elk habitat effectiveness analysis should be conducted. In those areas where there is a desire to improve and/or maintain elk habitat use, we recommend maintaining or decreasing road densities that correspond with the desired habitat effectiveness level per Christensen et al. (1993).

Travel planning should also take into consideration the location of motorized routes and minimize motorized use in highly productive elk summer range habitats (see discussion).

For simplicity, route density is used as the proxy for habitat effectiveness. In reality, elk habitat effectiveness may be influenced by other factors than motorized routes. Other factors should be considered at the site-specific project level.

Discussion:

A desired condition expressed by the agency participants was to maintain elk use of National Forest land during the highly productive summer months (per Canfield et al. 1999). This recommendation focuses on elk distribution and is related to effective use of habitat for optimal fat accumulation on summer range (the time of year when forage conditions allow for weight gain needed to survive the winter and other stresses (such as the rut, and/or calf production and recruitment). Habitat effectiveness is defined as the percentage of available habitat that is usable by elk outside the hunting season (Lyon and Christensen 1992). This is the measure of success in meeting elk needs on summer range. Habitat effectiveness is directly related to motorized route densities (Christensen et al. 1993), but route locations may also be important. Recent studies (e.g. Wisdom et al. 2004; Proffitt et al. 2012) validate Lyon's (1983) finding that elk avoid open motorized routes and therefore, areas with high motorized route densities have the effect of reducing the available habitat. At 2 miles/section, elk use potential is about 50% (see **Appendix B**).

The agency participants acknowledged that not all FS lands would be managed with elk as the sole or primary consideration. The agency participants also acknowledged that travel management decisions often have to strike a balance between resource protection, public recreation opportunities, and other multiple use considerations. However, the potential to revisit existing travel plans in order to improve elk habitat effectiveness where there are concerns should be considered.

Where travel management decisions have been made, and consideration was given to habitat effectiveness in the overall decision, habitat effectiveness analysis at the project level should focus on the additional temporary project routes and their potential to reduce elk use of important summer habitat.

Although Christensen et al. (1993) suggest that any motorized vehicle use on roads will reduce habitat effectiveness, a review of the published literature provides strong evidence that elk use declines as traffic volume increases (Johnson et al. 2000, Edge and Marcum 1991, Rumble et al. 2005, Stubblefield et al. 2006, Montgomery et al. 2013). Johnson et al. found that elk avoided roads that had 2-4 vehicles per 12 hours or higher. Rowland and Wisdom (2012) also found that the strongest predictor of elk use in late summer was distance to open public roads. Therefore, low intensity, occasional administrative travel and management activity on routes closed to the public could be reasonably excluded in habitat effectiveness analyses. However, consistent, frequently-used non-public routes or temporary roads would detract from habitat effectiveness if such roads are used during the summer. The specific situation should be addressed at the project level and needs to be interpreted relative to potential for displacement of elk (see Lyon et al. 1985).

The group acknowledged that HE is an average condition, and that a range of habitat effectiveness can occur over a landscape. Some areas will have fairly homogenous route systems; other areas will have concentrations of open routes. In those situations, it is important to manage some areas with seasonal or yearlong restrictions such that route density is close to zero (100% habitat effectiveness).

The location of motorized routes may potentially negatively impact elk habitat as much as actual motorized route densities. Travel planning should consider minimizing or restricting motorized use on routes that impact high quality habitat, such as heads of drainages, saddles, moist areas, wet meadows, etc. (Lyon et al. 1985, page 7), and other important foraging habitats (e.g. aspen, riparian, grassland meadows, shrublands).

6. Recommendation for Spring/Summer/Fall Forest Cover

The recommendation is that within the non-winter portion of the EAU, coniferous cover will be managed within the ecological context or NRV for the geographic area (e.g. mountain range) where the EAU is located (see ecological context section). Cover assessments conducted for a project level analysis within the EAU should include a consideration of the condition (health), quantity, location, and configuration of desirable cover conifer species (see discussion). Specific to elk, the amount and types of available cover should accommodate the needs of elk relative to an assessment of limiting habitat factors (forage, cover) within the EAU.

Discussion:

The agency participants discussed the need to have any restrictions regarding cover management for elk, given the lack of research regarding the quantity of cover that is optimal. Additionally there is the general trend on the four forests (since the Forest Plans were written) of less timber harvest overall, smaller timber sales, and greater cover retention in treated areas (thinning rather than clear cutting). In addition, fire suppression has probably resulted in more cover than would be supported under a more natural fire regime. However, the agency participants recognized that the timber harvest levels could increase in the future and that large forest fires could significantly change the amount of available timber cover on any landscape in the future.

Current cover requirements in Forest Plans distinguish between cover for hiding and cover needed for thermal regulation. While we recognize the importance to elk of having cover available that provides hiding and/or thermal regulation benefits among other attributes, the agency participants did not see the need for managing for two different types of cover with different definitions. We did, however, recognize value in differentiating between the functions of cover on winter vs. spring/summer/fall ranges. We agreed that generally both hiding and thermal functions on spring-summer-fall ranges could be provided by conifer stands with at least 40% canopy cover (as determined from available map products). Other functions of cover on spring-summer-fall range (e.g. forested foraging areas) could be satisfied with lower canopy cover stand conditions.

40% is not TC

A previous interagency group that discussed management direction for elk in 2004 concluded that travel management and forage production were more important as management emphases to elk, and that cover is naturally dynamic relative to fire and the levels could be managed within natural fire regimes on most elk range. They felt that cover, where it occurred on elk winter range, was critical and should be maintained (Firebaugh et al. 2004). For this current effort, agency participants acknowledged that forested cover could function to reduce elk vulnerability during the hunting seasons; otherwise, forested cover may influence the way animals use habitat, and the ability of habitat to meet elk needs for growth and welfare requirements (Christensen et al. 1993). For example, Rowland and Wisdom (2012) found that late summer forage quality for elk in the Blue Mountains was highest where it was within a forest cover type.

Security

Shade

The agency participants discussed instances where forest cover was important for holding elk on public land even when security (due to route density) was lacking. The agency participants agreed that there may be legitimate reasons to maintain more cover in certain situations while reducing cover in other situations depending on the management issues in question. We ultimately agreed that it would be best to manage cover levels with respect to the ecological context of the geographic area (e.g. mountain range) in which the EAU occurs (see ecological context section).

The function of cover during spring, summer and fall may include thermal regulation, lengthening the season of succulence and palatability where adequate understory forage exists and the overstory provides shades, seclusion, and buffering of special features such as licks and moist areas (Christensen et al. 1993). Forested cover influences the way animals use habitat, and the ability of habitat to meet elk needs for growth and welfare requirements (Christensen et al. 1993).

Limber

In some areas, juniper, limber pine, whitebark pine, or ponderosa pine may function as cover for elk, but agency participants agreed that thinning or removal of these species may have more value, in the

Local circumstances result in varying effects on elk (e.g. elk in Arizona may experience a very different habitat reality than elk in Montana). On NFS lands in the Northern Region, elk are coming out of the most forage restricted time of year in the spring, are in the last trimester of pregnancy, and are beginning to nurse. Summer forage is key to abundant milk production as nursing of the growing calf continues. Summer is also the time of year when fat reserves are accumulated, which affect breeding success, pregnancy rates, parturition synchronization, the energy to avoid predators in the winter, and general winter survival (physiological demands of cold temperatures, travelling in snow during a time of greatly reduced forage availability and quality). Fall forage provides some maintenance energy between the summer and the winter when fat reserves supply much of their energy. Fall forage also helps bulls recover from stress and energy expenditures associated with the breeding season. Winter forage provides some maintenance energy and roughage for rumination which produces body heat. In addition, protein levels in some shrubs may be sufficient even in winter to meet elk protein requirements.

High quality forage is often found in areas with limited coniferous cover (e.g., meadows, brushfields, and aspen stands). Riparian and wetland areas can provide disproportionate amounts of succulent vegetation (particularly in more arid conditions). Forests can also provide important forage, particularly where they are open enough to allow adequate sunlight, water, and nutrients to get to the forest floor. Forage within forested areas can have a longer green and succulent season, when more open areas cure out in the late summer sun.

Dry out.

Any management action affecting vegetation can affect forage quality and abundance (e.g., wildfire management, fire suppression, prescribed fire, thinning, timber harvest, noxious weed management, livestock grazing, and revegetation). Management actions related to motorized vehicles or even non-motorized uses which disturb or displace elk can affect the ability of elk to utilize available forage.

Factors such as fire suppression, noxious weeds, or some livestock grazing practices can alter vegetative composition, structure, and successional stage. Forage can often be produced or improved by addressing the departure from range of natural variation of vegetative composition and structure (e.g., fire management, mechanical treatments, improved livestock management, aspen or riparian treatments, and noxious weed treatment). Habitat management projects designed to address departure from NRV generally benefit a broad suite of fauna adapted to that landscape. ?

The main focus in this collaborative effort is on travel management and cover requirements for big game. However, agency participants agreed that at some future time, management of forage would need to be addressed in some detail. However, that discussion would need to include other FS and MDFWP specialists.

The agency participants recognized that forage availability and quality plays a critical role in big game habitat use and big game distribution. It was also recognized that more importance should be placed on forage availability and quality during the summer and early fall, as that is often the determining factor in whether or not animals make it through the winter (Canfield et al. 1999) and the effects on body condition during the rut likely influences other factors such as pregnancy rates and synchronized calving. High quality forage may be seasonally unavailable in some areas due to

United States Department of the Interior

FISH AND WILDLIFE SERVICE
334 Parsley Boulevard
Cheyenne, Wyoming 82007

In Reply Refer to:

January 17, 2023

Memorandum

To: Assistant Regional Director, U.S. Fish and Wildlife Service, Ecological Services, Lakewood, Colorado

From: *for* Field Supervisor, U.S. Fish and Wildlife Service, Wyoming Field Office, Cheyenne, Wyoming NATHAN DARNALL Digitally signed by NATHAN DARNALL Date: 2023.01.17 14:40:48 -07'00'

Subject: Standing Analysis for Effects to Whitebark Pine (*Pinus albicaulis*) from Low Effect Projects and Whitebark Pine Restoration and Recovery Activities within Montana and Wyoming

On December 15, 2022, the U.S. Fish and Wildlife Service (Service) published a final rule to list the whitebark pine (*Pinus albicaulis*) as threatened under the Endangered Species Act of 1973, as amended (Act; 16 U.S.C. 1531 *et seq.*) (87 FR 76882). The attached Standing Analysis was developed to expedite the consultation process for ongoing maintenance and management actions, new development or construction projects that will damage or kill small numbers of live whitebark pine, and for activities that are beneficial for the restoration or recovery of whitebark pine within Montana and Wyoming. The Standing Analysis considers the range-wide status of the species, the status of whitebark pine within Montana and Wyoming, including threats and the impact of ongoing activities and conservation efforts. The Standing Analysis evaluates these factors and the effects to whitebark pine from recovery activities and other actions considered to have low effects, along with cumulative effects. Based on the Standing Analysis, we developed a determination key to be used in the Service's online Information for Planning and Consultation (IPaC) system.

Federal agencies or their designated representatives may use the determination key in IPaC for consultation. Information submitted online (e.g., project description and location, responses to questions about the project) is evaluated by a determination key in IPaC to ensure that projects fall within the scope of what is analyzed in the Standing Analysis. The Standing Analysis provides the basis for Service concurrence with "may affect but not likely to adversely affect" determinations. The Standing Analysis documents the Service's analysis for a wide variety of small effect actions and activities, and the Standing Analysis can be readily updated with new information.

The determination key in IPaC will be used to evaluate potential effects to whitebark pine from individual projects. Based on the information provided in IPaC, the determination key may confirm that a project either (1) is not anticipated to have any effect on the whitebark pine, or (2) "may affect but is not likely to adversely affect" the whitebark pine. Projects falling into these categories will receive a response either acknowledging the "no effect" determination or concurring with the "may affect but is not likely to adversely affect" determination. Projects that fall outside the bounds of the Standing Analysis and projects that "may affect and are likely to adversely affect" whitebark pine, even if they are addressed in the Standing Analysis, are not part of the online determination key in IPaC at this time.

No Effect (NE) Determination

Based on the Standing Analysis, projects within the area of influence (species range) but for which whitebark pine are not present and projects involving only dead trees (no live seedling, sampling, or mature trees) will have no effect (NE) on whitebark pine. Concurrence from the Service is not required for NE determinations; however, IPaC will deliver a letter documenting the outcome of the determination key.

May Affect, Not Likely to Adversely Affect (NLAA) Determination

The Standing Analysis concludes that projects occurring more than 10 meters (33 feet) from any live whitebark pine trees (seedling, sapling, and mature trees) will result in insignificant, discountable, or wholly beneficial effects to the whitebark pine, resulting in a "may affect, not likely to adversely affect" (NLAA) determination (more specifically described in the Standing Analysis). We also anticipate that agencies will implement the appropriate conservation measures from the Standing Analysis and adhere to the most recent best management practices to minimize impacts to whitebark pine.

It is the federal agency's responsibility to make a final effect determination for potential impacts resulting from actions they conduct or permit and to confirm that the project is consistent with the determination key's questionnaire and output. This letter only addresses whitebark pine. If other listed species or critical habitat occur within a project area or may be affected by a project, a separate consultation with the Service may be necessary.

Questions regarding this memo or responsibilities under the Act should be directed to the Wyoming Ecological Services Field Office at the letterhead address, by email at WyomingES@fws.gov, or by phone at (307) 772-2374.

EXECUTIVE SUMMARY

On December 15, 2022, the U.S. Fish and Wildlife Service (Service) published the final rule to list the whitebark pine (WBP) (*Pinus albicaulis*) as a threatened species (87 FR 76882) under the Endangered Species Act of 1973, as amended (Act; 16 U.S.C. 1531 et seq.). The Service also published a final 4(d) rule for WBP that identified actions necessary for the conservation and recovery of the species and included prohibited acts as well as a limited number of exceptions to the prohibited acts (87 FR 76882). Previously, on December 2, 2020, the Service published a proposed rule to list the WBP as threatened with a proposed 4(d) rule under the Act (85 FR 77408). No critical habitat was proposed for WBP. Section 4(d) rules cannot and do not absolve federal agencies of their consultation requirements under section 7 of the Act.

This Standing Analysis provides a basis for section 7 consultation for activities that are not prohibited by the 4(d) rule (see Appendix A for more information on the 4(d) rule). The analysis in this Standing Analysis (SA) evaluates effects to WBP from a variety of actions across Montana and Wyoming, including ongoing maintenance and management actions, new development or construction projects that will damage or remove no more than 125 total WBP trees (seedlings, saplings, and mature trees combined), livestock grazing, right-of-way maintenance, and activities that are beneficial for the restoration or recovery of WBP. Specifically, activities considered in the framework include:

a. Livestock management and range improvements: livestock management and range infrastructure improvements (*e.g.*, grazing, gathering and moving, fencing, stock ponds and tanks, and spring development);
b. Infrastructure actions: infrastructure maintenance, upgrades, and replacement activities for existing pipelines, communication towers, utility lines, renewable energy facilities, trails, and highway infrastructure (*e.g.*, to roads, bridges, culverts, bike and pedestrian facilities, fencing, lighting, and all other distinct aspects of a highway project); construction of new pipelines, communication towers, utility lines, renewable energy projects, and highway infrastructure;
c. Mineral and conventional oil and gas exploration and development: maintenance of existing locatable and leasable mining and conventional oil and gas projects; new development of existing mineral and conventional oil and gas leases; and new development of new mineral and conventional oil and gas leases;
d. Vegetation management actions: vegetation (forest) management activities that take into account age class prescriptions, genetic variability within and among WBP populations, and activities that may impact the adaptive potential and the forest trajectories of WBP (salvage harvests and pest control that remove dead and diseased trees and encourage natural WBP recruitment and are not in existing WBP restoration areas); timber harvest projects, hazardous fuel removal (in Wildland Urban Interface (WUI) areas), and precommercial thinning and group selection projects;
e. Recreation activities: maintenance of existing trail systems (hiking and biking specifically); maintenance of existing recreational development projects (*e.g.*, ski resorts and campgrounds), and other recreational activities (*e.g.*, off-highway vehicles (OHV) and over-snow vehicles (OSV) use); upgrades, replacement, expansion, or new construction outside the existing disturbance footprint of existing recreation projects, and outfitter and guide permit programs;

may remove no more than 125 trees, thus providing an additional buffer. However, we will continue to evaluate development projects that remove more than 125 WBP trees separately (outside of this SA), even if the aggregate number of trees removed per year is less than 12,500 (of which, 5,000 are assumed to be rust-resistant). We anticipate the recovery efforts (*e.g.*, planting rust-resistant trees) will either remain the same or are likely to increase after the species has been listed under the Act; however, we will continue to use the approach described above and its thresholds rather than increasing the number of projects and trees. We will re-evaluate this SA at least annually and will reconsider its application, use, thresholds, and efficacy at a minimum of five years after implementation.

2.f. Conservation Measures

To fully understand the extent of the effect that individual project activities analyzed by the SA will have on WBP trees (seedlings, saplings, and mature trees), and to avoid and minimize these impacts, the SA provides a detailed list of successfully-implemented conservation measures from which project applicants can choose. We consider the damage of a WBP to include, but is not limited to, soil tilling, disking, plowing, excavating, raking, sod rolling, soil compaction, soil disturbance, weed management, revegetation, crushing, bumping, and scraping WBP, root damage, mycorrhizal damage, nicks and opening wounds on WBP bark, and pruning.

The following conservation measures are considered part of the actions under the SA. The commitments and implementation of the following conservation measures and exploration of additional opportunities to reduce impacts made by the federal action agencies will reduce adverse effects of the proposed actions to the WBP. These conservation measures will become commitments by federal action agencies where appropriate and practicable. Many of these conservation measures are to be applied throughout the action area, though others specify the location or distance from WBP trees for application. Not all conservation measures are applicable or reasonable for all actions that are covered under this SA. The Determination Key includes a section where project applicants must select which of the following conservation measures they plan to implement as part of their action, which will be repeated back to the applicant in our IPaC generated response letter or memo.

General

CM 1. Conduct pre-project surveys to identify WBP individuals of all age classes. If not feasible, conduct surveys using appropriate agency protocols to estimate the number of WBP individuals of all age classes.

CM 2. When marking is appropriate, all mature WBP trees or clusters of other age class trees will be marked in a manner that does not cause damage to the tree or introduce disease.

CM 3. Damaging or killing a plus, elite, or phenotypically resistant tree will only occur in situations where human health and safety are at risk or when restoration actions such as pruning are occurring.

3. Status of the Species

This SA provides a general background on the status of the species and its habitat, relying on the Service's 2021 SSA (USFWS 2021, entire). No critical habitat has been designated for this species. A detailed status of the WBP's biology, range, habitat, needs, stressors, and conservation can be found in Appendix C.

The WBP is a five-needle conifer species placed in the subgenus *Strobus,* which also includes other five-needle white pines. Recent phylogenetic studies (Liston *et al.* 1999; Syring *et al.* 2005, 2007; as cited in Committee on the Status of Endangered Wildlife in Canada (COSEWIC) 2010) showed no difference in monophyly (ancestry) between subsection *Cembrae* and subsection *Strobi* and merged them to form subsection *Strobus.* No taxonomic subspecies or varieties of WBP are recognized (COSEWIC 2010). Based on this taxonomic classification information, we recognize WBP as a valid species (USFWS 2021).

There are four stages in the life cycle of the WBP: seed, seedling, sapling and mature trees, also referred to as reproductive adults. Seeds are produced in female cones and once on the ground may take two years or more, up to 11 years in some cases, to germinate. Germinated seeds become seedlings that are between 3 to 4 inches tall (8 and 10 centimeters) with a taproot that can measure between 5 to 7 inches (13 to18 centimeters), with 7 to 9 cotyledons, also known as the embryonic first leaves, as documented by (Arno and Hoff 1989). WBP seedlings may persist for multiple years, depending on growing conditions, until reaching the sapling stage of the life cycle. WBP saplings persist for few to many years, depending on growing conditions, until they produce male and female cones. Mature reproductive WBPs contain both female and male cones, which is known as monoecious reproduction, and can survive on the landscape for hundreds of years. This slow-growing long-lived tree with a life span between 500 years and 1,000 years (Arno and Hoff 1989; Perkins and Swetnam 1996), provided it is located in an area with lower competition, such as a more open canopy with low litter depth and high rock cover (Maloney 2014). Therefore, in addition to the four general needs for all life stages, mature WBP trees require a more open canopy, dispersal of seeds by Clark's nutcracker, two summers of suitable temperatures and precipitation for pollinated cones to mature, as well as levels of nitrogen and phosphorus that are adequate to restore values after being depleted in masting year (USFWS 2021).

The WBP is a five-needle pine that lives in windy, cold, high-elevation or high-latitude environments across the western United States and southern Canada. The WBP pine has a broad range both latitudinally, occurring from a southern extent of approximately 36° north in California to 55° north latitude in British Columbia, Canada, and longitudinally, occurring from approximately 128° in British Columbia, Canada to an eastern extent of 108° west in Wyoming. It also occurs in scattered areas of the warm and dry Great Basin. As a result, many stands are geographically isolated as documented by (Arno and Hoff 1989).

Most current management and research focus on producing and planting white pines (including WBP) with genetic resistance to white pine blister rust, but also include natural regeneration (e.g., those areas identified in Appendix B, Tables A, B, and C) and silvicultural treatments, such as appropriate site selection and preparation, pruning, and thinning (Zeglen *et al.* 2010).

5.d. Forest (Vegetation) Management Actions

Forest (vegetation) management includes a variety of methods and techniques used to manage healthy forests, and known previous projects that fall within this action type are summarized in Table D of Appendix C. These types of forest management activities include timber harvest (using chainsaws and using machinery which may create skid trails) and management/hazardous fuels reduction (using chainsaws and using machinery which may create skid trails, to remove dead and dying trees and understory vegetation that may carry wildfire), salvage harvest (removing dead trees either by hand or using machinery which may create skid trails), pest control (use of Verbenone or Carbaryl insecticide), precommercial thinning (thinning trees that are too small for a commercial timber harvest), silviculture stand improvement projects (using a planned set of treatments such as thinning, harvesting, planting, pruning, prescribed burning, and site preparation designed to change the current stand structure and composition to one that meets a management goal), and silvicultural reforestation activities (planning for natural regeneration or tree planting).

Forest management related road construction, maintenance, and use may also be part of vegetation management projects. Harvest of WBP has not been well tracked as records often group it with other species and incorrectly identify it as another species. Silviculture approaches create a system that excludes regeneration opportunities and increases competition by planting faster-growing species, and consequently, stands that contain WBP prior to harvest are not routinely replanted with WBP.

Projects that implement resetting the successional stage of the forest stands need to be carefully thought out and planned to increase WBP recruitment. Campbell and Antos (2003) noted that successional patterns in WBP forests are more complex than others have reported, finding that subalpine fir readily established after fire in their British Columbia study areas, and although subalpine fir density was increasing in older WBP stands with relatively open canopies, they estimated that succession to subalpine fir would take more than 500 years. Campbell and Antos (2003) reported that WBP in their study area was stress-tolerant (able to persist under conditions that restrict production), was capable of surviving long periods of suppressed growth, and was able to release upon reaching the main canopy after more than 150 years of low growth rates. The results of these studies indicate that the loss of WBP due to succession to subalpine fir and Engelmann spruce in some areas may be an extremely slow process and that WBP may be more shade-tolerant and resilient to suppression than previously suggested. Further, thinning and timber harvest projects intended to improve WBP recruitment may increase WBP susceptibility to mountain pine beetle infestation, if the beetles do not have their preferred food sources during outbreak years. The densification of and succession of subalpine fir and Engelmann spruce co-occurred with WBP mortality caused by bark beetle outbreaks and/or blister rust; therefore, disentangling the effects of blister rust- and bark beetle-mortality on succession from the effects of fire suppression in these studies is difficult (Hartwell *et al.* 1997; Arno *et al.* 1993 in Keane *et al.* 1994; Flanagan *et al.* 1998).

Projects including those in WUI, salvage harvests, and pest control efforts remove dead and diseased trees, and may encourage natural WBP recruitment. In large acreages of dead trees, salvage harvest and firewood cutting projects can be designed to avoid damaging or killing live

WBP, which may be resistant to blister rust. Projects where the removal of surface and ladder fuels through hand cutting, piling of project generated materials, and burning the piles with the purpose of increasing stand resilience to fire may also be beneficial for the recruitment of WBP. Felling trees and creating skid trails for salvage harvests may damage or kill WBP seedlings and saplings and compress the soil and undetected seeds. Implementation of the conservation measures (*e.g.*, CM 1-10, 12-14, and 16-21) in the project design that avoid impacts to WBP seedlings, saplings, and live mature trees, and that minimizes soil disturbance and compaction that may destroy microsites for cached seeds, interrupts drainage, and limits tree rooting will have beneficial long-term impacts to WBP.

Vegetation management includes many project types (*e.g.*, WUI, salvage harvest of dead trees, harvest of Christmas trees, pest control, firewood collection) and sizes (less than 1 acre to thousands of acres). In this SA, we evaluated the effects of smaller forest management projects that damage or kill fewer than 125 live WBP of all age classes. Effects of larger project will be addressed by a standalone consultation or may be covered by a future standing analysis. We have elected to use a limit of 125 WBP of all age classes as a threshold for forest (vegetation) management projects, based on our understanding of the stressors to WBP, the level of ongoing restoration efforts, and our commitment to track and re-evaluate project impacts and restoration efforts for the life of this SA. While forest management projects will result in adverse effects to WBP, these should not result in population level effects for the reasons described above.

5.e. Recreation Development and Activities

The following recreational activities commonly occur in WBP habitat: construction and maintenance of hiking trails and roads (analyzed in the Infrastructure section); motorized use of trails year-round; (snow machines, all-terrain vehicles (ATV), utility task vehicles (UTV), motorcycles, electric bikes, and mountain bikes); operation of facilities (snow making, lift chairs analyzed in the Infrastructure section); firewood consumption; special use permits (hunting, photography); and horseback riding.

There are 91 recreation sites within WBP habitat in the action area, including developed campsites, horse corrals, trail heads, parking areas, toilets, staging areas, scenic overlooks, and primitive campsites. Back country campers and hikers may burn WBP for campfires, cause ground compression, climb on trees, or remove WBP when clearing trails. Motorized recreation activities, hiking, use of pack animals, and construction equipment used for trail maintenance and construction, may cause soil disturbance and compaction, destroy microsites for cached seeds, interrupt drainage, limit tree rooting, and damage seedlings. Over snow vehicles (OSV) could break the tops of trees or could damage branches or seedlings and saplings. We acknowledge that there may be some damage and death to WBP seedlings and saplings from authorized and unauthorized off-road motorized recreation activities which could affect individuals or local areas. Overall, impacts from all recreation activities could affect less than one percent of the species wide range (based on IUCN threats summary for WBP in Canada) (USFWS 2021) and are not considered a significant threat to WBP.

We conclude that, while not all adverse effects can be avoided, the implementation of the conservation measures (*e.g.*, CM 1-14, and 16-21) will minimize impacts to WBP and that between one and 29 years of age and, on average, are 1.37 m (4.5 ft), which is the height assessed at dbh (Tomback and Pansing 2018). WBP seedlings have highly variable survival rates; seedlings originating from nutcracker caches ranged from 56 percent survival over the first year to 25 percent survival by the fourth year (Tomback 1982). Seedlings of WBP may persist for multiple years, depending on growing conditions, until reaching the sapling stage of the life cycle. Saplings are non-reproductive trees greater than 1.37 m (4.5 ft) in height; for WBP, the average age of reproductive maturity is 29 to 40 years of age (Tomback and Pansing 2018). Saplings of WBP persist for few to many years, depending on growing conditions, until they produce male and female cones and are considered reproductive at approximately 60 years of age. Some WBP individuals are capable of producing limited amounts of seed cones at 20 to 30 years of age, although large cone crops usually are not produced until 60 to 80 years (Krugman and Jenkinson 1974, as cited in McCaughey and Tomback 2001), with average earliest first cone production at 40 years (Tomback and Pansing 2018). Therefore, the generation time of WBP is approximately 40 to 60 years (Tomback and Pansing 2018; COSEWIC 2010). Mature WBP trees require two summers of suitable temperatures and precipitation for fertilized cones to mature (Rapp *et al.* 2013). Years with high seed production (mast years) typically occur once every three to five years; however, that time interval can vary by geographic location and health condition of the stand (McCaughey and Tomback 2001). After such a masting year, each individual WBP is depleted of nitrogen and phosphorus and so those nutrients must be replaced during the three to five years between masting events (Sala *et al.* 2012). Mature reproductive WBP trees contain both female and male cones (*i.e.*, monoecious reproduction), and can survive on the landscape for hundreds of years.

The WBP is the only stone pine (so-called for their stone-like seeds) in North America of the five species worldwide (McCaughey and Schmidt 2001). Characteristics of stone pines include five needles per cluster, indehiscent seed cones (scales remain essentially closed at maturity) that stay on the tree, and wingless seeds that remain fixed to the cone and cannot be dislodged by the wind. Because WBP seeds cannot be wind disseminated, primary seed dispersal occurs almost exclusively by Clark's nutcrackers in the avian family Corvidae (whose members include ravens, crows, and jays) (Lanner 1996; Schwandt 2006). Consequently, Clark's nutcrackers facilitate WBP regeneration and influence its distribution and population structure through their seed caching activities (Tomback *et al.* 1990).

The WBP may occur as a climax species, early successional species, or seral (mid-successional stage) codominant associated with other tree species. Although it occasionally occurs in pure or nearly pure stands at high elevations, it more typically occurs in stands of mixed species in a variety of forest community types. WBP is typically 5 to 20 m (16 to 66 ft) tall with a rounded or irregularly spreading crown shape. On higher density conifer sites, WBP tends to grow as tall, single-stemmed trees, whereas on open, more exposed sites, it tends to have multiple stems (McCaughey and Tomback 2001). Above the tree line, it grows in a krummholz form (stunted, shrub-like growth) (Arno and Hoff 1989). Production of male and female cones in mature trees will begin sometime from June to September depending on environment (McCaughey and Tomback 2001; Sala *et al.* 2012). Female cones take 2 years to fully develop (Weaver 2001). Its characteristic dark brown to purple seed cones are 5 to 8 cm (2 to 3 in.) long and grow in clusters of 2 to 4 cones at the outer ends of upper branches (Hosie 1969).

Ensuring future generations of WBP are genetically resistant to white pine blister rust is the most critical action for achieving long-term recovery of this species (Mahalovich and Dickerson 2004; Perkins *et al.* 2016). Genetic management of white pine blister rust is actively conducted for WBP, including the USFS white pine blister rust resistance screening programs (Mahalovich 2016; Sniezko 2016). Seeds and pollen sourced from plus trees or elite trees are used for screening and selective breeding for white pine blister rust resistance (not immunity), molecular genetics studies, assessing levels of inbreeding, growing compatible rootstock for grafting in seed orchards, clone banking and gene conservation, and identifying genetic macro-refugium (Mahalovich 2016; Perkins *et al.* 2016; Sniezko 2016).

Eventually, the long-term goal is to establish WBP seed orchards in situ across the spectrum of WBP habitat to provide reliable and accessible sources of genetically resistant seed (Mahalovich 2017). Scions (*i.e.*, living branches) taken from trees with proven genetic resistance to white pine blister rust are grafted onto established root stocks, enabling them to develop the capability to produce cones much sooner than the time required for out planted seedlings to reach reproductive maturity (approximately 60 years).

Seeds from cone collections in the northwest (*i.e.*, Washington, Oregon) are now stored at the United States Department of Agriculture (USDA) National Center for Genetic Resources Preservation in Fort Collins, Colorado for long-term ex situ gene conservation (Sniezko and Kegley 2017). Seven separate white pine blister rust screening trials are occurring at the USFS Coeur d'Alene Nursery in Idaho using over 100,000 seedlings (Mahalovich 2016, 2017).

4.c. Fuel Reduction Treatment

Silvicultural practices, such as thinning, are frequently employed to treat existing stands of WBP to modify surface and ladder fuels and improve their chances of surviving fire. Table D of Appendix C describes the silvicultural and restoration activities in WBP habitat within Montana between 2009 and 2021 and Wyoming between 1991 and 2020. Most thinning treatments are designed to mimic non-lethal mixed-severity fire (Keane and Arno 2001), reduce or eliminate competition from other conifer species such as subalpine fir (*Abies lasiocarpa*), and to increase regeneration space for potentially rust-resistant seedlings. Approaches include creating openings wherein all trees except live WBPs are cut within a 1- to 5-ac opening to provide open growing space for WBP regeneration and existing WBP trees (Keane and Arno 2001; Keane and Parsons 2010); thinning of all non-WBP trees below a certain diameter (Chew 1990); and fuel enhancement treatments where other competing trees are directionally felled to modify fire behavior and reduce fire intensity (Keane and Arno 2001; Keane and Parsons 2010). Reducing tree density within WBP stands may result in increased vigor (*i.e.*, growth rate) of remaining sapling to mature-class trees (Keane *et al.* 2007; Retzlaff *et al.* 2018); however, counterintuitively, increased WBP vigor may not impart increased resistance to mountain pine beetle, as some evidence suggests that mountain pine beetles select faster growing trees for attack (Six *et al.* 2021). In addition to or in place of treating fuels within WBP stands, managers should consider conducting fuel reduction treatments in non-WBP stands adjacent to WBP, thereby reducing the intensity of fire as it moves from the adjacent stand into the WBP stand.

Considered a keystone, or foundation species in western North America, WBP it increases biodiversity and contributes to critical ecosystem functions (Tomback *et al.* 2001). As a pioneer or early successional species, it may be the first conifer to become established after disturbance, subsequently stabilizing soils and regulating runoff (Tomback *et al.* 2001). At higher elevations, snow drifts around WBP trees, thereby increasing soil moisture, modifying soil temperatures, and holding soil moisture later into the season (Farnes 1990). These higher elevation trees also shade, protect, and slow the progression of snowmelt, essentially reducing spring flooding at lower elevations. The WBP also provides nutritious seeds for a number of birds and mammals (Tomback *et al.* 2001).

3. Population Dynamics, Status and Distribution

The WBP has persisted in high elevation sites in western North America for the past 8,000 years (McCaughey and Schmidt, 2001). WBP has a broad range both latitudinally (occurring from approximately 36 degrees in south California to 55 degrees north latitude in British Columbia, Canada) and longitudinally (occurring from approximately 128 degrees in British Columbia, Canada to 108 degrees east in Wyoming). For the SSA, we developed an updated WBP range map based on the best available occurrence and distribution data (Figure 2 of the SA). This range map is at a coarse scale but encompasses the known distribution of the species' occurrences.

The WBP typically occurs on cold and windy high-elevation or high-latitude sites in western North America, although it also occurs in scattered areas of the warm and dry Great Basin. As a result, many stands are geographically isolated (Arno and Hoff 1989; Keane *et al.* 2010). The distribution of WBP includes coastal and Rocky Mountain ranges that are connected by scattered populations in northeastern Washington and southeastern British Columbia (Arno and Hoff 1990; Keane *et al.* 2010). The coastal distribution of WBP extends from the Bulkley Mountains in northwestern British Columbia to the northeastern Olympic Mountains and Cascade Range of Washington and Oregon, to the Kern River of the Sierra Nevada Range of east-central California (Arno and Hoff 1990). Isolated stands of WBP are known from the Blue and Wallowa Mountains in northeastern Oregon and the subalpine zone of mountains in northeastern California, south-central Oregon, and northern Nevada (Arno and Hoff 1990; Keane *et al.* 2010). The Rocky Mountain distribution of WBP ranges from northern British Columbia and Alberta to Idaho, Montana, Wyoming, and Nevada (Arno and Hoff 1990; Keane *et al.* 2010), with extensive stands occurring in the Yellowstone ecosystem (McCaughey and Schmidt 2001).

In general, the upper elevational limits of WBP decrease with increasing latitude throughout its range (McCaughey and Schmidt 2001). The elevational limit of the species ranges from approximately 900 m (2,950 ft) at its northern limit in British Columbia to 3,660 m (12,000 ft) in the Sierra Nevada (McCaughey and Schmidt 2001). WBP is typically found growing at the subalpine treeline or with other high-mountain conifers just below the treeline and subalpine zone (Arno and Hoff 1990; McCaughey and Schmidt 2001). In the Rocky Mountains, common associated tree species include lodgepole pine (*P. contorta* var. *latifolia*), Engelmann Spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and mountain hemlock (*Tsuga mertensiana*). Common associated tree species are similar in the Sierra Nevada and Blue and Cascade Mountains, except lodgepole pine is present as Sierra-Cascade lodgepole pine (*P.*

contorta var. murrayana) and mountain hemlock is absent from the Blue Mountains (Arno and Hoff 1990; McCaughey and Schmidt 2001).

Rangewide, WBP occurs on an estimated 32,616,422 ha (80,596,935 ac) in western North America (Figure 2 of the SA). Roughly 70 percent of the species' range occurs in the United States, with the remaining 30 percent of its range occurring in British Columbia and Alberta, Canada (USFWS 2018). In Canada, the majority of the species' distribution occurs on federal or provincial crown lands (COSEWIC 2010). In the United States, approximately 88 percent of land where the species occurs is federally owned or managed (Figure 2 of the SA). The majority is located on USFS lands (approximately 74 percent, or 17,391,455 ha (42,975,220 ac)). The bulk of the remaining acreage is located on NPS lands (approximately 10 percent, or 2,275,746 ha (5,623,490 ac)). Small amounts of WBP also can be found on BLM lands (approximately 4 percent, or 1,002,152 ha (2,476,371 ac)). The remaining 12 percent of the range is under non-federal ownership, on State, private, and Tribal lands. In the United States, 29 percent of the range is designated as wilderness under the Wilderness Act of 1964 (16 U.S.C. 1131-1136). This designation limits management options and conservation efforts in those areas to some degree.

Populations are typically defined by the potential for genetic exchange among their members, to the exclusion of members of other populations (in the absence of immigration or emigration). For WBP, genetic exchange is limited by the dispersal distance of pollen, which is carried by wind, and the seed caching behavior of Clark's nutcracker (Lorenz *et al.* 2011; Keane *et al.* 2017b). Both pollen dispersal and Clark's nutcracker seed dispersal can occur at a scale of few to many kilometers (km) (*e.g.*, up to 30 km in the case of Clark's nutcracker seed dispersal). To promote a greater than 75 percent probability of occurrence of Clark's nutcracker at a site, recommended management plans that achieves landscape composition of a minimum 12,500-25,000 ha of cone bearing WBP habitat within a 32.6-km radius. The optimal Clark's nutcracker habitat mosaic includes moderate levels of Douglas-fir habitat (Douglas-fir seeds are another important food source for Clark's nutcracker in the GYE) (Schaming 2016 and Schaming and Sutherland 2020). WBP is a long-lived species that exhibits masting, where years of high seed production are synchronized within a population approximately every 3 to 5 years (McCaughey and Tomback 2001). This masting strategy is an adaption to heavy seed predation; during masting years seed consumers are satiated, resulting in excess seeds that escape predation (Lorenz *et al.* 2008). WBP populations need a certain density of reproductive individuals to produce sufficient pollen clouds that facilitate the synchronization of masting, and thus increased probability of regeneration (Rapp *et al.* 2013).

4. Conservation Actions and Restoration Strategies

Most current management and research focus on producing and planting white pines (including WBP) with genetic resistance to white pine blister rust, but also include natural regeneration (e.g., those areas identified in Appendix B, Tables A, B, and C) and silvicultural treatments, such as appropriate site selection and preparation, pruning, and thinning (Zeglen *et al.* 2010). Genetic management of white pine blister rust is actively conducted for several five-needle white pine species breeding programs (Sniezko 2016, Mahalovich 2015, Mahalovich 2010, Shelley 2016) including the USFS resistance screening programs for WBP. High-elevation pines such as WBP also present management challenges to restoration due to remoteness, difficulty of access, a

WBP both as a stressor that can cause mortality of all life stages of WBP and as a mechanism that may affect forest succession (Arno 2001; Shoal *et al.* 2008; Keane and Parsons 2010). Although WBP has been described as fire-adapted, there is uncertainty surrounding the specifics of these adaptations, including the species' ability to survive fires of differing intensity, the role of low-severity fire, and how fire suppression interacts with fire return intervals to affect forest succession across the range of WBP.

Fire regimes in WBP systems are often characterized as being of mixed severity (Arno *et al.* 2000; Arno 2001, Campbell and Antos 2003; Larson *et al.* 2009). However, some WBP systems are dominated by high-severity fire events (Romme 1982; Campbell and Antos 2003). Low-severity surface fires may also occur in WBP stands, particularly at higher elevations (Barrett 1994). Clark's nutcracker ecology provides further insight into the typical fire regime in WBP ecosystems. The Clark's nutcracker serves as the main dispersal agent for WBP (Tomback *et al.* 2001). Eighty-five percent of the WBP seeds cached are above ground in the trunks of live trees. In most cases Clark's nutcracker avoids caching seeds in forest openings (Brodin and Clark 2007; Lorenz 2011; Lorenz et al 2012. Clark's nutcrackers have been found dispersing seeds farther than the wind-dispersed seeds of other conifers, allowing for the establishment of WBP seedlings in the interior of large patches of high severity fire effects and over broad geographic areas (McCaughey *et al.* 1985; Tomback *et al.* 1990, 1993 in Keane and Parsons 2010). To promote a greater than 75 percent probability of Clark's nutcracker occurrence at a site containing WBP, the landscape composition must contain a minimum 12,500 to 25,000 ha cone bearing WBP within a 32.6-km radius (Schaming and Sutherland 2020). Clark's nutcrackers feed equally on Douglas-fir seeds and WBP seeds, so the optimal habitat mosaic includes moderate levels of Douglas-fir habitat (Schaming 2016).

Although some experts have suggested that WBP is phenotypically adapted to survive low-intensity fire, Stevens *et al.* (2020) found that WBP had relatively thin bark compared to other conifer species and, based on a systematic ranking of numerous traits associated with fire resistance in western conifers, WBP was found to have one of the lowest fire resistance scores of the 29 conifers examined in the study. Others have also observed that WBP trees can be sensitive to bole (main steam of the tree) scorching, resulting in cambium injury or death, even from low-intensity fire (Hood *et al.* 2008). Keane *et al.* (2020) noted several recent reports of prescribed fire and low-intensity fire killing WBP trees, despite pre-fire site preparation activities implemented to reduce or modify surface and ladder fuels and protect the residual WBP trees. Keane and Parsons (2010) studied the effects of seven different fuel treatment combinations on WBP at five treatment sites in Montana and Idaho and found that WBP mortality from low-intensity fire was comparable to subalpine fir under all treatment combinations. As a result, empirical evidence shows that low-intensity fire in WBP can result in higher-severity fire effects. In summary, although it is clear that WBP individuals are capable of surviving some low-intensity fire, based on the presence of multiple fire scars in some areas, the biotic and abiotic (*i.e.*, terrain, weather, and fuel) conditions under which the species is most likely to survive such fires remain largely unknown.

Determining if periodic fire is necessary to maintain ecosystem integrity in WBP systems may be as important as understanding the conditions under which WBP trees are most likely to survive fire. Experts have suggested that, without periodic low-severity fire in some subalpine forests where WBP co-occurs with subalpine fir and Engelmann spruce, successional pathways can lead to climax communities dominated by these shade-tolerant conifers and the loss of WBP (Arno 1980; Arno 2001; Keane et l. 2017; Keane and Parsons 2010; Flanagan *et al.* 1998). It has further been suggested that, in these WBP systems, fire suppression policies over the past 90 years have resulted in WBP declines due to succession to subalpine fir and Engelmann spruce (Arno 1980; Arno 2001; Keane *et al.* 2017; Keane and Parsons 2010; Flanagan *et al.* 1998). This is supported by the presence of multiple fire scars in WBP trees at some locations, which shows they are capable of surviving repeated low-intensity fires and maintaining dominance or co-dominance in stands for long-periods of time when these fires are occurring periodically (Morgan and Bunting 1990, Barrett 1994). Additional support for the successional theory is based on documented densification of subalpine fir and Engelmann spruce in stands where WBP was once prevalent (Hartwell *et al.* 1997; Arno *et al.* 1993 in Keane *et al.* 1994; Flanagan *et al.* 1998). However, in these studies, the authors noted that the densification of and succession to subalpine fir and Engelmann spruce co-occurred with WBP mortality caused by bark beetle outbreaks and/or blister rust; therefore, disentangling the effects of blister rust- and bark beetle-mortality on succession from the effects of fire suppression in these studies is difficult.

The idea that fire suppression in some whitebark systems has resulted in densification and loss of WBP has been a predominant theory in the WBP literature (Arno 1980; Arno 2001; Keane *et al.* 2017; Keane and Parsons 2010; Flanagan *et al.* 1998). However, some have recently called into question the idea that fire suppression has led directly to forest succession and the loss of WBP. For example, Larson and Kipfmueller (2012) suggested there is uncertainty in the effects of fire suppression on WBP and a relative lack of data supporting the hypothesis. Larson and Kipfmueller (2012) noted that age structure data in their study showed that many of the small subalpine fir trees occurring below the WBP, trees that visually appeared to be young saplings, were more than 100 years of age, suggesting that size class data should not be used as a surrogate for tree age or to determine the rate of succession. Campbell and Antos (2003) also noted that successional patterns in WBP forests are more complex than others have reported, finding that subalpine fir readily established after fire in their British Columbia study areas, and although subalpine fir density was increasing in older WBP stands with relatively open canopies, they estimated that succession to subalpine fir would take more than 500 years. Campbell and Antos (2003) reported that WBP in their study area was stress-tolerant (able to persist under conditions that restrict production), was capable of surviving long periods of suppressed growth, and was able to release upon reaching the main canopy after more than 150 years of low growth rates. The results of these studies indicate that the loss of WBP due to succession to subalpine fir and Engelmann spruce in some areas may be an extremely slow process and that WBP may be more shade-tolerant and resilient to suppression than previously suggested.

The broad range of fire return intervals in WBP ecosystems further complicates theories that fire suppression has caused succession in WBP systems. Fire history studies in WBP forests have identified fire return intervals ranging from 33 years (Morgan and Bunting 1990) to greater than 400 years (Campbell and Antos 2003). Several authors have noted that mean fire return intervals in subalpine forests that include WBP can be much longer than contemporary fire suppression policies (Dolanc *et al.* 2013; Meyer and North 2019; Sibold *et al.* 2006). Over an 80-year period, Dolanc *et al.* (2013) documented an increase in the number of small diameter trees, including WBP, in subalpine forests of the central Sierra Nevada. However, Dolanc *et al.* (2013)

Beetle activity in the phloem mechanically girdles the host tree, disrupting nutrient and water transport and ultimately killing it. Additionally, mountain pine beetles carry symbiotic blue stain fungi on their mouthparts, which are introduced into the host tree upon feeding. These fungi also inhibit water transport and further assist in killing the host tree (Raffa and Berryman 1987; Keane *et al.* 2012).

Mountain pine beetles are considered an important component of natural forest disturbance regimes (Raffa *et al.* 2008; Bentz *et al.* 2010). At endemic, or more typical levels, mountain pine beetles remove relatively small areas of trees, changing stand structure and species composition in localized areas. However, when conditions are favorable (abundant hosts and favorable climate), mountain pine beetle populations can erupt to epidemic levels and create stand-replacing events that may kill 80 to 95 percent of suitable host trees (Berryman 1986 as cited in Keane *et al.* 2012). Such outbreaks are episodic, can have a magnitude of impact on the structure of western forests greater than wildfire (the other major component of natural forest disturbance), and are often the primary renewal source for mature stands of western pines (Hicke *et al.* 2006; Raffa *et al.* 2008; Six *et al.* 2014). Mountain pine beetle outbreaks typically subside only when suitable host trees have been exhausted or temperatures are sufficiently low to kill larvae and adults (Gibson *et al.* 2008).

The range of the mountain pine beetle completely overlaps with the range of the WBP, and mountain pine beetle epidemics affecting WBP have occurred throughout recorded history (Keane *et al.* 2012). Recent outbreaks occurred in the 1930s, 1940s, and 1970s, and numerous ghost forests of dead WBP still dot the landscape as a result (Arno and Hoff 1989; Perkins and Swetnam, 1996, Ward *et al.* 2006). The most recent mountain pine beetle epidemic began in the late 1990s and continues to be a measurable but much reduced source of mortality for WBP (MacFarlane *et al.* 2013; Mahalovich 2013; Shelly 2014) (Figure A).

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Science FINDINGS

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"Science affects the way we think together."
Lewis Thomas

Woodpecker Woes: The Right Tree Can Be Hard to Find

Teresa Lorenz

A female black-backed woodpecker perches at a nest excavated in a live ponderosa pine in central Washington. Although the tree diameters used by this species for nesting differed (ranging from 8 to 34 inches), it consistently selected nest sites with softer interior wood.

IN SUMMARY

Woodpeckers and other cavity-excavating birds worldwide are keystone species. These birds excavate their nests out of solid wood, and because their nests are often well protected against predators and the environment, other species use and compete for their old, vacant nests. The presence of cavity-excavating birds in forests has far-reaching effects on species richness and ecosystem health.

Given the species' importance, Teresa Lorenz, a research wildlife biologist with the U.S. Forest Service Pacific Northwest Research Station wanted to find out why cavity-excavating birds do not use many trees seemingly suitable for nesting. This puzzle has eluded researchers for decades. Lorenz and her colleagues also wanted to know what role wood hardness plays in the birds' nest site selection.

The resulting study in the eastern Cascades of Washington found that cavity-excavating birds preferred to nest in trees with significantly softer interior wood. The researchers also found that at-risk species were nesting within burned areas where up to 96 percent of the trees had unsuitably hard wood. This suggests that many trees and snags previously considered suitable for cavity-excavating birds actually may not be.

In dry forests, prescribed mixed-severity fire may be a useful tool for creating suitable nesting habitat for cavity excavators.

"In order to see birds, it is necessary to become part of the silence."
—John Keats

Somewhere in Washington's eastern Cascade Range a bird perches motionless against a burned tree. It's an American three-toed woodpecker (*Picoides dorsalis*), sporting black back feathers, a white mustache stripe, and a yellow cap. About the size of a large banana, the bird has evenly spaced white flecks on the sides of its chest and on its primary and secondary feathers.

The bird was extracting beetle larvae from another burned tree when a larger, louder black-backed woodpecker (*P. articus*) drove it away. But here, on this burned tree, it drums, "This one is mine. Stay away."

Soon it will pick a spot on the tree, say 30 feet above the forest floor, and bore rapidly into the wood with its chisel-like bill until it excavates a deep pocket. The bird and its mate will roost and nest in this safe place until its favorite food runs out, perhaps in 4 or 5 years. By then, another forest fire might provide a new supply of wood-boring insects.

In the woodpeckers' absence, other birds or small mammals like owls, squirrels, and bats move in. The new tenants disperse seeds and spores that help new plants and fungi grow. When the dead, burned trees—also called snags—fall to the ground, wood-living bees and wasps take up residence. Fungi and lichens flourish on the fallen tree's gnarly surface, and occasionally a marten hides behind the log.

Because of the woodpeckers' crucial role in maintaining healthy forests, researchers want to understand how these cavity-excavating birds choose trees to nest in. When forest managers know what these birds like, they can take steps to ensure that forests have these kinds of trees.

"Woodpeckers are ecosystem engineers," says Teresa Lorenz, a research wildlife biologist with the Pacific Northwest Research Station. "Many small animals compete for their vacated nests because they are so protected from the elements and other predators."

Ecosystem engineers are organisms that significantly modify, maintain, or destroy a habitat. In the woodpeckers' case, the species help build and maintain a habitat for other species by excavating nests in live and dead trees. When forest managers find many different woodpecker species in a forest, it often signals the presence of a thriving and diverse community of plants and animals.

KEY FINDINGS

- Across 818 snags in Yakima, Kittitas, and Chelan Counties in Washington's eastern Cascade Range, trees not used by birds had wood five times harder than trees that were. Such trees could not be used by birds simply because their wood was too hard for the birds to excavate.
- Within burns used by at-risk woodpeckers, the majority (86 to 96 percent) of seemingly suitable trees contained unsuitably hard wood; wood hardness limits nest site availability for these declining species.
- Researchers found no reliable visual cues to distinguish between suitable and unsuitable trees. Currently, the most effective management solution is to provide large numbers of snags, which can be difficult without the aid of fire.
- In dry forests, a mixed-severity fire that kills trees is an important but underappreciated strategy for providing enough snags for cavity-dependent species. Low-severity prescribed fires may not provide enough snags for these species.
- Suitable snags are limited, such that snag availability drives landscape-level habitat-selection by some species. For example, white-headed woodpeckers selected severely burned patches for nesting, which was initially puzzling because this species does not characteristically forage in burns.

Cavity-nesting birds typically nest in recent burns. More than 50 years ago, studies suggested that woodpeckers picked trees solely based on external tree- or habitat-level factors, like the size of the tree, the tree species, or what the vegetation cover is like near the nest.

Other studies pointed to wood decay, but used only visual indicators to assign a snag decay class as a proxy for measuring this. Lorenz felt there was a missing piece: few studies actually measured wood inside the trees where cavity-excavating birds had chosen to nest.

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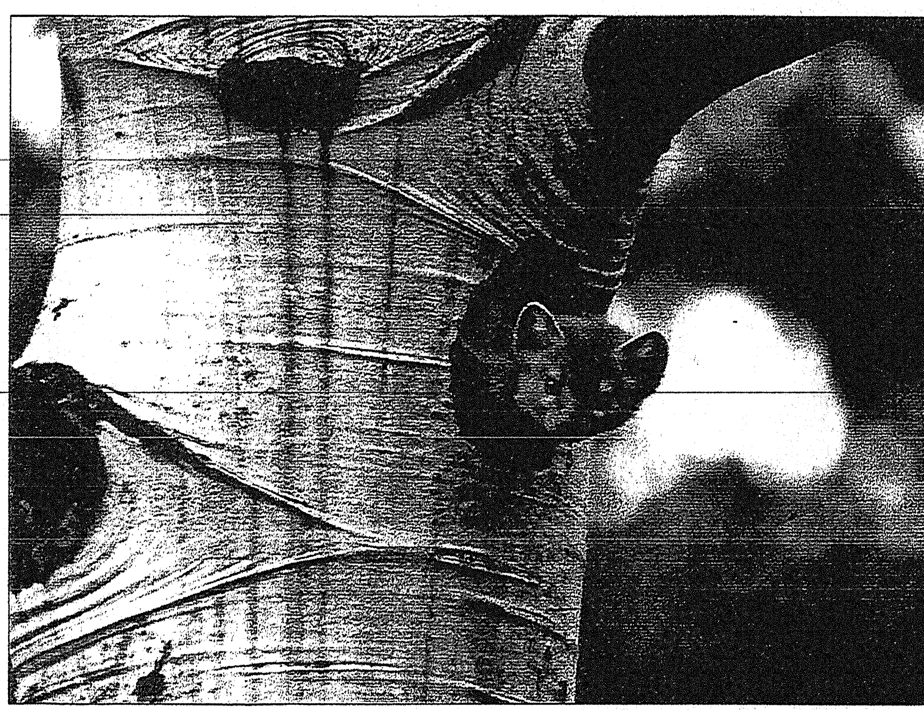


Teresa Lorenz

A pine marten looks out of a pileated woodpecker nest cavity. Many species require or prefer vacated woodpecker cavities, but cannot excavate the cavity themselves. Thus, they rely on woodpeckers to construct habitat for them.

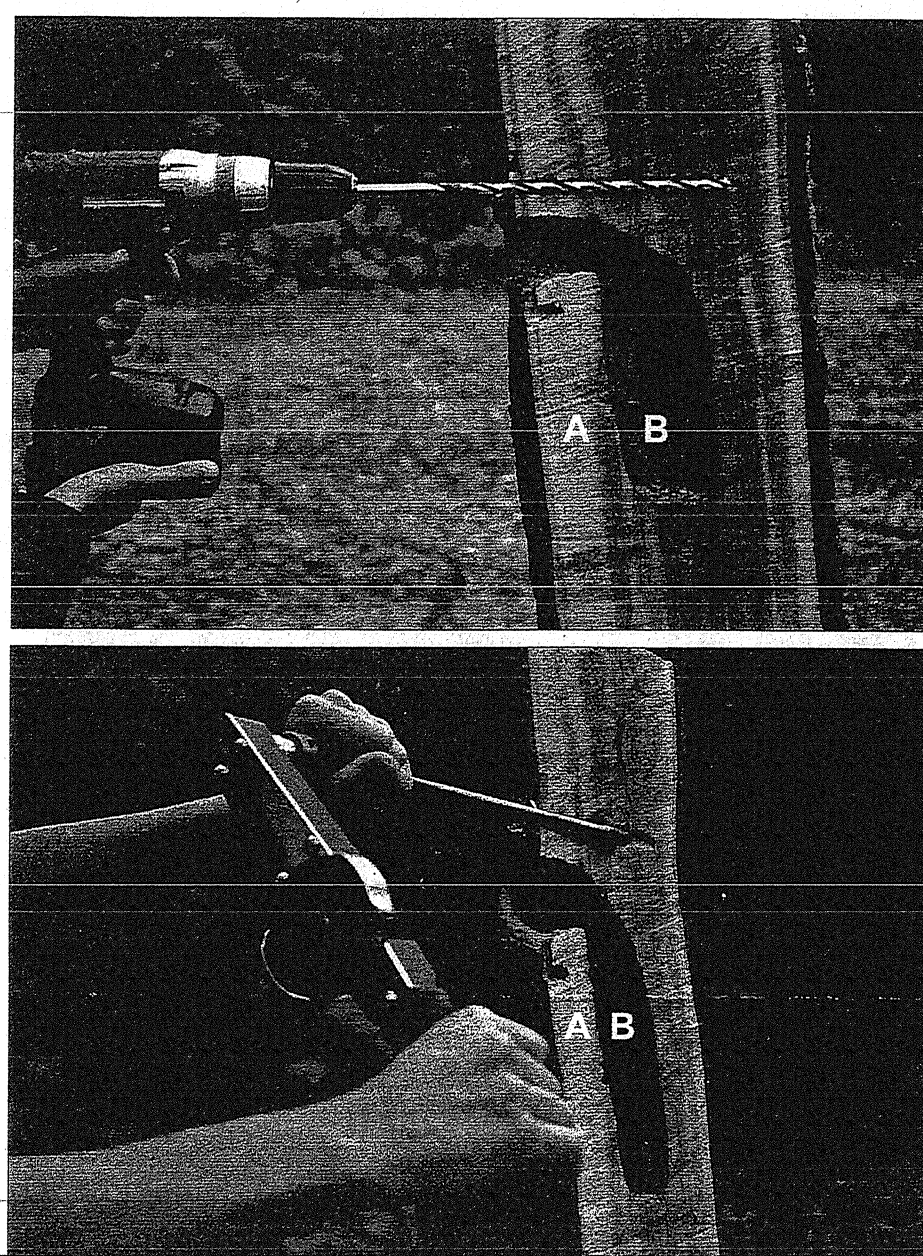
Matsuoka's Method

The idea for the study came to Lorenz and her colleagues in summer 2011, when they were investigating how far woodpecker nestlings wandered from their nests soon after they fledge. Using hole saws, they drilled into trees to get to the nestlings, to band them. "We noticed an enormous variation in the thickness of the wooden plug," Lorenz says. "Sometimes the nestling would be just inside the bark of the tree. Other times we had to drill through 4 inches of wood. We had all these hypotheses as to what was going on."

A colleague suggested that the variation could be due to the properties of the wood in the tree. Intrigued, they checked the literature. What if wood hardness had something to do with the birds' preference when looking for trees to nest in? Lorenz and her colleagues did not find many studies that addressed this question. "We were surprised that nobody had considered that this wood inside the tree was influencing where the birds were nesting," Lorenz says. So, she decided to investigate the question for her Ph.D. dissertation at the University of Idaho.

The problem was there weren't many reliable methods for measuring wood hardness in a forest setting. "But we found an obscure article in the *Japanese Journal of Ornithology* and this guy had figured out an ingenious way to measure the hardness of wood inside a tree, using a cordless drill, a foot-long drill bit, and an increment borer, without destroying the nest or harming the tree," Lorenz says.

An increment borer is a tool used by foresters and other researchers to extract a section of wood tissue so they can count a tree's annual rings and determine its age. Japanese researcher Shigeru Matsuoka's method involves inserting this tool into a pre-drilled hole above the nest cavity opening. The torque required to spin the increment borer would represent wood hardness. With the most important aspect of her field methods squared away, Lorenz set out with colleague Philip Fischer to gather field data in the eastern Cascade Range in Washington.



Teresa Lorenz

Teresa Lorenz

This photo shows the longitudinal section of an American three-toed woodpecker (Picoides dorsalis) nest and the procedure the study used to quantify wood hardness. First, the researchers use a drill to create a small hole about one-third inch in diameter above the nest cavity opening (top). Then they record the torque required to spin an increment borer into the predrilled hole (bottom). The area marked "A" represents the nest sill, and the area marked "B" represents the nest cavity body.

Lorenz had selected six species of cavity-excavating birds for the study. She picked three strong excavators: the three-toed woodpecker, the black-backed woodpecker, and the hairy woodpecker. She also picked a weaker guild of excavators, namely the northern flicker (*Colaptes auratus*), the white-headed woodpecker (*Picoides albolarvatus*), and Williamson's sapsucker (*Sphyrapicus thyroideus*).

They searched for nests in all major forest types for 3 years from March to July, 2011 through 2013, hiking through ponderosa pine, Douglas-fir, and western hemlock in lower elevations and through higher elevation forests of grand fir, subalpine fir, and western larch. Lorenz and Fischer located adult birds by broadcasting playbacks of their calls and drumming. "Then, we followed the adults until they led us to their nest cavities," Lorenz says. They marked the locations of occupied nests on portable GPS units. If they noted fresh woodchips on the ground surrounding the nests, they marked those as current-year excavations. They found nests in forest patches ranging from unburned to severely burned.

Lorenz and Fischer covered a lot of ground. Each of their 10 study sites were 1,483 to 7,413 acres in size. "You get in really good shape," Lorenz says.

Soft Pockets

Each carrying about 30 pounds of equipment, Lorenz and Fischer returned in summer to make their measurements. After ensuring the nests had been vacated by the birds, the researchers would scale a tree on climbing ladders to measure wood hardness. They drilled 2 inches above the nests because Matsuoka's study suggested this spot would be representative of the excavated wood.

"In the beginning, we had assumed that wood at the nests would be entirely soft, but that wasn't the case," Lorenz says. "There was some variability when we first started drilling into the trees. Some had hard surfaces. But when you get into the tree a bit, that's when you find these soft pockets. Other times, the soft pockets were close to the surface of the tree, and other times the tree was soft from the get go."

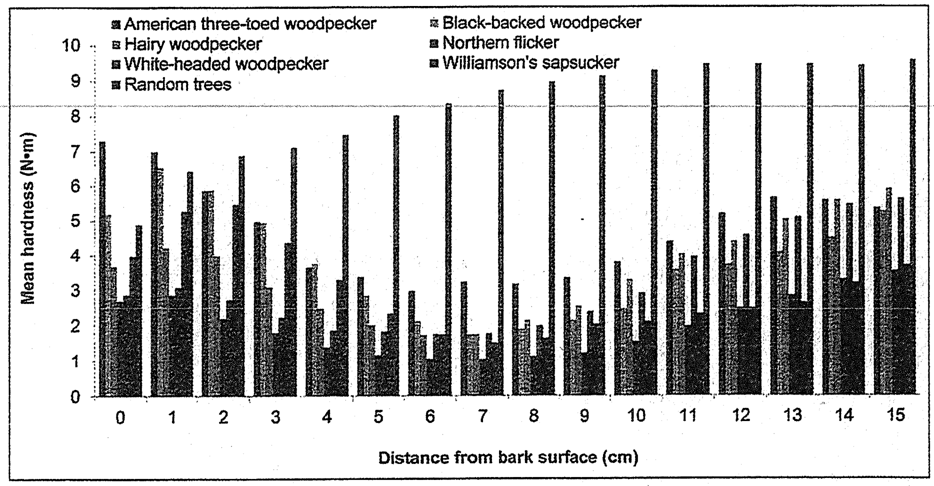
To determine if a tree's external features are reliable indicators of interior wood hardness, the researchers measured wood hardness in random trees within different decay classes.

Then the researchers measured vegetation features that were hypothesized to influence cavity-excavating birds' nest site selection in past studies, such as diameter at breast height of the nest tree; nest and tree height; orientation of the nest cavity entrance; proportion of the ground covered by shrubs; tree and snag density at the nest sites; and tree species. For nests found in snags, they examined the remaining bark, tree growth form, and other features to determine the tree's species.

"It was a pretty brutal field season," Lorenz says. "We worked from 6 a.m. to 4 p.m. And it got really hot in these sites."

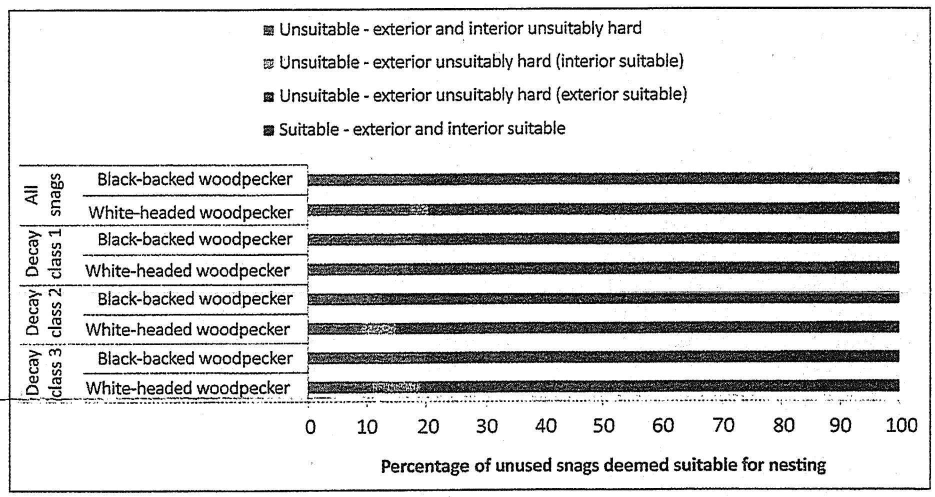
Lorenz and Fischer also used remote sensing data. To estimated prefire canopy cover at nest sites found in snags, they used gradient nearest neighbor classified Landsat satellite imagery captured between 2 and 8 years prior to each fire.

Lorenz was concerned about seeing an effect in nature, measuring it, but finding a lot of variability. "What if we went to another tree and that pattern didn't hold?" Lorenz says. "But in this study, we were able to measure so many trees that we were confident that the effect we were seeing was real. And it has major management implications."



Lorenz et al. 2015

Mean hardness at nests for six species of woodpecker compared to random sites in central Washington, from 2011 to 2013.



Lorenz et al. 2015

A small percentage of snags are suitable for woodpecker nesting. This chart shows the percentage of 360 unused snags in black-backed woodpecker and white-headed woodpecker nesting territories that were deemed suitable for nesting, based on wood hardness, in central Washington, from 2011 to 2013.

Drawn to Snags

Lorenz and her colleagues found that cavity-excavating birds preferred to nest in trees with significantly softer interior wood. They found that across 818 snags, trees not used by birds had wood five times harder than trees that were. These trees could not be used by birds simply because of the birds' physical limitations in pecking ability.

They found that at-risk species, namely the black-backed and the white-headed woodpeckers, were nesting within burns that contained 86 to 96 percent of trees with unsuitably hard wood. This suggests that past studies that did not measure wood hardness counted many sites as available to cavity-excavating birds when actually they were unsuitable. "By not accounting for wood hardness, managers may be overestimating the amount of suitable habitat for cavity-excavating bird species, some of which are at risk," Lorenz says.

In their study plots, the researchers did not find reliable visual cues to distinguish between suitable and unsuitable trees. Snag decay class was a poor indicator of internal wood properties—and this was not the first study to demonstrate that fact, although it was the first study to do so in ornithology (past studies had been done by foresters and published in forestry journals).

"Currently, the best solution we can recommend is to provide large numbers of snags for the birds, which can be difficult without fire," Lorenz says. According to the researchers' calculations, if one of every 20 snags (approximately 4 percent) has suitable wood, and there are five to seven species of woodpeckers nesting in a given patch, approximately 100 snags may be needed each year for nesting sites alone. This does not account for other nuances, like the fact that most species are territorial and will not tolerate close neighbors while nesting, or the fact that species like the black-backed woodpecker need more foraging options. Overall, more snags are needed than other studies have previously recommended.

Based on their results, Lorenz and her colleagues see the critical role that mixed-severity fires play in providing enough snags for cavity-dependent species. Low-severity prescribed fires often do not kill trees and create snags for the birds. In addition to benefiting the cavity nesters, a mixed-severity prescribed burn could also aid species that, historically, did not prefer to nest in snags. During their field work, Lorenz and Fischer observed that white-headed woodpeckers, a species traditionally associated with old-growth forest, used prescribed burns presumably because of shortages in suitable nest sites.

"I think humans find low-severity fires a more palatable idea. Unfortunately or fortunately, these birds are all attracted to high-severity burns," Lorenz says. "The devastating fires that we sometimes have in the West almost always attract these species of birds in relatively large numbers."

Many studies have shown that a severely burned forest is a natural part of western forest ecosystems. Snags from these fires attract insects that love to burrow beneath charcoal bark. And where there are insects, there are birds that love eating these insects.

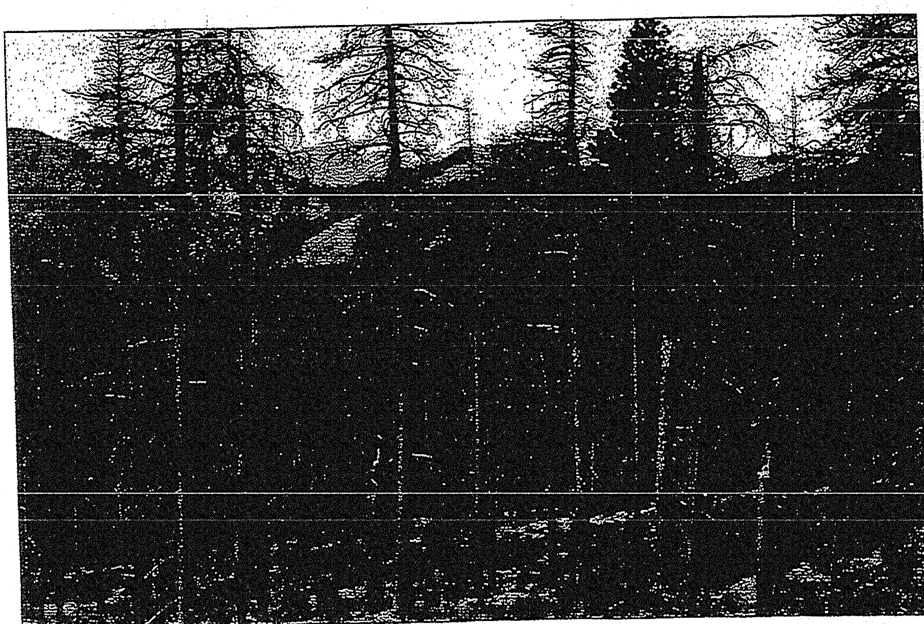
Lorenz and her colleagues stress that providing snags that woodpeckers can excavate is crucial for forest ecosystem health in the Pacific Northwest, where more than 50 wildlife species use woodpecker-excavated cavities for nesting or roosting.

"This study was just the beginning of what I hope will be decades of research," Lorenz says. Picking up on what she learned in this study, Lorenz is now collaborating with Forest Service colleague Michelle Jusino to investigate a kind of fungi that follows forest fires. "There is some mechanism by which the death of a tree somehow enables some species of fungi to decay the wood," she says. The research is bound to take her back into the forests of the Pacific Northwest, where woodpeckers follow forest fires in a timeless cycle of change.

"The powerful play goes on and you may contribute a verse."
—Walt Whitman, *Leaves of Grass*

LAND MANAGEMENT IMPLICATIONS

- Providing snags that woodpeckers can excavate is important for forest ecosystem health in the Pacific Northwest, where more than 50 wildlife species use woodpecker-excavated cavities for nesting or roosting.
- Mixed-severity prescribed-fire may be useful in creating breeding habitat for the white-headed woodpecker, a species traditionally associated with old-growth forest.


Teresa Lorenz

The prescribed burn conducted in this stand flared up in patches, creating a mosaic of burned and unburned trees (a mixed severity burn) that is still used by white-headed woodpeckers and black-backed woodpeckers 10 years after the fire. Tree death is usually required to initiate the process of wood decay—which softens the wood, making it suitable for woodpecker excavation.

For Further Reading

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Writer's Profile

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Doc. 2153

DEFINING OLD-GROWTH DOUGLAS-FIR FORESTS OF CENTRAL MONTANA AND USE OF THE NORTHERN GOSHAWK (*Accipiter gentilis*) AS A MANAGEMENT INDICATOR SPECIES

Diam of snags = 20

by

Thomas C. Whitford

B.S., University of Montana, 1986

A 27- acre stand was used to calculate OG.

Presented in partial fulfillment of the requirements for the degree of

Master of Science

UNIVERSITY OF MONTANA

1991

Approved by:

F. Earl Willard

Chairman, Board of Examiners

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Dean, Graduate School

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Date

20 snags/ac 7/10''

24

Whitford, Thomas C., M.S., August 1991 Wildlife Biology

Defining Old-growth Douglas-fir Forests of Central Montana and Use of the Northern Goshawk (*Accipiter gentilis*) as a Management Indicator Species (62 pages)

Director: Dr. E. Earl Willard E.E.W.

Old-growth Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) forest stands and Douglas-fir northern goshawk (*Accipiter gentilis*) nest stands were investigated in summers 1989 and 1990. Data were collected from 21 old-growth stands and 12 goshawk nest stands. Objectives of this study were to recommend refinements to an old-growth forest definition used by the Lewis and Clark National Forest (LCNF), to compare old-growth Douglas-fir stands with Douglas-fir goshawk nest stands in order to evaluate the appropriateness of the goshawk as a management indicator species (MIS) for old-growth Douglas-fir forests on the LCNF, and to examine the applicability of the nesting habitat portion of a goshawk habitat suitability model for the LCNF. Results indicated that old-growth Douglas-fir stands could be identified with minimum age and minimum dbh used as descriptors. Hence, simplification of old-growth definitions and development of definitions for each forest type were recommended for the LCNF. Differences between old-growth Douglas-fir stands and Douglas-fir goshawk nest stands were significant. The northern goshawk was a poor old-growth forest MIS on the LCNF. Land managers must identify a valid old-growth MIS or employ other methods in order to identify and manage old-growth forests. Index values produced by the goshawk habitat suitability model for each old-growth stand and goshawk nest stand verified that the model was successful in rating the nest stands higher than the old-growth stands. However, index values were virtually impossible to interpret so further refinement is necessary if the model is to be useful.

C). The two largest (>20 cm dbh) Douglas-fir trees in each of the 0.04 ha plots were cored and aged. If two large Douglas-fir trees were not located in the plot, large trees (>20 cm dbh) closest to the plot were used. Volume of downed logs (>10 cm 4" bottom diameter) in the 0.04 ha plots was estimated by measuring log length and the log's top and bottom diameters. Volume was calculated using the frustrum of a cone formula (J. Brown pers. comm.).

Small trees, seedlings, shrubs, tree canopy, and ground cover types were considered secondary old-growth components. Trees (0-20 cm dbh), seedlings (>50 cm tall) and shrubs (>50 cm tall) were tallied by species on the 0.007 ha plots. Tree canopy, seedling, shrub, and ground cover were calculated, on the first, third, and fifth 0.04 ha plots, using a method developed by Hejl (1989). Seven points, placed at 0.0, 3.9, 6.0, 7.5, 8.7, 9.8, and 10.8 m intervals, were located in each cardinal direction (N, E, S, and W) from plot center. The type of cover and species were noted after sighting upward and downward from each point. Percent cover was calculated by multiplying the number of points blanketed by each cover type by four. Ground cover types were litter, rock, soil, downed wood, grass, forb, water, and other.

Overall mean values, 95% confidence intervals, and ranges were calulated using the averages for each variable by stand (N = 21).

within the Pseudotsuga menziesii series but included other climax series habitat types as well. A complete list of all recorded plant species is in Table 1 (See Appendix A for scientific names).

Primary Old-Growth Stand Components

8"

Live trees. A total of 1,395 trees (> 20 cm dbh) were sampled on the study sites, 1,217 (87%) of which were Douglas-fir. Stem diameters (dbhs) ranged from 20.0 cm to 94.0 cm. Mean dbh was 36.0 cm, with a 95% confidence interval of 34.5 to 37.5 (Table 2). Large tree densities, by stand, ranged between 85 and 205 per 0.4 hectare (0.4 ha = 1.0 acre). Average large tree density was 130 per 0.4 ha with a 95% confidence interval of 116.5 to 143.5. Mean percent canopy cover was calculated for each old-growth stand. Mean canopy cover ranged from 41% to 77% with an overall mean of 55%. A 95% confidence interval for mean canopy cover was 50.2% to 59.8%.

7 8"

avg CC = 55%

Rang 41 – 77

Table 1. List of recorded plant species and numbers in each measured group. Species are listed according to prevalence.

Plant Species		Classification				
		Large[a]	Small[b]	Seed[c]	Shrub[d]	Note[e]
Tree Species						
Douglas-Fir		1217	382	677	---	---
Lodgepole Pine		64	13	17	---	---
Engelmann Spruce		52	10	26	---	---
Subalpine Fir		35	39	175	---	---
Ponderosa Pine		21	0	8	---	---
Limber Pine		6	4	6	---	---
Western Redcedar		0	9	6	---	---
Mountain Maple		0	0	48	---	---
	Total	1395	457	963		
Shrub Species						
Common Snowberry		---	---	---	876	---
Common Juniper		---	---	---	267	---
Mountain Gooseberry		---	---	---	193	---
White Spiraea		---	---	---	67	---
Buffaloberry		---	---	---	56	---
Wood's Rose		---	---	---	42	---
Blue Huckleberry		---	---	---	23	---
Mountain Snowberry		---	---	---	16	---
Chokecherry		---	---	---	10	---
Ninebark		---	---	---	4	---
Elderberry		---	---	---	1	---
	Total				1555	
Other Recorded Species						
Creeping Oregon Grape		---	---	---	---	X
Kinnikinnick		---	---	---	---	X
Red Raspberry		---	---	---	---	X
Twinflower		---	---	---	---	X
Dogwood		---	---	---	---	X

[a] Trees > 20 cm dbh.
[b] Trees 0-20 cm dbh.
[c] Seedlings 50 cm $\geq$ 0 $\leq$ 137 cm in height.
[d] Shrubs $\geq$ 50 cm in height.
[e] Plant species noted but numbers were not counted.

4"

Snags. A total of 481 coniferous snags ($\geq$ 10 cm dbh) were sampled. Stem diameters (dbhs) ranged from 10 cm to 89 cm. Mean snag dbh was 28 cm with a 95% confidence interval of 24.4 to 31.6 (Table 2). Snag heights ranged between 1.2 m and 27.0 m. Snag heights averaged 8.2 m

11"

19

with a 95% confidence interval of 7.7 to 8.7. Snag densities, by stand, ranged between 15.0 and 105.0 snags per 0.4 ha (Table 2). Average snag density was 45.0 per 0.4 ha with a 95% confidence interval of 35.0 to 55.0.

avg snag = 45/ac

Snag decomposition classes used were described by Cline et al. (1980) (Appendix B). Of snags recorded on each old-growth stand, an average of 25% were assigned to decomposition class 1, 24% to class 2, 23% to class 3, 21% to class 4, and 7% to class 5.

Downed logs. A total of 2,167 downed logs (> 10 cm bottom diameter) were tallied across all stands. Mean log density, calculated by stand, was 195.0 per 0.4 ha with a 95% confidence interval of 151.0 to 239.0 (Table 2). Log densities ranged from 35.0 to 385.0 per 0.4 ha. Average downed log volume, again calulated by stand, was 100.0 m^3 per 0.4 ha. The 95% confidence interval was 72.0 to 128.0. Average downed log volumes ranged between 4.0 and 166.0 m^3 per 0.4 ha.

Downed log decomposition classes used were described by Thomas et al. (1979) (Appendix C). Of downed logs measured on each old-growth study site, an average 3% were assigned to decomposition class 1, 13% to class 2, 40% to class 3, 25% to class 4, and 17% to class 5.

Secondary Old Growth Components

Tree canopy. The number of canopy layers was not measured in this study. However, the occurrence of two or more canopy layers can be noted for an old-growth stand based on the assumption that when small and large diameter trees occur together, two or more canopy layers are present in the stand. Therefore, those sample stands with at least 10 small trees (0-20 cm dbh) and 10 large trees [> 36 cm dbh (mean tree dbh)] recorded were tallied as old-growth forest stands with more than one canopy layer. A total of 13 of 21 (62%) sample old-growth stands exhibited more than one canopy layer.

Shrubs. Mean percent shrub cover, calulated by stand, was 16% with a 95% confidence interval of 10.6 to 21.4 (Table 2). Percent shrub cover ranged between 0.0 and 51.0%. A total of 1,555 shrubs (≥ 50 cm in height) were tallied on the study sites. Using mean shrub densities for each sample stand, shrub density averaged 715.0 per 0.4 ha. The 95% confidence interval was 287.4 to 1,142.6. Densities ranged between 41.0 to 4,235.0 per 0.4 ha.

Small trees. A total of 457 small trees (0-20 cm dbh) were tallied. Small tree densities, by stand, averaged 206 small trees per 0.4 ha (Table 2). The 95% confidence interval for small tree density was 31.4 to 380.6. Densities ranged from 0.0 to 1,787.5 small trees per 0.4 ha.

Seedlings. A total of 963 seedlings (50 to 137 cm in height) were tallied across the stands. Average seedling density, by stand, ranged from 0.0 to 2,213.8 per ha (Table 2). Mean seedling density was 413.0 per 0.4 ha with a 95% confidence interval of 168.2 to 657.8.

Ground cover. A total of 1,532 ground cover recordings were taken, 502 (34%) were recorded as litter, 436 (29%) as forb, 233 (15%) as downed wood, 216 (14%) as grass, 110 (7%) as other (moss, lichen, and seedlings), 27 (2%) as rock, and 8 (1%) as soil.

Table 2. Primary and secondary old-growth study stand components.

	N	$\overline{X}$	95% CI	Range
78 Live trees		14		
dbh[a]	21	36.0	36.0 ± 1.5	20.0-94.0
height[b]	21	17.0	17.0 ± 0.2	6.0-34.0
age[c]	21	268.0	268.0 ± 36.3	70.0-456.0
no./0.4 ha[d]	21	130.0	130.0 ± 13.5	85.0-205.0
Snags				
dbh	21	28.0	28.0 ± 3.6	10.0-89.0
height	21	8.2	8.2 ± 0.5	1.2-27.0
no./0.4 ha	21	45.0	45.0 ± 10.0	15.0-105.0
Downed logs				
no./0.4 ha	21	195.0	195.0 ± 44.0	35.0-385.0
vol/log	21	0.8	0.8 ± 0.15	0.02-11.6
vol/0.4 ha[f]	21	100.0	100.0 ± 28.0	4.0-166.0
Trees				
canopy cover[e]	21	55.0	55.0 ± 4.8	41.0-77.0
Shrubs				
cover	21	16.0	16.0 ± 5.4	0.0-51.0
no./0.4 ha	21	715.0	715.0 ± 427.6	41.0-4,235.0
Small trees				
no./0.4 ha	21	206.0	206.0 ± 174.6	0.0-1,787.5
Seedlings				
no./0.4 ha	21	413.0	413.0 + 244.8	0.0-2,213.8

[a] All diameters were measured in centimeters.
[b] All heights were measured in meters.
[c] Age was in years.
[d] All densities were in numbers per 0.40 ha (1.0 acre).
[e] Covers were expressed in percentages.
[f] Volume was expressed in square meters per 0.4 ha.

Comparison of Old Growth Study Results and Old Growth Specifications Currently Used

All 21 of the old-growth study sites fulfilled the currently used old-growth criteria of containing one or more dominant coniferous species which are climax or long-lived seral dominants. However, a total of 13 of the 21 (62%) old-growth sites displayed two or more canopy layers or age classes. Therefore, single canopy layered old-growth Douglas-fir stands were not uncommon, as 38% (8/21) of the study stands had single canopy layers.

55%

The tree canopy closure for the old-growth sample stands averaged 55%, which fell short of the ≥ 60% parameter of the currently used old-growth specifications. In fact, only 7 of 21 stands (33%) had mean canopy closures of 60% or more.

A total of 48% of the measured trees exceeded 33 cm (13 inches) dbh and 52% were greater than or equal to 15 m (50 feet) in height. ~~Overall, approximately 50% of the old-growth sample stands met the currently used criteria where dominant trees generally exceeded 33 cm~~ (13 inches) dbh and 15 m (50 feet) in height.

Indicated by the densities of snags and downed logs, decadence was ~~present and obvious in all 21 old-growth stands. Fifty-two percent of~~ the recorded snags were classified in decomposition classes three, four, and five (Appendix B). Eighty-eight percent of the recorded logs were classified in decomposition classes three, four, and five (Appendix C).

All sample old-growth stands except 3 (14%) had at least two snags per 0.4 ha (1.0 acre) of 25cm (10 inches) dbh or greater. In fact the

~~5~~ 10"

≥ 1 ac

20 snag/ac
710"

old-growth stands averaged 20 snags per 0.4 ha of this size class with a 95% confidence interval of 15.0 to 25.0.

Contrary to currently used old-growth specifications, a variety of shrubs were common and herbaceous plants were very common throughout the old-growth study sites.

Downed logs, woody debris, and litter were common throughout all of the old-growth stands. Table 3 contains the comparison of the currently used old-growth standards and those derived from this study.

Table 3. Comparison of values used in current LCNF old-growth definition and those resulting from this study.

Variable	Currently Used Value	Study Results
Climax or long-lived seral dominants	$\geq$ 1 species present	$\geq$ 1 species present
Canopy layers or age classes	$\geq$ 2	13/24 (54%) stands
Canopy closure	$\bar{X} \geq 60\%$	$\bar{X} = 55\%$
Dominant trees		
dbh	$\bar{X} \geq$ 13 inches	$\bar{X}$ = 14 inches
~~height~~	~~$\bar{X} \geq$ 50 feet~~	~~$\bar{X}$ = 56 feet~~
decadence	present and obvious	present and obvious
Snags	2 $\geq$ 10 in. dbh/acre	$\bar{X}$ = 5 snags 10 in. dbh/acre
Shrubs and herbaceous veg.	sparse	common
logs and downed material	common and well distributed	common and well distributed

All old-growth sample stands fully met the following parameters of currently used old-growth criteria: presence of one or more coniferous species which are climax or long-lived seral dominants, dominant trees exceed 33 cm dbh and 15 m in height, signs of decadence present and obvious, at least 2 snags per 0.4 ha of 25 cm dbh or greater, and logs and other downed material common and well distributed. Parameters not met were two or more layers or age classes, tree canopy closure that averages 60 percent or more, and sparse understory shrub and herbaceous vegetation.

DISCUSSION AND MANAGEMENT IMPLICATIONS

Old-growth forests are difficult to define. When establishing a usable definition, one needs to consider the variables that are likely to have ecological significance as well as being easily measured. With a basic understanding of a specific forest type, land managers need to consider the logical criteria on which to base a area and forest type specific old-growth definition. Important factors that must be addressed are the number of descriptors to be included in the definition and how narrow or flexible they are.

As expected, there are good aspects and bad aspects to old-growth definitions that are narrow or flexible. One advantage of a definition with flexible descriptors would be that old-growth forests could be "created" ~~in~~ areas where they are rare or nonexistent by allowing existing forests to develop into old growth with time (Hunter 1989). In this manner, those stands meeting the minimum requirements of a flexible definition can be preserved and allowed to evolve into a "true" old-growth stand. Another advantage of a flexible definition is that forest stands representing early, middle, and late old growth stages could be identified and preserved. A disadvantage of a flexible definition is that the amount of remaining old growth in an area may be over-estimated. This may lead land managers to believe that current harvest rates are acceptable when in fact they might be too high for sustained yield management practices.

Advantages and disadvantages of old-growth definitions with narrow descriptors also occur. Possibly the "best" old-growth stands will be identified and preserved when implementing a narrow definition, therefore eliminating or reducing over-estimation errors. A disadvantage of a narrow definition is that representative stands of early, middle, and late old-growth stages may not be preserved. Consequently, the number and variety of plant and animal species that inhabit or utilize old-growth forest ecosystems would possibly be reduced. Narrow definitions may also reduce the possibility of "creating" old-growth stands (Hunter 1989) in areas where they are rare or nonexistent. Obviously, care must be taken when creating old-growth forest definitions.

After working with several different Forest Service and private industry data gathering crews, I firmly support the simplistic approach of data gathering designs, especially after utilizing information stemming from different sources. The primary advantage of this approach is that it reduces biased information, especially since in most cases the information is supplied by individuals with varying backgrounds.

Undoubtedly it is difficult, if not impossible, to link a specific list of criteria to encompass all old-growth forest stands found in even a small area. Fortunately all the old-growth study sites had two parameters in common: they all supported large diameter, old trees. With this fact in mind, I propose that old-growth Douglas-fir forest definition on the Lewis and Clark National Forest be based primarily on the number of live trees of a specific dbh per unit area. These dominant trees must be at least 200 years.

The principal justification for such reasoning is that all structural characteristics of old-growth forests were linked to age, and the presence of large diameter trees was a reflection of age (≥ 200 years) in this study. Therefore if large, old trees are present, other old-growth structural attributes will theorectically evolve with time. Development of large diameter trees in central Montana is truly the time-consuming factor. Second, a major purpose of old growth definitions is to provide a standard criteria for land managers to identify old-growth stands and subsequently to formulate old-growth management strategies. Finally, a usable definition should be as simple as possible in order to limit implementation costs, bias, and confusion.

On the LCNF, old-growth forest definitions should be broken down by forest types and when necessary site conditions which can be deliniated through habitat types. For the Douglas-fir forest type (Table 4) further breakdown is unnecessary because there were no pronounced structural differences between old-growth Douglas-fir forests representing different Habitat Types (site conditions) in this study. Lower values from the 95% confidence intervals, calculated for the variables measured in this study, were used to define old-growth Douglas-fir forests. Sample old-growth stands averaged 74 trees ≥ 34 cm dbh (low CI value for live tree dbh) per 0.4 hectare with a 95% CI of 62 to 86. Tree ages averaged 268 years with a 95% confidence interval of 232 to 304. Therefore, old-growth Douglas-fir forests will contain a minimum of 62 live trees ≥ 34 13" cm dbh per 0.4 hectare and dominant trees (largest trees in the stand) are ≥ 200 years. Table 4 displays the proposed old-growth Douglas-fir forest definition for the LCNF.

Inclusion of other structural attributes in the definition are important in determining whether an old-growth stand is in early, middle, or late stages of development. When possible, management of old-growth forests in all stages of maturation is desirable because plant and animal species associated with old-growth ecosystems are likely to be linked to specific developmental stages and attributes.

Table 4. Minimum values for components included in the Douglas-fir forest type old-growth definition.

Site[a] Cond	For[b] Type	No.[c] TPH.dbh	Age[d]
All	DF	62>34cm (13)	≥200

62/ac
7 13"

[a] Old-growth forest component values may need further breakdown by site condition in some forest types. Ex. Lodgepole pine (LP) types.
[b] Described by dominant overstory species present.
[c] Minimum number of trees per 0.4 hectare of stated dbh.
[d] Minimum age of dominant live trees.

Again it is important to stress that old-growth forest ecosystems are extremely complex and thus difficult to define. This complexity compounds the difficulties of defining old-growth forests. Nevertheless, functional definitions are necessary for determining the amount of old-growth forests that remain in an area and subsequent management practices that will ensure their preservation.

Complicated definitions using a wide array of variables have been implemented in the past and newly formulated definitions continue with this trend. These definitions have provided limited success to the old-growth inventory process being initiated in central Montana as well as elsewhere. Undoubtedly the definitions can be simplified as the proposed old-growth Douglas-fir forest definition outlined in this paper suggests. Before more irreversable destruction and fragmentation occurs in these unique, complex forest ecosystems, usable definitions must be devised and agreed upon by land managers.

logs such as snag condition (after Cline et al. 1980, see Appendix B), and log condition (after Thomas et al. 1979, see Appendix C). The two largest (>20 cm dbh) Douglas-fir trees in each of the 0.04 ha plots were cored and aged. If two large Douglas-fir trees were not located in the plot, large trees (>20 cm dbh) closest to the plot were used. Volume of downed logs (>10 cm bottom diameter) in the 0.04 ha plots was estimated by measuring log length and the log's top and bottom diameters. Volume was calculated with the frustrum of a cone formula (J. Brown pers. comm.).

Small trees, seedlings, shrubs, tree canopy, and ground cover types were considered secondary old-growth components. Trees (0-20 cm dbh), seedlings (>50 cm tall), and shrubs (>50 cm tall) were tallied by species on the 0.007 ha plots.

Tree canopy, seedling, shrub, and ground cover were calculated, on the first, third, and fifth 0.04 ha plots, using a method developed by Hejl (1989). Seven points, placed at 0.0, 3.9, 6.0, 7.5, 8.7, 9.8, and 10.8 m intervals, were located in each cardinal direction (N, E, S, and W) from plot center. The type of cover and species were noted after sighting upward and downward from each point. Percent cover was calculated by multiplying the number of points blanketed by each cover type by four. Ground cover types were litter, rock, soil, downed wood, grass, forb, water, and other.

T tests were used to test the hypothesis that mean values of the measured variables that were derived from the old-growth stands were equal to the mean values from the goshawk nest stands.

on the old-growth stands and 82% were Douglas-fir on the nest stands. Of the measured live trees, 28% were ≥ 43 cm dbh on the old-growth stands, whereas only 10% were ≥ 43 cm dbh on the goshawk nest stands (Figure 5).

Mean live tree dbh was 36 cm for old-growth stands and 31 cm dbh for nest stands (Table 5). The t-test determined that these mean live tree dbhs were significantly different at the 1% level. Mean live tree ages were also calculated for each study stand. Using mean values from each stand, overall mean live tree age was calculated at 268 years for old-growth stands and 203 years for goshawk nest stands (Table 5). Mean live tree ages were significantly different at the 1% significance level. Overall live tree density was 130 per 0.4 ha for old growth and 190 per 0.4 ha for goshawk nest stands (Table 5). Difference in tree density between stand types was statistically significant at the 1% level

12" mean dbh nest

203 yr.

goha more dense

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Conservation implications of a multi-scale study of Flammulated Owl habitat use in the northern Rocky Mountains, USA.

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Abstract.—Our multi-scale analysis of Flammulated Owl (*Otus flammeolus*) habitat use in the northern Rocky Mountains indicates some landscapes may be unsuitable for this species. As a result, there may be less habitat available for Flammulated Owls than thought based on the results of microhabitat studies. Thus, we suggest Flammulated Owl habitat conservation measures be based on the results of landscape-level, as well as microhabitat studies. Habitat conservation and restoration efforts in the ponderosa pine ecosystem should retain large trees, large snags, understory tree thickets, and grassland openings within landscapes that contain an abundance of suitable forest types.

INTRODUCTION

Effective conservation strategies cannot be designed without understanding the distributions of rare species. Bird distributions are heavily dependent on habitat distribution (reviews in Cody 1985), partly because populations are limited by the availability of suitable habitat. Thus, identifying and maintaining adequate amounts of suitable habitat are critical to supporting the population sizes and population structures necessary for long-term species viability.

Flammulated Owls in the central Rocky Mountains (Hayward 1986, Reynolds and Linkhart 1992) and Blue Mountains (Bull et al. 1990) predominantly nest and forage in old-growth ponderosa pine (*Pinus ponderosa*) forests, suggesting the species depends on the ponderosa pine ecosystem for population viabililty in some geographic areas. This ecosystem has been heavily altered by past forest management in the northern Rocky Mountains. Specifically, the removal of overstory ponderosa pine since the early 1900's and nearly a century of fire suppression have led to the replacement of most old-growth ponderosa pine forests by younger forests with a greater proportion of Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) than ponderosa pine (Habeck 1990). Clearcut logging and subsequent reforestation have converted many older stands of ponderosa pine/Douglas-fir forest to young structurally-simple ponderosa pine stands (Wright and Bailey 1982).

Fire scar evidence in the northern Rocky Mountains indicates that ponderosa pine forests burned approximately every 1-30 years prior to fire suppression, preventing contiguous understory development and, thus, maintaining relatively open ponderosa pine stands (Arno 1988, Habeck 1990). In old forests that retain a ponderosa pine overstory, a century of fire exclusion has permitted development of a more contiguous dense Douglas-fir understory (Mutch et al. 1993). Those U.S. Forest Service personnel entrusted with the management of national forests in the northern Rocky Mountains are currently investigating techniques to remove understory Douglas-fir and return pre-European-settlement fire regimes to ponderosa pine ecosystems (Mutch et al. 1993). National Forests such as the Bitterroot and Lolo National Forests in west-central Montana are proposing to restore old-growth ponderosa pine

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rather than the xeric grassland understory used by Flammulated Owls in our study area. Owls in those areas might have been solicited through tape playbacks from adjacent xeric stands, or they might use more mesic habitat types in the moister landscapes of northern Idaho and northwestern Montana.

Thus, specific results of our study may not be applicable in regions with different habitats, including areas with aspen or areas without xeric ponderosa pine/Douglas-fir forest. Two types of ponderosa pine forest that existed in our study area were not surveyed during our study. Old-growth ponderosa pine forests occur along many south-facing slopes in the Bitterroot Mountains. These slopes were too steep and rocky to safely traverse at night, and the creek noise from spring runoff was too loud to survey these areas from gentler slopes high above the canyons. The understory vegetation on these slopes was sparse, and may represent lower quality foraging habitat than under the more contiguous ponderosa pine forests that occur on gentler slopes. Additionally, Flammulated Owls in the southeastern region of the study area used home ranges with a lower slope gradient, and it is possible these slopes are too steep to be suitable. Thus, forests on these south-facing slopes represent a different, unsurveyed habitat type that may or may not be suitable. Ponderosa pine also occurred in association with black cottonwood along terraces of the Bitterroot River (Habeck 1990). Based on the presence of cottonwoods, which often have an abundance of cavities, such forests would be expected to contain an abundance of suitable nest trees. Most of these terraces in the study area occur on private land, and many of the large ponderosa pine were removed when the land was settled in the early 1900's (Habeck 1990). Intact examples of this forest type along the Bitterroot River were rare and were not surveyed for Flammulated Owls. Thus, study results are not applicable to these forest types.

Understory Vegetation

There was no significant difference in the amount of understory Douglas-fir in occupied and unoccupied plots in the BEMRP study in west-central Montana. However, other researchers have noted the importance of understory thickets to Flammulated Owls. For instance, while stands of dense young trees in New Mexico or Oregon (McCallum and Gehlbach 1988, Bull 1990) were not suitable as nest sites, thickets of dense vegetation were present near all nests, and were used for roosting and singing in New Mexico (McCallum and Gehlbach 1988). Reynolds and Linkhart (1992) also observed males singing within dense clumps of foliage, and Flammulated Owls in eastern Oregon predominantly roosted in dense stands with >50% canopy cover. Mean stem density in roost sites observed by Goggans (1986) was 2016 trees/ha (SD = 1378, n = 31, range 509-5346), with mean basal area of 129 m^2 (SD = 48.5, n = 31, range 21-239). Flammulated Owl use of dense forest thickets was also recorded by Bull and Anderson (1978) and Marcot and Hill (1980).

Because Flammulated Owls use both ponderosa pine and Douglas-fir dominated forest types, the recent floristic change in many ponderosa pine forests to predominantly Douglas-fir might not be expected to affect Flammulated Owl occupancy of stands. However, there are no data on reproductive success in the two forest types. The change in forest structure, from a low canopy cover forest with openings and patchy understory thickets, to a contiguous high canopy forest with fewer openings, might decrease food availability for Flammulated Owls. Densities of orthopteran prey in grassland are greater than in forest, and ponderosa pine/Douglas-fir forests with open canopies have greater food availability than continuous forests (Goggans 1986). For instance, based on insect window trap stations in eastern Oregon, 2.7 times as many prey items occurred in ponderosa pine/Douglas-fir forest, and 8.7 times as many prey items occurred in grassland, than in mixed conifer forest (Goggans 1986). Thus, stands with dense understories probably contain less prey, and hinder foraging maneuverability (Goggans 1986).

While the elimination of some understory forest would be expected to maintain the grassland openings used by foraging owls, management activities that eliminate all understory Douglas-fir may remove thickets important for roosting and singing, for drop-pounce foraging perches, and for predator protection cover. Flammulated Owls roosted an average of 53 m from nests during the nesting period, and <20 m from nests prior to juvenile fledging; therefore, Goggans (1986) suggested that suitable nest-sites may include patches of dense forest for roosting, as well as openings for foraging.

174' 66'

2004 a

Snow depth
energetic stress

Effects of Mid-winter Snow Depth on Stand Selection by Wolverines, *Gulo gulo luscus,* in the Boreal Forest

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Wright, Jonathan D., and Jessica Ernst. 2004. Effects of mid-winter snow depth on stand selection by Wolverines, *Gulo gulo luscus*, in the boreal forest. Canadian Field-Naturalist 118(1): 56-60.

Wolverines (*Gulo gulo luscus*) in a study area in the boreal upland forests of northwestern Alberta and northeastern British Columbia (approximately 57°N) were noted to be limited to upland landscapes, despite abundant food in adjacent lowland landscapes. Snow-tracking suggested that the species was selecting for the densest climax conifer stands for travel in search of food. It was hypothesized that snow depth was a limiting factor for Wolverines in the boreal forest during midwinter, and that they selected for this stand-type because of the buffering effect of this type of canopy on ground snow-depths. A series of snow-depth measurements were collected. Snow depths collected along Wolverine trails were very significantly lower than random snow depths collected under upland canopy ($F = 32.84$, df = 1, $P << 0.010$). There was a significant buffering effect on snow depth indicated for upland canopy ($F = 11.1$, df = 1, $P < 0.010$), while adjacent lowland canopy had no significant buffering effect on snow depth ($F = 3.45$, df = 1, $P > 0.05$). Wolverines were hypothesized to be limited to upland landscapes in the study area because of the buffering effect on snow-depth of the stand types found there, and not for reasons of food availability. Climax conifer stands were interpreted as being of high importance to Wolverine survival during winter. Conservation implications include the detrimental effect on Wolverine populations likely to result from current timber harvesting practices in the boreal forest.

Key Words: Wolverine, *Gulo gulo luscus,* snow depth, boreal forest, stand, landscape, climax, buffering effect, timber harvesting, Alberta, British Columbia.

The Wolverine, *Gulo gulo luscus,* remains among the least understood of the world's forest mesocarnivores. Recent research focusing on the life history of the species in cordilleran landscapes is beginning to fill knowledge gaps. However, landscape features documented as being of high importance to Wolverines in mountainous regions – for example, the alpine zone (Eric Lofroth, personal communication [see Acknowledgments section for affiliation]) – are unavailable to forest populations.

In the boreal forest, Wolverines are believed to be declining from densities that appear to be lower than in any of the other landscapes in which they occur (Banci 1994). As they are less observable in the forest due to the unbroken forest canopy, Wolverine study in the forest ecoprovinces is an even greater challenge than in alpine habitats. It is therefore not surprising that Banci (1994) has suggested that stand-level and landscape scale "habitat use by Wolverines in forests has not been adequately investigated", and that research is needed to study the "habitat needs of Wolverine in forests, because there is no sound basis for developing habitat management prescriptions at the stand level". In forested areas of the northwest, the Wolverine is the furbearer about which wildlife agencies have the greatest concern (Bill Johnson, personal communication).

There has been agreement among researchers that Wolverine "habitat is probably best defined in terms of adequate year-round food supplies in large, sparsely inhabited wilderness areas, rather than in terms of particular types of topography or plant associations" (Kelsall 1981)*. Seasonal shifts by Wolverine in cordilleran landscapes from the alpine zone in the summer to subalpine forest in the winter have been attributed to the availability of food (Banci 1994), or are thought to be related to avoidance of high temperatures or humans (Hornocker and Hash 1981). While snow depth has been investigated as a limiting factor for Fishers, *Martes pennanti* (Krohn et al. 1995; Raine 1983), it has not been implicated as a factor affecting habitat selection in the Wolverine.

This paper presents the hypothesis that Wolverines in the boreal forest are limited by mid-winter snow conditions, which in turn affects stand, and in this case, landscape selection.

Methods

Snow tracking is being increasingly recognized as a reputable scientific tool in wildlife studies, management and conservation, with efforts being made to establish standardized terminology, institute university courses on the subject, and to establish networks of trackers throughout the North American continent (Rezendes 1999; Zielinski and Kucera 1995). Snow tracking may be the only practical way of learning details of Wolverine habits and habitat use, as such details are not adequately provided by radio-telemetry studies (John Krebs, personal communication; Eric Lofroth, personal communication).