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Terrestrial Species Viability Assessments for National Forests in Northeastern Washington

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Abstract

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We developed a process to address terrestrial wildlife species for which management for ecosystem diversity may be inadequate for providing ecological conditions capable of sustaining viable populations. The process includes (1) identifying species of conservation concern, (2) describing source habitats, and other important ecological factors, (3) organizing species into groups, (4) selecting surrogate species for each group, (5) developing surrogate species assessment models; (6) applying surrogate species assessment models to evaluate current and historical conditions, (7) developing conservation considerations, and (8) designing monitoring and adaptive management. Following the application of our species screening criteria, we identified 209 of 700 species as species of concern on National Forest System lands east of the Cascade Range in Washington state. We aggregated the 209 species of conservation concern into 10 families and 28 groups based primarily on their habitat associations (these are not phylogenetic families). We selected 32 primary surrogate species (78 percent birds, 17 percent mammals, 5 percent amphibians) for application in northeastern Washington, based on risk factors and ecological characteristics. Our assessment documented reductions in habitat capability across the assessment area compared to historical conditions. We combined management considerations for individual species with other surrogate species to address multiple species. This information may be used to inform land management planning efforts currently underway on the Okanogan-Wenatchee and Colville National Forests in northeastern Washington.

Keywords: Viability assessments, northeastern Washington, surrogate species, conservation considerations.

Summary

Regulations and directives associated with enabling legislation for management of national forests require maintenance of viable populations of native and desired nonnative wildlife species. Broad-scale assessments that address ecosystem diversity may cover assessment of viability for most, but not all, species. We developed a process to address those species for which management for ecosystem diversity may be inadequate for providing ecological conditions capable of sustaining viable populations. The process includes (1) identifying species of conservation concern; (2) describing source habitats and other important ecological factors; (3) organizing species into groups; (4) selecting surrogate species for each group; (5) developing surrogate species assessment models; (6) applying surrogate species assessment models to evaluate current and historical conditions; (7) developing conservation considerations; and (8) designing monitoring and adaptive management strategies. Following the application of our species screening criteria, we identified 209 of 700 species as species of concern on National Forest System lands east of the Cascade Range in Washington. We aggregated the 209 species of conservation concern into 10 families and 28 groups based primarily on their habitat associations (these are not phylogenetic families). We selected 32 primary surrogate species (78 percent birds, 17 percent mammals, 5 percent amphibians) for application in northeastern Washington, based on risk factors and ecological characteristics. Our assessment documented reductions in the viability outcomes for all surrogate species compared to historical conditions. The species for which current viability outcomes were most similar to historical viability outcomes included the golden eagle, harlequin duck, northern goshawk, peregrine falcon, and Wilson's snipe. Species for which current viability outcomes have departed the most from historical viability outcomes and are of greatest concern included the eared grebe, fox sparrow, sage thrasher, western bluebird, and white-headed woodpecker. To address such changes, we identified conservation considerations for each surrogate species that included habitat protection and restoration and amelioration of risk factors. We combined individual species with other surrogate species and with management proposals for other resources (e.g., recreation, fire, and fuels management) to develop multispecies, multiresource management considerations. The information generated from our approach could be used to inform land management planning to help improve the probability that desired population outcomes will be achieved. However, practitioners should note that a conservation planning process, such as ours, cannot remove all uncertainty and risk to species viability, warranting an adaptive management approach.

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Chapter 1: Terrestrial Species Viability Assessments: Process and Overall Results

Introduction

The National Forest Management Act (NFMA) of 1976 (Public Law 94-588) and the Multiple-Use Sustained-Yield Act of 1960 (Public Law 86-517) require maintenance of diversity and sustainability of plant and animal communities on National Forest System lands throughout the United States (Marcot and Murphy 1996). Associated regulations and directives call for providing viable populations of native and desired nonnative wildlife with an emphasis on those species considered to be at risk (Suring et al. 2011). The assessment of population viability for species of concern is one component of an evaluation of ecosystem sustainability (Linder et al. 2004). Comprehensive analyses of ecosystem sustainability may be accomplished through a hierarchical approach that addresses ecosystem diversity and species diversity. Guidance on the assessment of species diversity within the Pacific Northwest Region of the U.S. Department of Agriculture, Forest Service was developed to help improve efficiencies, reduce costs by eliminating redundancy in analyses, provide a forum for a rigorous science review of the process, and provide consistency across the region as national forests or groups of national forests revise their land and resource management plans (USDA FS 2006). This document presents the application of the regional guidance to the northeast Washington state assessment area, a cluster of two national forests that are among the first in the Pacific Northwest Region to revise their land and resource management plans. We defined the assessment area to include all of the watersheds (5^{th}-level hydrologic unit code [HUC]) that contained any amount of Forest Service land under the management of the Okanogan-Wenatchee or Colville National Forests. We defined the plan area as the combined land managed by the Okanogan-Wenatchee and Colville National Forests, a subset of the assessment area (fig. 1).

This document presents a process that was developed under the 1982 Planning Rule (Section 219.27, as amended in 1983). However, the process is consistent with the 2012 Planning Rule (36 CFR 219.9(b)) and associated directives (FSH 1909.12 chapter 10, 12.5).

The initial focus of our application of the assessment process was an evaluation of ecosystem diversity, which considered the maintenance of functioning native ecosystems within the assessment area, and the extent to which maintaining ecosystem diversity will also sustain populations of animal species within their ranges in the plan area (Samson 2002, Samson et al. 2003). This is referred to as a "coarse-filter" approach to conservation (Baydeck et al. 1999, Hunter et al. 1988,

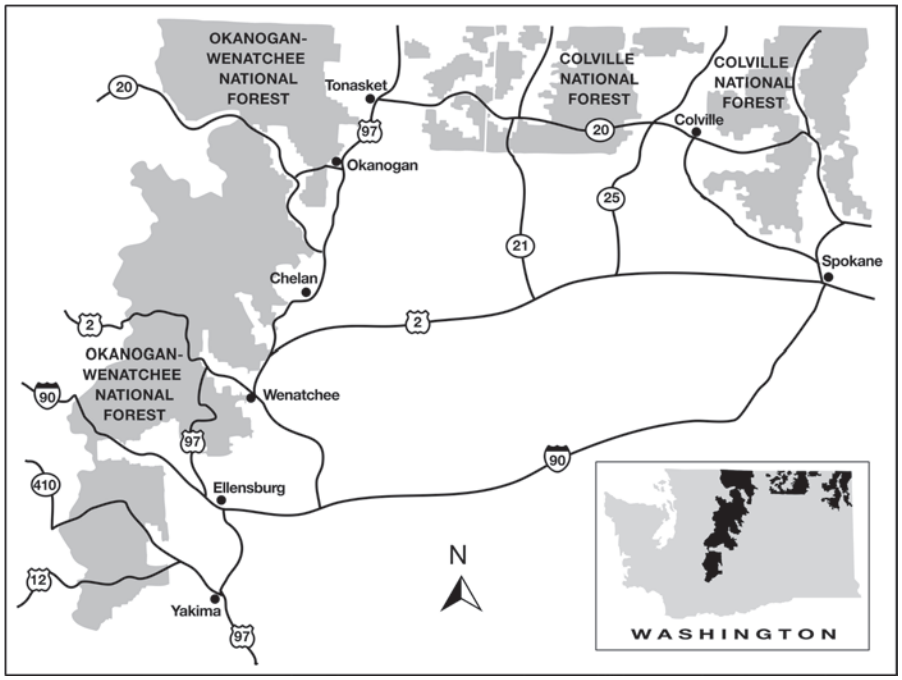

Figure 1—Area of the Okanogan-Wenatchee and Colville National Forests that makes up the northeast Washington planning area.

Landres et al. 1999, Samson 2002, Samson et al. 2003). The coarse-filter evaluation of ecosystem diversity generally compares the amount and distribution of existing vegetation communities to a set of reference conditions (e.g., pre-European settlement, historical range of variability [HRV]) to evaluate current representation of vegetation communities across the plan area (Samson 2002). For national forests located within the interior Columbia Basin, coarse-filter assessments were completed as part of the Interior Columbia Basin Ecosystem Management Project (ICBEMP) (Hann et al. 1997, Hessburg et al. 1999a). These assessments included evaluations of existing vegetation communities compared to the HRV and of changes that have occurred in the amount, effectiveness, and connectivity of habitats for surrogate wildlife species (Hessburg et al. 1999a, Lehmkuhl et al. 2001, Raphael et al. 2001, Wisdom et al. 2000).

The 2012 planning rule uses a coarse-filter approach by managing for "ecological sustainability" (36 CFR 219.8) along with "ecosystem plan components" to maintain and restore ecosystem integrity (36 CFR 219.9 (a)). The directives associated with the 2012 Rule suggest the use of natural range of variation (NRV) as the ecological reference model (FSH 1909.12 ch10 12.14a). This coarse-filter approach is intended to provide appropriate conditions to maintain most species.

A complementary approach to a coarse-filter analysis is necessary for species for which ecological conditions needed to maintain populations may not be completely provided for by merely maintaining ecosystem diversity (Samson 2002). For example, species associated with fine-scale ecosystem components (Samson 2002) or habitat generalists influenced by human activities such as roads (Carroll et al. 2001) may not be adequately addressed by a broad-scale assessment of vegetation conditions (Cushman et al. 2007). In these cases, a species-specific approach to the analysis and establishment of plan direction may be necessary. The assessment of individual species is a "fine-filter" approach to conservation (Andelman et al. 2001, 2004; Holthausen et al. 1999, Holthausen 2002, Samson et al. 2003). Holthausen (2002) and Andelman et al. (2001) provided valuable suggestions on how to conduct assessments of species diversity. In addition, Forest Service regulations (36 CFR 219.9) and directives, (FSH 1909.12 chapter 10, 12.5) and regional guidance (USDA FS 2006) provide guidelines for conducting assessments of the viability of species of conservation concern.

This document details the terrestrial species assessment process, results of the assessment, and the management considerations for the conservation of key elements of terrestrial diversity.

This document details the terrestrial species assessment process, results of the assessment, and the management considerations for the conservation of key elements of terrestrial diversity. The document is divided into four chapters: chapter 1 presents the assessment process and summarizes overall results, chapter 2 presents results and management considerations for the individual surrogate species, chapter 3 brings together results of the individual surrogate species assessment into multispecies assessment, and chapter 4 discusses priorities for monitoring based on assessment results, general monitoring methods, and how results can be used in an adaptive management framework.

Terrestrial Species Viability Assessment Process

The process we used to assess the viability of terrestrial species included the following major steps (Suring et al. 2011):

- Identify species of conservation concern
- Define source habitats for species of conservation concern
- Group species of conservation concern
- Identify the ecological relationships of species of conservation concern
- Select surrogate species
- Develop surrogate species assessment models
- Assess viability outcomes for surrogate species
- Evaluate habitat conditions for conservation planning

Identifying Species of Conservation Concern

A process for identifying the full set of species for a geographic area that may be at risk because of future management actions is very complex, and a universally accepted approach is not available (Holthausen et al. 1999, Raphael and Marcot 1994). Numerous approaches are applied by natural resource management agencies to classify species according to their risk of extirpation or extinction at regional (Breininger et al. 1998, Lunney et al. 1996, Millsap et al. 1990), national (Czech and Krausman 1997, Molloy and Davis 1992), and international (IUCN 2000) scales. Andelman et al. (2004) recommended using the global species ranks from the Natural Heritage Program (Master 1991, Master et al. 2000) as a system appropriate for use by the Forest Service to address the agency's legal requirements (Holthausen et al. 1999, Raphael and Marcot 1994). They recommended using these rankings because many of the species that occur on National Forest System lands have been evaluated, the database with species ranks is readily available, and the Natural Heritage Program process may be the most suitable existing protocol for identifying species of concern. However, Andelman et al. (2004) noted that the initial protocol used by the Natural Heritage Program to rank species (Master 1991, Master et al. 2000) did not explicitly incorporate weightings for threats.

We developed a process to identify species of conservation concern (Suring et al. 2011). Our process is consistent with the 2012 planning rule and associated directives for identifying species-at-risk and species of conservation concern (SCC) (FSH 1909.12 chapter 10, 12.52).

We identified the following criteria to identify SCC:

1. Species listed as endangered, threatened, candidate, or proposed under the U.S. Endangered Species Act.
2. Species that had been petitioned for listing under the U.S. Endangered Species Act and have received a determination of "may be warranted" or "warranted but precluded."
3. Species with the following ranks from the Natural Heritage Program as documented on NatureServe (2009):
    a. G-1 through G-3.
    b. Intraspecific (subspecific) taxa with ranks of T-1 through T-3.
    c. S-1 through S-3.
4. Species listed by Washington Department of Fish and Wildlife as threatened or endangered.
5. Species on the U.S. Fish and Wildlife Service birds of conservation concern national priority list (U.S. Fish and Wildlife Service 2002).

6. Bird species in the Partners in Flight species assessment database (http://www.rmbo.org/pif/pifdb.html) with scores indicating a moderate to large population decline or severe to extreme threats to populations (Carter et al. 2000).
7. Species identified as a "terrestrial vertebrate species of focus" from the ICBEMP (Lehmkuhl et al. 1997, Wisdom et al. 2000). Species from the ICBEMP list were included unless they met one or more of the following criteria:
   a. Wisdom et al. (2000) concluded a positive or no change in source habitats in the ecological reporting units (ERU) overlapping Washington, and no other published reasons for concern were found.
   b. If a more recent assessment of populations (e.g., breeding bird survey data) or expert opinion was available that indicated there was not a current reason for concern.
8. Species described by Raphael et al. (2001) as having fragmented populations that are currently vulnerable to extirpation or extinction, especially if they were abundant historically.[1]
9. Species listed by Washington state as strategy species in its comprehensive wildlife conservation strategies.

The focus in the application of this process was on species that are of regional or local conservation concern as indicated by documented risks to populations or habitats. All native terrestrial vertebrates known to occur on land managed by the Forest Service east of the crest of the Cascade Range in Washington were evaluated. Accidental species were not included nor were extirpated species without near-term plans or opportunities for reintroduction. Note that this process does not include species of public interest for hunting, trapping, or other consumptive or commercial uses unless their populations were determined to be at risk (e.g., bighorn sheep).

Definition of Source Habitats

Concerns have been raised about using habitat as an indicator of how populations may respond to environmental changes. For example, Cushman et al. (2007) evaluated the use of cover type or successional stage to predict the abundance of

[1] Species with a population outcome of D or E under the current condition scenario, or outcome C with a decline from an historical outcome of A or B on National Forest System or Bureau of Land Management lands as defined in Raphael et al. (2001).

birds in a forested environment and found that either variable used alone was a poor predictor. When they used cover type and successional stage in combination, as we did, they found more reliable predictions, although the accuracy varied among bird guilds. They concluded that while habitat-relationship models are a necessary guide for management and conservation, they do not provide an effective surrogate for measuring population levels. We addressed the concerns raised by Cushman et al. (2007) by including variables, such as fine-scale habitat variables (e.g., snags, downed logs) and risk factors (e.g., roads, invasive species), in addition to cover type and structural stage in our evaluation of surrogate species viability. We evaluated our models using independent data on species distribution and abundance (for those species we could), and we included options for monitoring species populations and distribution for surrogate species with poor viability outcomes. We concur with Cushman et al. (2007) that monitoring habitat alone may not be an effective replacement for monitoring species population size and distribution.

We defined source habitats as those providing characteristics of macrovegetation that contribute to stationary or positive population growth (Wisdom et al. 2000). Source habitats are distinguished from habitats simply associated with species occurrence; such habitats may or may not contribute to long-term population persistence (Wisdom et al. 2000). The macrohabitats used by each of the species considered in our assessment were described using cover type and structural stage. We included habitats used for reproduction, movement, and cover (e.g., protection, thermoregulation) as described by Johnson and O’Neil (2001), other primary literature, and professional judgment.

Vegetation for the east side of Washington was classified using a combination of cover types and structural classes (tables 1 and 2) similar to those described in Johnson and O’Neil (2001). Some of the Johnson and O’Neil (2001) cover types were combined to better match the vegetation classification used by the Forest Service. Also, a postfire cover type was included to identify vegetation that occurred immediately following stand-replacing fires. Six types of riparian habitat were also described.

The 26 classes of Johnson and O’Neil (2001) were condensed into 14 to reduce the types to a manageable list for our assessment (table 1). Two canopy closure breaks, open (equal to or less than 50 percent canopy closure) and closed (more than 50 percent) were used because they most effectively characterized the habitat relationships of wildlife species of conservation concern based on extensive review of the literature. Structural condition classes were described only for upland forested habitats (table 2).

Table 1—Cover types used to describe source habitats and their relation to the Johnson and O'Neil (2001) habitat type classification

Cover type	Habitat types[a]
Open water	Open water
Marsh	Herbaceous wetlands
Wet meadow	Herbaceous wetlands
Coniferous riparian	Montane coniferous wetlands east-side (interior) riparian wetlands
Deciduous riparian/shrub wetland	Montane coniferous wetlands east-side (interior) riparian wetlands
Alpine	Alpine grassland and shrublands
Grasslands	East-side (interior) grasslands
Shrublands	East-side (interior) canyon shrublands shrub-steppe dwarf shrub-steppe desert playa and salt scrub shrublands
Juniper woodlands	Western juniper and mountain mahogany woodlands
Montane mixed-conifer forest	Montane mixed-conifer forest
East-side mixed-conifer forest	East-side (interior) mixed-conifer forest
Lodgepole pine forest	Lodgepole pine forest and woodlands
Ponderosa pine forest	Ponderosa pine and east-side white oak forest and woodlands
Subalpine	Subalpine parkland

[a] Source: Johnson and O'Neil 2001.

Table 2—Structural stages used to describe source habitats and their relationship to the Johnson and O'Neil (2001) structural condition classes

Structure stage	Canopy cover[a]	Definition of structure stage	Structural condition classes
Grass/forb	Open	Herbaceous seral stage in forested habitats	Grass/forb–open Grass/forb–closed
Postfire	Open	First 10 years post-stand-replacing fire, abundant standing dead trees	None
Sapling	Open or closed	Earlier seral stages in forested habitats from shrub stage through closed forest of trees <10 inches diameter at breast height (d.b.h.)	Shrub/seedling–open Shrub/seedling–closed Sapling/pole–open Sapling/pole–moderate Sapling/pole–closed
Small tree	Open	Primarily mid-seral stages May be later seral on harsher sites Forested stages with trees 10 to 15 in d.b.h.	Small tree–single story–open Small tree–single story–moderate Small tree–multistory–open Small tree–multistory–moderate
Small tree	Closed	Primarily mid-seral stages May be later seral on harsher sites Forested stages with trees 10- to 15 in d.b.h.	Small tree–single story–moderate Small tree–single story–closed Small tree–multistory–moderate Small tree–multistory–closed
Medium tree	Open	Usually mid- to late-seral stages Forested stages with trees 15 to 20 in d.b.h.	Medium tree–single story–open Medium tree–single story–moderate Medium tree-multistory–open Medium tree multistory–moderate
Medium tree	Closed	Usually mid- to late-seral stages Forested stages with trees 15 to 20 in d.b.h.	Medium tree–single story–moderate Medium tree–single story–closed Medium tree–multistory–moderate Medium tree–multistory–closed
Large tree	Open	Usually late-seral stages Forested stages with trees >20 in d.b.h.	Large tree–single story–open Large tree–single story–moderate Large tree–multistory–open Large tree–multistory–moderate Giant tree–multstory
Large tree	Closed	Usually late-seral stages Forested stages with trees >20 in d.b.h.	Large tree–single story–moderate Large tree–single story–closed Large tree–multistory–moderate Large tree–multistory–closed Giant tree–multistory

[a] A break in canopy closure at 50 percent was used to separate open from closed canopy.

Grouping Species of Conservation Concern

While managing species habitats and populations using a species-by-species approach has intuitive ecological merit, the sheer number of species of conservation concern often makes such an approach untenable. In many cases, the ecological understanding and resources needed to manage all species on an individual basis are not available. More importantly, attempting to manage for species of concern on an individual basis may not result in holistic management of the needs of all species because management focus is often fine scale, piecemeal, and without explicit understanding of the commonalities and differences in species needs among large sets of species (Wisdom et al. 2002).

Tremendous efficiencies are gained from managing groups of species. The idea that efficiency is gained while maintaining effectiveness in accounting for all species needs is a central premise to grouping approaches (Coppolillo et al. 2008, Suring et al. 2011). Grouping species based on one or more ecological factors provides a strong foundation for developing conservation strategies for species of conservation concern because the conservation strategies are ordered around ecological principles.

Species were grouped primarily based on habitat associations using cover type and structural stage (Suring et al. 2011, Wisdom et al. 2000). A cluster analysis was performed to describe groups of species based on their habitat associations. In the cluster analysis, 53 habitat variables were used consisting of six forest cover types, five tree size classes, and two canopy closure categories, three nonforest land cover types, six riparian/water land cover types, and a cave category.

We sequentially examined sets of clusters, with increasing numbers of clusters in each set, to find an aggregation that was consistent with our understanding of species ecological relationships at the macrohabitat scale (as done by Wisdom et al. (2000). We also evaluated similarities among species and clusters using the Ochiai index of similarity (Ludwig and Reynolds 1988).

Based on our knowledge of ecological relationships of the species evaluated, we chose the smallest number of groups possible that still allowed a meaningful aggregation of species and habitats. Groups were then combined into families (categorical not phylogenetic) to help describe how similar groups of species are related to each other.[2] Families include one or more groups that were associated with similar broad-scale macrohabitat conditions. These generalized habitat conditions

[2] Note that the term “families” does not have a taxonomic meaning, but instead identifies robust similarities in habitat requirements among large groups of species, regardless of taxonomic relation (Wisdom et al. 2002).

were often used by managers to interpret broad-scale patterns and trends (Suring et al. 2011, Wisdom et al. 2000). By using a hierarchical evaluation of species, groups, and families, the analysis process addressed single and multispecies needs and identified patterns of habitat change similar to the process followed by Wisdom et al. (2000).

Identifying the Ecological Relationships of Species of Conservation Concern

We reviewed scientific information, in addition to defining source habitats, to more thoroughly understand the ecological requirements of the species of conservation concern. Additional information was compiled on risk factors, fine-scale habitat features, home-range size, and species ranges for each species of conservation concern. We followed the recommendations of Andelman et al. (2001) when determining what ecological information to compile for each species. Compiling this information was important for determining which species were best suited to be surrogate species, and to model relationships between species, habitats, and risk factors.

Risk Factors

Through literature review, we identified factors for species of conservation concern that potentially increased the risk of reducing (1) habitat availability, (2) habitat effectiveness (e.g., roads that reduce the probability of use of a habitat), and (3) population size and fitness (table 3).

The reviews by Wisdom et al. (2000), and Singleton and Lehmkuhl (1998) that addressed road-related factors, and Gaines et al. (2003a) that addressed recreation-related factors were expanded to include risk factors associated with the management of vegetation, fire, grazing, and invasive species.

Fine-Scale Habitats

In addition to broad-scale habitat relationships, we noted from Johnson and O'Neil (2001) whether a species used specific fine-scale habitats such as water features (e.g., springs and seeps), topographic features (e.g., talus slopes), within-stand features (e.g., logs, decayed trees), or other physical features (e.g., serpentine soil).

Home Range and Dispersal Information

Both the typical size of home range used by a species and the species' dispersal capabilities influence which species best represent ecological requirements of other species (Coppolillo et al. 2008, Lambeck 1997, Noss et al. 1997). This information was compiled for all species of conservation concern.

Table 3—Risk factors assessed for each surrogate species through literature reviews and used to develop surrogate species assessment models

Risk factor	Effects of the risk factor
Hunting and trapping	Mortality from hunting and trapping as facilitated by road and trail access
Poaching	Increased illegal take of animals, as facilitated by road and trail access
Collisions	Death or injury resulting from a motorized vehicle running over or hitting an animal as facilitated by road access
Negative human interactions	Increased direct mortality of animals (e.g., shooting) as a result of increased contact with humans, as facilitated by road and trail access and increased building density
Movement barrier or filter	Alteration of dispersal or other movements as a result of human activities (e.g., road or road networks, large openings from clearcuts, increased building density)
Displacement or avoidance	Spatial shifts in animal populations or individuals away from human activities
Habitat loss and fragmentation	Loss and fragmentation of source habitat resulting from human activities or fire
Edge effects	Changes to habitat associated with human-induced edges
Snag and downed log reduction	Snag and downed log reduction associated with their removal along roads, trails, in timber harvest units, and during timber salvage operations
Collection	Collection of live animals for use as pets (e.g., amphibians, reptiles), as facilitated by the physical characteristics of roads and trails or by road and trail access
Access for competitors or predators	A physical human-induced change in the environment that provides access for competitors or predators that would not have existed otherwise (e.g., snow compaction by snowmobiles facilitates movements of coyotes and bobcats into lynx habitat)
Disturbance at a specific site	Displacement of individual animals from a specific location that is being used for reproduction and rearing young
Snow compaction	Direct mortality of animals crushed or suffocated as a result of snow compaction from snowmobile routes or groomed ski trails
Physiological response	Increase in heart rate or level of stress hormones as a result of proximity to a human activity
Reduction of food or cover	Reduction in the availability of food or cover as a result of human activities (e.g., cover reduction from forest thinning, forage reduction from domestic grazing)
Nest parasitism	An increase in the potential for nest parasitism as a result of human activities (e.g., nest parasitism by cowbirds facilitated by proximity to agricultural lands)

Species Range Across the Assessment Area

Range information is helpful in determining which species may best represent the ecological requirements of other species across the assessment area (e.g., species with nonoverlapping ranges will poorly represent each others' requirements). We define a species' range as the polygon or polygons that encompass the outer boundaries of a species' geographic occurrence within the assessment area. In addition to actual boundaries of species ranges, we categorized the extent of the species distribution within the assessment area as shown in table 4.

Table 4—Categories used to describe a species distribution within the assessment area

Distribution category	Descriptions
Endemic	Species whose entire distribution was restricted to the assessment area
Peripheral	Species with only a small portion of their population that occurs within the assessment area
Inherently rare	Species with low population numbers and not naturally well distributed across the assessment area
Large, interacting	Broadly distributed species with one interacting population, the range is depicted as one large polygon that may encompass both used and unused areas
Large, disjunct	Commonly occurring species with disjunct populations; range maps reflect the outer extent of individual populations; and the ranges consist of two or more separate polygons within the planning area, representing two or more separate populations that have limited interaction or do not interact
Small, isolated; and small, fragmented	Locally endemic species or species with small, scattered populations that can have ranges expressed as one small polygon (one small, isolated population), or a series of small populations (a set of small, fragmented populations)

Selection of Surrogate Species

Wiens et al. (2008) summarized the pros and cons of using surrogate species as proxies for a broader set of species when the number of species of concern is too great to allow each species to be considered individually. In addition, they described the spatial scale at which a surrogate approach is likely most effective. They suggested that with small-size planning areas (hundreds of acres), the number of species may be small enough to allow individual species to be assessed, while for very large areas (regional or continental), surrogate species may not adequately represent the variety of taxa or habitats present. The area in-between was termed the surrogate zone where a surrogate-species approach might be most useful (Wiens et al. 2008). They also suggested that the most appropriate use of the surrogate approach would be when the management objective was to conserve or recover many species (e.g., >50) or when biological diversity conservation was broadly considered (Wiens et al. 2008). We met both of these criteria in that the size of our assessment area fell within the surrogate zone (intermediate between hundreds of acres and regional) and the management objectives were to address a broad array of species as required under the NFMA (about 200 species of conservation concern in our case). Wiens et al. (2008) then went on to describe a process for selecting surrogate species, which we closely followed in our selection of a set of surrogate species.

The surrogate species approach is an attempt to streamline the assessment of ecological systems by monitoring a subset of species. It is a pragmatic response to dealing with ecosystem complexity (Noon 2003, Roberge and Angelstam 2004) and is a rigorous way to deal with assessments that involve large numbers of species (Adelman et al. 2001, Roberge and Angelstam 2004). The key characteristic of a surrogate species is that status and trend of habitat conditions provide insights to the integrity of the larger ecological system to which it belongs (Andelman et al. 2001, Lambeck 1997, Noon 2003, Noss et al. 1997). Surrogate species may serve as an umbrella function in terms of encompassing habitats needed for other species, being sensitive to the ecological changes likely to occur in the area, or otherwise serve as an indicator of ecological sustainability (Andelman et al. 2001, COS 1999, Lambeck et al. 1997, Noss et al. 1997, Wegner 2008). In addition, it is assumed that a surrogate species has more demanding requirements for factors putting other group members at greater risk of extinction than the rest of species in the group (Andelman et al. 2001). Surrogate species are intended to represent ecological conditions that provide for sustainable ecosystems, and it is not expected that the population dynamics of a surrogate species would necessarily represent the population dynamics of another species.

The concept of surrogate species differs from management indicator species (MIS) described in the regulations written to implement the NFMA (36 CFR 219.19). The use of MIS was considered a means of evaluating the effects of management actions on a suite of species in that their population trends were assumed to reflect the changes in habitat amount and quality owing to the effects of the management actions (Suring et al. 2011). The MIS concept has been questioned in the literature over the past two decades (Andelman et al. 2001, Landres et al. 1988). The MIS concept evolved to the concept of surrogate species, in the late 1990s (Lambeck 1997). Surrogate species are considered a more appropriate approach to addressing species viability (Wiens et al. 2008).

Lindenmayer et al. (2002) pointed out some of the limitations of the surrogate species concept, including that the approach is data-intensive, that scientific understanding is lacking for many species, and were concerned that there is a lack of testing to validate the approach. Lindenmayer et al. (2002) suggest that the surrogate species approach not be the only approach used to guide landscape restoration. The surrogate species approach has recently been tested for wide-ranging carnivores (Carroll et al. 2001), birds (Drever et al. 2008, Watson et al. 2001), and fish

(Wenger 2008). In addition, Roberge and Angelstam (2004) reviewed the umbrella species concept and concluded that the surrogate species approach seems the most promising because it provides a systematic procedure for selecting umbrella species.

However, the risks and uncertainties involved in using the surrogate species concept must be recognized and acknowledged (Ficetola et al. 2007, Freudenberger and Brooker 2004). Development of a logical foundation for surrogate species selection is critical but poorly developed at this time (Noon 2003); however, advances have been made (see Freudenberger and Brooker 2004). In some cases, the use of surrogate species may fail to account for key requirements of individual species (Ficetola et al. 2005). This risk is highest when surrogate species identified through a process at a broad scale are then used in finer scale applications (Wisdom et al. 2002).

The goal for our assessment was to have a manageable number of surrogate species (about 30) to assess while still maintaining a reliable inference for providing appropriate ecological conditions for nonsurrogate species. After species were clustered into groups based on habitat relationships and other environmental factors, a single or small set of surrogate species was identified within each group. The intent was to select a set of species that represented the full array of potential responses of species to management activities (Raphael et al. 2001). We used the following criteria to select surrogate species:

1. Represent source habitats: Species habitat use represents others in the group, and, in some cases, the family. If there were important differences in used source habitats among species within a group, multiple surrogate species were selected to represent the full array of source habitats used by the group.
2. Risk factors: Species were selected that were affected by all or key combinations of risk factors identified for the group or family.
3. Fine-scale habitats: Species were selected to represent fine-scale habitat features identified for the group or family. For example, if some species within the habitat-based group used snags, then a species with the most demanding or limiting snag requirements was selected as a surrogate species.
4. Home range and dispersal information: We selected species with large home space-use requirements (Gaines et al. 2003b, Noss 1990). Knowledge of dispersal capabilities was lacking for most species, although, where possible, we selected species with the most limited dispersal capability as surrogate species.

5. Species range across the assessment area: Species with the widest distribution across the assessment area were given priority in the selection of surrogate species.

Four types of surrogate species were identified:

1. **F** indicated a surrogate species for the group that should be addressed in the development of management actions.
2. **F*** indicated that there was a choice of which surrogate species to use. Managers from different areas may choose different species primarily based on the distribution of the species.
3. **f** indicated species that had localized populations that were confined to very specific habitats. Proposed management alternatives for these species were applied only to local areas.
4. **CS** indicated a conservation strategy or recovery plan was in place, usually developed by the U.S. Fish and Wildlife Service under the Endangered Species Act. In some cases, the conservation strategy encompassed the range of other species in the group, and therefore other species in this group with similar source habitats and risk factors benefited from the conservation strategy.

Development of Surrogate Species Assessment Models

Assessing the viability of each surrogate species required the development of credible and repeatable analysis processes. This was accomplished through the use of Bayesian belief networks (BBNs) (Marcot et al. 2001, Raphael et al. 2001, Rieman et al. 2001). The use of Bayesian statistics, specifically BBNs, is one way to combine scientific data and information with expert knowledge and experience (Lehmkuhl et al. 2001; Marcot et al. 2001, 2006a, 2006b; Marcot 2006; Wade 2002; Wegner 2008). This is especially important when trying to assess multiple species, many of which have limited empirical data available. A BBN is an influence diagram that depicts the relationships among ecological factors (such as habitat and risks) that influence the likelihood of the outcome of some parameter(s) of interest, such as forest condition or wildlife species viability (Marcot et al. 2001). This approach provided a conceptual model outlining the interconnections among ecosystem components and how a species was anticipated to respond to risk factors. This represented an important step in the application of the surrogate species approach intended to provide insights into ecosystem processes and functions (Noon 2003, Ogden et al. 2003, Wegner 2008).

We followed the guidelines suggested by Marcot et al. (2006a) to develop our surrogate species assessment models, which included the following steps: create an influence diagram of key factors affecting the viability of a species, develop an alpha-level BBN model from the influence diagram, revise the model with input from expert reviewers, test and calibrate the model with case files to create a beta-level model, and evaluate the model.

Surrogate species assessment models were used to assess response of surrogate species to changes in habitat conditions and risk factors resulting from proposed management actions. The BBN models provide a structured tool for integrating several sources of information to make comparisons among management alternatives on how well the conservation of surrogate species was addressed (Marcot et al. 2001). The BBN modeling approach was selected for the following reasons (Marcot et al. 2001, Marcot 2006, Raphael et al. 2001):

1. Major influences on population persistence and/or quality of habitat is displayed.
2. Linkages between features of a proposed management action and the predicted response of a species are represented.
3. Empirical data and expert judgment are combined.
4. Models are easily rerun with different management actions or new model assumptions.
5. Predicted outcomes are based on probabilities and are presented as probabilities.
6. Model results included measures of uncertainty and sources of variation.
7. Model results are spatially explicit.

Surrogate species assessment models were developed for application at two spatial scales: the watershed (5^{th}-field HUC) scale and the entire assessment area scale using information from each watershed. At the watershed scale, we developed the watershed index (WI), and a weighted watershed index (WWI). The WI provided a measure of change of source habitat (HRV compared to current conditions), and the influences of habitat quality (e.g., patch size) and risk factors (e.g., road density) for each watershed. The WWI was calculated from the WI by weighting it with the amount of source habitat that was currently available in each watershed. The WWI provided a measure of the capability of each watershed to contribute to the viability of the surrogate species. At the assessment area scale, we developed a viability outcome index (VOI) for each surrogate species. The VOI calculated an overall index of the potential capability of the assessment area to provide for the viability of the surrogate species. The VOI model used aggregated data from the watershed-scale models, and, for some species, an assessment of how well habitats are connected (how this was assessed is described later) across the assessment area.

Once the surrogate species assessment modeling framework was established, a method for objectively assessing the quality and quantity of habitat available for surrogate species was chosen. We compared the current area of source habitat for a surrogate species within each watershed to estimate the HRV for that species' source habitat (Lehmkuhl et al. 1997, Suring et al. 2011, Wisdom et al. 2000). The HRV refers to the composition, structure, and dynamics of ecosystems before Euro-American settlement (Fule´ et al. 1997, Landres et al. 1999, Morgan et al. 1994, Swanson et al. 1994). By comparing the current condition of source habitats with the HRV, insights were gained into the capability of each watershed to provide habitat that would contribute to the viability of surrogate species (Wisdom et al. 2000). We recognized that the HRV is likely to change as global and regional climates change (Gartner et al. 2008, Millar and Woolfenden 1999, Westerling et al. 2006), which has implications for conservation of biological diversity (Lawler and Mathias 2007). However, we contend that understanding both the current condition and the HRV provide important information for managers to consider, along with climate change projections, in determining desired conditions for wildlife habitats (Wiens et al. 2012). Additionally, the HRV provides an objective measure of habitat sustainability and allows habitat restoration opportunities to be identified (Gaines 2000, Society for Ecological Restoration 1993, Wisdom et al. 2000). We used published estimates for the HRV to develop reference conditions for surrogate species habitats (Agee 2003; Harrod et al. 1998; Hessburg et al. 1999b, 2000, 2005; Wright and Agee 2004).

The use of ecological thresholds is highly controversial and difficult to validate (Bestelmayer 2006, Lindenmayer and Luck 2005, Lindenmayer et al. 2006, Muradian 2001, Tear et al. 2005), yet they are continually being applied to address conservation issues (Groves 2003, Huggett 2005, Noss et al. 1997, Rompre et al. 2010, Svancara et al. 2005, Tear et al. 2005). We conducted a review of the literature to identify a habitat threshold that we could apply to evaluate the number, distribution, and connectivity of watersheds across the assessment area in order to identify those watersheds that are in relatively good condition and may make important contributions toward the viability of surrogate species. We used the threshold to aid in priority setting for watershed restoration (e.g., Suding et al. 2004). Note that we did not use a threshold as a conservation goal; rather the threshold was used as a metric in the evaluation of species viability. We chose 40 percent as a minimum amount of source habitat after reviewing approaches used in other conservation assessments and empirical studies (Denoel and Ficetola 2007, Groves 2003, Noss et al. 1997, Olson et al. 2004, Radford et al. 2005, Rompre et al. 2010, Svancara et al. 2005, Tear et al. 2005, Zuckerberg and Porter 2010). Svancara et al. (2005) showed that conserving a minimum of 40 percent of total available habitat maintained representation, resiliency, and redundancy in the remaining habitat and associated wildlife populations. Representation, resiliency, and redundancy were elements we considered important to maintaining or restoring species viability (Groves 2003, Shaffer and Stein 2000).

We developed surrogate species assessment models for each surrogate species using findings reported in the literature and from expert knowledge. The primary variables in the WI and WWI models included (1) reference conditions (e.g., estimates of HRV of source habitats), (2) estimates of the current amount and distribution of source habitats, (3) factors that influenced the quality of the source habitat (e.g., patch size, fine-scale habitat features, habitat connectivity), and (4) risk factors (e.g., road density, recreation routes, domestic grazing, invasive species). The approaches used to gather information to address the variables used in the surrogate species assessment models are described below. Details of the models developed for each surrogate species are provided in chapter 2.

Reference Conditions for Source Habitats

The current condition of source habitat within each watershed for a surrogate species was compared to reference conditions (e.g., HRV) for that species source habitat. The reference condition estimates within each watershed were based on the results of published analyses (table 5) (Agee 2003, Hann et al. 1997, Hessburg et al. 1999a).

Table 5—Estimated reference conditions for forested habitats in the northeast Washington assessment area[a]

	Reference condition by structural stage group										
Potential vegetation group	**Postfire**	**Early_ Open**	**Early_ All**	**Mid_ Open**	**Mid_ Closed**	**Late_ Mid_ Open**	**Late_ Mid_ Closed**	**Late_ Single_ Open**	**Late_ Single_ Closed**	**Late_ Multi_ Open**	**Late_ Multi_ Closed**
	Percent										
Dry	10-18	10-22	10-25	18-32	5-8	18-32	5-8	11-31	2-5	1-8	0-1
Mesic	15-22	15-27	15-35	2-5	8-20	2-5	8-20	0-3	0-12	4-9	21-42
Cold-moist	10-14	10-20	10-36	1-4	9-27	1-4	9-27	0-1	0-5	3-7	23-59
Cold-dry	10-22	10-30	10-52	5-7	18-26	5-7	18-26			3-7	12-33

[a] Estimates of open/closed reference conditions were derived from table 3-3 in the *Interagency Fire Regime Condition Class Guidebook V1.0.5.*

Reference Conditions of Forested Source Habitats

We estimated reference conditions for forested source habitats using the following steps:

Step 1. Using information in table 5, we identified the low and high percentages of forest group(s), structural stage(s), and canopy closure(s) that corresponded best to our description of source habitats for the surrogate species.

Step 2. We then determined the area of each watershed that is potential source habitat based on the potential natural vegetation group (PVG). Potential source habitat was a combination of PVGs that had the capability of providing source habitat given the appropriate structure stage and canopy closure were present.

Step 3. We used the percentages derived from step 1 and the area estimates from step 2 to calculate a range of high and low area estimates of the predicted amount of source habitat for each watershed.

Step 4. We then divided the range of area estimates from step 3 by the area (size) of each watershed that corresponded to the appropriate PVG to get estimates of the percentage of each watershed that historically had the potential to provide source habitat for the surrogate species. Each watershed had a high and low percentage generated at this step. We used the absolute low and absolute high across all watersheds to bound our estimated reference condition for each species. We also calculated the median percentage of the potential of all the watersheds.

Step 5. We then classified the range into four equal categories between the absolute low and the median, and four equal categories between the median and the absolute high (fig. 2).

Reference Conditions of Postfire Source Habitats

Reference conditions for postfire source habitats were derived from the information presented in table 6; the proportion of the landscape that was in an early-seral reference condition for each forest type. Forest and fire ecologists were asked to estimate how much of the early-seral reference condition would be in a ≤10 years postfire condition. Ten years was derived from descriptions of postfire habitat use

	Low				Median				High
Categories	-4	-3	-2	-1	0	1	2	3	4

Figure 2—Departure classes were created using low, median, and high projected estimates of the amount of source habitat for each watershed.

Table 6—Estimated reference conditions for postfire habitats by forest group in the northeast Washington assessment area

Forest group	Percentage of forested landscape in postfire source habitat ($\leq$10 years postfire)
Dry	10 to 18
Mesic	15 to 22
Cold-moist	10 to 14
Cold-dry	10 to 22

by woodpeckers (Lehmkuhl et al. 2003, Saab and Dudley 1998) and postfire snag fall rates (Everett et al. 1999, Harrod et al. 1998).

Reference Conditions for Species Associated With Nonforested Source Habitats

As described above, in forested communities we have estimated proportions of different PVGs and structural stages that were likely to occur at any given time considering succession and disturbance across the landscape. In the nonforested shrublands and riparian environments, we did not have published estimates of HRV to use as reference conditions. Therefore, to evaluate the relative amount of upland nonforest source habitat within watersheds, we assumed that land currently occupied by agriculture and urban areas within the assessment area historically supported shrub-steppe, grasslands, or wet meadows. The proportions of shrub-steppe, grasslands, and wet meadows cover types in the existing mapped shrub-steppe vegetation zone were multiplied by the area currently in agriculture and urban areas, and the result added to the area currently in these cover types to obtain an estimate of the reference conditions of source habitat for surrogate species (table 7).

Historical grasslands = {[current grass/ (current grass + current shrub)] X (urban + agriculture)} + current grass

Historical shrub = {[current shrub/ (current shrub + current grass)] X (urban + agriculture)} + current shrub

We then created nine classes of departure to measure the relative amount of source habitat within each watershed: 16 to 30 percent below the median = -1; 31 to 45 percent below the median = -2; 46 to 60 percent below the median = -3; >60 percent below the median = -4; 16 to 30 percent above the median = +1; 31 to 45 percent above the median = +2; 46 to 60 percent above the median = +3; >60 percent above the median = +4; 0 to 15 percent above or below the median = 0 departure.

Table 7—An example of estimates of the relative amount of upland nonforest source habitat within two watersheds

Huc5 name	Current urban_ agri._	Habitats current grassland_	Current shrub-steppe_	Current total	Historical grass	Historical shrub
American River	0	140	0	140	140	0
Boulder/Deadman	110	3,665	10,565	14,339	3,693	10,646

Huc5 = hydrological unit code level 5.

Reference Conditions for Wetland and Riparian Source Habitats

Numerous reports describe how human activities (e.g., those associated with dams, diversions, agriculture conversion, stream channelization, road construction, etc.) have permanently altered large areas of wetland habitat. Brinson et al. (1981) estimated that 9.3 million ha (3.2 million ac) of the original flood-plain forest was converted to urban and cultivated agricultural land uses in the United States. Klopatek et al. (1979) estimated that northern floodplain forests have decreased 69 percent in area from their potential, and Hirsh and Segelquist (1978) estimated that 70 to 90 percent of all natural riparian areas was subjected to extensive alteration. Little is known about the extent and status of mountain riparian ecosystems, which are affected primarily by impacts associated with other natural resource uses (e.g., timber harvest, recreation, livestock grazing) although federal and state surveys have found that 50 percent of all fish habitats on public and private lands in western Oregon have been altered since 1960 (Kadera 1987). Dahl (1990) described that about 47.3 million ha (11.6 million ac) or 53 percent of all U.S. wetlands have been lost since the 1780s. Based on these studies, we assumed that the current amount of source habitat for wetland and riparian deciduous surrogate species in the assessment area was about 70 percent of the historical amount in each watershed (Dahl 1990, Peters 1990). In the WI source habitat departure node, we used the [(-1) – (-2)] category for every watershed to reflect our assumption that the availability of these habitats was near 70 percent of their historical median.

Reference Conditions for Streamside Riparian and Cliff Source Habitats

For these habitats, we assumed that their availability has not changed from their historical amounts. Therefore, our assessment focused on factors that could influence the quality of these habitats. In the WI source habitat departure node, we used the 0 to 1 quartile for every watershed to reflect our assumption that the availability of these habitats was near the historical median.

In summary, we used the following approach to standardize how we estimated habitat departure for each surrogate species (table 8).

Step 1. We identified combinations of spatial data that best represented the source habitat for the surrogate species.

Step 2. We determined the amount of source habitat within each watershed that was located within the assessment area through geographic information system (GIS) processes.

Step 3. We compared the current estimates of source habitat to our estimates for the reference conditions for each watershed and determined the degree of departure based on table 8. For example, if 1,000 ac of source habitat occurred in the watershed and that value falls between 2 and 3 categories below the median (-3Q to -2Q), a likelihood of 100 percent is entered in the -3 to -2 category reference condition node (see example below).

If there currently is no source habitat in the watershed and source habitat did not occur in the watershed historically, then we entered zero in the habitat potential node that resulted in a 100 percent likelihood of zero habitat in the habitat amount versus reference condition node.

Factors That Influenced Habitat Quality

Several factors were identified from our literature review that influence habitat quality for the surrogate species source habitats and were incorporated into the surrogate species assessment models.

Patch size—

We used the average patch size of source habitat in each watershed to assess the current condition for those species that might be affected by patch size.

Snags—

Snags were an important component of source habitat or were important determinants of habitat quality for a number of surrogate species. (Mellen-McLean et al. 2009, Rose et al. 2001). We used information from Ohman and Gregory (2002) to estimate the current density of snag habitat, overlayed this with our source habitat data, and then summarized the availability of snag habitat within source habitat for each watershed. Ohman and Gregory (2002) used snag data from the Forest Inventory and Analysis (FIA) plots and a gradient nearest neighbor (GNN) analysis to estimate the density of snags. These data are not accurate at small spatial scales (Ohman and Gregory 2002), which is why we chose to summarize the snag information for all source habitat within a watershed.

Table 8—Summary of how habitat departure was evaluated for each habitat group

	Departure category								
Habitat group	**-4**	**-3**	**-2**	**-1**	**0**	**1**	**2**	**3**	**4**
Forested	Absolute low				Median				Absolute high
Nonforested	RC .4	RC .55	RC .70	RC .85	RC median				
Wetlands	RC 4	RC .55	RC .70++	RC .85	RC median				
Stream riparian					RC = current				

RC = reference conditions.

We developed reference conditions for snag densities for surrogate species using information from Harrod et al. (1998) for the dry forests. We used the tolerance levels from Mellen-McLean et al. (2009) for snag density from unharvested inventory plot data (including plots with no snags) for east-side mixed-conifer (EMC-NCR), large tree vegetation type for pileated woodpecker and fringed myotis, averaged snag densities at the three tolerance levels across EMC-NCR, ponderosa pine/Douglas-fir (PPDF), and montane mixed-conifer (MMC), small-medium tree vegetation types for black-backed woodpecker (table 9).

We compared the watershed mean snag density estimates to the reference conditions to determine the current condition scores for snag habitat (low, moderate, high, very high) (table 9) for each watershed.

Table 9—Historical reference condition classes for snags by surrogate species

Surrogate species	**Forest type**	**Snag size**	**Low**	**Moderate**	**High**	**Very high**
		D.b.h.	- - - - - - - - - - - *Per hectare* - - - - - - - - - - -			
Lewis's woodpecker[a]	Dry	>50 cm	<2.47	2.47 to -<3.1	3.1 to 3.7	>3.7
White-headed woodpecker[a]	Dry	>50 cm	<2.47	2.47 to 3.1	3.1 to 3.7	>3.7
Western bluebird[a]	Dry	>37.5 cm	<5.19	5.19 to <6.9	6.9 to <8.65	>8.65
Black-backed woodpecker[b]	Dry, mesic Cool-moist, Cold-dry	>25 cm	<9.13	9.13 to 17.9	18.0 to 45.0	>45.0
Pileated woodpecker[b]	Mesic, dry	>50 cm	<1	1.1 to 3.6	>3.6 to 39.3	>39.3
Fringed myotis[b]	Mesic, dry	>50 cm	<1	1.1 to 3.6	>3.6 to 39.3	>39.3

D.b.h. = diameter at breast height.

[a] Based on Harrod et al. 1998.

[b] Based on Mellen-McLean et al. 2009.

Late-successional forest—

Several surrogate species were associated with late-successional forests or structural attributes associated with late-successional forest (e.g., large trees, down wood, etc.). We used a combination of forest cover types, medium and, large tree size classes (>15 in quadratic mean diameter [QMD]), and canopy closure (>70 percent canopy closure) to define late-successional forests.

Denning—

For many species, denning habitat comprised fine-scale habitat features beyond our ability to spatially evaluate. However, potential wolverine denning habitat was mapped using land-type associations (USDA FS 2000) that correspond to alpine cirques with the type of structure typically used by wolverines for natal dens (Copeland 1996) and are likely to have adequate snow cover also important for denning (Copeland 2010). These included land type associations Ha7, Ha8, Hb9, and Hi9 (USDA FS 2000).

In addition, down woody debris is an important component of lynx denning habitat (Koehler 1990, Mowat et al. 2000, Organ et al. 2008, Squires and Laurion 2000). We could not specifically identify lynx denning habitat, but we used the availability of down woody debris as a way of assessing the quality of the source habitat and its potential to provide this component of lynx denning habitat. The down wood density values calculated from the FIA data through the GNN analysis (Ohman and Gregory 2002) were used to quantify the availability of down wood within source habitat for lynx and summarized for each watershed.

Open landscape—

We used all vegetation types with less than 10 percent tree canopy closure to identify portions of the landscape that were considered open. This variable was used to evaluate habitat for each watershed for species that require relatively high levels of forest cover.

Shrub cover—

For species that are affected by the amount of shrub cover, such as the fox sparrow, we used the GNN data from Ohmann and Gregory (2002) to identify the percentage of shrub cover in source habitat for each watershed.

Cliffs—

Cliffs provided important habitat features for several surrogate species and were identified using a digital elevation model. We found that a slope break of 38 degrees identified most of the cliff structures used by the surrogate species we were assessing.

Riparian and wetland habitats—
In addition to providing source habitat for some of the surrogate species, riparian and wetland habitats were also an important determinant of habitat quality for other species. Therefore, we mapped riparian and wetland habitats using the national wetlands inventory.

Risk Factors

We conducted a literature review to identify the risks that most likely influenced surrogate species persistence and to develop indices (e.g., road density, zone of influence) that could be used to spatially evaluate levels of risk. Application of these indices relied on the availability of spatial data describing factors such as roads, trails, and human population centers.

Recreation routes and sites—
Recreation routes such as roads, trails, snowmobile routes, and groomed ski trails were identified as risk factors for a number of surrogate species. We summarized road and trail densities from current maps into the following categories: no roads, <1 m/m^2, 1 to 2 m/m^2, and >2 m/m^2 (based on Gaines et al. 2003, Wisdom et al. 2000). For several surrogate species, we used proximity of source habitat to roads and trails (also referred to as zone of influence) to evaluate the effects of disturbance on surrogate species. Distance buffers placed on roads and trails to evaluate proximity were based on the literature review of Gaines et al. (2003). We also used locations and densities of recreation sites, such as campgrounds and boat launches, when literature showed these features were important risk factors for surrogate species.

Domestic grazing—
Grazing by domestic livestock was evaluated as a risk factor for several surrogate species. This was determined by overlaying the location of active grazing allotments on maps of source habitat. Only currently active allotments were used to assess risks to surrogate species. We were not able to evaluate the intensity of the grazing that occurred within the grazing allotments as this information was not available spatially.

Invasive species—
Stocking of nonnative fish species was identified as a risk factor for some surrogate species. This practice occurs throughout the assessment area and was documented for each water body by gathering stocking information from the Washington Department of Fish and Wildlife.

Housing density—

We obtained spatial information on housing density to index the effects of human development on surrogate species source habitats and habitat effectiveness.

Viability Outcomes

The VOI model was developed for each surrogate species to incorporate information from the WI scores; distribution of source habitats across the assessment area; and for some species, how well habitats were connected across watersheds (fig. 3, table 10). The VOI is a large-scale index of population abundance and distribution (based on habitat and risk factors) across the landscape, not an actual prediction of population occurrence, size, density, or other demographic characteristics. We assumed that species with high VOI scores have a high probability of having populations that are self-sustaining and well distributed throughout their historical ranges in the assessment area.

The VOI model incorporated the WWI score (described earlier); a habitat distribution index; and for some species, a habitat connectivity index that assessed how well habitats were connected across watersheds. Each variable of the VOI model is described in detail below.

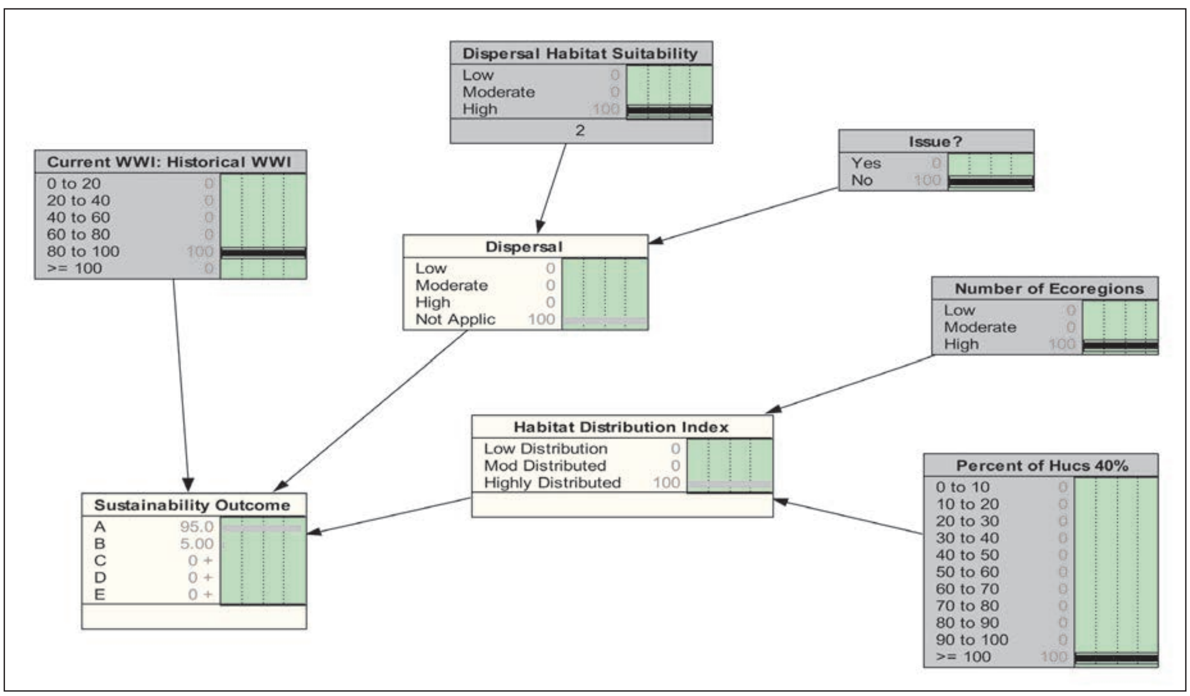

Figure 3—Viability outcome Bayesian belief network model for the northeast Washington assessment area. WWI = weighted watershed index, Huc = hydraulic unit code.

Table 10—Key indices used in the surrogate species assessment models and viability outcome model

Index	Definition
Watershed index (WI)	A measure of amount (reference condition vs. current) of source habitat, and the influence of habitat quality and risk factors for each watershed in the assessment area. Provides an index of the capability of the watershed to contribute to the viability of the surrogate species.
Weighted watershed index (WWI)	The WI weighted by the amount of source habitat in the watershed. Provides a relative measure of the potential capability of the watershed to contribute to the viability of the surrogate species.
Current WWI: historical WWI	Provides a measure of the current capability of the assessment area to contribute to the viability of the surrogate species compared to the historical capability.
Dispersal habitat suitability	Calculated for Canada lynx, American marten, bighorn sheep, and wolverine. Provides a measure of the relative permeability of the landscape considering habitat characteristics (e.g., cover type, vegetation structure) and risk factors (e.g., road density, housing density) during current and historical conditions.
Ecoregions[a] with watersheds greater than minimum habitat amount	This node was calculated for each surrogate species to assess the distribution of watersheds with current source habitat amounts that were >40 percent of the historical median of source habitat. We used the estimates of the reference conditions of source habitat to calculate the median amount of source habitat across all watersheds that occurred within the distribution of the surrogate species in the assessment area. We then determined the number of ecoregions that contained ≥1 watershed that exceed the habitat minimum.
Percentage of watersheds greater than minimum habitat amount	An assessment of the proportion of watersheds with current source habitat amounts that were >40 percent of the median reference condition of source habitat. We used the estimates of the reference conditions of source habitat to calculate the median amount of source habitat across all watersheds that occurred within the distribution of the surrogate species in the assessment area.
Viability outcome index	An index of how well the assessment area contributes to the viability of the surrogate species.

[a] Ecoregions were paired 4th-field subbasins that were identified by vegetation ecologists to create relatively similar land units for vegetation modeling.

Weighted Watershed Index Calculation

The WWI was incorporated into the VOI model by calculating the ratio of current WWI to historical WWI to assess the current capability of the assessment area to provide for the viability of surrogate species. The WWI score was calculated using the following method (table 11):

Step 1. We determined the current WIs for each watershed in the assessment area through application of the surrogate species assessment models.

Step 2. We determined the historical WIs for each watershed in the assessment area by setting all human influence nodes to zero, assuming the amount of source habitat is one category above the reference condition median, and assuming the snag variable (for those species that have it) was at the 50 to 80 percent tolerance level (Mellen-McLean et al. 2009) (table 11).

Step 3. We weighted the current and historical WIs using the current amount and historical estimates of source habitat within each watershed.

Step 4. We then summed the current (W) and historical (X) amount of source habitat across all watersheds and divided the sum of the weighted WI values for all watersheds by this number. This resulted in an overall weighted WI value for both current (Y/W) and (Z/X) historical conditions (table 11).

Step 5. We determined the ratio of the current WWI:historical WWI to determine the relationship between current and historical conditions for each surrogate species in the assessment area.

Table 11—Hypothetical example showing the calculations for the overall weighted watershed index (WI) value

	WI score current	WI score historical	Source habitat current	Source habitat historical	Weighted WI score current	Weighted WI score historical
			- - - - - *Acres* - - - - -			
Watershed A	1.3	3.0	50	80	65	240
Watershed B	2.1	3.0	70	75	147	225
Watershed C	2.7	3.0	90	95	243	285
Totals			W = 210	X = 250	Y = 455	Z = 750
Overall weighted WIs					Y/W = 2.17	Z/X = 3.0
Current: historical						2.17/3.0 = 0.72

Habitat Distribution Index

This index assessed how watersheds with relatively high amounts of source habitat were distributed across the assessment area. The habitat distribution index was calculated by the interaction of two variables: number of ecoregions (see footnote 3) with at least one watershed that met a threshold for the amount of source habitat, and percentage of the total number of watersheds that met the threshold for amount of source habitat. The threshold amount of source habitat within a watershed was at least 40 percent of the historical median of source habitat (see page 17 for further explanation).

We estimated the habitat distribution index for historical conditions as well. We did this by determining which of the watersheds had historical estimates of source habitat amounts that were >40 percent of the median of the historical amount across all watersheds, and then used those watersheds to calculate the number of ecoregions with at least one watershed above the 40 percent threshold, and percentage of watersheds with source habitat above the 40 percent threshold.

We categorized the habitat distribution index for both current and historical conditions as follows: low habitat distribution equals less than or equal to one ecoregion with at least one watershed above the 40 percent source habitat threshold; moderate = two to three ecoregions with at least one watershed above the 40 percent source habitat threshold; and high = four to five ecoregions with at least one watershed above the 40 percent source habitat threshold. We categorized the percentage of watersheds within the assessment area that met the 40 percent threshold habitat amounts under both current and historical conditions into 10 equal categories from 0 to 100 percent (10 percent increments) (see fig. 3).

Dispersal Habitat Suitability

We evaluated dispersal habitat suitability for surrogate species whose dispersal patterns were appropriate to assess at the spatial scale of our assessment area. Our analysis was based on the idea that resistance to movement across a landscape can be mapped by assigning resistance values to habitat attributes. These values depict the relative “cost” for an animal to move across areas (Singleton et al. 2002, WWHCWG 2010). Areas with “good” habitat characteristics (i.e., forested land cover, low road densities, and low human population densities) have low costs of movement, whereas areas with “poor” habitat characteristics (i.e., agriculture land cover, high road densities, and high human population densities) have high movement costs.

The criteria used to determine which species to evaluate included (1) moderate to large (>2,471-ac) home range size, (2) relatively large dispersal distances (>6.2

mi), (3) knowledge of potential dispersal barriers, and (4) dispersal limited (e.g., many surrogate bird species were not dispersal limited) and habitat limited (not a habitat generalist).

Canada lynx, wolverine, bighorn sheep, and American marten were identified as appropriate species to evaluate. Methods similar to those used by Singleton et al. (2002) and WHCWG (2010) were used to model dispersal habitat suitability within the assessment area.

We compiled datasets of land cover types, canopy closure, vegetation zones, road density, motorized and nonmotorized trail density, human population density, slope, and elevation. These attributes were assigned resistance values ranging from 0.1 (high cost of movement) to 1, (low cost of movement) based on extensive literature review and expert knowledge (table 12). All datasets for each surrogate species assessed were combined using math algebra resulting in an overall score between 0 (low permeability) and 1 (high permeability). This analysis resulted in a map that depicts the cumulative energetic cost for an animal to move across the landscape, expressed as "dispersal habitat suitability." The relative importance of each parameter was reflected in the permeability value it was assigned. Parameters with more influence were attributed with coefficients of lower values and a higher range of scores (i.e., 0.1 to 1) than parameters with less influence (i.e., 0.6 to 1). The overall permeability score was determined by calculating the percentage of the assessment area in three dispersal habitat suitability classes (high = >0.5, moderate = 0.1 to 0.5, and low = 0.0 to 0.1). All spatial analysis was done using ArcInfo 9.0 (ESRI 2004) in a Windows NT environment.

We evaluated historical dispersal habitat suitability by "turning off" the effects of roads and trails, and housing density variables. We did not attempt to evaluate the influences of changes in the vegetation-related variables (vegetation zone, cover type, canopy cover) between current and historical conditions as we did not have information on the spatial distribution of historical vegetation available.

Viability Outcomes for Surrogate Species

Environmental outcomes defined in Raphael et al. (2001) were used as a basis to describe five viability outcomes. These outcomes were calculated for current and historical conditions for each surrogate species to assess changes in habitat conditions. The term "suitable environment" refers to a combination of source habitat and risk factors that influence the probability of occupancy and demographic performance of a surrogate species. The viability outcomes are based on departure from historical conditions. The five viability outcomes we used were:

Table 12—Habitat variables and resistance values used to evaluate dispersal habitat suitability for American marten, bighorn sheep, Canada lynx, and wolverine within the northeast Washington assessment area

	Resistance values for surrogate species			
Habitat variable	**American marten**	**Bighorn sheep**	**Canada lynx**	**Wolverine**
		Vegetation zone		
Alpine	0.1	0.1	0.8	1.0
Parkland	0.3	0.1	0.9	1.0
Subalpine fir	1.0	0.1	1.0	1.0
Mountain hemlock	1.0	0.1	1.0	1.0
Pacific silver fir	1.0	0.1	0.8	1.0
Western hemlock	1.0	0.1	0.8	1.0
Grand fir	1.0	0.1	0.7	0.8
Douglas-fir	0.7	1.0	0.7	0.8
Oregon white oak	0.3	0.1	0.6	0.5
Ponderosa pine	0.5	1.0	0.6	0.8
Shrub steppe	0.1	1.0	0.5	0.5
		Cover type		
Mixed conifer	0.7		0.8	1.0
Douglas-fir	0.7		0.7	1.0
Engelmann spruce	1.0		0.8	1.0
Grand fir	1.0		0.7	1.0
Lodgepole pine	0.7		1.0	1.0
Mountain hemlock	1.0		1.0	1.0
Pacific silver fir	1.0		0.8	1.0
Parkland	0.5		0.9	1.0
Subalpine fir	1.0		1.0	1.0
Western hemlock	1.0		0.8	1.0
Western larch	0.7		0.7	1.0
Western redcedar	1.0		0.8	1.0
Meadow	0.1		0.8	0.8
Nonvegetated	0.0		0.7	0.8
Ponderosa pine	0.4		0.6	0.8
Riparian deciduous	0.6		0.9	0.8
Wet meadow	0.1		0.8	0.8
Dry meadow	0.1		0.6	0.6
Grassland	0.1		0.5	0.5
Shrub	0.2		0.5	0.5

Table 12—Habitat variables and resistance values used to evaluate dispersal habitat suitability for American marten, bighorn sheep, Canada lynx, and wolverine within the northeast Washington assessment area (continued)

	Resistance values for surrogate species			
Habitat variable	**American marten**	**Bighorn sheep**	**Canada lynx**	**Wolverine**
Shrub steppe	0.1		0.5	0.5
Oregon white oak	0.5		0.5	0.5
Urban/agriculture	0.0		0.1	0.1
Water	0.0	0.1	0.1	0.1
Ponds/lakes	0.0	0.1	0.1	0.1
	Roads and motorized trail density			
(km/km^2)				
0 to 0.1	1.0	1.0	1.0	1.0
0.1 to 1.6	0.8	1.0	1.0	1.0
1.6 to 3.2	0.6	0.8	1.0	0.8
3.2 to 6.4	0.5	0.6	1.0	0.6
6.4 to 9.7	0.4	0.5	0.8	0.5
9.7 to 12.9	0.3	0.4	0.7	0.4
12.9 to 16.1	0.2	0.2	0.3	0.2
>16.1	0.1	0.1	0.1	0.1
	Nonmotorized trail buffer			
Meters				
<200		0.6		
>200		1.0		
	Housing density			
Ac/unit				
0 or no data	1.0	1.0	1.0	1.0
>80	1.0	1.0	1.0	1.0
50 to 80	1.0	0.8	1.0	0.8
40 to 50	0.8	0.6	0.8	0.6
30 to 40	0.7	0.6	0.7	0.6
20 to 30	0.5	0.4	0.5	0.4
10 to 20	0.3	0.2	0.3	0.2
1.7 to 10	0.2	0.1	0.2	0.1
0.6 to 1.7	0.1	0.1	0.1	0.1
<0.6	0.1	0.1	0.1	0.1

Table 12—Habitat variables and resistance values used to evaluate dispersal habitat suitability for American marten, bighorn sheep, Canada lynx, and wolverine within the northeast Washington assessment area (continued)

	Resistance values for surrogate species			
Habitat variable	American marten	Bighorn sheep	Canada lynx	Wolverine
		Elevation		
Meters				
0 to 1000	0.8		0.8	0.6
1000 to 1500	1.0		1.0	0.8
>1500	1.0		1.0	1.0
		Slope (degrees)		
0 to 20	1.0		1.0	1.0
0 to 31		0.4		
20 to 40	0.8		0.8	0.8
31 to 38		1.0		
>40	0.6		0.6	0.6
>38		0.6		
		Canopy cover (percent)		
Percent				
0 to 40		1.0		
40 to 60		0.6		
>60		0.1		

1. **Outcome A**—Suitable environments are broadly distributed across the historical range of the species throughout the assessment area. Habitat abundance is high relative to historical conditions. The combination of distribution and abundance of environmental conditions provides opportunity for continuous or nearly continuous intraspecific interactions for the surrogate species.
2. **Outcome B**—Suitable environments are broadly distributed across the historical range of the species. Suitable environments are of moderate to high abundance relative to historical conditions, but there may be gaps where suitable environments are absent or present in low abundance. However, any disjunct areas of suitable environments are typically large enough and close enough to permit dispersal among subpopulations and to allow the species to potentially interact as a metapopulation. Species with this outcome are likely well distributed throughout most of the assessment area.

3. **Outcome C**—Suitable environments moderately distributed across the historical range of the species. Suitable enviroments exist at moderate abundance relative to historical conditions. Gaps where suitable environments are either absent or present in low abundance are large enough such that some subpopulations may be isolated, limiting opportunity for intraspecific interactions especially for species with limited dispersal ability. For species for which this is not the historical condition, reduction in the species' range in the assessment area may have resulted. Surrogate species with this outcome are likely well distributed in only a portion of the assessment area.
4. **Outcome D**—Suitable environments are low to moderately distributed across the historical range of the species. Suitable environments exist at low abundance relative to their historical conditions. While some of the subpopulations associated with these environments may be self-sustaining, there is limited opportunity for population interactions among many of the suitable environmental patches for species with limited dispersal ability. For species for which this is not the historical condition, reduction in species' range in the assessment area may have resulted. These species may not be well distributed across the assessment area.
5. **Outcome E**—Suitable environments are highly isolated and exist at very low abundance relative to historical conditions. Suitable environments are not well distributed across the historical range of the species. For species with limited dispersal ability there may be little or no possibility of population interactions among suitable environmental patches, resulting in potential for extirpations within many of the patches, and little likelihood of recolonization of such patches. There has likely been a reduction in the species' range from historical conditions, except for some rare, local endemics that may have persisted in this condition since the historical period. Surrogate species with this outcome are not well distributed throughout much of the assessment area.

Habitat Conditions for Conservation Planning

For each surrogate species, we classified each watershed into one of five habitat conditions (table 13). The habitat conditions were based on the WI scores and whether the current amount of source habitat was above or below 40 percent of the historical median. This allowed the identification of a basic set of management options to consider:

Table 13—Definitions for habitat conditions (HC) and their primary management options

Habitat condition	Definition	Potential management options to consider
Habitat condition 1a, 1b	The quality and quantity of source habitat is relatively unchanged from historical conditions. Watershed index >2.0 and the amount of source habitat is (a) >40 percent of the historical median or (b) <40 percent of the historical median.	Protection of existing source habitat, especially in habitat condition 1a. Restoration can occur elsewhere as needed.
Habitat condition 2a, 2b	The quality and quantity of source habitat has been moderately reduced (WI 1.0 to 2.0), and the amount of potential source habitat is (a) >40 percent of the historical median or (b) <40 percent of the historical median.	Restoration of source habitats. Protection of existing source habitat Restoration of habitat condition 2a.
Habitat condition 3a, 3b	The quality and quantity of source habitat has been severely reduced (WI <1.0) and the amount of potential source habitat is (a) >40 percent of the historical median or (b) <40 percent of the historical median.	A combination of protection and restoration depending on the juxtaposition of these watersheds in relation to HC1 and HC2 watersheds.
Habitat condition 4	Connectivity or habitat distribution indices identify significant gaps in the distribution of watersheds with >40 percent of the historical median of source habitats.	Manage for dispersal habitat that provides for habitat connectivity.
Habitat condition 5	The amount of source habitat in the watershed for the surrogate species <25 percent on federal ownership.	Land ownership limits the strategies that can be used to contribute to species viability.

- Habitat condition 1—Protection of habitat in watersheds that were in good condition
- Habitat condition 2—Restoration of habitat within watersheds that were in moderate condition but could be raised to a habitat condition 1 with a reasonable amount of effort
- Habitat condition 3—Watersheds have been severely degraded and may require substantial commitment of resources to improve the condition
- Habitat condition 4—Watersheds that were important for connectivity owing to their location
- Habitat condition 5—Watersheds with limited federal land ownership

We evaluated more than 700 species (67 percent birds, 23 percent mammals, 5 percent amphibians, 5 percent reptiles) documented to occur in the Pacific Northwest Region (Oregon and Washington).

Overall Results of the Surrogate Species Assessments

We evaluated more than 700 species (67 percent birds, 23 percent mammals, 5 percent amphibians, 5 percent reptiles) documented to occur in the Pacific Northwest Region (Oregon and Washington) (Suring et al. 2011). Following the application

We identified 52 surrogate species for the national forests east of the Cascades Range in Oregon and Washington (Suring et al. 2011), which included 67 percent birds, 17 percent mammals, 14 percent amphibians, and 2 percent reptiles.

of the screening criteria, we identified 209 species of conservation concern on National Forest System lands east of the crest of the Cascade Range. We aggregated these species into 10 habitat families and 28 habitat groups based on habitat associations. We identified 52 surrogate species for the national forests east of the Cascades Range in Oregon and Washington (Suring et al. 2011), which included 67 percent birds, 17 percent mammals, 14 percent amphibians, and 2 percent reptiles based on risk factors and ecological characteristics. We selected 32 of the surrogate species for evaluation in the northeastern Washington assessment area. The species identified as species of conservation concern are shown in appendix 2. These species represent the full range of habitats and risk factors.

We developed surrogate species assessment models for 27 of the 32 surrogate species used in the northeast Washington assessment area (table 14). Five surrogate species (northern bog lemming, larch mountain salamander, western gray squirrel, Townsend's big-eared bat, and fringed myotis) had such limited distributions within the assessment area that we did not develop formal models. For these species, we summarized available literature on their habitats, risk factors, and conservation status to offer management considerations.

All surrogate species that we assessed showed lower viability outcomes under current conditions compared to historical conditions (table 15). The species for which current viability outcomes are most similar to historical viability outcomes include the golden eagle, Harlequin duck, northern goshawk, peregrine falcon, and Wilson's snipe. Species for which current viability outcomes have departed the most from historical viability outcomes and are of greatest concern included the eared grebe, bighorn sheep, fox sparrow, sage thrasher, western bluebird, and white-headed woodpecker. Results of WI and VOI models and considerations for improving the viability for each species are provided in detail in chapters 2 and 3.

Surrogate-Species Assessment Model Evaluation

To evaluate our surrogate species assessment models, we conducted three levels of peer review and evaluated the scores from the WI models with independent data for a subset of surrogate species. For the peer reviews, we first convened a science team (app. 3) to provide input on our process, including the use of habitat relationships data, clustering procedure, surrogate species selection, and development and application of surrogate species assessment models. Second, we consulted species experts (app. 3) to help with the development of the species-specific models. These experts helped us determine which variables were most important to include in our

Table 14—Surrogate species for the northeast Washington assessment area

Surrogate species	Family/group association	Focal type	Okanogan	Wenatchee	Colville
Water vole	Boreal forest	F	X	X	X
Northern bog lemming	Boreal forest	f	X	X	X
Canada lynx	Boreal forest	F*	X	X	X
Northern goshawk	Medium-large trees/all forest communities	F	X	X	X
Cassin's finch	Medium-large trees/all forest communities	F	X	X	X
Larch mountain salamander	Medium-large trees/cool-moist forest	f		X	
Pileated woodpecker	Medium-large trees/cool-moist forest	F	X	X	X
American marten	Medium-large trees/cool-moist forest	F	X	X	X
White-headed woodpecker	Medium-large trees/dry forest	F	X	X	X
Western bluebird	Open forest/all forest communities	F	X	X	X
Fringed myotis	Open forest/all forest communities	F	X	X	X
Fox sparrow	Open forest/early successional	F	X	X	X
Western gray squirrel	Open forest/pine/oak (medium-large trees)	f	X	X	
Lewis's woodpecker	Open forest/postfire	F	X	X	X
Black-backed woodpecker	Open forest/postfire	F	X	X	X
Peregrine falcon	Habitat generalist/cliff	F	X	X	X
Wolverine	Habitat generalist	F	X	X	X
Golden eagle	Woodland/grass/shrub	F	X	X	X
Lark sparrow	Woodland/grass/shrub	F	X	X	X
Sage thrasher	Shrub	F*	X	X	X
Tiger salamander	Grass/shrub	f	X	X	X
Bighorn sheep	Grass/shrub	f	X	X	X
Northern harrier	Grassland	F*	X	X	X
Townsend's big-eared bat	Chambers/caves	f	X		
Inland tailed frog	Conifer riparian	F	X	X	
Wood duck	Snag/open water	F	X	X	X
Harlequin duck	Riparian/large tree	f	X	X	X
Bald eagle	Riparian/large tree	F	X	X	X
Red-naped sapsucker	Shrubby/deciduous riparian	F	X	X	X
MacGillivray's warbler	Shrubby/deciduous riparian	F	X	X	X
Columbian spotted frog	Pond/small lake/backwater	F*	X	X	X
Wilson's snipe	Marsh/wet meadow	F	X	X	X
Eared grebe	Marsh/open water	F	X	X	X

F = a surrogate species for the habitat group that should be addressed in the development of management actions, F* = a choice of surrogate species for managers to use primarily based on the distribution of the surrogate species, f = a surrogate species with localized populations that were confined to very specific habitats.

X = viability for the surrogate species was assessed. Whether or not a species was assessed for each forest was based on the distribution of the species (e.g., the inland tailed frog does not occur on the Colville National Forest).

Table 15—Current (Cur) and historical (Hist) viability outcomes for surrogate species assessed in the northeastern Washington assessment area

	Probability of viability outcome									
	A		B		C		D		E	
Surrogate species	Cur	Hist	Cur	Hist	Cur	Hist	Cur	Hist	Cur	Hist
American marten	1	59	37	30	36	8	22	2	4	0
Bald eagle	0	76	23	16	73	7	4	1	0	0
Bighorn sheep	0	51	0	33	53	12	45	4	2	0
Black-backed woodpecker	9	81	25	13	40	5	26	1	0	0
Canada lynx	4	71	65	19	26	9	6	1	0	0
Cassin’s finch	0	81	0	13	40	5	60	1	0	0
Columbia spotted frog	0	71	22	19	72	9	6	1	0	0
Eared grebe	0	0	0	20	0	47	28	33	72	0
Fox sparrow	0	86	0	10	0	3	20	1	80	0
Golden eagle	32	81	56	13	8	5	4	1	0	0
Harlequin duck	34	81	57	13	6	5	3	1	0	0
Lark sparrow	0	71	3	19	45	9	52	1	0	0
Lewis’s woodpecker	0	76	0	16	60	7	40	1	0	0
MacGillivray’s warbler	0	76	22	16	73	7	5	1	0	0
Marsh wren	0	59	21	27	71	14	8	2	0	0
Northern goshawk	28	81	54	13	12	5	6	1	0	0
Northern harrier	0	71	22	19	72	9	6	1	6	0
Peregrine falcon	32	76	56	16	8	7	4	1	0	0
Pileated woodpecker	0	81	21	13	71	5	8	1	0	0
Sage thrasher	0	67	0	21	10	10	50	2	40	0
Tailed frog	0	76	23	16	73	7	4	1	0	0
Tiger salamander	0	67	21	22	71	10	8	2	0	0
Western bluebird	0	76	0	16	6	7	66	1	28	0
White-headed woodpecker	0	76	0	16	3	7	50	1	47	0
Wilson’s snipe	28	57	54	27	12	14	6	2	0	0
Wolverine	5	79	68	15	22	5	5	1	0	0
Wood duck	0	71	22	19	72	9	6	1	0	0

models, the best way to try to quantify the relationship between the variables, and likely outcomes for the species detailed in the conditional probability tables. Finally, we convened teams of field biologists (app. 3) familiar with the habitat conditions for surrogate species to provide feedback on the relative ranks of watersheds, as assigned by the models, to contribute to the conservation of several of the surrogate species. Following each of these reviews, the surrogate species assessment models were adjusted to better reflect scientific understanding of the relationship between the surrogate species and important variables that influenced their viability.

We statistically tested the WI models for a subset of surrogate species for which we had adequate species occurrence data. We obtained data from the Washington Department of Fish and Wildlife Heritage Database (WDFW 2006). This database contains occurrence data for a wide variety of wildlife species. We only assessed species for which we had a minimum of 30 verified locations that have occurred since 1990. This enabled us to evaluate models for the northern goshawk (674 records), tailed frog (279 records), bald eagle (153 records), golden eagle (296 records), wolverine (64 records), and peregrine falcon (33 records). For the white-headed woodpecker (88 records), we used information on species locations compiled by Mellen-McLean et al. (2013). We compared the WI scores associated with the locations of the recorded occurrences to an equal number of random locations. Our assumption was that the mean WI values from the occurrence points would be greater than the mean WI values generated from the random points if our models were operating as intended. We used 2-sample *t*-tests for unequal variances to compare the average values of the WI scores associated with the species occurrence data to those of the random points (Snedecor and Cochran 1989).

Results from five of the seven surrogate species assessment models showed statistically significant support for the hypothesis that the mean WI values generated from the species occurrence points were greater than those generated from random points (table 16). An additional model (bald eagle) had mean WI values that were greater from the occurrence points compared to the random points, but this result was not statistically significant. The final model we evaluated was for the white-headed woodpecker. We believe that the WI values were so low for all watersheds that there was an insufficient distribution of values to make our statistical approach meaningful. These results suggest that our modeling approach worked well overall, adding confidence in results from species we did not have occurrence data to evaluate.

Table 16—Results of surrogate species model evaluations

Species	Surrogate species occurrences	Random point	Mean WI occ.	Mean WI rand.	*t, P*
	- - - - - - - *Number*	- - - - - - -			
Tailed frog	279	146	1.89	1.71	1.97, 0.0008
Northern goshawk	674	674	1.72	1.56	1.96, <0.0001
Peregrine falcon	33	33	1.89	1.33	2.00, 0.004
Wolverine	64	63	2.01	1.58	1.98, <0.0001
Golden eagle	296	296	1.251	0.905	-8.827, <0.001
Bald eagle	153	153	1.612	1.588	-0.494, 0.622
White-headed woodpecker	88	88	0.16	0.17	1.97, 0.78

WI = watershed index.

Chapter 2: Individual Surrogate Species Assessments

American Marten

Introduction

Significant declines in the distribution of many carnivore species have occurred across North America since the arrival of Europeans (Giblisco 1994, Laliberte and Ripple 2004) with major reductions in marten populations resulting from the fur trade and timber harvest (Giblisco 1994). Despite protection of martens from trapping since 1953, continued habitat loss has led to increased concern for martens in the West (Ruggiero et al. 1994, 2007; Zielinski et al. 2001). American martens have a wide distribution across the western and eastern portions of the assessment area (Johnson and Cassidy 1997) and large home ranges, making them a good surrogate species to represent landscape characteristics of the cold-moist forests group in the medium/large trees family. Martens are associated with large trees, snags, and coarse woody debris (CWD), all of which are affected by timber management practices. Martens also have risk factors associated with human disturbance and roads. American martens were year-round residents of the assessment area; this assessment was for year-round habitats.

Model Description

Source habitat—

For the purpose of this analysis, source habitat for both current and historical conditions was considered to be cold-moist and cold-dry forests (i.e., subalpine fir [*Abies lasiocarpa*], grand fir [*A. grandis*], Pacific silver fir [*A. amabilis*], Engelmann spruce [*Picea englemanni*], western hemlock [*Tsuga heterophylla*], mountain hemlock [*T. mertensiana*], and western redcedar [*Thuja plicata*]) with multistories, large-tree structure, quadratic mean diameters (QMDs) >16 in and closed canopies (i.e., >50 percent). This designation of source habitat was based on associations of medium and large trees (i.e., >14 in diameter at breast height [d.b.h.]) and closed-canopy overstory vegetation in coniferous forests with martens, which were reported in the literature (Bull and Heater 2000, Buskirk et al. 1989, Campbell 1979, Gosse et al. 2005, Kirk and Zielinski 2009, Koehler et al. 1975, Martin 1987, Nams and Bourgeois 2004, Wilbert et al. 2000).

This source habitat described above is assumed to have high snag and CWD densities. Marten habitat values associated with varying snag densities have been documented in several studies (Gilbert et al. 1997, Martin and Barrett 1991, Payer and Harrison 1999, Ruggiero et al. 1998). Martin and Barrett (1991) found 16 logs per acre within habitats used by martens. Woody structures used for resting were large, with a mean d.b.h. of 36.7 in for live trees, 37.3 in for snags, and 34.7 in maximum diameter for logs with a mean density of CWD of 5.3 per acre (Buskirk et al. 1989, Slauson and Zielinski 2009). The mean age of trees at 24 of the resting sites was 339 years (range 131 to 666 years). Natal den sites were found to have 47 pieces of CWD per acre, and maternal den site had 36 pieces of CWD per acre (Ruggiero et al. 1998). Gilbert et al. (1997) found 61 logs per acre at den and rest sites. Marten avoided plots with low densities of CWD, whereas plots with high to very high densities were selected by martens (Spencer et al. 1983). Log densities of 8 to 29 per acre were considered optimum (Martin 1987). Andruskiw et al. (2008) showed that the frequency of prey encounter, prey attack, and prey kill were higher in old uncut forests for American martens, despite the fact that small-mammal density was similar to that in younger logged forests. These differences in predation efficiency were linked to higher abundance of CWD, which seems to offer sensory cues to martens, thereby increasing the odds of hunting success.

Patch size—

Snyder and Bissonette (1987) reported limited use by martens of patches <6 ac of suitable habitat. Patches used by resident martens were 18 times larger (median = 11 ac) than patches that were not used (median = 0.2 ac) and were closer to adjacent forest preserves (Chapin et al. 1998). Median size of largest forest patch in martens' home ranges was 61 ac for females and 100 ac for males (Chapin et al. 1998). Similarly, Slauson et al. (2007) reported a minimum patch size used by American martens of >33 ac with a mean patch size of 73 ac. Potvin et al. (2000) recommended that uncut forest patches be >40 ac to maximize core area and to minimize edge. Generally, more habitat, larger patch sizes, and larger areas of interior forest were important predictors of occurrence (Chapin et al. 1998, Hargis et al. 1999, Kirk and Zielinski 2009, Potvin et al. 2000). Based on those findings, the following classes were used to describe the mean patch size of source habitat within watersheds (fig. 4):

- Low: <6 ac mean patch size of source habitat within a watershed
- Moderate: 6 to 40 ac mean patch size of source habitat within a watershed
- High: >40 ac mean patch size of source habitat within a watershed

Riparian habitat—

Martens select for riparian habitats throughout their range (Anthony et al. 2003, Baldwin and Bender 2008, Buskirk et al. 1989, Martin 1987) and habitats near water (Bull et al. 2005). Fecske et al. (2002) characterized this relationship by distinguishing areas that were less than and greater than 330 ft from streams. The

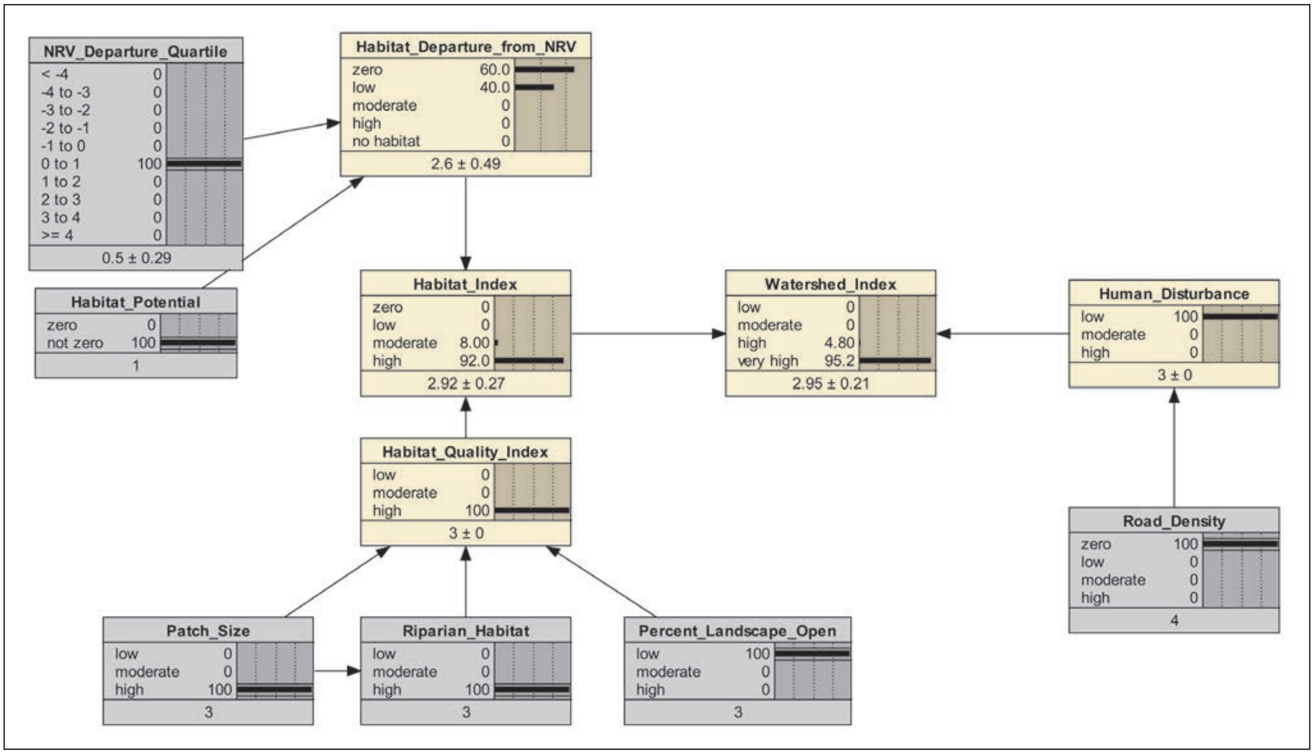

Figure 4—Surrogate species assessment model for American marten.

suitability of riparian habitat was evaluated in this analysis by determining what percentage of the total area within 330 ft of streams (i.e., perennial, orders 3 to 8) was source habitat on a watershed basis and then placing watersheds in the following classes (fig. 4):

- Low: <25 percent of a watershed within 330-ft buffers was source habitat
- Moderate: 25 to 50 percent of a watershed within 330-ft buffers was source habitat
- High: >50 percent of a watershed within 330-ft buffers was source habitat

Percentage of landscape open—

Percentage of the landscape in openings was a primary factor in determining the quality of American marten habitat. Hargis and Bissonette (1997) and Hargis et al. (1999) reported very little use by martens in landscapes with 25 percent or greater in openings. Potvin et al. (2000) also reported that marten home ranges contained less than 30 to 35 percent clearcut openings. Clearcuts supported 0 to 33 percent of population levels of martens compared to nearby uncut forest (Snyder and Bissonette 1987, Soutiere 1979, Thompson et al. 1989). Marten population reductions of 67 percent were reported following removal of 60 percent of forest (Soutiere 1979) and 90 percent with 90 percent forest removal (Thompson 1994). Chapin et al. (1998) reported that martens tolerated 20 percent (median value) of their home range in regenerating forest. More recently, Dumyahn et al. (2007) demonstrated that American martens did not establish home ranges unless ≥70 percent of an area was suitable habitat. Also, Broquet et al. (2006) found that the movement of American marten individuals and gene flow through logged landscapes did not follow a linear, shortest-path movement. Rather, movement corresponded to a least-cost path that avoided openings. The following classes were developed from those findings and used to characterize watersheds (fig. 4):

- Low: 0.0 to 10.0 percent of a watershed in open condition
- Moderate: 10.1 to 30.0 percent of a watershed in open condition
- High: >30 percent of a watershed in open condition

Vegetation types were considered "closed" for this analysis if they had a tree canopy (i.e., ≥10 percent tree cover in the overstory layer). "Open" vegetation classes were all vegetation types without a tree canopy.

Habitat effectiveness—
Hodgman et al. (1994) reported 90 percent of marten mortality resulted from trapping on an area with a road density of 0.42 mi/mi^2. Thompson (1994) also reported that trapping was the major source of mortality for martens. He also observed that predation and trapping mortality rates were higher in logged forests (with road development) than in uncut forests. Alexander and Waters (2000) observed avoidance by martens of areas within 160 ft of roads. Roads also facilitate the removal of snags as firewood and for safety considerations (Bates et al. 2007, Gaines et al. 2003, Wisdom and Bate 2008). The findings of Godbout and Ouellet (2008) indicated that increasing road density results in lower quality habitat for American martens. Webb and Boyce (2009) showed that increased disturbance, particularly road access and oil and gas well sites, negatively affected habitats of American martens and reduced trapper success. The following density classes were summarized within source habitat by watershed (fig. 4):

- Zero: <0.1 mi/mi^2 of open roads
- Low: 0.1 to 1.0 mi/mi^2 of open roads
- Moderate: 1.1 to 2.0 mi/mi^2 of open roads
- High: >2.0 mi/mi^2 of open roads

Calculation of historical conditions—
Values of the model variables were set with the following values to estimate historical habitat conditions:

- Departure of source habitat from natural range of variability (NRV)—0.5
- Patch size: Class increase from current condition
- Riparian habitat: Same as current condition
- Percentage of landscape open: Same as current condition
- Road density: Class zero

Geographic information systems (GIS) databases used—

- Cover type
- Source habitat (departure from NRV)
- Patch size
- Riparian
- Percentage of landscape open
- Canopy cover
- Quadratic mean diameter
- Streams
- Riparian
- Road density

The relative sensitivity of the watershed index (WI) values to the variables used in the American marten model are shown in table 17.

Watershed scores—
This analysis indicated that 64 of 72 (89 percent) watersheds within the assessment area provided habitat for American martens (fig. 5). Forty-eight (75 percent) of the watersheds with habitat were well below their historical median levels of source habitat (i.e., < -1 class), while 16 (25 percent) of the watersheds were at or above historical median levels (i.e., $\geq$ -1 class) (fig. 5). Watersheds that currently have the

Table 17—Relative sensitivity of watershed index values to variables in the model for American marten

Variable	Sensitivity rank
Habitat departure (amount)	1
Road density	2
Percentage of landscape open	3
Patch size	4
Riparian habitat	5

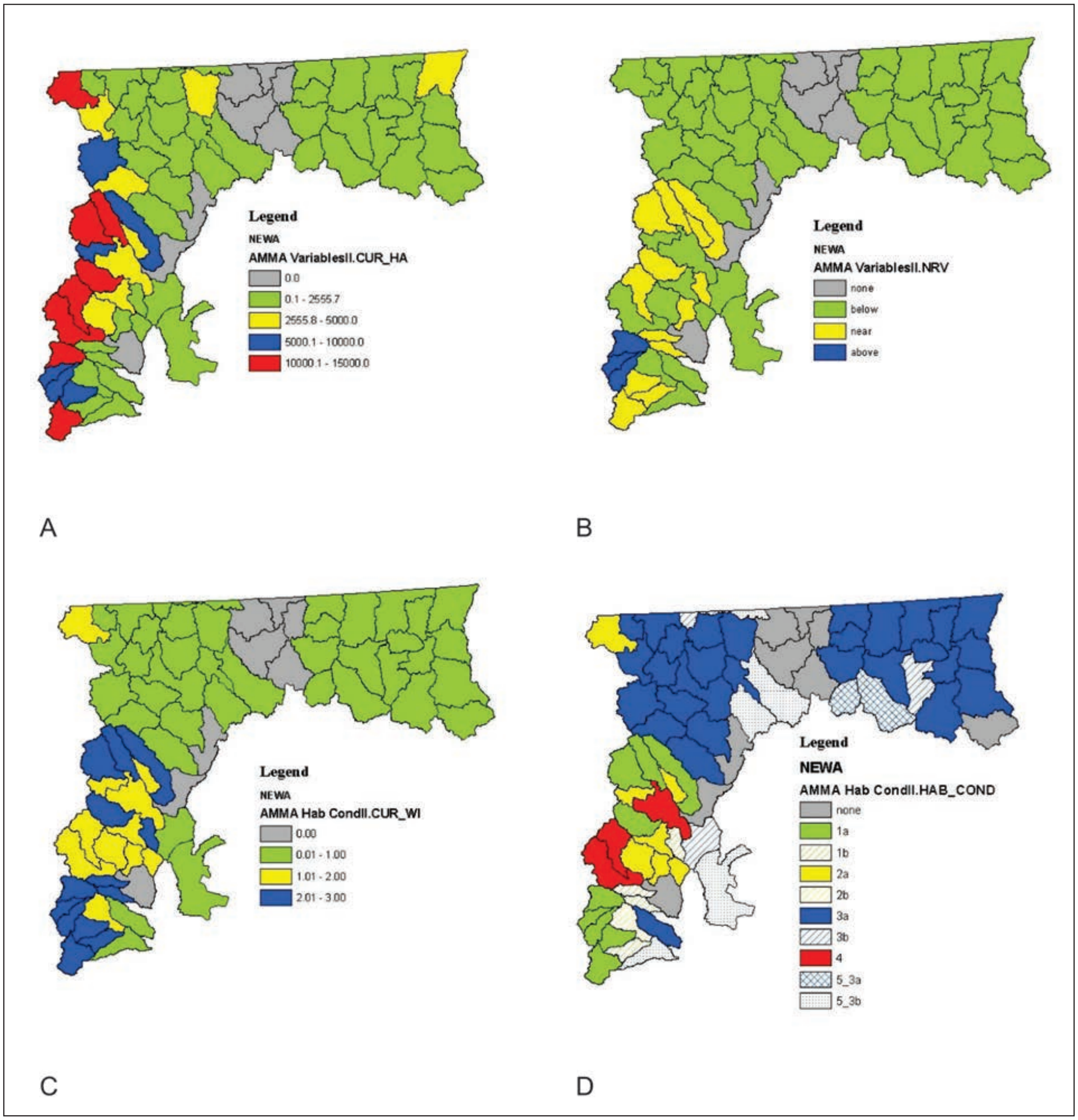

Figure 5—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, (D) and habitat condition class for American marten (AMMA) by watershed in the northeast Washington assessment area (NEWA).

greatest amount of source habitat included Cle Elum River, Chiwawa River, Icicle Creek, Little Naches River, Ross Lake, Upper Tieton River, Upper Yakima River, and White-Little Wenatchee (fig. 5). The watersheds with the least amount of source habitat were located across the eastern and northern portions of the assessment area. Wisdom et al. (2000) also reported strongly negative trends in the amount of source habitat across the northern portion of the North Cascades ecological reporting unit (ERU) (northern Okanogan National Forest) and across the Northern Glaciated Mountains ERU (Colville National Forest).

Currently, 20 percent (n = 13) of the watersheds with source habitat had WI scores that were high (i.e., ≥2.0) (fig. 5). All of these were located on the Wenatchee National Forest portion of the assessment area. Eleven (17 percent) additional watersheds had WI scores that were moderate (i.e., >1.0 to <2.0). Again, these were distributed on the Wenatchee National Forest portion of the assessment area.

We assessed the effect of size of patches of source habitat by calculating the mean patch size of source habitat within each watershed. This analysis showed that average patch sizes within watersheds ranged from <0.4 to >2.4 ha. All watersheds, but one (Icicle Creek) had low (<37 ac) average source habitat patch sizes. In addition, the percentage of the landscape classified as open in this analysis ranged from <2 to >87 percent. Thirteen of the watersheds had low (<10 percent open) levels of open landscape, 24 moderate (10.1 to 30.0 percent open), and 27 high (>30 percent open).

The percentage of 330-ft stream buffers in source habitat ranged from <1 to >30 percent across all watersheds in this analysis. All watersheds but six (American River, Bumping River, Chiwawa River, Little Naches River, Stehekin, and White-Little Wenatchee) had low (i.e., <25 percent) amounts of source habitat within the riparian zone.

Six of the watersheds in the assessment area did not have any open roads in marten source habitat. The percentage of source habitat in watersheds with high open road densities ranged from 0.0 to 100.

Viability outcome scores—
The viability outcome index (VOI) model incorporated the weighted watershed index (WWI) scores (described earlier), a habitat distribution index, and a habitat connectivity or permeability index. The WWI scores indicated that the current habitat capability within the assessment area for American martens was 76 percent of the historical capability. The ability of American martens to disperse across the assessment area was considered an issue for this species, thus we calculated permeability across the assessment area. The results were 41 percent of the assessment area had a low permeability for dispersal, 39 percent moderate, 20 percent high.

In summary, under historical conditions, there was a high probability that viable populations of American martens and other species associated with the cold moist forests group in the medium/ large trees family were well distributed throughout the assessment area.

All five ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (the median was calculated across all watersheds with source habitat). Thirty-one percent (n = 22) of watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 37.0 percent probability that the current viability outcome for American marten was class B and a 35.8 percent probability the outcome was class C (fig. 6). It is likely that the other species associated with the cold-moist forests group in the medium/large trees family had similar outcomes.

Dispersal across the assessment area was also an issue for this species historically (38 percent of the assessment area had a low permeability for dispersal, 34 percent moderate, 28 percent high). Historically, all five ecoregions contained ≥1 watersheds with >40 percent of the median amount of historical source habitat (the median was calculated across all watersheds with source habitat). Seventy-six percent (n = 55) of watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 58.6 percent probability that the historical viability outcome for American marten was class A and a 30.5 percent probability that the historical outcome was class B (fig. 6).

In summary, under historical conditions, there was a high probability that viable populations of American martens and other species associated with the cold-moist forests group in the medium/large trees family were well distributed throughout the assessment area. The effects of development and habitat change across the assessment area has led to a lower probability that populations of American martens and all other species associated with the cold-moist forests group in the medium/ large trees family were viable and a finding that they were likely well-distributed in only a portion of the assessment area.

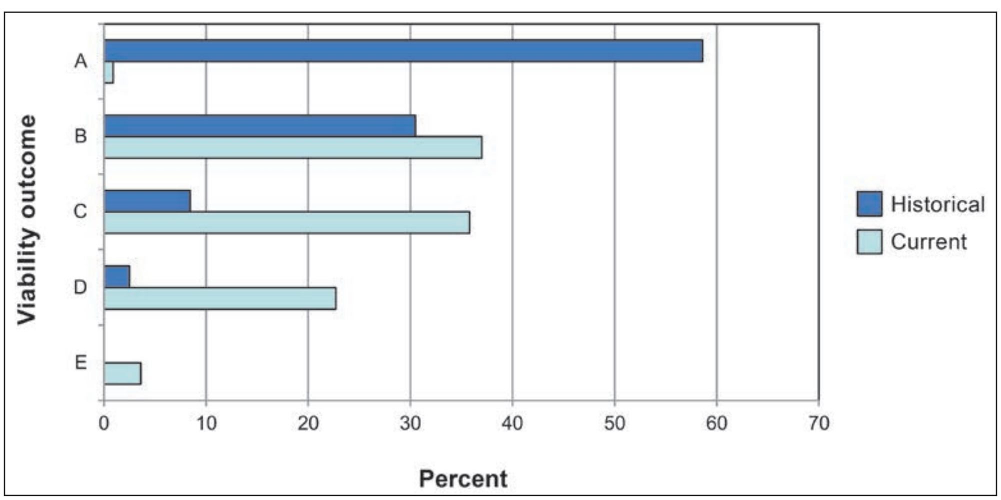

Figure 6—Current and historical viability outcomes for American martens in the northeast Washington assessment area.

Our results for this species were similar to those reported in the broad-scale habitat analysis by Wisdom et al. (2000) in the Interior Columbia Basin Ecosystem Management Project (ICBEMP). According to the ICBEMP terrestrial vertebrate habitat analyses, historical source habitats for American martens included portions of the Northern Cascades and the Northern Glaciated Mountains ERUs that overlap our assessment area (Wisdom et al. 2000). Within this historical habitat, declines in source habitats have been extensive, -60 percent in the Northern Cascades and -88 percent in the Northern Glaciated Mountains according to Wisdom et al. (2000).

Management Considerations

The following management issues for American martens and all other species associated with the cold-moist forests group in the medium/large trees family were identified during this assessment and from the published literature for considerations by managers:

1. Reduction and fragmentation of old-forest habitats.
2. Negative effects of roads in source habitats.
3. The sustainability of dry forest habitats adjacent to source habitats for species associated with the cold-moist forests group in the medium/large trees family is a concern owing to risk of fire spreading from dry habitats into these source habitats (Townsley et al. 2004).

Bald Eagle

Introduction

The bald eagle was chosen as a surrogate species for the riparian family and the large tree or snag/open water group. Bald eagles were recently removed from U.S. Fish and Wildlife (USFWS) federal list of threatened and endangered species (Stinson et al. 2007). The primary risk factor identified for the bald eagle is human disturbance. In Washington, bald eagles nest primarily west of the Cascade Range, with scattered breeding areas along major rivers in the eastern part of the state (Stinson et al. 2007, Watson and Rodrick 2000). Wintering eagles occur along the upper and lower Columbia River and its tributaries, with major wintering concentrations located along rivers with salmon runs (Stinson et al. 2007, Watson and Rodrick 2000).

Model Description

Source habitat—

Breeding territories for bald eagles are established in upland woodlands and lowland riparian stands with a mature conifer or hardwood component (Anthony and Isaacs 1989, Garrett et al. 1993, Watson and Pierce 1998). Territory size and configuration are influenced by factors such as density of breeding bald eagles (Gerrard and Bortolotti 1988), quality of foraging habitat, and the availability of prey (Watson and Pierce 1998). The three main factors that influence the location of nests and territories include proximity of water and availability of food; availability of nesting, perching, and roosting trees; and the density of breeding-age bald eagles in the area (Stalmaster 1987). Anthony and Isaacs (1989) reported that nest sites in older contiguous forest habitats with low levels of human disturbance resulted in higher levels of bald eagle productivity.

We modeled source habitat for bald eagles using a combination of tree structure (e.g., canopy layers and canopy closure) and size class, elevation, and proximity to water bodies described in the literature listed above. Our model of source habitat included the following:

- Elevation: <3,000 ft
- Waterbody: Waterbodies >5 ac (including large stream reaches)
- Distance from suitable-sized waterbody: 984-ft buffer
- Tree structure and size classes: Single and multistory, >15 in d.b.h. calculated as the QMD

The current habitat departure class for the bald eagle was set at -1 class for all watersheds (see page 17 calculation of reference condition).

Late-successional forest—

Several studies have reported the importance of late-successional forests in defining quality of nesting habitat and influencing productivity of bald eagles (Anthony and Isaacs 1989, Garrett et al. 1993). We included the amount of potential source habitat that was in a late-successional forest condition as a factor that affected habitat quality (fig. 7). We used the following GIS data layers to map late-successional forest as a subset of the total potential source habitat:

- Single and multistory forests, >20 in d.b.h. (calculated as QMD)
- Canopy closure >50 percent

We then used the following categories in our model to categorize the proportion of source habitat within a watershed composed of late-successional habitat:

- Zero: No source habitat is late successional
- Low: >0 to 20 percent of the source habitat is composed of late-successional forest
- Moderate: >20 to 50 percent of the source habitat is composed of late-successional forest
- High: >50 percent of the source habitat is composed of late-successional forest

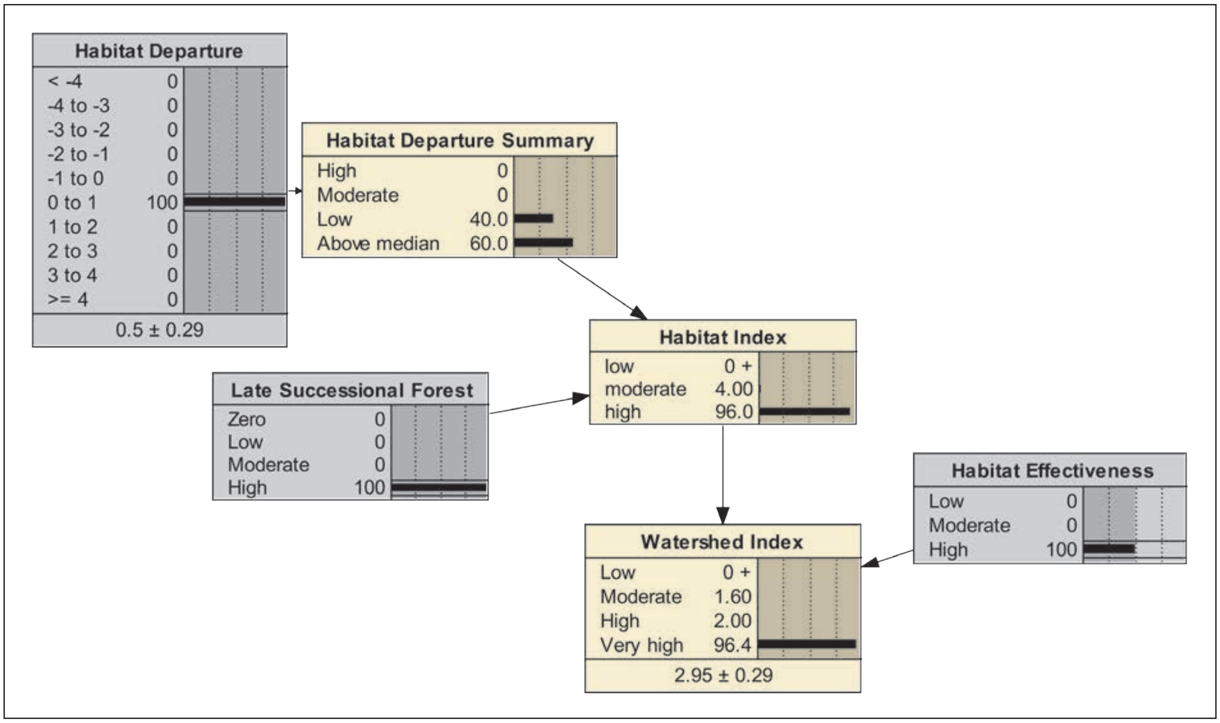

Figure 7—Surrogate species assessment model for the bald eagle.

Habitat effectiveness—

Reported responses of bald eagles to human disturbances have ranged from spatial avoidance of the activity to reproductive failure (Anthony et al. 1995, Buehler et al. 1991, McGarigal et al. 1991, Watson 1993), although in some cases, bald eagles tolerate human disturbances (Harmata and Oakleaf 1992). Bald eagles seem to be more sensitive to humans afoot than to vehicular traffic (Grubb and King 1991, Skagen et al. 1991, Stalmaster and Newman 1978). Fletcher et al. (1999) reported that the abundance of bald eagles was lower in riparian habitats with nonmotorized trails compared to riparian habitats without trails. Recommended buffer distances to reduce the potential for disturbance to bald eagles during the nesting period have ranged from 984 to 2,624 ft (Anthony and Isaacs 1989, Fraser et al. 1985, Stalmaster 1987). Grubb and King (1991) evaluated the influence of pedestrian traffic and vehicle traffic on bald eagle nesting activities and recommended buffers of 1,800 ft for pedestrians and 1,500 ft for vehicles.

We included a habitat effectiveness variable in the bald eagle model to assess the potential influence of human activities on source habitats. We used the bald eagle nesting habitat disturbance index described in Gaines et al. (2003a). To do this, we buffered roads and motorized trails by 1,500 ft on each side and nonmotorized trails by 1,800 ft on each side to establish zones of influence. Next, we intersected this with our maps of source habitat to determine the proportion of source habitat within each watershed that was inside a zone of influence. We then developed the following categories to assess the potential influences of increasing proportions of source habitat within a zone of influence (fig. 7):

- Low habitat effectiveness: <30 percent of the source habitat outside of a zone of influence.
- Moderate habitat effectiveness: 30 to 50 percent of the source habitat outside of a zone of influence.
- High habitat effectiveness = >50 percent of the source habitat outside of a zone of influence.

Calculation of historical conditions—

- Historical habitat departure: Departure class 1 assumed no departure from historical conditions.
- Late-successional forest: Moderate
- Habitat effectiveness: High

The relative sensitivity of WI values to variables in the model for bald eagles is shown in table 18.

Model evaluation—

We used 153 bald eagle occurrence points and 153 random points to evaluate the surrogate species assessment model. The mean WI value derived from the occurrence points was 1.612 and the mean WI value derived from the random points was 1.588. While the trend showed slightly better WI values for the occurrence points, this was not a statistically significant result (t = -0.494, P = 0.622).

Table 18—Relative sensitivity of watershed index values to variables in the model for the bald eagle

Model variables	Order of variable weighting
Source habitat	1
Late-successional forest	2
Habitat effectiveness	3

Assessment Results

Watershed scores—

This analysis showed that 60 of 72 (83 percent) watersheds within the assessment area had habitat. Thirteen percent (n = 8) of the watersheds assessed had high WI scores ($\geq$2.0), and an additional 62 percent (n = 39) had moderate scores (>1.5 to 2.0). The watersheds with a high WI score were Curlew, Upper Okanogan River, Mill, Sinlahekin Creek, Stensgar/Stranger, Lower Okanogan River, Ross Lake, and Stehekin River.

The median amount of source habitat across all watersheds was 238.5 ac. Watersheds with the most source habitat (>1,200 ac) included Cle Elum, Entiat, White-Little Wenatchee, Wenatchee, Chiwawa, and Upper Yakima Rivers. Because this is a riparian species, habitat departure for all watersheds was classified in the -1 departure class.

The percentage of late-successional forest within source habitat was low and reduced the habitat quality in most watersheds. One-third (n = 21) of the watersheds did not have late-successional forest within source habitat, 54 percent (n = 34) had low levels (>0 to 20 percent of the source habitat), 8 percent (n = 5) had moderate levels (>20 to 50 percent of the source habitat), and only 5 percent (n = 3) had high (>50 percent) levels of late-successional forest in source habitat. The watersheds with high levels of late-successional forests are Lightning Creek, Ruby Creek, and Ross Lake.

Model results indicate that human activities are having a large impact on the effectiveness of source habitat for bald eagles across the assessment area. Activities associated with roads and trails have reduced habitat effectiveness to low levels in 68 percent (n = 43) of the watersheds and moderate levels in 19 percent (n = 12) of the watersheds. Only 13 percent (n = 8) of the watersheds assessed had a high level of habitat effectiveness within bald eagle source habitat.

Viability outcome scores—

The VOI model incorporated the WWI scores, and a habitat distribution index. The WWI scores indicated that the current habitat capability for bald eagles within the assessment area is 56 percent of the historical capability. This score is largely influenced by the effects of human activities within or adjacent to bald eagle habitat. Seventy-two percent of the watersheds (n = 52) had both current and historical amounts of source habitat >40 percent of the historical median, and they were distributed across all ecoregions.

A reduction in the availability of suitable environments for the bald eagle may have occurred in the assessment area compared to the historical distribution and condition of their habitats.

There is a 73 percent probability that the current viability outcome for the bald eagle is C, which suggests that suitable environments for bald eagles are distributed frequently as patches or exist in low abundance (fig. 8). Species with this outcome are likely well-distributed in only a portion of the assessment area. Historically, there was a 76 percent probability that the viability outcome was A, where suitable environments were more broadly distributed or of high abundance. In addition, the suitable environments were better connected, allowing for interspecific interactions. A reduction in the availability of suitable environments for the bald eagle may have occurred in the assessment area compared to the historical distribution and condition of their habitats.

Gaines et al. (2003a) found that roads and other human activities have had a disproportionately high impact on riparian habitats owing to their proximity to riparian areas. This result is the same for the bald eagle as well as other species associated with streamside forested riparian habitats such as the harlequin duck and tailed frog (see pages 118 and 185 of this assessment).

Management Considerations

The following issues were identified during this assessment and from the published literature regarding the viability of populations of bald eagle and likely other species in the riparian family and the large tree or snag/open water group for the considerations of managers:

1. Low availability of late-successional forest within riparian source habitat.
2. Human activities reduced the effectiveness of source habitats.

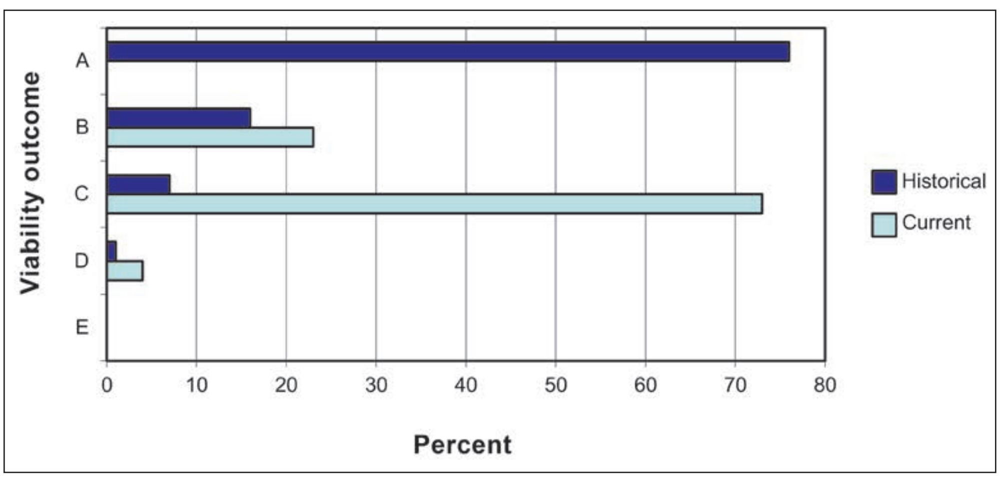

Figure 8—Current and historical viability outcomes for the bald eagle in the northeast Washington assessment area.

Bighorn Sheep

Introduction

Bighorn sheep were selected as surrogate species to represent the grassland/shrubland group. They are distributed across the assessment area in nine herds, each with a limited range. Many of these herds are a result of efforts to reintroduce bighorn sheep populations throughout eastern Washington following their extirpation. Historically, California bighorn sheep occurred on the eastern slopes of the Cascade Range from the Canadian border south to the Columbia River. Most of the herds were gone before 1900; the last known survivors, on Chopaka, died in 1925 (Johnson 1999). Rocky Mountain bighorn sheep occurred historically in the Selkirk Mountains but were extirpated from this area by the late 1800s (Johnson 1999). Currently, there are around 550 to 675 Rocky Mountain and California bighorn sheep within the assessment area (table 19); this assessment was for year-round habitats.

Model Description

Source habitat—

Previous studies have used GIS to model bighorn sheep habitat (Cassirer et al. 1997, Johnson and Swift 2000, Smith et al. 1991, Zeigenfuss et al. 2000). Source habitat for bighorn sheep was modeled by Begley (2008) using logistic regression and telemetry data from the Swakane bighorn sheep herd located in the central portion of the assessment area. The model was then extrapolated across the assessment area,

Table 19—Estimated sizes of the bighorn sheep populations in the herds located in the northeast Washington assessment area[a]

Herd name	Estimated numbers	Adjacent national forest
Tieton	33 to 41	Okanogan-Wenatchee
Clemens Mountain	140 to 172	Okanogan-Wenatchee
Umtanum	156 to 190	Okanogan-Wenatchee
Swakane	48 to 58	Okanogan-Wenatchee
Lake Chelan	41 to 51	Okanogan-Wenatchee
Sinlahekin	27 to 33	Okanogan-Wenatchee
Mount Hull	59 to 72	Okanogan-Wenatchee
Vulcan Mountain	21 to 27	Colville
Hall Mountain	26 to 33	Colville

[a] Based on WDFW (2008).

and telemetry data from the Tieton and Vulcan herds were used to test the extrapolated model. The variables that were identified in the model included vegetation zones, canopy closure, and escape terrain (Begley 2008). Based on this, source habitat within the assessment area was mapped using the following spatial data sources (fig. 9).

- Vegetation zones: Douglas-fir, ponderosa pine, and shrub-steppe cover types within 1,600 ft of escape terrain
- Canopy closure: ≤60 percent
- Escape terrain: Area with slope between 31 and 85 degrees and >4.0 ac

Patch size—

Several studies have shown that the size of a patch of suitable habitat can influence bighorn sheep occupancy, habitat use, success of reintroduction efforts, and population demographics (Cassirer et al. 1997, Gross et al. 2000, Johnson and Swift

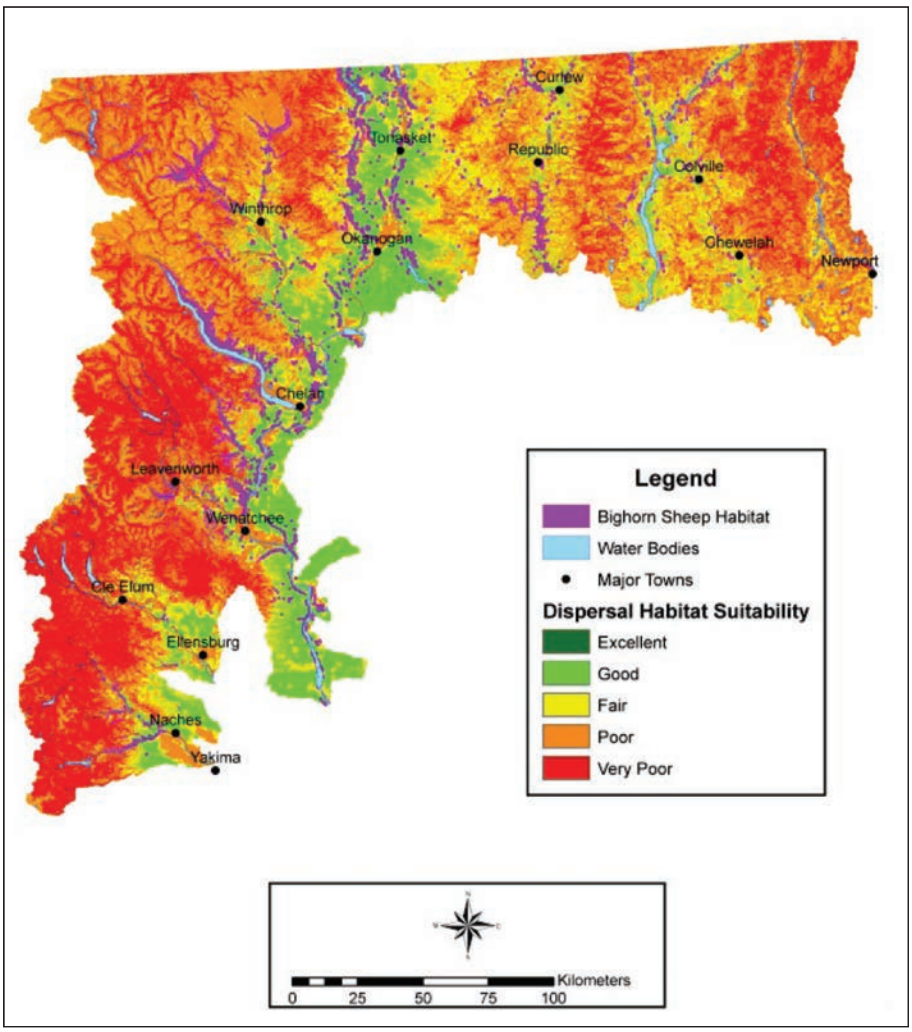

Figure 9—Bighorn sheep source habitat and dispersal habitat suitability.

2000, Singer et al. 2000, Zeigenfuss et al. 2000). Begley (2008) used an area (4 ac of escape terrain plus a 1,600-ft buffer) of about 200 ac in the habitat model. Based on this, a range of size categories was used to assess the quality of habitat patches within a watershed.

- Low: Average source habitat patch size in the watershed <100 ac
- Moderate: Average source habitat patch size in the watershed 100 to 400 ac
- High: Average source habitat patch size in the watershed >400 ac

Domestic sheep grazing—
Domestic sheep overlapping or in proximity to bighorn sheep have been reported to result in the spread of *Pasturella* among bighorn sheep, with subsequent die-offs in bighorn sheep populations (Foreyt 1989, Foreyt and Jessup 1982, Foreyt et al. 1994, Schommer and Woolever 2008). Gross et al. (2000) showed that the risk of disease spread was a more important variable in determining the extinction rates of bighorn sheep than habitat restoration. Therefore, to evaluate the potential for disease spread, we assessed the area of active domestic sheep grazing allotments within 1.0 mi of bighorn sheep source habitat for each watershed. This variable was then scaled from 0 percent overlap to 100 percent overlap in increments of 10 percent (fig. 10). Conditional probability tables were calibrated so that when >20 percent of the 1-mi-buffered source habitat within a watershed was in an active domestic sheep allotment, the risk of disease spread to bighorn sheep was considered high.

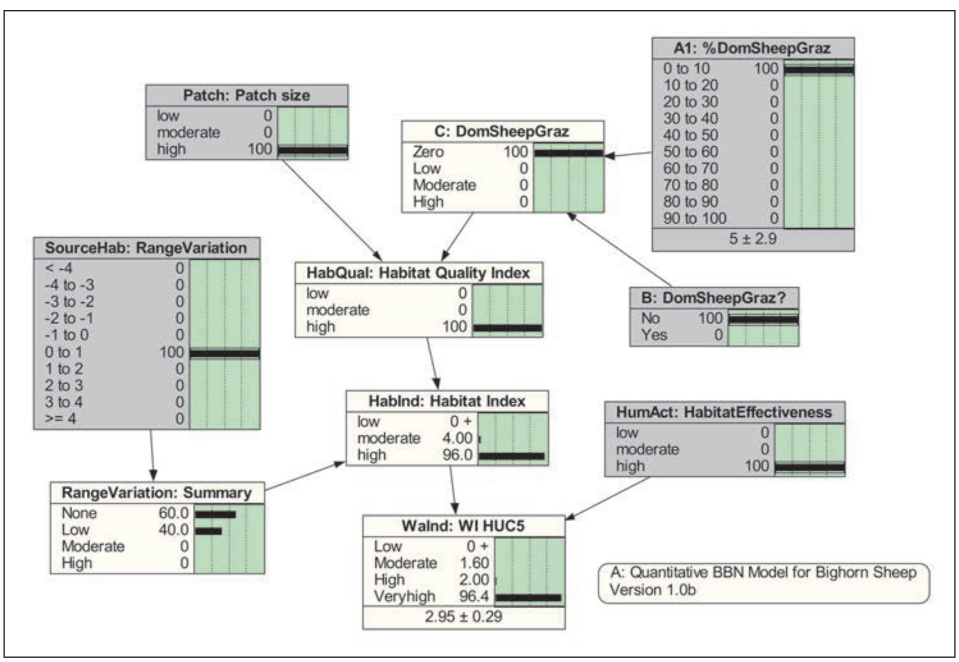

Figure 10—Surrogate species assessment model for bighorn sheep.

Habitat effectiveness—

Bighorn sheep have been reported to respond to human disturbance (Hicks and Elder 1979; King and Workman 1986; Leslie and Douglas 1980; MacArthur et al. 1979, 1982; Papouchis et al. 2001; Smith et al. 1991). MacArthur et al. (1979) showed that the heart rate of bighorn sheep varies inversely with the distance from a road. MacArthur et al. (1982) reported that sheep were affected by a human approaching within 160 ft, and Papouchis et al. (2001) found that bighorn sheep respond to hikers at an average distance of 650 ft. They also showed avoidance of roads was greater for high-use (5 to 13 vehicles per hour) than low-use (1 vehicle per hour) roads. On average, radio-collared sheep were located 1,600 ft from high-use roads compared to 1,200 ft from low-use roads (Papouchis et al. 2001). Smith et al. (1991) developed a habitat suitability model for bighorn sheep and considered areas within 330 ft of low to moderate human use (<500 visitors per year) trails and roads as unsuitable, and areas within 490 ft of high human use (>500 visitors per year) trails and roads as unsuitable. Based on this information, Gaines et al. (2003a) developed a bighorn sheep habitat disturbance index that we used to assess habitat effectiveness. The index was based on a zone of influence on each side of roads or trails that was overlaid with the bighorn sheep source habitat to assess the proportion of source habitat within a zone of influence (table 20).

We then categorized the amount of source habitat within the zone of influence to assess habitat effectiveness for each watershed as follows (based on Gaines et al. 2003a) (fig. 10):

- High habitat effectiveness: >70 percent of the source habitat outside a zone of influence
- Moderate habitat effectiveness: 50 to 70 percent of the source habitat outside a zone of influence
- Low habitat effectiveness: <50 percent of the source habitat outside a zone of influence

Table 20—Zone of influence applied to each side of a trail or road based on road type and use level for bighorn sheep

Trail or road type and status	Zone of influence[a]
	Feet
Nonmotorized trial (ski or hiking)	650
Motorized trail	1,150
Road ≤1 vehicle per day	1,150
Road >1 vehicle per day	1,640

[a] Based on Gaines et al. (2003).

Calculation of historical conditions—

Values of the surrogate species model variables were set with the following values to estimate historical habitat conditions:

- Source habitat: Set at the 1 departure node
- Patch size: Assumed to be one category larger than current in order to estimate the effects of fire exclusion on the canopy-closure variable
- Domestic sheep grazing: Class zero
- Habitat effectiveness: Class high

The relative sensitivity of WI values to model variables for bighorn sheep is displayed in table 21.

Assessment Results

Watershed scores—

Most, or 73 percent (n = 52) of the watersheds had moderate WI scores (1.0 to 2.0), while 8 percent (n = 6) had high (>2.0), and the remainder (18 percent, n = 13) had low (<1.0) scores.

Watersheds with the greatest amount of source habitat (>19,000 ac) currently included the Upper Methow, Entiat, Upper Columbian-Swamp Creek, Sinlahekin, Columbia River-Lynch Coulee, Lake Entiat, Okanogan River-Omak Creek, and Lower Lake Chelan. Of these watersheds, the Lower Lake Chelan, Entiat River, and Upper Methow River had more than 60 percent of the source habitat on National Forest Systems lands. The median amount of source habitat across all watersheds was 5,379 ac.

Average patch sizes of source habitat were high (>400 ac) in 14 percent (n = 10) of the watersheds, moderate (100 to 400 ac) in 24 percent (n = 17), and low (<100 ac) in 62 percent. Fire exclusion has resulted in an increase in the density of trees in formerly open stands, reducing forage quality and causing bighorns to avoid these areas because of reduced visibility (Wisdom et al. 2000).

Table 21—Relative sensitivity of watershed index values to variables in the model for bighorn sheep

Model variables	Order of variable weighting
Source habitat	1
Domestic sheep grazing	2
Habitat effectiveness	3
Patch size	4

Habitat effectiveness, indexed by assessing the amount of source habitat within a distance buffer of roads and trails was ranked low in 61 percent (n = 43) of the watersheds, moderate in 25 percent (n = 18), and high in only 14 percent (n = 10) of the watersheds.

The potential for disease spread from domestic sheep to bighorn sheep was assessed by identifying the amount of source habitat within 1.0 mi of an active domestic sheep allotment. This assessment showed that 14 (20 percent) of the watersheds had ≥20 percent of the source habitat within 1.0 mi of an active grazing allotment and had the greatest effect on reducing the watershed scores. Watersheds with the highest proportions (>60 percent) of source habitat near or within active grazing allotments included the Bumping River, Little Naches, Mad River, Naches, Rattlesnake Creek, Sinlahekin Creek, and Wenas Creek.

Viability outcome scores—
The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI portion of the viability outcome model showed that the current habitat capability was at 57 percent of the historical habitat capability when calculated across all of the watersheds within the assessment area. The major factors that contributed to the decline in the habitat capability were the influence of human access on habitat effectiveness and the potential impact of disease transmission from domestic sheep grazing.

Forty-seven (66 percent) of the watersheds had source habitat amounts above the 40 percent median (2,150 ac), and all of the ecoregions had one or more watersheds with more than 40 percent of the median. The habitat distribution index showed that 88 percent of the watersheds were classified as low permeability and 12 percent as moderate (fig. 9).

The current viability outcome for bighorn sheep across the assessment area had a probability of 52 percent for outcome C and a 44 percent probability of outcome D, indicating that suitable environments are frequently isolated and that this species is not well distributed across the assessment area (fig. 11). Historically, the viability assessments showed a probability of 51 percent for outcome A and 33 percent for outcome B, indicating that bighorn sheep populations and habitat were well distributed and better connected across the assessment area compared to contemporary landscapes.

Wisdom et al. (2000) found large declines in the amount of bighorn sheep habitat throughout the interior Columbia Basin including eastern Washington. Additionally, Raphael et al. (2001) showed a similar viability outcome using a similar habitat

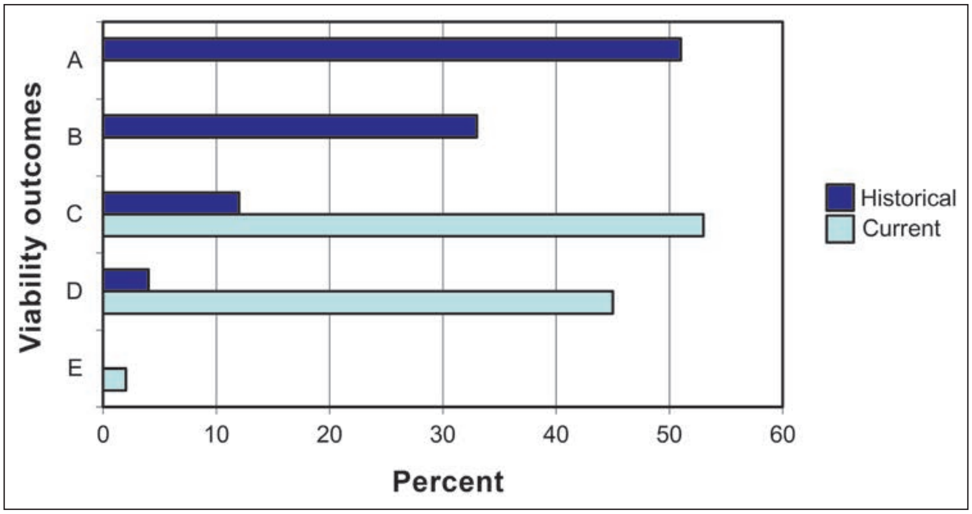

Figure 11—Current and historical viability outcomes for bighorn sheep in the northeast Washington assessment area.

In summary, under historical conditions, there was a high probability that viable populations of bighorn sheep, and other species associated with the grassland/shrubland group, were well distributed throughout the assessment area. Currently, populations are reasonably well distributed but relatively small and isolated across the assessment area.

model, and reported downward trends in habitat quantity and quality for bighorn sheep for the entire interior Columbia Basin.

In summary, under historical conditions, there was a high probability that viable populations of bighorn sheep, and other species associated with the grassland/shrubland group, were well distributed throughout the assessment area. Currently, populations are reasonably well distributed but relatively small and isolated across the assessment area.

Management Considerations

The following issues were identified during this assessment and from the published literature regarding the viability of populations of bighorn sheep across the assessment area for the consideration of managers:

1. Fire exclusion has resulted in an increase in the density of trees in formerly open stands, reducing forage quality and causing bighorns to avoid these areas because of reduced visibility.
2. The proximity of domestic sheep allotments adjacent to bighorn sheep source habitat has resulted in a potential to spread disease to some populations.
3. Human activities have reduced the effectiveness of bighorn sheep source habitat in several watersheds.

Black-Backed Woodpecker

Introduction

Black-backed woodpeckers were chosen as surrogate species for the postfire group. They represent postfire habitat with a relatively high density of trees and snags, as compared to other species in the group (e.g., Lewis's woodpecker). Black-backed woodpeckers have been reported to exist at higher densities and are more productive in postfire habitats than in other forest conditions in which they occur. They range across the assessment area and are sensitive to salvage activities, making them a good surrogate species. These birds are resident throughout the assessment area.

Model Description

Source habitat—

Black-backed woodpeckers are associated with boreal and montane coniferous forests, especially in areas with standing dead trees such as burns (Dixon and Saab 2000). This bird is extremely restricted in its use of habitat types and is strongly associated with recently burned forests (Gentry et al. 2007, Hutto 2006, Nappi et al. 2003, Raphael and White 1984, Saab and Dudley 1998). In the northern Rocky Mountains of the United States, a regionwide landbird survey and extensive literature review revealed that the species is almost exclusively associated with recently burned forests (<5 years), although it is occasionally observed in mixed-conifer, lodgepole pine, Douglas-fir, and spruce-fir forests (Hutto 1995a, 1995b). Several studies have found that in recently burned forests, black-backed woodpecker nest sites were found at higher densities and had higher nest success in areas that were not salvage logged and had high densities of standing snags (Haggard and Gaines 2001, Saab and Dudley 1998, Saab et al. 2009).

In California, these woodpeckers occurred in burned sites 6 to 8 years after fire, but were not recorded during surveys 15 to 19 years and 21 to 25 years postfire, although they were present in very low densities during all periods in unburned control plots (Raphael et al. 1987). Hutto (1995b) suggested that a mosaic of recently burned forests may best represent source habitat, where local reproduction exceeds mortality. Several researchers have suggested that the low densities of woodpeckers in unburned forests may indicate sink populations that are maintained by birds that move into these areas as conditions on postfire habitats become less suitable over time (Hutto 1995, Murphy and Lehnhausen 1998, Nappi et al. 2003, Saab et al. 2005). However, Goggans et al. (1988) suggested that this species be an indicator species for mature and old-growth lodgepole pine stands in Oregon, and Trembly et al. (2009) suggested the use of black-backed woodpecker as an indicator species in mature and overmature coniferous stands in northeastern North America.

For this species, we considered both a primary and secondary source habitat in all forested potential vegetation types. Primary source habitat was considered any postfire habitat from 1994 to 2003 that had not been salvage harvested. Secondary habitat was defined as forests with >10 in d.b.h. and ≥50 percent canopy closure. In addition, we included forested areas with a high degree of insect outbreak (e.g., bark-beetle-killed forests) over the past 10 years. Areas identified on the insect and disease map with >5.6 snags/acre and not harvested since 1994 were also included as secondary habitat (http://www.fs.fed.us/r6/nr/fid/as/index.shtml). The area within 200 ft surrounding open roads was not considered primary or secondary habitat owing to the likelihood of reduced snag densities that result from firewood cutting and hazard tree felling (Bate et al. 2007, Wisdom and Bate 2008).

We identified primary source habitat as:

- Potential vegetation types: dry, mesic, cold-moist, and cold-dry forests
- Postfire habitat 1994–2003, that had not been salvage harvested in all forested cover types

We identified secondary source habitat as:

- Potential vegetation types: Dry, mesic, cold-moist and cold-dry forests
- Cover types: All forested types
- Tree size: ≥25 in QMD
- Tree canopy closure: ≥50 percent

Snag densities—

Black-backed woodpeckers nest in both live and dead trees but may require heart-rot for nest excavation (Goggans 1989). Nests are usually in a conifer such as pine, spruce, fir, or Douglas-fir (Scott et al. 1977). In Idaho, used nest trees averaged 12.7 in d.b.h. (N = 15; Saab and Dudley 1998). In a study in the Sierra Nevada, California, black-backed woodpeckers favored partially dead trees and hard snags for nesting; used nest trees >16 in d.b.h. and >42.6 ft tall in both burned and unburned forest (Raphael and White 1984). Mean d.b.h. of nest trees reported in this study was 16 in, nest-tree height 92 ft, and nest-cavity height averaged 36 ft. Of 15 nests in northeast Oregon, 9 nests were located in snags and 6 in live trees; most (10) were in ponderosa pine, 4 in lodgepole pine, and 1 in western larch (Bull et al. 1986). They also reported that nest trees averaged 14.5 in d.b.h., 62 ft in height, and 16 ft at nest-cavity height.

This woodpecker forages predominantly on wood-boring beetles, engraver beetles, and mountain pine beetles (Dixon and Saab 2000, Goggans et al. 1988, Harris 1982, Villard and Beninger 1993). In central Oregon, they foraged predominantly on lodgepole pine trees with a mean d.b.h. of 14 in ± 4.7 standard deviation (SD) (range 2 to 39, n = 330); dead trees were used in greater proportion than available, and most were recently dead; 81 percent of forage trees were infested with mountain pine beetle; mean foraging height 16 ft ± 11 SD (range 0 to 59, n = 339; Goggans et al. 1988). In a burned, mixed-conifer forest in northeast Washington, black-backed woodpeckers foraged on dead trees 99 percent of the time (Kreisel and Stein 1999). Woodpeckers as a group (included black-backed woodpeckers) selected trees or snags greater than 17 in d.b.h. to forage on in a foraging study on the Wenatchee National Forest (Lyons et al. 2008).

We assumed that snag densities preferred by the species were available in primary habitat (unsalvaged postfire forest). In secondary habitat, we calculated the percentage of area of source habitat within each watershed that had snag densities (>10 in d.b.h.) in the following classes based on data from Mellen-McLean et al. (2009): low ≤4/ac, moderate 4.0 to 7/ac, high 7 to 18/ac, and very high ≥18 snags/ac (fig. 12). The breaks between classes are based on averaged DECAID (Mellen-McLean et al. 2009) data for ponderosa pine/Douglas-fir, mesic, and montane forests snags >10 in and expert opinion.

Road density—

Snag numbers adjacent to roads are often lower owing to the felling of snags for safety considerations, firewood cutters, and other management activities (Bate et al. 2007, Wisdom and Bate 2008). Other literature has reported the potential for reduced snag abundance along roads (Gaines et al. 2003a, Wisdom et al. 2000). To account for reduced snag density along all roads, we calculated the percentage of secondary source habitat in the following road density classes by watershed (fig. 12):

- Zero: <0.1 mi/mi^2 open roads in a watershed
- Low: 0.1 to 1.0 mi/mi^2 open roads in a watershed
- Moderate: 1.1 to 2.0 mi/mi^2 open roads in a watershed
- High: >2.0 mi/mi^2 open roads in a watershed

Historical inputs for surrogate species assessment model—

- Primary habitat departure: Class 1
- Secondary habitat departure: Class 1
- Snag density: High

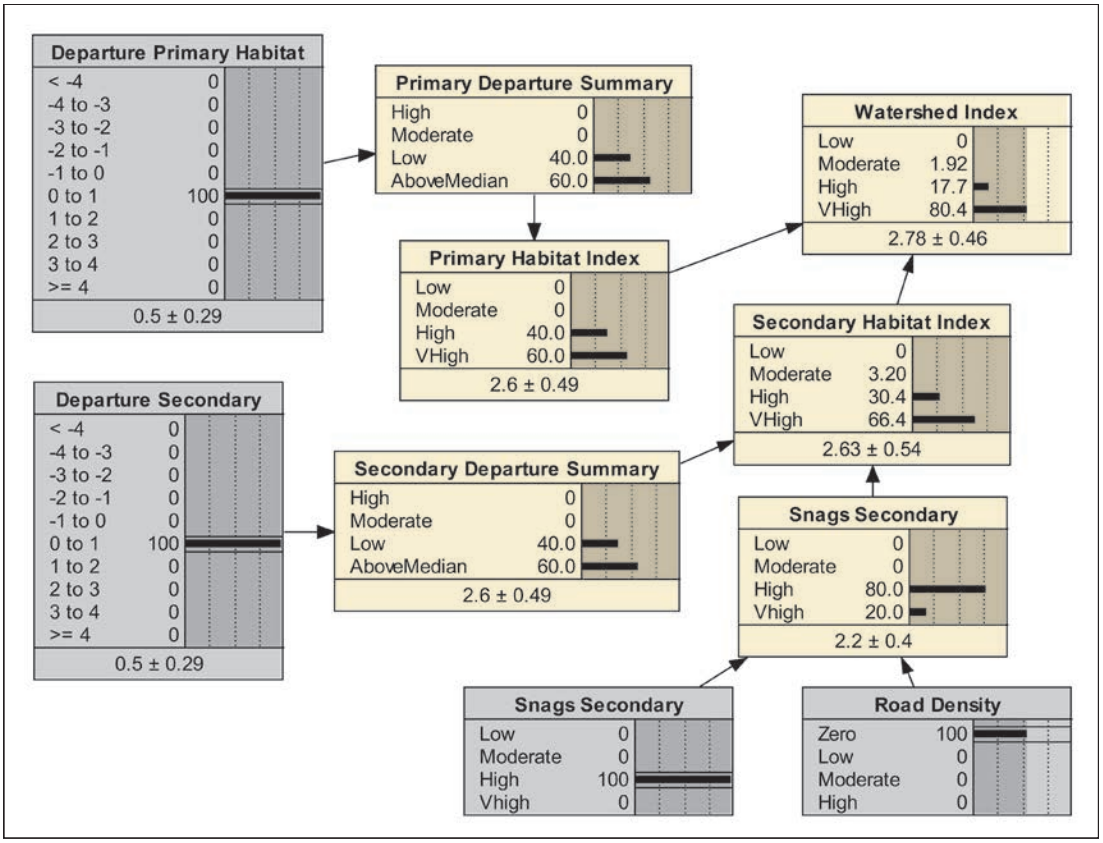

Figure 12—Surrogate species assessment model for black-backed woodpeckers.

The relative sensitivity of the WI values to variables in the model for the black-backed woodpecker are shown in table 22.

Assessment Results

Watershed scores—

Our analysis indicated that primary habitat was below the historical median in most watersheds (n = 61, 85 percent) (fig. 13) indicating a lack of recent stand-replacing wildfires. The remaining 11 watersheds all experienced recent wildfires. The Lower Lake Chelan (32,120 ac) and Upper Chewuch (49,420 ac) watersheds have much higher amounts of primary source habitat than any other watersheds (fig. 13). It is likely that past fire management policies have negatively affected this species by reducing the number of large, high-intensity wildfires that create suitable conditions for the black-backed woodpecker (Dixon and Saab 2000).

Table 22—Relative sensitivity of watershed index values to variables in the model for black-backed woodpecker

Variable	Sensitivity rank
Primary habitat departure	1
Secondary habitat departure	2
Snags secondary	3
Road density	4

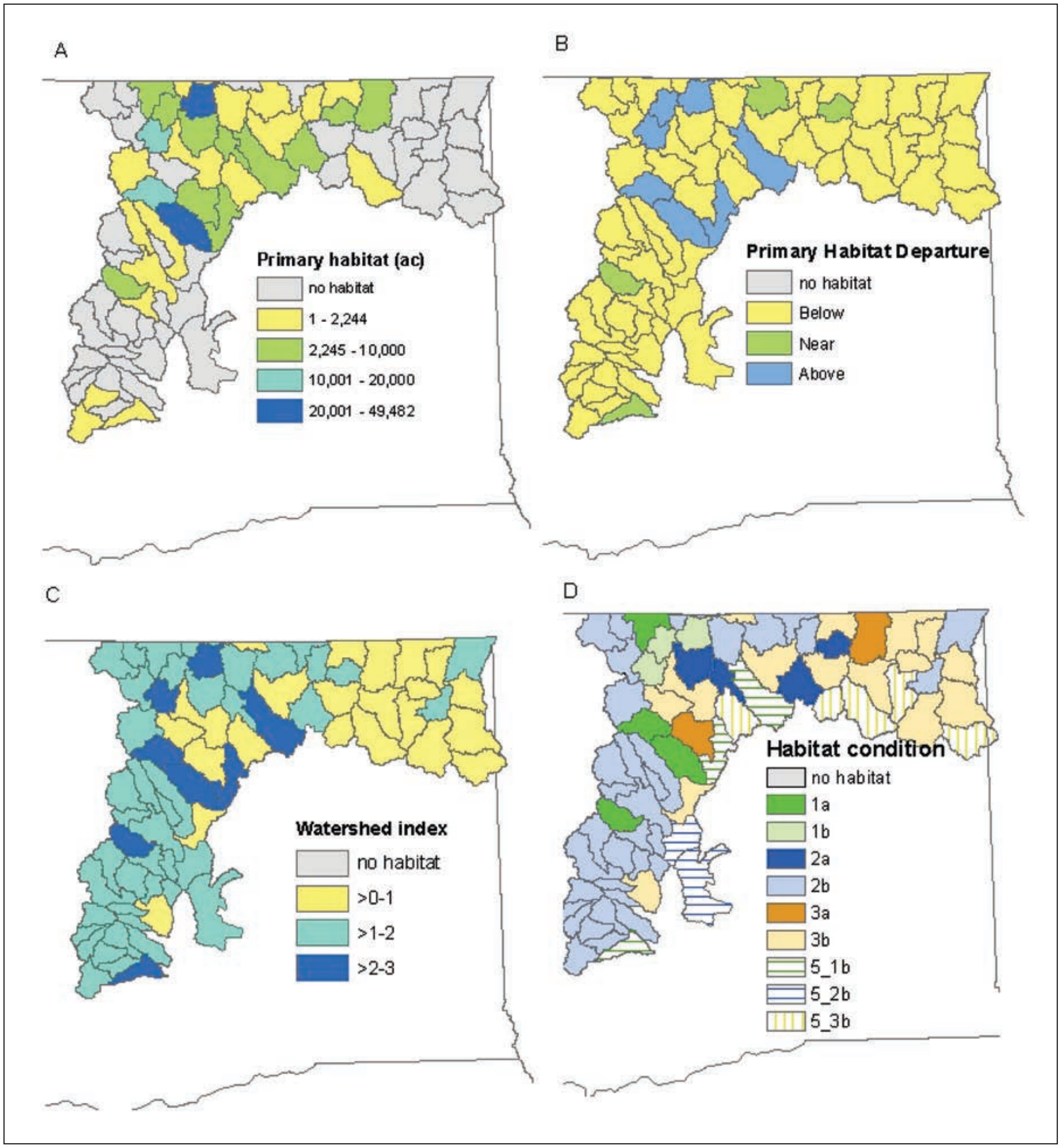

Figure 13—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D), habitat condition class for black-backed woodpeckers by watershed in the northeast Washington assessment area.

The departure of secondary source habitat from historical conditions was more mixed, 44 percent (n = 32) watersheds were below the median range of variation, 38 percent were near the median (n = 27), and 18 percent (n = 13) were above (fig. 13). Declines in secondary source habitat are associated with decreases in the abundance of trees ≥10 in d.b.h. with >50 percent canopy closure. Wisdom et al. (2000), in the evaluation of source habitat for black-backed woodpeckers across the interior Columbia Basin, found similar overall declines in the habitat area of the Colville National Forest. However, they found an overall decline along the Cascade Range, where we found departure to be closer to the historical reference conditions. This difference likely can be explained by differences in definition of source habitats, except in the lodgepole pine types. Wisdom et al. (2000) defined source habitat in larger tree structural stages, while in this analysis, we included forests with >10 in d.b.h.

Overall, snag densities in secondary source habitat were in the low class (≤4/ac): 57 percent (n = 41) of the watersheds had greater than half their source habitat in the low class. Seven watersheds had more than 70 percent of their source habitat in the low class: Columbia tributaries, upper Columbia–Swamp Creek, Middle Sanpoil, lower Okanogan River–Omak Creek, Columbia River-Lynch Coulee, and Cowiche. No watersheds had a majority of secondary habitat in the medium, high, or very high snag classes.

Overall road densities were not high in secondary source habitat, though 15 percent (n = 11) of watersheds showed greater than half the habitat area in high road density, with two watersheds having >70 percent of habitat in that watershed in a high class (Wenas Creek and Middle Yakima River). Twenty-nine percent (n = 21) of the watersheds had >50 percent of source habitat in the zero road density class.

Because primary postfire habitat is well below the historical median in most watersheds, the amount of secondary habitat is likely playing an important role. In 31 percent (n = 22) of the watersheds where both primary and secondary habitat were below the historical median, the WI value currently was <1 (fig. 13). Watersheds with an index >2.0, are thought to be less departed in the amount of habitat, and contain good quality habitat. Ten watersheds (14 percent) in the study area meet these criteria with a WI value ≥2.0.

Viability outcome scores—
The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for black-backed woodpeckers within the assessment area is 81 percent of the historical capability. Currently, four of five ecoregions contain at least one watershed

Fire suppression efforts likely reduced the amount and distribution of primary habitat for this species, and likely other species in the postfire group, leading to reduced viability.

with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Fifteen of seventy-two (21 percent) watersheds had >40 percent of the historical median amount of source habitat.

Currently, the viability outcome falls primarily within outcomes C (40.5 percent), D (25.5 percent), and B (24.5 percent), which indicates that suitable environments are frequently patchily distributed, and source habitat is in low abundance (fig. 14). Historically, the outcome was primarily an A (80.8 percent) indicating that suitable environments were once abundant, broadly distributed, and better connected.

Historically, all five ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Sixty-three watersheds (88 percent) had >40 percent of the median amount of historical source habitat.

The main factor leading to a lower viability outcome from historical was the decreased percentage of watersheds with recent wildfire activity (primary source habitat). Historically, 88 percent (n = 63) of the watersheds contained >40 percent of the historical median amount of primary habitat while primary habitat currently occurs in this quantity in only 21 percent (n = 15) of the watersheds. Fire suppression efforts likely reduced the amount and distribution of primary habitat for this species, and likely other species in the postfire group, leading to reduced viability.

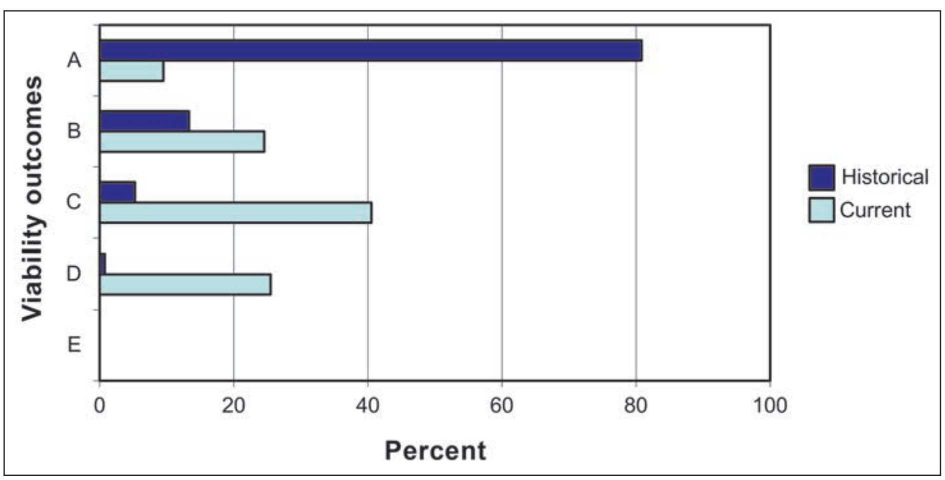

Figure 14—Current and historical viability outcomes for the black-backed woodpecker in the northeast Washington assessment area.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Low abundance of recent unsalvaged postfire habitat throughout the assessment area.
2. Decline in secondary source habitat in some areas.
3. Snag densities in secondary habitat were primarily low.

Canada Lynx

Introduction

The Canada lynx was selected as a surrogate species to represent the boreal forest group owing to its close association with boreal forests (Aubry et al. 2000, ILBT 2013, Koehler and Aubry 1994, Maletzke 2004, von Keinast 2003) and because human disturbance was identified as a risk factor for lynx (Buskirk et al. 2000, ILBT 2013, Koehler and Aubry 1994, Koehler et al. 2008, Ruediger et al. 2000). The distribution of Canada lynx within the assessment area has been stratified into core, secondary, and peripheral habitat areas based on known records of their occurrences (ILBT 2013, USFWS 2005). Core areas occur on the Okanogan-Wenatchee National Forest, on the Methow Valley and Tonasket Ranger Districts, and in the Kettle Mountains on the Colville National Forest. The remainder of the Colville National Forest and the portion of the Okanogan-Wenatchee National Forest between Highway 2 and Lake Chelan are secondary, and the remainder of lynx habitat is peripheral.

Model Description

Source habitat—

The southernmost extent of the boreal forest supports Canada lynx and overlaps with the northeastern Washington assessment area (McKelvey et al. 2000). In the contiguous United States, these boreal forests transition into other vegetation communities and become more patchily distributed. In North America, the distribution of lynx is nearly coincident with that of snowshoe hares (Bittner and Rongstad 1982, McCord and Cardoza 1982). Lynx survivorship, productivity, and population dynamics are closely related to snowshoe hare density in all parts of its range (USFWS 2005). Quality habitat for lynx occurs where understory stem densities and other forest structures provide forage and cover needs of snowshoe hares (Agee 2000, Hodges 2000, Koehler 1990). Good snowshoe hare habitat has a common denominator—dense, horizontal vegetative cover 3 to 10 ft above the ground or snow level (Hodges 2000). These characteristics include a dense, multilayered understory that maximizes cover and browse at both ground level and at varying snow depths throughout the winter. Such habitat structure is common in early-seral stages but may also occur in coniferous forests with mature but relatively open overstories (Hodges 2000).

Primary vegetation that contributes to lynx habitat is lodgepole pine, subalpine fir, and Engelmann spruce (Aubry et al. 2000). Habitat selection by Canada lynx has been studied on the Okanogan portion of the assessment area. Lynx selected for Engelmann spruce and subalpine fir forest, moderate canopy cover, flat to moderate slopes, and relatively high elevations; and selected against Douglas-fir and ponderosa pine forests, forest openings, recent burns, sparse canopy and understory, and relatively steep slopes (Koehler et al. 2008, Maletzke et al. 2008). Probability of use by lynx was 19.4 times greater in spruce and subalpine fir forests than other vegetation types, 4.9 times greater in areas with moderate canopy cover than for other cover classes, 5.0 times greater at elevations ranging from 5,000 to 6,000 ft than other elevations, and 48.8 times greater on flat to moderate slopes than on steep slopes (Koehler et al. 2008).

An important component of lynx habitat is areas that are used for denning (Moen et al. 2008, Ruediger et al. 2000). The common component of natal den sites appears to be large woody debris, either down logs or root wads (Koehler 1990, Mowat et al. 2000, Slough 1999, Squires and Laurion 2000, Squires et al. 2008); these structures are often associated with late-successional forests. These den sites may be located within older regenerating stands (>20 years since disturbance) or in mature conifer or mixed-conifer-deciduous (typically spruce/fir or spruce/birch) forests (Koehler 1990, Slough 1999). Stand structure appears to be of more importance than forest cover type (Mowat et al. 2000).

To estimate these elements of lynx habitat for this assessment, we mapped both early- and late-successional forests within the subalpine fir vegetation zone. Source habitat was identified using the following GIS data layers:

- Cover types within the subalpine fir vegetation zone: Engelmann spruce, lodgepole pine, Pacific silver fir, and subalpine fir
- Tree size and canopy closure: 1 to 10 in d.b.h. QMD and >50 percent canopy closure for early-successional forests, >15 in d.b.h. QMD and >50 percent canopy closure for late-successional forests

Grazing—

Grazing can reduce the density of shrubs that create foraging habitat for snowshoe hares (Ruediger et al. 2000), which are the primary prey of the Canada lynx (Aubry et al. 2000, Koehler and Aubry 1994). We categorized the amount of snowshoe hare foraging habitat in an active grazing allotment using 10 percent increments from 0 to 100 percent, with increasing poorer habitat outcomes as the proportion of foraging habitat in an active allotment increased (fig. 15).

Down wood—

Down wood is an important component of Canada lynx denning habitat (Koehler 1990, Mowat et al. 2000, Slough 1999, Squires and Laurion 2000, Squires et al. 2008). To assess the availability of down wood, we intersected lynx source habitat with the down wood estimates available from Ohmann and Gregory (2002) using the gradient nearest neighbor approach. We categorized the availability of down wood within source habitat as follows (fig. 15):

- Low: 10 m^3 (354 ft^3) of down wood >20 in
- Moderate: 10 to 25 m^3 (354 to 882 ft^3) of down wood >20 in
- High: >25 m^3 (882 ft^3) of down wood >20 in

Winter recreation route density—

Several researchers have expressed concerns over the effects of winter recreational activities on Canada lynx (Bunnell et al. 2006, Buskirk et al. 2000, Koehler and Aubry 1994, Kolbe et al. 2007). Specifically, snow compaction associated with grooming for snowmobiling and cross-country skiing may provide travel routes for

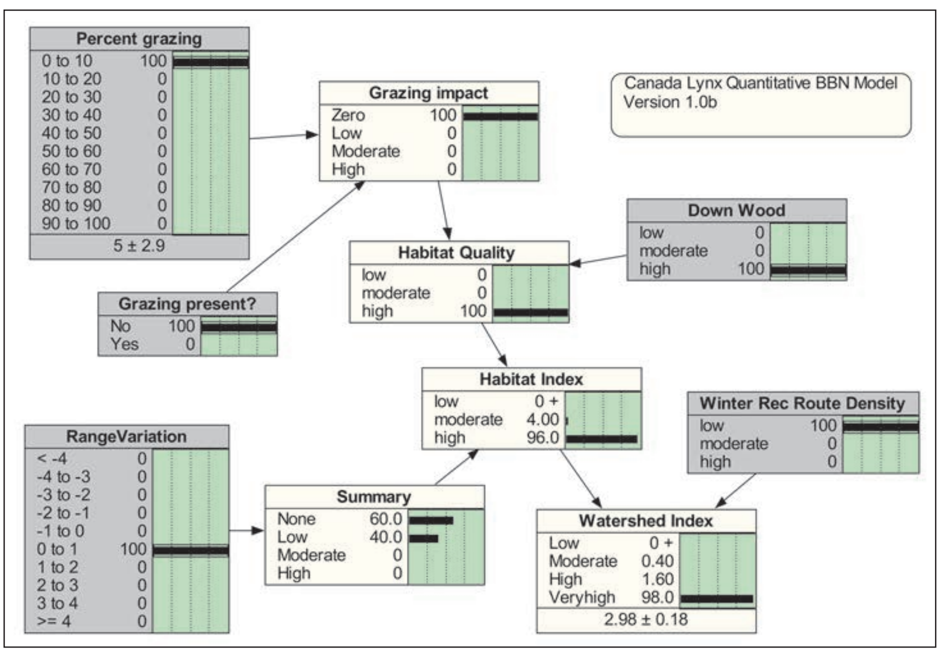

Figure 15—Surrogate species assessment model for the Canada lynx.

competitors and predators such as coyotes, bobcats, and mountain lions (Bunnell et al. 2006, Buskirk et al. 2000, Koehler and Aubry 1994). We assessed the influence of groomed snow routes on Canada lynx source habitat using existing information on the location of groomed and designated routes and using the index described in Gaines et al. (2003a). Other groomed routes and snow play areas were known to exist but were not available in a digital format. Thus our assessment of the influences of winter routes on Canada lynx source habitat likely underestimated the true impacts in many of the watersheds. Using the digital data we had on snowmobile trails, we used a moving windows analysis with a 0.9-km (2,953-ft) radius circular window (based on Gaines et al. 2003a), and categorized these effects as follows (fig. 15):

- Low influence on Canada lynx source habitat ≤25 percent of source habitat with winter route densities <1 mi/mi^2
- Moderate influence on Canada lynx source habitat ≥25 percent of source habitat with winter route densities 1 to 2 mi/mi^2
- High influence on Canada lynx source habitat ≥25 percent of source habitat with winter route densities >2 mi/mi^2

Calculation of historical conditions—

- Source habitat: Departure class 1
- Grazing: 0 percent
- Down wood: Moderate
- Winter recreation route density: Low

The relative sensitivity of watershed index values to variables in the model of Canada lynx are shown in table 23.

Table 23—Relative sensitivity of watershed index values to variables in the model for the Canada lynx

Variable	Sensitivity rank
Habitat departure	1
Down wood	2
Grazing	3
Winter recreation	4

Assessment Results

Watershed scores—

Sixty-two percent (n = 44) of the watersheds had high WI scores (>2.0), 30 percent (n = 21) had moderate WI scores (1.0 to 2.0), and 8 percent (n = 6) had low (<1.0) WI scores (fig. 16). Departure of source habitat from the historical median across all watersheds was the variable that had the greatest influence on the outcomes of the WI score (fig. 16). The majority (49 percent, n = 35) were well above their historical median levels of source habitat (i.e., >2 class), 28 percent (n = 20) were at or near historical levels, and 23 percent were well below their historical median levels of source habitat (i.e., < -1 class).

Watersheds that currently had the greatest amount of source habitat (>10,000 ac) are the Sinlahekin Creek and Pasayten River. Additionally, the following watersheds have 3,000 to 10,000 ac: Mad River, Lost River, Naneum, Lower Lake Chelan, Upper Chewuch, Twisp River, Entiat, Middle Methow, Lower Chewuch (fig. 16).

Wisdom et al. (2000) reported similar results for the source habitats in the North Cascades and Northern Glaciated Mountains ERUs. In the Northern Glaciated Mountains, a strong increase in mid-seral montane forests, along with increases in early- and mid-seral subalpine forests accounted for an overall increase in source habitat trend. In the North Cascades, increases in early-seral montane and subalpine forests were offset by decreases in mid- and late-seral subalpine forests.

Fires are a significant disturbance process in boreal forests of North America, and large areas burned throughout Washington during the 19^{th} and 20^{th} centuries (Agee 2000). Our assessment accounted for several of these fires; however, the 148,300-ac Tripod Fire was not accounted for in our vegetation data and burned much of the Meadows area in 2006, which had been considered the best and most extensive lynx habitat in Washington (Koehler et al. 2008, Stinson 2001).

Twenty-one percent (n = 21) of the watersheds had source habitat with a down wood rating of low (<10 m^3/ha [350 ft^3/ac] >20 in diameter), 59 percent (n = 42) had moderate levels (10 to 25 m^3/ha [883 ft^3/ac] >20 in diameter), and 20 percent (n = 14) had high levels of down wood (>25 m^3/ha [883 ft^3/ac] >20 in diameter).

Based on information that was available on known winter route locations, the influence of winter routes on Canada lynx source habitat was rated as low (<25 percent of source habitat in a watershed with winter route densities <1 mi/mi^2) in all watersheds. As we discussed earlier, our assessment of the influences of winter routes on Canada lynx source habitat likely underestimated the true impacts in many of the watersheds owing to data availability.

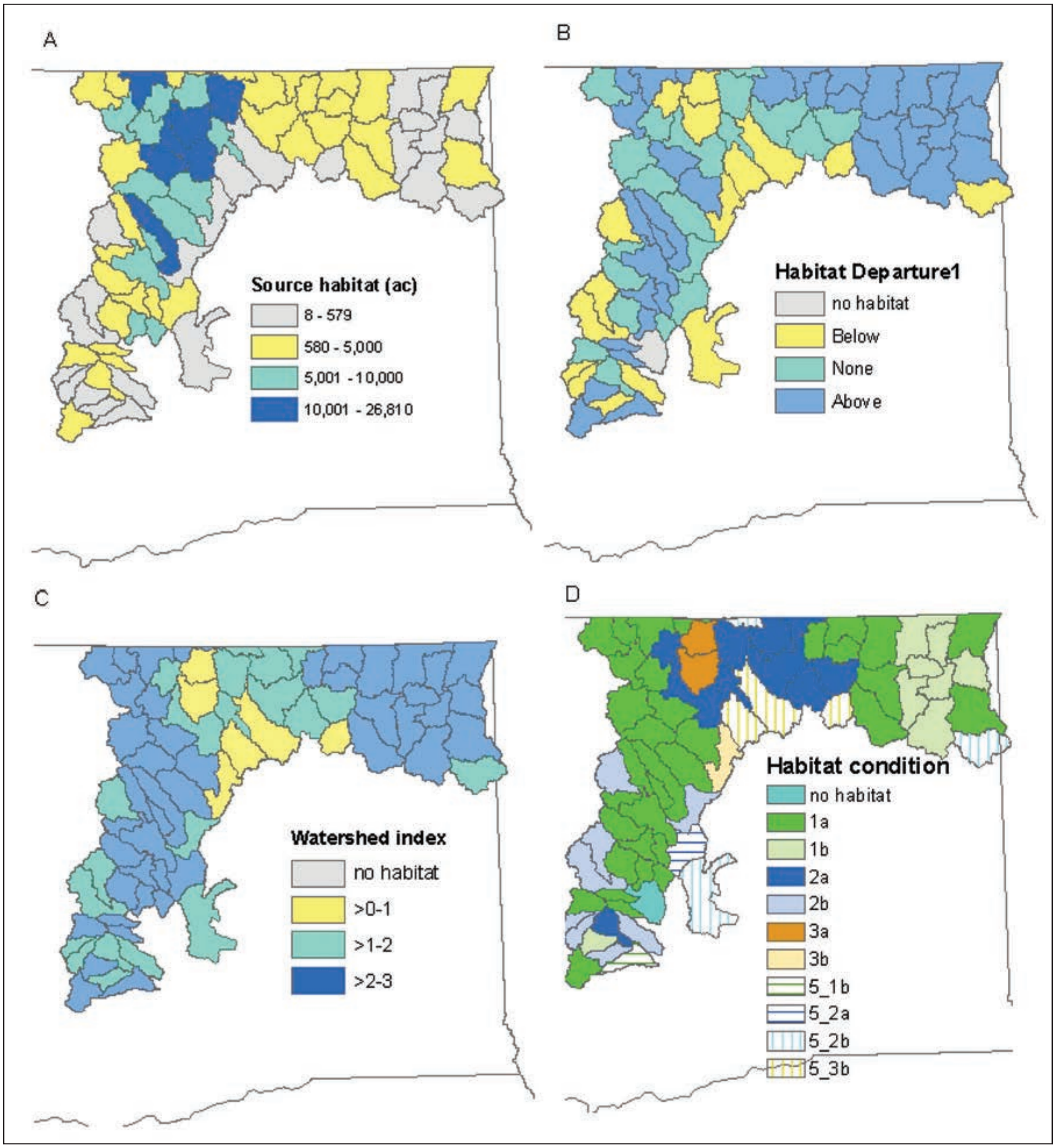

Figure 16—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D) habitat condition class for the Canada lynx by watershed in the northeast Washington assessment area.

We found that most source habitat for Canada lynx was in an active livestock grazing allotment. Forty-two percent (n = 30) of the watersheds had <10 percent of the source habitat in an active grazing allotment, 11 percent (n = 8) of the watersheds had 10 to 50 percent, and 47 percent (n = 33) had >50 percent of the source habitat in an active allotment.

Viability outcome scores—
The viability outcome scores (VOI) model incorporated the WWI scores (described earlier), a habitat distribution index, and a habitat connectivity or permeability index (see pages 29–32 in chapter 1). The WWI scores indicated that the current habitat capability for Canada lynx was 75 percent of its historical capability. Currently, 68 percent (n = 48) of the watersheds had source habitat amounts >40 percent of the historical median. The watersheds with >40 percent were distributed across all of the five ecoregions.

Currently, 7 percent of the assessment area was rated as low permeability, 60 percent rated as moderate, and 33 percent rated as high. These results are similar to the dispersal habitat suitability reported by Singleton et al. (2002) for lynx in the same general area. Singleton et al. (2002) identified "fracture zones," or sizeable gaps in dispersal habitat (usually because of low elevations and human development) that occur within the assessment area and warrant careful management attention. These include the upper Columbia-Pend Oreille, southern Okanogan, Stevens Pass-Lake Chelan, and Okanogan Valley.

Currently, there is a 64.5 percent probability that the current VOI for the assessment area was B and 25.5 percent probability of outcome C, which indicates that suitable environments for the Canada lynx are broadly distributed and of high abundance, but there are gaps where suitable environments were absent or only present in low abundance (fig. 17). However, the disjunct areas of suitable environments are typically large enough and close enough to each other to permit dispersal among subpopulations and to allow the species to potentially interact as a metapopulation. Again, exceptions to this may occur along the identified fracture zones.

Historically, dispersal across the assessment area was assumed to be high. All ecoregions and 66 percent (n = 47) of the watersheds contained greater than 40 percent of the median amount of habitat historically. Historically, there was a 71.2 percent probability that the viability outcome for Canada lynx was A and a 18.8 percent probability of a B outcome. This indicates that suitable environments were of high abundance and were better connected, allowing for interspecific interactions (fig. 17). We estimated that a reduction in the availability of suitable environments for the Canada lynx likely occurred in the assessment area compared to the

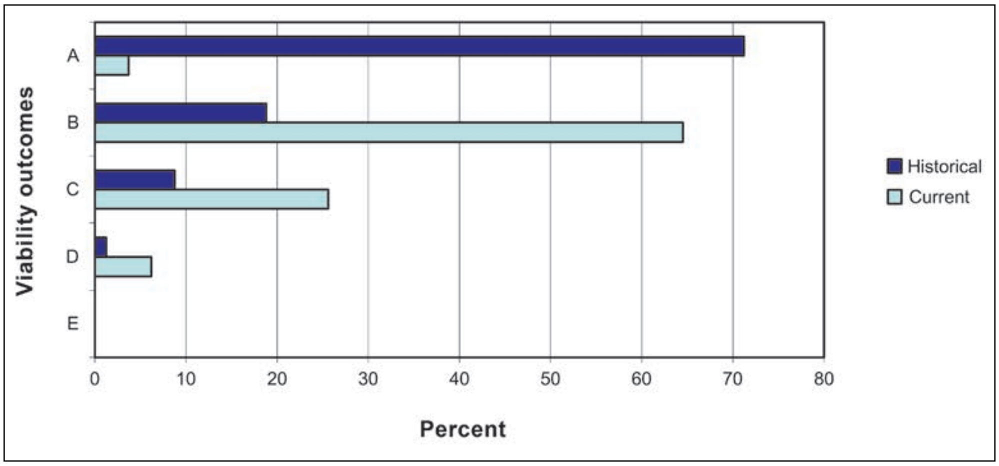

Figure 17—Current and historical viability outcomes for the Canada lynx in the northeast Washington assessment area.

Returning disturbance regimes toward the NRV measured at the landscape scale would provide Canada lynx habitat components that are distributed across the landscape in a sustainable fashion.

historical distribution and condition of their habitats. In summary, under historical conditions, there was a high probability that viable populations of Canada lynx and all other species associated with the boreal forest group were well distributed.

Management Considerations

The following issues were identified during this assessment and from the published literature for considerations by managers:

1. Returning disturbance regimes toward the NRV measured at the landscape scale would provide Canada lynx habitat components that are distributed across the landscape in a sustainable fashion (Agee 2000, Wisdom et al. 2000).
2. Additional information is needed on the location and extent of snowmobile routes and snow play areas that we were unable to evaluate owing to lack of data. Additional data would help to fully evaluate the effects of these activities on Canada lynx habitat.
3. Fracture zones, or sizeable gaps in dispersal habitat (usually as a result of low elevations and human development), occur within the assessment area and warrant careful management attention. These include the Upper Columbia-Pend Oreille, Southern Okanogan, Stevens Pass-Lake Chelan, and Okanogan Valley.

Cassin's Finch

Introduction

Cassin's finch is a surrogate species for medium and larger tree forests in the all forest communities group. This finch was chosen as a surrogate species primarily to represent the risk of grazing that other species in this group share with Cassin's finch. In addition, in contrast to pileated woodpecker and American marten, also surrogate species for larger trees, this species is primarily associated with open-canopied forests. Source habitats for this species overlap with species in the dry forest group as well. This species is distributed year-round across the assessment area and occurs in all the forested communities.

Model Description

Source habitat—

Cassin's finches breed primarily in open, mature coniferous forests of lodgepole and ponderosa pine, aspen, supalpine fir, grand fir, and juniper woodlands (Gaines et al. 2007, 2010a; Gashwiler 1977; Hahn 1996; Huff and Brown 1998; Lehmkuhl et al. 2007; Schwab et al. 2006; Sullivan et al. 1986). In the Blue Mountains, these finches were negatively associated with habitat variables representing increasing crown cover and down woody debris, and were positively associated with canopy height (Sallabanks 1995). Gaines et al. (2007) reported that Cassin's finch abundance was positively influenced by thinning and burning restoration treatments within dry forests that retained large trees but reduced canopy closure.

On both the Fremont and Winema National Forests, these finches were more abundant in salvage-logged stands where dead and down lodgepole pine were removed than in unharvested control stands (Arnett et al. 1997). This research also found that the probability of presence of Cassin's finches was negatively associated with the number of live and dead trees, number of live trees <32.8 ft tall, percentage of seedling cover, percentage of shrub and grass forb cover, foliage area of live trees, and percentage of canopy cover. The probability of presence of Cassin's finches was positively associated with number of trees >11.8 in d.b.h. and the amount of ground debris (Arnett et al. 1997). The presence of Cassin's finches was negatively associated with understory vegetation (Hutto 1995a). The more open structure was preferred for nesting and allowed them to forage on the ground (Bettinger 2003).

Hutto (1995a) found Cassin's finches abundant 1 year postfire in the Rocky Mountains, although their numbers dropped off in the second year following fire.

This species occupies burned forests as well, though this is usually restricted to 1 year postfire (Hutto 1995b, Smucker et al. 2005), suggesting that this species may be responding to short-term increases in the availability of seeds after wildfire (Jewett et al. 1953, Hutto 1995b, Kotliar et al. 2002, Saab and Dudley 1998, Sallabanks 1995, Smucker et al. 2005).

We identified source habitat as:

- Potential vegetation types: Dry, mesic, cold-moist and cold-dry forests
- Cover types: All forested types
- Tree size: ≥15 in QMD
- Canopy closure: ≤70 percent

Grazing—
Saab et al. (1995) summarized the results of five studies that evaluated the effects of livestock grazing on Cassin's finch. Three of the five studies found that Cassin's finches responded negatively to grazing (Page et al. 1978, Schulz and Leininger 1991, Taylor 1986), one found a neutral effect (Medin and Clary 1991), and one found a positive relationship (Mosconi and Hutto 1982). The amount of source habitat in an active grazing allotment was categorized using 10 percent increments from 0 to 100 percent, with increasing poorer habitat outcomes as the proportion of source habitat in an active allotment increased (fig. 18). We calibrated the overall negative effect of this risk factor to be relatively small owing to the mixed research results.

Historical inputs for focal-species assessment model—

- Departure of source habitat from departure class: Class 1
- Grazing: 0 percent

The relative sensitivity of watershed index values to model variables for the Cassin's finch is shown in table 24.

Assessment Results

Watershed scores—
Habitat for Cassin's finch was estimated to be generally well below the historical median amount of source habitat in nearly all watersheds (n = 68, 94 percent) (fig. 19). The four watersheds that had close to the historical median are the Lower Methow River, Ross Lake, Ruby Creek, and Stehekin. These watersheds also had the largest amount of current source habitat, as did the Middle Methow River, which is departed somewhat more from the historical median (fig. 19).

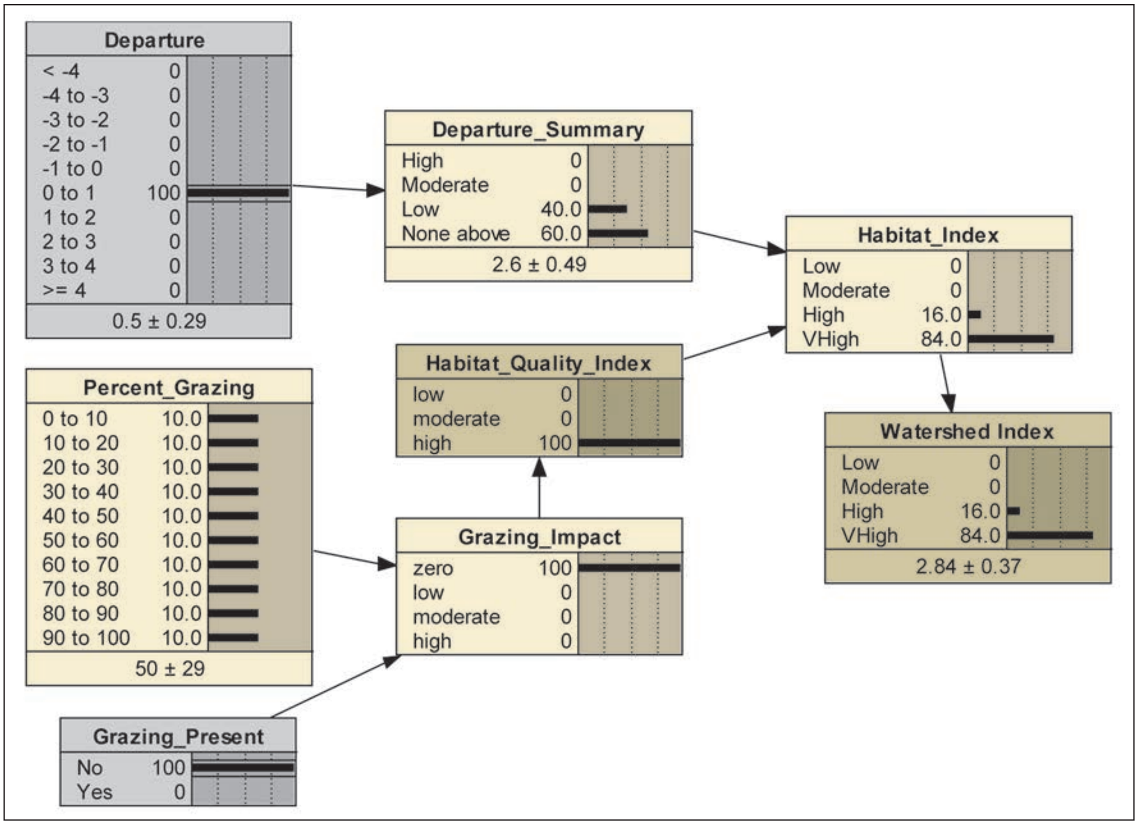

Figure 18—Surrogate species assessment model for the Cassin's finch.

Table 24—Relative sensitivity of watershed index values to variables in the model for Cassin's finch

Variable	Sensitivity rank
Habitat departure	1
Grazing	2

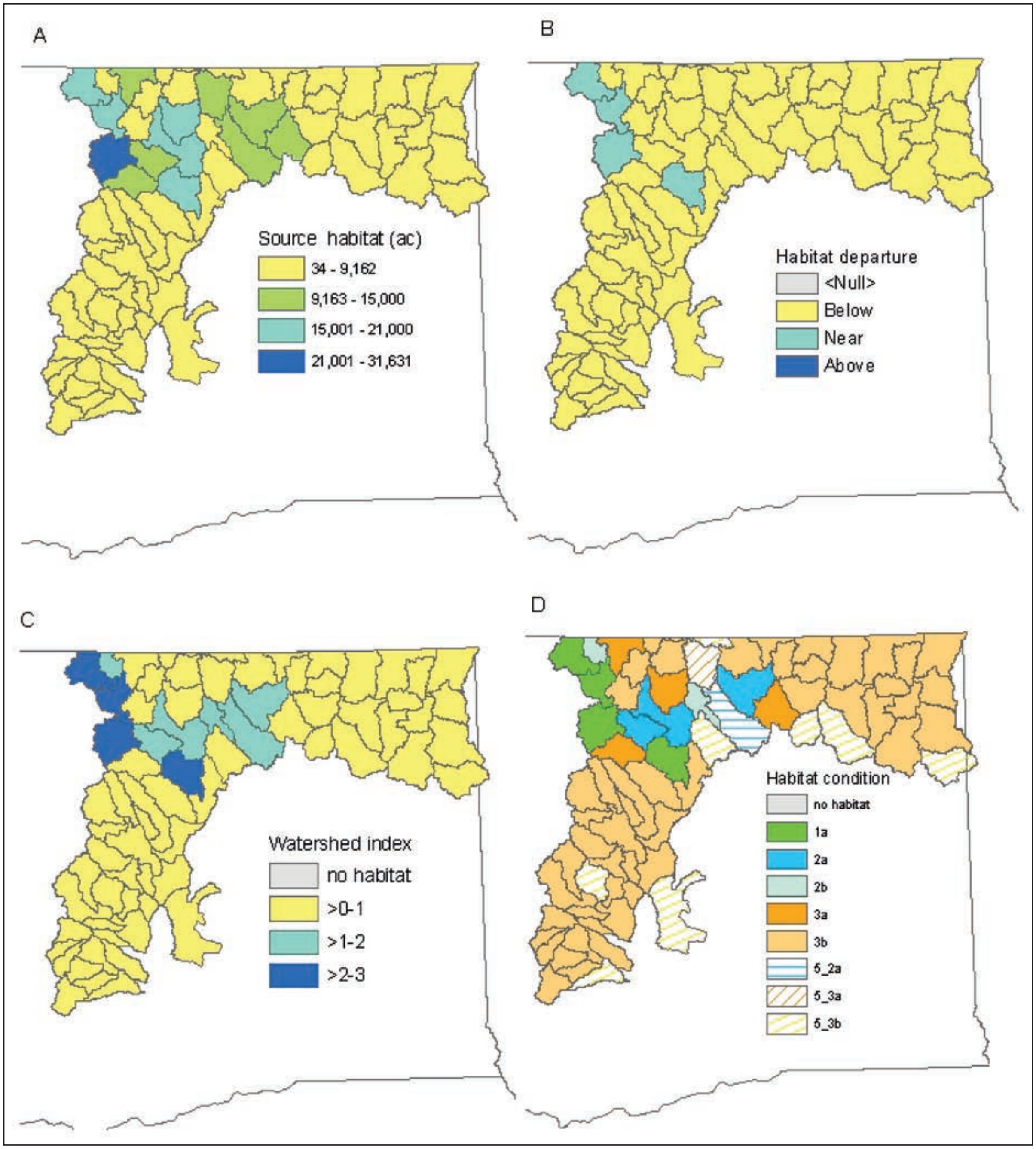

Figure 19—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D) habitat condition class for the Cassin's finch by watershed in the northeast Washington assessment area.

The area of source habitat in an active grazing allotment was mixed across all watersheds: 19 percent (n = 14) were not grazed, 43 percent (n = 32) had 1 to 50 percent of their source habitat in an active grazing allotment, and 30 percent (n = 26) had >50 percent of their source habitat in an active grazing allotment.

Owing to the extensive departure in the amount of source habitat from the historical amount in nearly all the watersheds, the WI values were fairly low. Eighty-six percent of the WI values were low (<1) (n = 62), 8 percent (n = 6) were moderate (≥1 and <2), while 6 percent (n = 4) were high (≥2) (fig. 19). The four watersheds with the highest WI values were those listed above that have departed the least from the historical amounts of habitat. The six watersheds that had a medium score (1-2) were Lightning Creek, Twisp River, Okanogan River-Bonaparte Creek, Middle Methow Creek, Salmon Creek, and Okanogan River-Omak Creek.

Viability outcome scores—
The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. Comparison of the average current WWI for the Cassin's finch to the average historical WWI showed that current conditions are 44 percent of the historical capability. Historically, 89 percent (n = 63) of the watersheds contained 40 percent of the historical median amount of habitat (7,927 ac), while currently 18 percent (n = 13) had at least that amount. The watersheds with >40 percent were distributed across two of the five ecoregions.

The current viability outcomes for the assessment area was C (40 percent) and D (60 percent), indicating that suitable environments for the Cassin's finches are frequently isolated or exist at low abundance (fig. 20). It is likely that historical conditions would have been characterized as an A outcome (80.8 percent) where habitats were broadly distributed and more abundant (fig. 20) than currently. Likely, other species in the medium and larger tree forests in the all forest communities group may have experienced similar declines in suitable environments and viability.

Decline in the amount of medium-large (>16 in) tree, open-canopy forests as source habitat for Cassin's finches across the assessment area.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Decline in the amount of medium-large (>16 in) tree, open-canopy forests as source habitat for Cassin's finches across the assessment area.
2. Nonnative ungulate grazing within the majority of the watersheds within the assessment area.

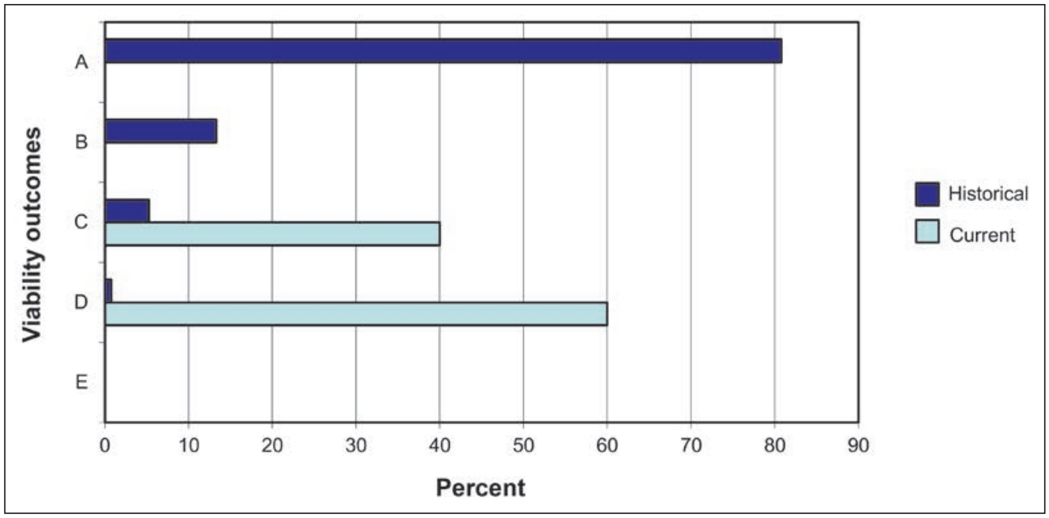

Figure 20—Current and historical viability outcomes for the Cassin's finch in the northeast Washington assessment area.

Columbia Spotted Frog

Introduction

Columbia spotted frog populations have declined precipitously across their range (e.g., they have been found at only 13 of 59 locations where they were present historically in Washington state (McAllister and Leonard 1997). Hayes (1997) suggested that Columbia spotted frogs occupied about 10 percent of its original range. Small population size and reproductive characteristics likely make Columbia spotted frog populations vulnerable to anthropogenic disturbance. The vulnerability of Columbia spotted frog populations to residential development at both local and regional scales may explain some of the declines seen in this species (Goldberg and Waits 2009, Reaser and Pilliod 2005). As a result, Columbia spotted frogs have been designated as a sensitive species by the USDA Forest Service, Pacific Northwest Region. They are relatively easy to survey and were found across the northeast Washington state assessment area except for high elevations along the western edge (Dvornich et al. 1997a). As a surrogate species, they represent species associated with the ponds/small lake/backwater group within the riparian family. Their source habitat and risk factors cover the other species within this group well where populations overlap. A variety of threats to the persistence of populations of Columbia spotted frogs have been identified, including wetland loss, introduced predators, mining, grazing, development, and diseases (Pearl et al. 2007a, Monello and Wright 1999, Reaser and Pilliod 2005, USDI Fish and Wildlife Service 1997a). Columbia spotted frogs are year-round residents of the assessment area (Reaser and Pilliod 2005); this assessment was for breeding and rearing habitat.

Model Description

Source habitat—

Columbia spotted frogs are highly dependent on aquatic habitats and require permanent and semipermanent wetlands that have aquatic vegetation and some deeper or flowing water for overwintering (Bull and Marx 2002, Pilliod et al. 2002). Breeding habitat for Columbia spotted frogs has been characterized, in general, as small silt or muck bottom ponds with emergent vegetation (Morris and Tanner 1969, Pearl et al. 2007b, Pilliod et al. 2002, Welch and MacMahon 2005). Wintering habitat was described as large (about 5 ac), deep (>10 ft) ponds and lakes (Bull and Hayes 2002, Pilliod et al. 2002). Munger et al. (1998) more specifically characterized the habitat associations of adult spotted frogs as still waters with associated shrublands and riverine conditions. They identified these areas as having National Wetland Inventory (NWI) classifications (Cowardin et al. 1979) associated with scrub-shrub and seasonally flooded wetlands. Presence of spotted frogs was negatively associated with areas classified with emergent vegetation and temporarily flooded. Specifically, adult spotted frogs were found more often than expected in palustrine, scrub-shrub, seasonally flooded (PSSC) wetlands and intermittent riverine, streambed, seasonally flooded (R4SBC) wetlands and less often than expected in palustrine, emergent, seasonally flooded (PEMC) wetlands and intermittent riverine, streambed, temporarily flooded (R4SBA) areas. Bull and Hayes (2001) also found adult Columbia spotted frogs associated with riverine habitats in the summer (<40 in deep, cobble substrate, without aquatic vegetation).

For this analysis, source habitat was considered to be PSSC and R4SBC, as described in the NWI (Cowardin et al. 1979) (fig. 21).

The current habitat departure for Columbia spotted frogs was set at -2 for all watersheds (see p. 19).

Invasive animals—

Introduced fish have been linked to the decline of ranid frog species in general across western North America (Hayes and Jennings 1986) and specifically to declines of Columbia spotted frogs (Monello and Wright 1999, Reaser 2000). The negative effects of fish introduced into previously fishless ponds and lakes were considerable for amphibians that required permanent water bodies for reproduction and overwintering (Knapp et al. 2001, 2003, 2005). These negative effects also extended to stream habitats with introduced salmonids (Bosch et al. 2006). Previously fishless lakes with introduced trout (*Oncorhynchus* spp.) populations had lower abundance and recruitment of spotted frogs than fishless lakes (Hirner and Cox 2007, Pilliod and Peterson 2001). However, Bull and Marx (2002) did not find

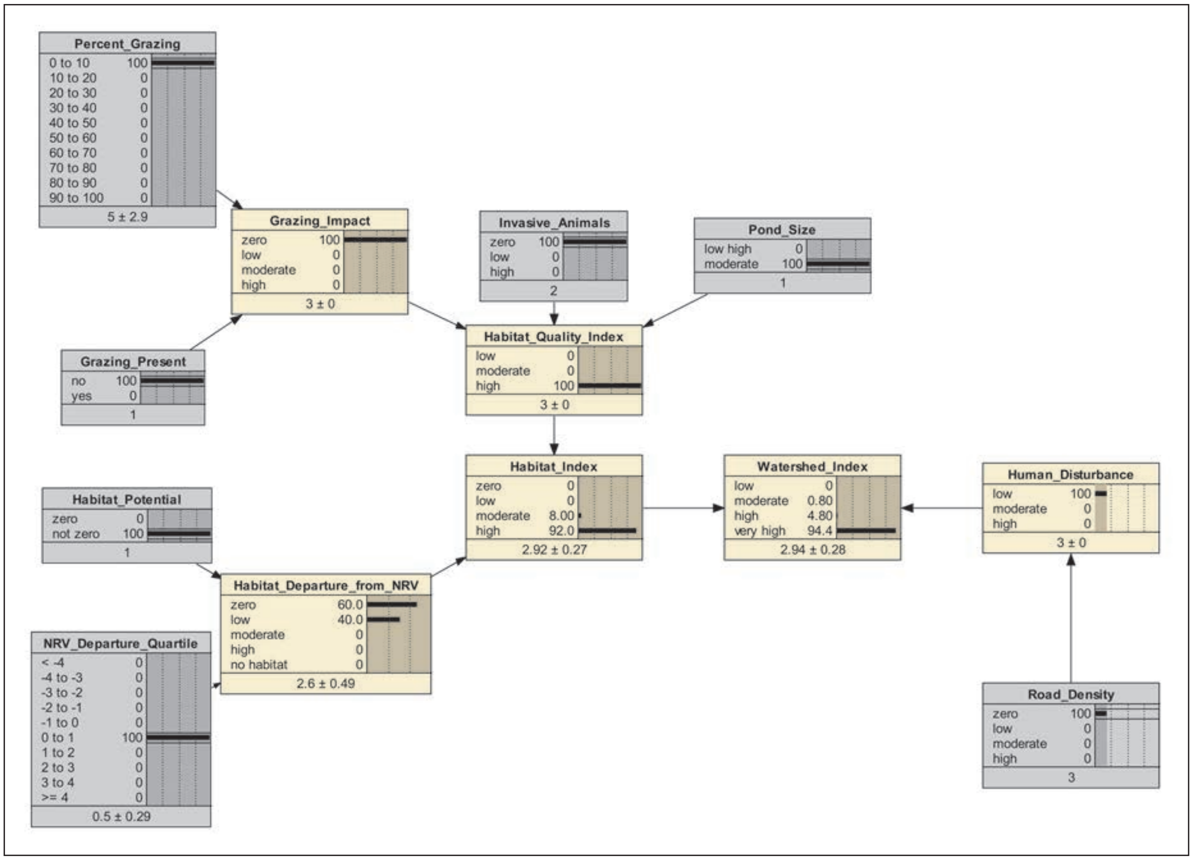

Figure 21—Surrogate species assessment model for Columbia spotted frog.

a strong relationship between the presence of introduced trout and the abundance of eggs and larvae of Columbia spotted frogs.

The following classes were used to evaluate the effect of introduced trout on Columbia spotted frogs (fig. 21):

- High: Introduced trout present in ≥50 percent of source habitat within a watershed
- Low: Introduced trout present in <50 percent of source habitat within a watershed
- Zero: Introduced trout not present in source habitat within a watershed

Grazing—

The results reported in the literature on the effects of grazing on Columbia spotted frogs were equivocal. Reaser (2000) found that cattle grazing was related to low recruitment and high mortality. These findings were supported by other studies (Cuellar 1994, Ross et al. 1999, Worthing 1993). Conversely, others (Adams et al.

2009, Bull and Hayes 2000) reported no differences in productivity of spotted frogs on grazed vs. ungrazed sites in northeast Oregon. However, there was an indication that grazed sites in northeast Oregon had reduced food abundance (Bull 2003a, Whitaker et al. 1983). Also, overgrazing could negatively affect reproduction if egg masses or recently metamorphosed frogs were directly trampled or if banks were collapsed along ponds or rivers that serve as overwintering sites (Bull 2005).

The impact of grazing on source habitat within a watershed was based on the percentage of source habitat in that watershed with an active cattle grazing allotment (i.e., sheep grazing allotments were not considered) (fig. 21). The amount of source habitat in an active grazing allotment was categorized using 10 percent increments from 0 to 100 percent, with the assumption that habitat outcomes became increasingly poorer as the proportion of source habitat in an active allotment increased.

Pond size—

Ponds reported used for breeding and during the summer ranged in mean size from 0.06 to 0.98 ac (Bull and Hayes 2001, Pilliod et al. 2002). Ponds used over winter ranged in mean size from 0.2 to 4.9 ac (Bull and Hayes 2002, Pilliod et al. 2002). Bull and Marx (2002) found that lake size was a significant factor in the prediction of the abundance of egg masses. Lakes evaluated in that study ranged in size from 0.98 to 86.0 ac. A negative relationship was found between productivity and lake size.

The following classes were used to evaluate the effect of pond and lake size on Columbia spotted frogs (fig. 21):

- Less than optimum: <0.062 or >4.9 ac mean size within a watershed
- Optimum: 0.062 to 4.9 ac mean size within a watershed

Road density—

Increasing densities of roads is expected to result in reductions of habitat quality for Columbia spotted frogs because of direct mortality, habitat fragmentation, and reduced water quality (Findlay and Houlahan 1997, Findlay and Bourdages 2000, Funk et al. 2005, Houlahan and Findlay 2003, Trombulak and Frissell 2000, Vos and Chardon 1998). Habitat fragmentation and associated reduction in connectivity of habitat has been associated with the disappearance of frog populations from occupied habitat (Cushman 2006, Knapp et al. 2003). Columbia spotted frogs have been reported to move from 1,640 ft (Bull and Hayes 2001, Hollenbeck 1974, Turner 1960) to 3,280 ft (Pilliod et al. 2002) between ponds. Therefore, the effects of roads were assumed to occur within 3,280 ft of source habitat.

The following density classes were based partially on the findings of Findlay and Houlahan (1997) and were applied to an area within 3,280 ft of source habitat within a watershed (fig. 21):

- Zero: <0.1 mi/mi^2 open roads
- Low: 0.1 to 1.0 mi/mi^2 open roads
- Moderate: 1.1 to 2.0 mi/mi^2 open roads
- High: >2.0 mi/mi^2 open roads

Variables considered but not included—

American bullfrogs (*Rana catesbeiana*) have been reported to be a factor in the decline of populations of ranid frogs (e.g., Doubledee et al. 2003) and may be associated with declines in spotted frog populations (Bull 2005, Monello et al. 2006). However, there was limited empirical evidence to implicate American bullfrogs as a cause of spotted frog population reduction or loss. There was also limited spatial data on the distribution of American bullfrogs across the assessment area. Because of these factors, we did not include potential effects of American bullfrogs on spotted frogs in this model.

Mining activities may affect wetlands and their biota directly through habitat destruction or runoff of sediments and contaminants generated during mining operations (Linder et al. 1991). Anecdotal evidence has indicated that mining operations may negatively affect habitat for spotted frogs. However, these effects have not been documented. Also, digital spatial information concerning locations of mining operations throughout the assessment area was generally unavailable. As a result, we did not include this variable in our assessment.

Calculation of historical conditions—

Values of the model variables were set with the following values to estimate historical habitat conditions:

- Current amount of habitat in each watershed was increased by 30 percent
- Departure of source habitat from HR: 0.5
- Invasive animals: Class zero
- Grazing: None
- Pond size: Same as current condition
- Road density: Class zero

GIS databases use—

- National Wetland Inventory
- Active cattle grazing allotments
- Lakes
- Roads

The relative sensitivity of WI values to variables in the model for Columbia spotted frog is shown in table 25.

Table 25—Relative sensitivity of watershed index values to variables in the model for Columbia spotted frogs

Variable	Sensitivity rank
Habitat departure	1
Pond size	2
Grazing impact	3
Invasive animals	4
Road density	5

Assessment Results

Watershed scores—

Major factors that influenced the WI scores included the amount of source habitat compared to levels historically available in the watersheds (fig. 22). We assumed that all watersheds had approximately 70 percent of the historical amount of habitat remaining based on the findings of Dahl (1990) and Peters (1990). Watersheds that currently have the greatest amount of source habitat included Chewelah, Stensgar/Stranger, Upper Little Spokane River, Upper Okanogan River, and Upper Pend Oreille (fig. 22). However, within all of these watersheds <25 percent of the source habitat was managed by federal agencies. The watersheds with the least amount of source habitat were located across the western portion of the assessment area.

Historically, 65 of 72 (90 percent) watersheds within the assessment area provided habitat for Columbia spotted frogs (fig. 22). However, 25 of those watersheds provided less than the historical median amount of habitat across all watersheds with habitat in the assessment area (fig. 22). This analysis indicated that 8 percent (n = 6) of watersheds with source habitat currently have high WI scores (>2.0) (fig. 22). The majority of watersheds (82 percent, n = 59) have WI scores that were moderate (>1.0 to <2.0). These were distributed across the assessment area.

Grazing affected suitability of habitat for Columbia spotted frogs in 43 percent of the watersheds (n = 31).

The size of wetlands affected suitability of habitat for Columbia spotted frogs. Mean size of ponds and wetlands within watersheds ranged from <0.25 to >124 ac. The mean sizes of habitats within 31 percent of the watersheds (n = 22) were within the optimum range.

Grazing affected suitability of habitat for Columbia spotted frogs in 43 percent of the watersheds (n = 31) (Reaser 2000). Percentage of source habitat grazed was highest in the northern and central portions of the assessment area.

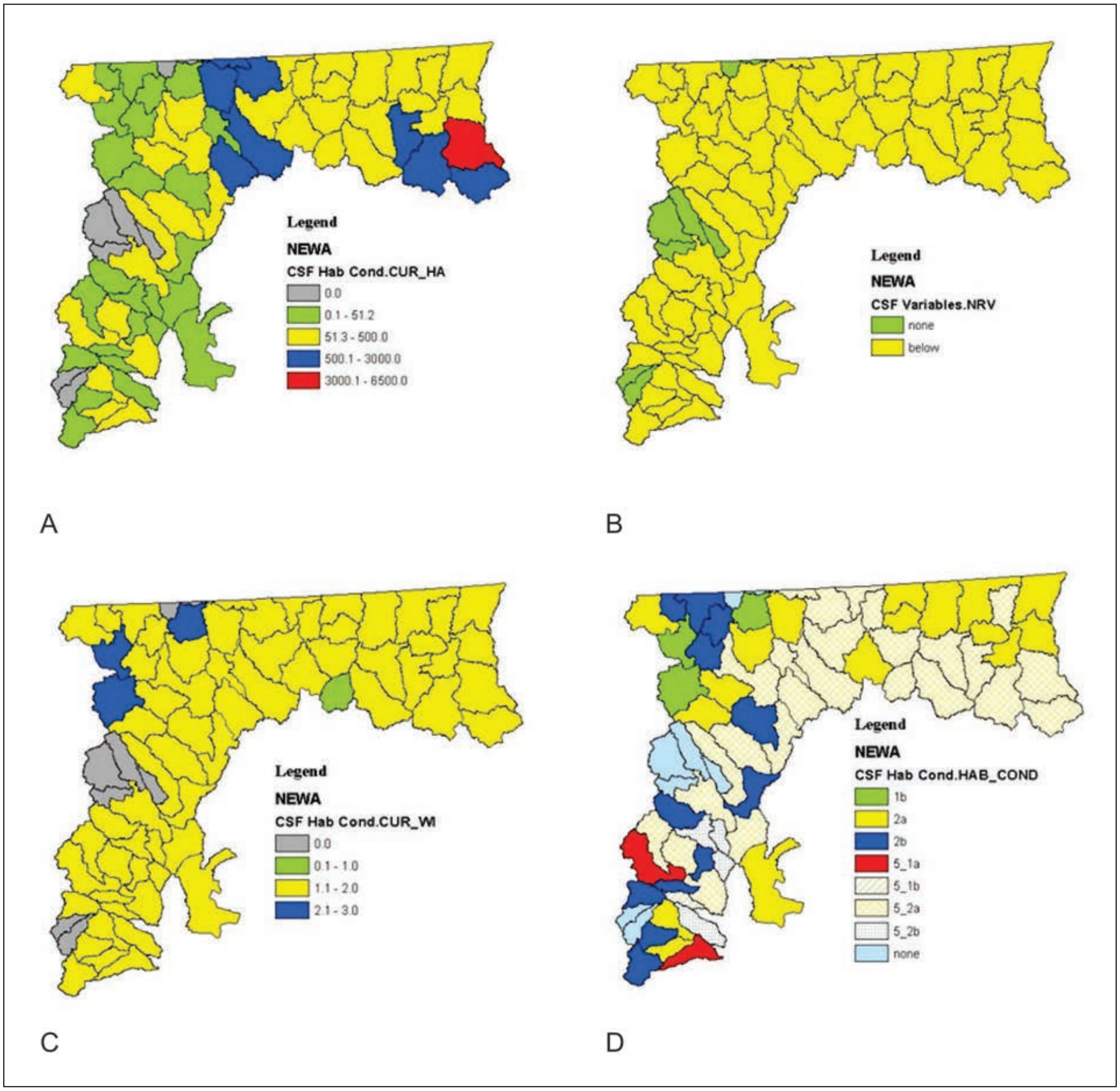

Figure 22—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D) habitat condition class for Columbia spotted frogs (CSF) by watershed in the northeast Washington assessment area (NEWA).

Road density also affected suitability of watersheds as habitat for Columbia spotted frogs (Trombulak and Frissell 2000). The percentage of source habitat with high road densities generally increased from the northeast to the southwest portion of the assessment area with low densities of roads dominating in the northwest.

Viability outcome scores—

The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for Columbia spotted frogs within the assessment area was 56 percent of the historical capability. Dispersal across the assessment area was not considered an issue for Columbia spotted frogs. Four of five ecoregions currently contained ≥1 watershed with >40 percent of the median amount of historical source habitat (the median was calculated across all watersheds with source habitat). Forty watersheds (56 percent) had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 72 percent probability that the current viability outcome for Columbia spotted frogs was C (fig. 23). It was likely that all other species included in the ponds/small lake/backwater group within the riparian family had similar outcomes.

Historically, dispersal across the assessment area was not considered an issue for Columbia spotted frogs. All ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat. Sixty-four percent (n = 46) of the watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 71 percent probability that the historical viability outcome for Columbia spotted frogs was A (fig. 23).

Historically, Columbia spotted frogs and other species in the ponds/small lake/backwater group within the riparian family were likely well distributed with viable populations across the assessment area. Changes in habitat conditions have resulted in the current situation where these species are likely well distributed in only a portion of the assessment area.

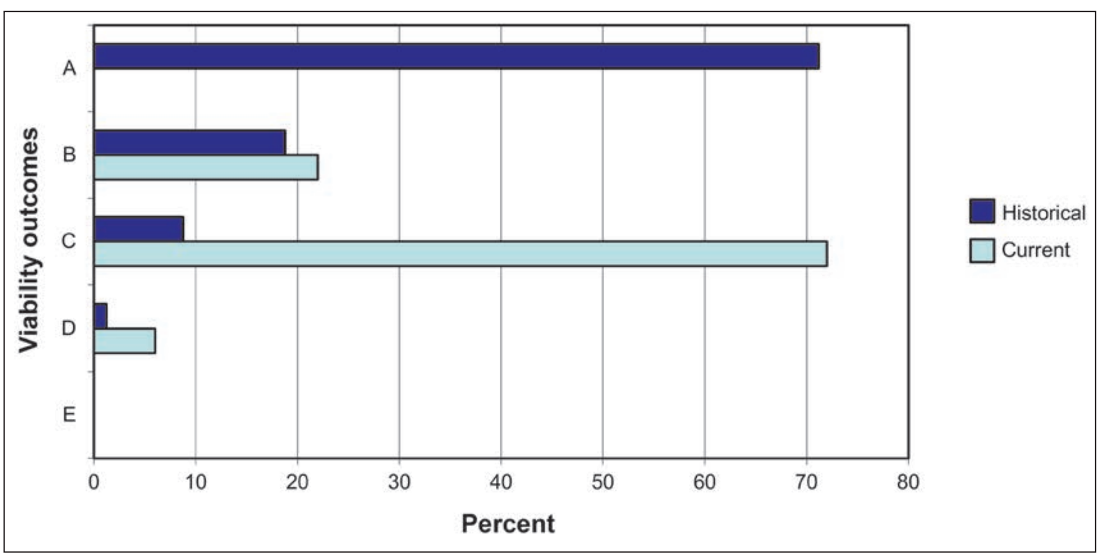

Figure 23—Current and historical viability outcome for Columbia spotted frogs in the northeast Washington assessment area.

Management Considerations

The following issues were identified for species in the ponds/small lake/backwater group within the riparian family during this assessment and from the published literature for considerations for managers:

1. Reduction of suitable wetland source habitats.
2. Negative effects of roads adjacent to source habitats.
3. Negative effects of introduced fish.
4. Degradation of source habitats by domestic livestock.

Eared Grebe

Introduction

Eared grebes were chosen as a surrogate species to represent species associated with the marsh/open water group in the wetland family. The main risk factors for all species associated with marsh habitat were draining, filling, and degradation of marshes; environmental contaminants; and disturbance. Eared grebes were chosen as the surrogate species for this group because they had widespread distribution in eastern Washington, their risk factors included those of the other species in this group, and they were not a hunted species. Habitats for eared grebes and other species in this group were not abundant on National Forest System lands in eastern Washington state, and they were patchily distributed across the northeast Washington assessment area with concentrations in the central and eastern portions (Smith et al. 1997). Eared grebes were breeding season residents of the assessment area (Cullen et al. 1999); this assessment was for breeding and rearing habitat.

Model Description

Source habitat—

Large, very open (i.e., 70 percent open water) wetlands, ponds, and lakes <9.8 ft deep were preferred colony sites for eared grebes (Boe 1992, Faaborg 1976, Savard et al. 1994). Boe (1992) went on to report that type 4 wetlands were preferred and type 5 wetlands were avoided. Kantrud and Stewart (1984) reported that 54 percent of eared grebe colonies were in seasonal wetlands, 36 percent in semipermanent wetlands, and 11 percent in permanent wetlands (n = 35). Naugle et al. (1999) and Savard et al. (1994) also noted that eared grebes avoided wetlands, ponds, and lakes with woody vegetation at the edges. These findings suggested that palustrine, emergent wetlands (PEM), as described in the NWI (Cowardin et al. 1979) with adjacent open water were preferred habitat for nesting eared grebes. We delineated habitat for this analysis by identifying all PEMs from NWI maps and adding a 1,640-ft buffer into adjacent open water, where it was present. Habitat below 5,900-ft elevation was considered as source habitat. Habitat above that elevation was not available for nesting in the spring because of persistent ice.

Eared grebes also used open water lakes with submergent vegetation, which was used as a base for nest building (Boe 1993). However, this condition was not characterized in the NWI maps so we did not include it in our description of source habitat. Also, although wetlands may have been created with development of reservoirs within the assessment area, wetlands were also inundated as reservoirs were filled (Yokom et al. 1958). Information was not available on the resulting net loss or gain, so this aspect was not addressed in these applications of the model.

The current habitat departure for eared grebes was set at -2 for all watersheds (see p. 21).

Pond/lake size—

Eared grebes require a long, running takeoff to take flight so they prefer large, very open ponds and lakes (Faaborg 1976, Johnsgard 1987). Increasing area of wetland was strongly related to suitability of a site for eared grebes (Naugle et al. 2001, Yokom et al. 1958). Ponds and lakes >75 ac were preferred (Boe 1992) although smaller water bodies (e.g., 50 ac) also were used (Faaborg 1976). Colony size was positively correlated with wetland size, and larger wetlands tended to be used more often in subsequent years than smaller wetlands (Boe 1992). We also assumed that the probability of a disturbance effect from human recreation activity was lower on large water bodies than on small water bodies. The following classes were used to evaluate the effect of habitat size on habitat quality (fig. 24):

- Small: <50 ac mean size within a watershed
- Medium: 50 to 75 ac mean size within a watershed
- Large: >75 ac mean size within a watershed

Emergent plant:open water ratio—

Access to open water was important for eared grebes because they move to open water when disturbed from their nests, and because they need a running start before taking flight (Boe 1992). The source habitat complex of wetland and open water with ≥50 percent open water was considered in this analysis to be higher quality habitat than wetlands with <50 percent open water. The following classes were used to evaluate the effect of emergent plant:open water ratio on habitat quality (fig. 24):

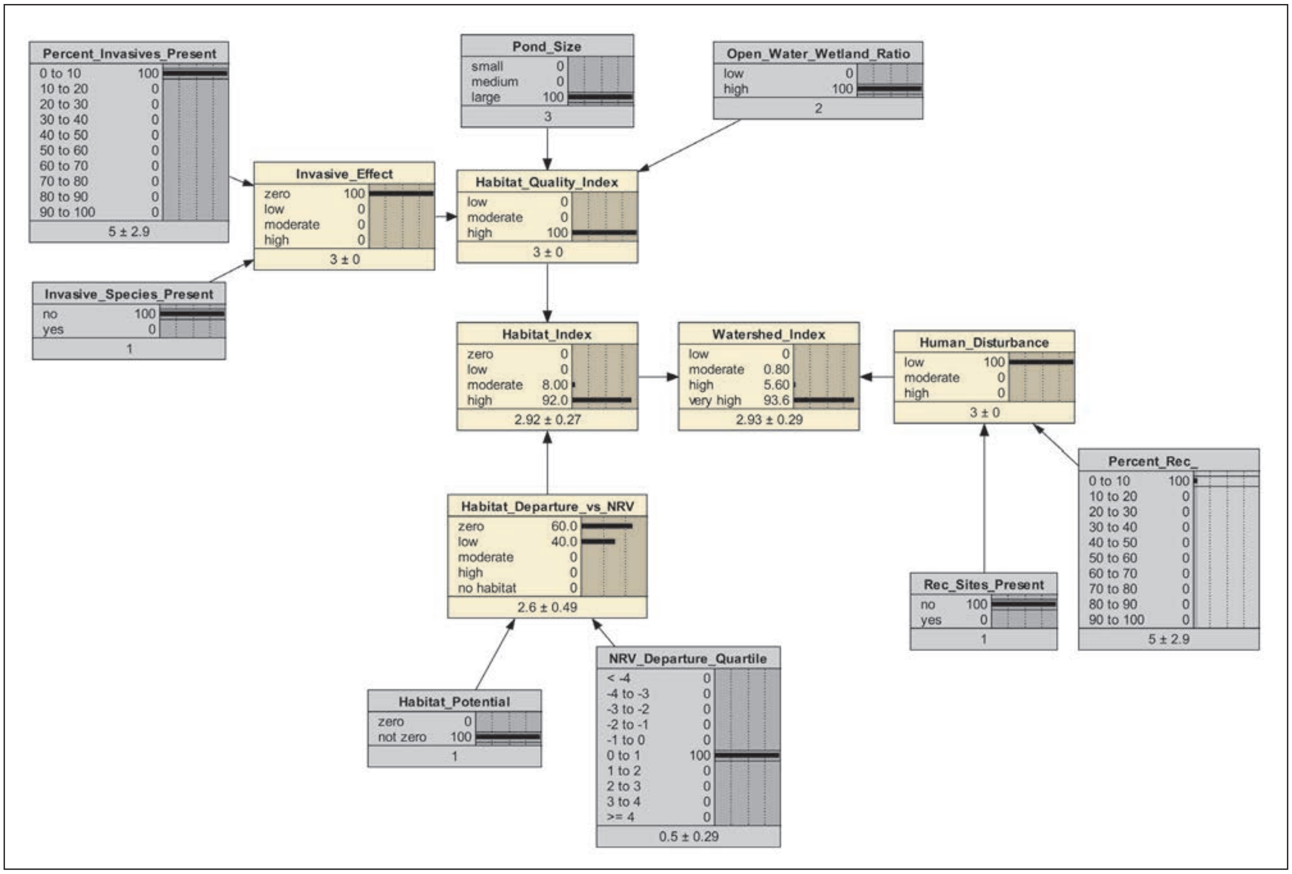

Figure 24—Surrogate species assessment model for eared grebe.

- Low: <50 percent open water mean size in wetland complexes within a watershed
- High: ≥50 percent open water mean size in wetland complexes within a watershed

Invasive animals—

Grass carp and common carp have been documented to have detrimental effects on aquatic vegetation in lakes and wetlands through uprooting of plants, increased herbivory, and decreased water quality resulting in a decrease in habitat quality for waterfowl (Bonar et al. 2002, Crivelli 1983, Fletcher et al. 1985, Roberts et al. 1995). The presence of carp in lakes and wetlands identified as source habitat for eared grebes was assumed to result in lower habitat quality.

The impact of carp on the quality of source habitat within a watershed was based on the percentage of source habitat in that watershed with carp present. The amount of source habitat with carp present was categorized using 10 percent increments from 0 to 100 percent, with increasing poorer habitat outcomes as the proportion of source habitat with carp increased (fig. 24). We used information from WDFW (2005) on fish surveys for this analysis to evaluate the likelihood of the presence of carp in source habitats in the assessment area.

Recreation sites—
Presence of boat-launch ramps and campgrounds on lakes and ponds was expected to result in reductions of habitat quality for eared grebes because of an increased potential for human disturbance and habitat fragmentation (Boe 1992, Hanus et al. 2002). Potential adverse effects include egg and nestling mortality, premature fledging or nest evacuation, and reduced body mass, or slower growth of nestlings (Rogers and Smith 1995, Skagen et al. 2001). Adult behavior also may be altered by disturbance, resulting in altered foraging patterns. Use of motorized watercraft near nests of eared grebes may result in increased disturbance, but the published literature was equivocal on this aspect (Rogers and Smith 1995, Titus and VanDruff 1981).

The impact of human disturbance on the quality of source habitat within a watershed was based on the percentage of source habitat in that watershed with associated recreation sites. The amount of source habitat associated with recreation sites was categorized using 10 percent increments from 0 to 100 percent, with increasing poorer habitat outcomes as the proportion of source habitat associated recreation sites increased (fig. 24).

GIS databases used—

- National Wetland Inventory
- Lakes
- Recreation sites
- Carp survey data

Calculation of historical conditions—
Values of the model variables were set with the following values to estimate historical habitat conditions:

- Current amount of habitat in each watershed was increased by 30 percent
- Departure of source habitat from HRV: 0.5
- Pond/lake size: Same as current condition
- Emergent plant: Open water ratio—Same as current condition
- Invasive animals: 0 percent of source habitat affected
- Recreation sites: 0 percent of source habitat affected

The relative sensitivity of WI values to variables in the model for eared grebe are shown in table 26.

Table 26—Relative sensitivity of watershed index values to variables in the model for eared grebes

Variable	Sensitivity rank
Habitat departure	1
Human disturbance	2
Pond/lake size	3
Invasive effect	4
Open water: wetland ratio	5

Assessment Results

Watershed scores—

Historically, 21 percent (n = 15) of the watersheds within the assessment area provided habitat for eared grebes. Currently, the same watersheds contained some habitat for eared grebes, although three had minimal amounts (i.e., <50 ac) (fig. 25). Watersheds with the largest amounts of habitat were located in the central, eastern, and southern portions of the assessment area. We assumed all watersheds had reductions in amount of habitat when compared to historical conditions (fig. 25). All watersheds with habitat had low WI scores (i.e., <1.0) (fig. 25).

Large wetlands were assumed to be higher quality habitat than small wetlands (Boe 1992). Thirteen of the 15 watersheds with source habitat had large mean sizes of wetland habitat; two had medium mean size of habitat. The emergent plant:open water ratio was high for all watersheds but one. The presence of carp in lakes and wetlands identified as source habitat for eared grebes was assumed to result in lower habitat quality; carp were assumed to be present in all source habitats. Presence of boat-launch ramps on lakes and ponds was expected to result in reductions of habitat quality for eared grebes (Boe 1992, Hanus et al. 2002). Within the 10 watersheds with boat launches associated with source habitat for eared grebes, >80 percent of the habitat was accessible to boats.

Viability outcome scores—

The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for eared grebe within the assessment area was 12 percent of the historical capability. Dispersal across the assessment area was not considered an issue for this species. Four of five ecoregions currently contained at least one watershed with >40 percent

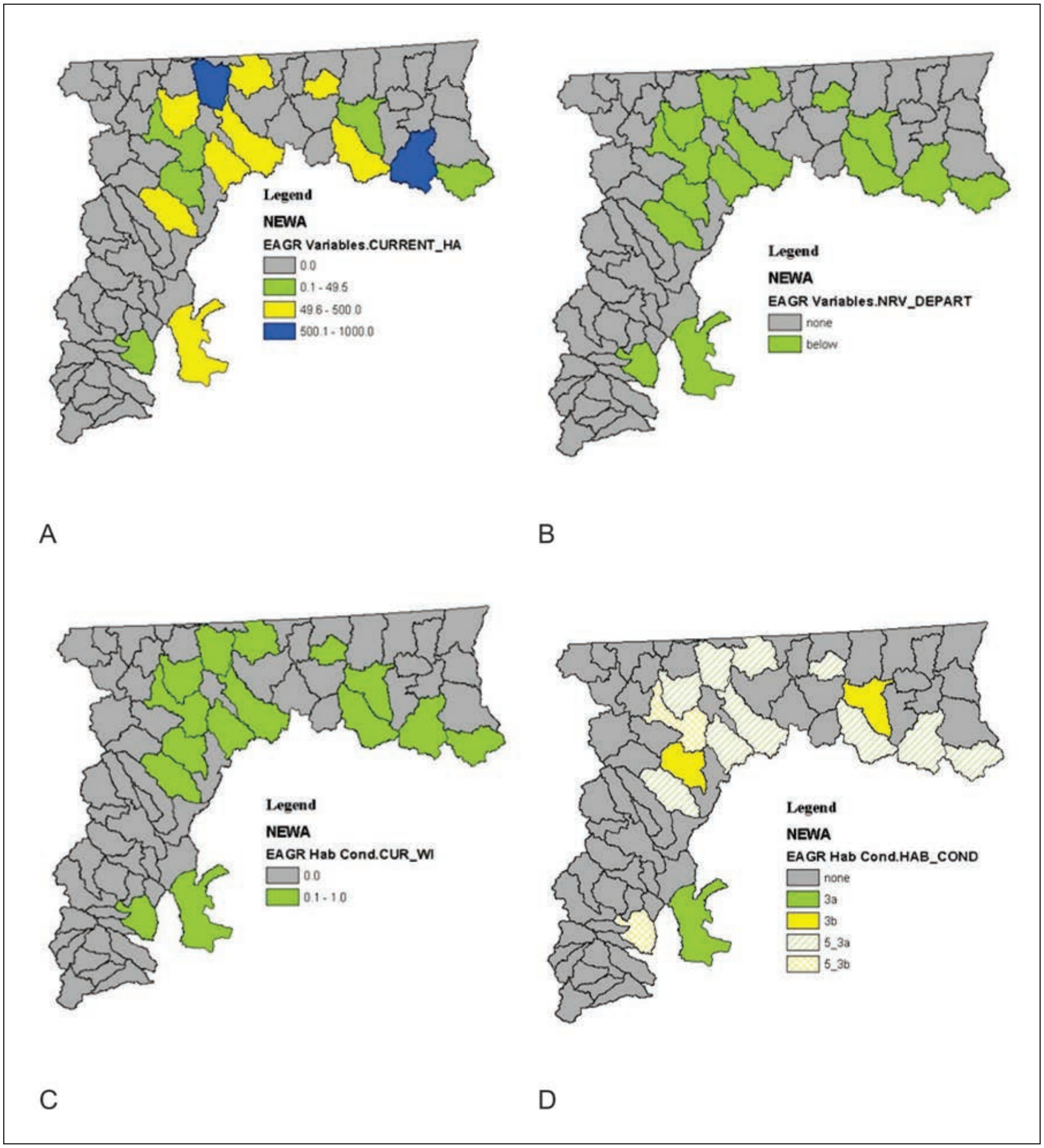

Figure 25—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D) habitat condition class, for the eared grebe (EAGR) by watershed in the northeast Washington assessment area (NEWA).

of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Fourteen percent (n = 10) of the watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 72 percent probability that the current viability outcome for

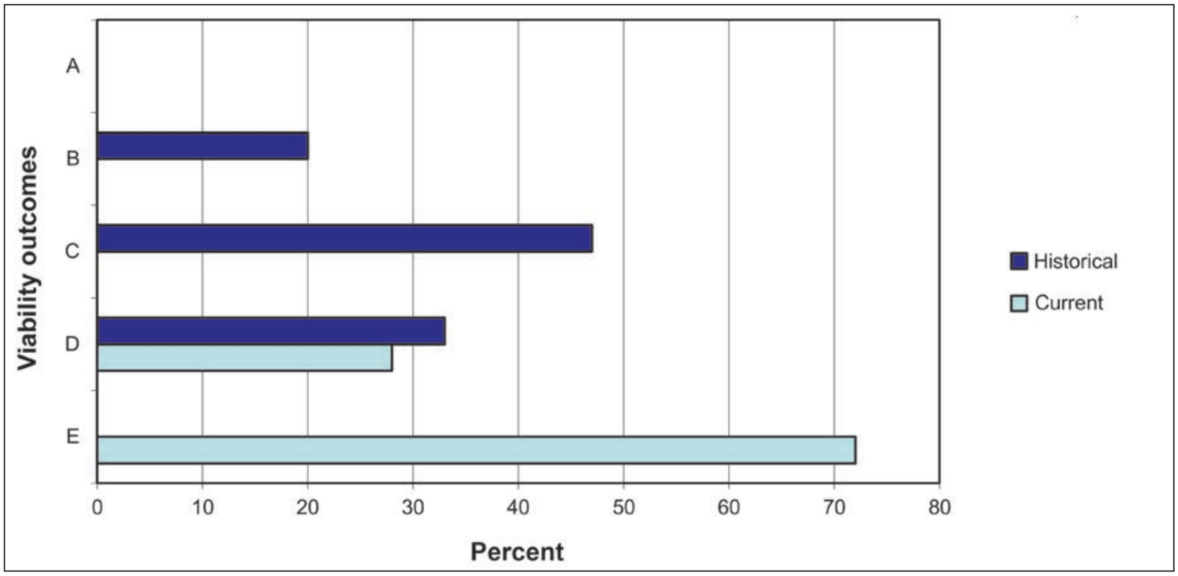

Figure 26—Current and historical viability outcomes for eared grebes in the northeast Washington assessment area.

Suitable habitat was highly isolated and in very low abundance. It is likely that other species associated with the marsh/open water group in the wetland family had similar outcomes.

eared grebe with the marsh/open water group in the wetland family was E, indicating that suitable habitat was highly isolated and in very low abundance (fig. 26). It is likely that other species associated with the marsh/open water group in the wetland family had similar outcomes.

Historically, dispersal across the assessment area was not considered an issue for this species. All ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Fifteen percent (n = 11) of watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 47 percent probability that the historical viability outcome for eared grebe was C and a 33 percent probability that the historical viability outcome for these species was D, indicating that this habitat had a patchy to isolated distribution and existed at low abundance (fig. 26).

Under historical conditions, eared grebes and other species associated with the marsh/open water group in the wetland family were likely well distributed in only a portion of the assessment area or were not well distributed throughout the assessment area. However, currently they are likely to face extirpations throughout the assessment area owing to loss of habitat and limited distribution of suitable environments.

Management Considerations

The following issues were identified during this assessment and from the published literature for species associated with the marsh/open water group in the wetland family for consideration by managers:

1. Loss and degradation of wetland habitats.
2. Negative effects of carp invasion in source habitats.
3. Negative effects of disturbance from water-based recreation.

Fox Sparrow

Introduction

Fox sparrows were chosen as a surrogate species to represent species in the early-successional group of the open forest family. They preferred dense, low shrub growth typical of such habitats and were susceptible to the effects of grazing by domestic livestock similar to other species in this group. The range of fox sparrows includes the western and eastern portions of the assessment area (Smith et al. 1997). Fox sparrows are breeding season residents of the assessment area (Weckstein et al. 2002); this assessment is for breeding and rearing habitat.

Model Description

Source habitat—

Fox sparrows were strongly associated with riparian shrubs (e.g., willow [*Salix* spp.], alder [*Alnus* spp.]) (Webster 1975), and the shrub stage of succession following fire and clearcut logging in mature forests (Banks 1970, Fontaine et al. 2009, Hagar 1960, Kirk and Hobson 2001, Machtans and Latour 2003, Simon et al. 2002, Weckstein et al. 2002). Densities of fox sparrows were reported highest in stands with heavy salvage logging following fire, intermediate in moderately salvaged stands, and lowest in the unsalvaged stands (Cahall and Hayes 2009). Although the early stages of the shrub successional stage were preferred (e.g., 3 to 15 years) (Hagar 1960, Meslow and Wight 1975), they also used shrub habitats for up to 30 years after disturbance (Simon et al. 2002). Residual trees remaining after clearcut logging (especially conifers) resulted in reduced densities of fox sparrows (Simon et al. 2002).

Abundance of fox sparrows was significantly correlated with mean shrub height (Anderson 2007, Olechnowski and Debinski 2008). Tall shrubs without tree cover were preferred, and tall shrubs with residual tree cover were used, but to a lesser extent. Densities of fox sparrows ($r = 0.80$) were positively correlated with shrub volume (Cahall and Hayes 2009). Cover types representing montane shrubs and forest reinitiation and regeneration following timber harvest and fire were included as source habitats (fig. 27). This included single- and multistory forested stands in mesic forest, cold-dry forest, cold-moist forest, and parkland potential vegetation conditions (nonforest and dry forest were not included) with <30 percent canopy cover or tree size <4 in QMD. Shrub-steppe (i.e., arid shrub) land cover classes were not included.

- Vegetation zones: Western hemlock, Pacific silver fir, mountain hemlock, subalpine fir, parkland
- Cover type: Conifer mix, Douglas-fir, Engelmann spruce, grand fir, lodgepole pine, mountain hemlock, Pacific silver fir, parkland, riparian and deciduous, montane shrubs, western hemlock, western larch, western redcedar
- Size class: <4 in QMD
- Tree layers: Single and multistoried

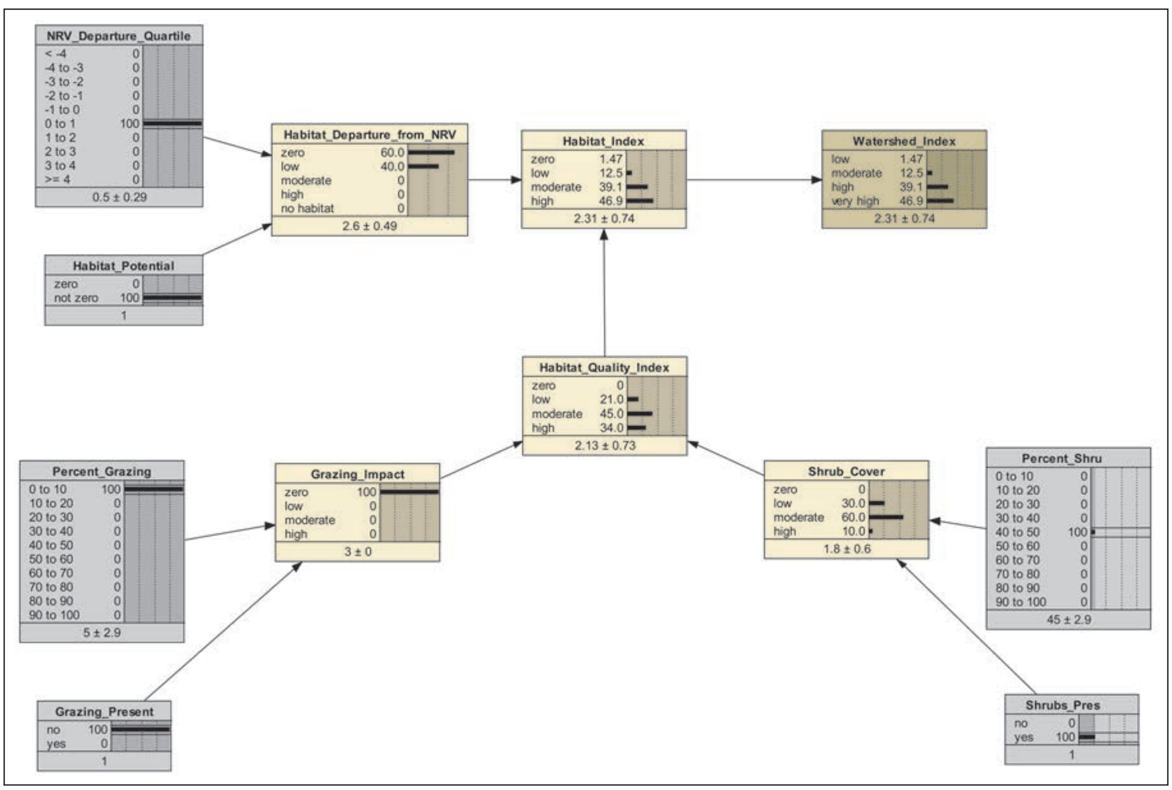

Figure 27—Surrogate species assessment model for fox sparrow.

Grazing—
The results reported in the literature on the effects of grazing on fox sparrows were unequivocal. Several studies reported a negative response from fox sparrows associated with cattle grazing (Knopf et al. 1988, Page et al. 1978, Schulz and Leininger 1991). Although fox sparrows were parasitized by brown-headed cowbirds (which were often associated with livestock grazing operations), it occurred infrequently (Friedmann 1963).

The impact of grazing on source habitat within a watershed was based on the percentage of source habitat in that watershed within an active grazing allotment. The amount of source habitat in an active grazing allotment was categorized using 10 percent increments from 0 to 100 percent, with increasing poorer habitat outcomes as the proportion of source habitat in an active allotment increased (fig. 27).

Shrub cover—
The amount of shrub cover was directly related to habitat quality for fox sparrow. Low shrub cover greatly diminishes the value of an area as habitat for fox sparrows. High shrub cover greatly increases the quality of habitat for fox sparrows. Fires tend to eliminate shrub cover and reduce habitat quality for fox sparrow in the short term (Samuels et al. 2005). This variable addressed the proportion of source habitat that had >70 percent shrub cover as determined from gradient nearest-neighbor analysis (Ohmann and Gregory 2002) (fig. 27).

Calculation of historical conditions—
Values of the model variables were set with the following values to estimate historical habitat conditions:

- Departure of source habitat from HRV: 0.5
- Grazing: None
- Shrub cover: Percentage of shrubs was set at 50 percent

The relative sensitivity of watershed index values to variables in the model for fox sparrow are shown in table 27.

Assessment Results

Watershed scores—
Historically, all 73 watersheds within the assessment area provided habitat for fox sparrows. Currently, 92 percent (n = 67) of the watersheds contain some habitat for fox sparrows, although several have minimal amounts (i.e., <50 ac) (fig. 28). All watersheds with habitat had low WI scores (i.e., <1.0) (fig. 28).

Table 27—Relative sensitivity of watershed index values to variables in the model for fox sparrows

Variable	Sensitivity rank
Shrub cover	1
Habitat departure	2
Grazing impact	3

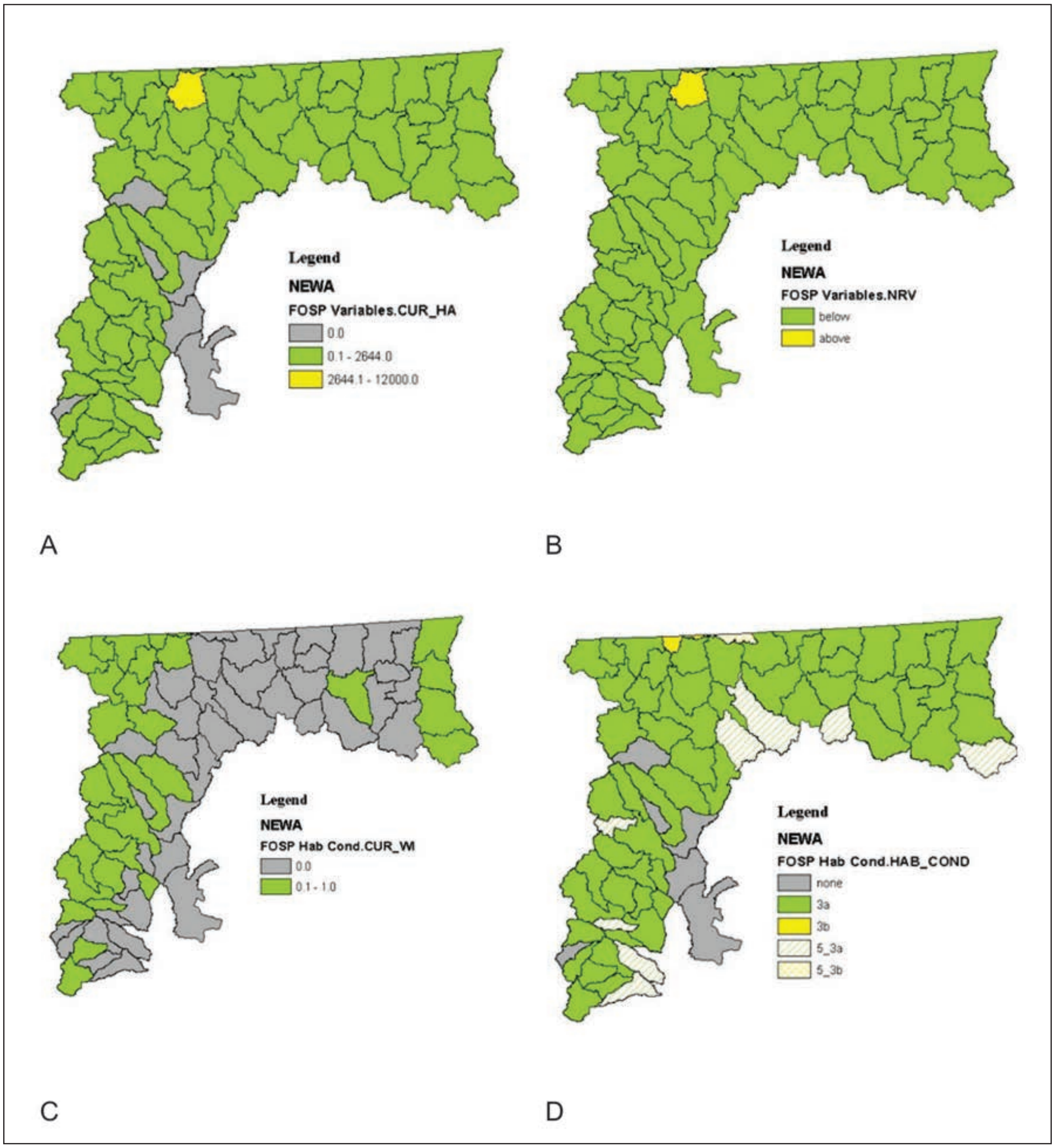

Figure 28—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D) habitat condition class for fox sparrows (FDSP) by watershed in the northeast Washington assessment area (NEWA).

Factors that influenced the WI scores included the amount of source habitat compared to levels historically available in the watersheds. All but one of the watersheds were well below their historical median levels of source habitat (i.e., class -4.0). The upper Chewuch River watershed was above its historical median levels (>4.0) (fig. 28).

The percentage of source habitat within an active grazing allotment was used to assess the impact of grazing to fox sparrows and ranged from 0 to 100 percent by watershed. Source habitat within 20 percent (n = 14) of the watersheds was not grazed. Twenty-four percent (n = 17) of the watersheds had <25 percent of source habitat grazed; 13 percent (n = 9) had 25 to 50 percent grazed; 11 percent (n = 8) had 50 to 75 percent grazed; and 32 percent (n = 23) had >75 percent grazed.

The amount of shrub cover was directly related to habitat quality for fox sparrow (Samuels et al. 2005). The percentage of source habitat with >70 percent shrub cover varied from 0 to 40 percent among watersheds. Seventeen percent (n = 12) of the watersheds had <1.0 percent, 57 percent (n = 41) of the watersheds had 1 to 10 percent, 21 percent (n = 15) had 10 to 20 percent and 5 percent (n = 3) had >20 percent.

Viability outcome scores—
The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for fox sparrows within the assessment area was only 9 percent of the historical capability. Dispersal across the assessment area was not considered an issue for this species. One of five ecoregions currently contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). One watershed had >40 percent of the median amount of historical source habitat (1 percent). Under those circumstances, there was an 80 percent probability that the current viability outcome for fox sparrow was E with the remaining in D (20 percent), indicating that habitat for these species was highly isolated and at very low abundance (fig. 29). Outcomes were likely similar for other species in the early-successional group of the open forest family.

Historically, dispersal across the assessment area was not considered an issue for this species. All ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Ninety-five percent (n = 69) of the watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 85.5 percent probability that the historical viability outcome for fox sparrow was A, indicating that habitat was broadly distributed and in high abundance (fig. 29).

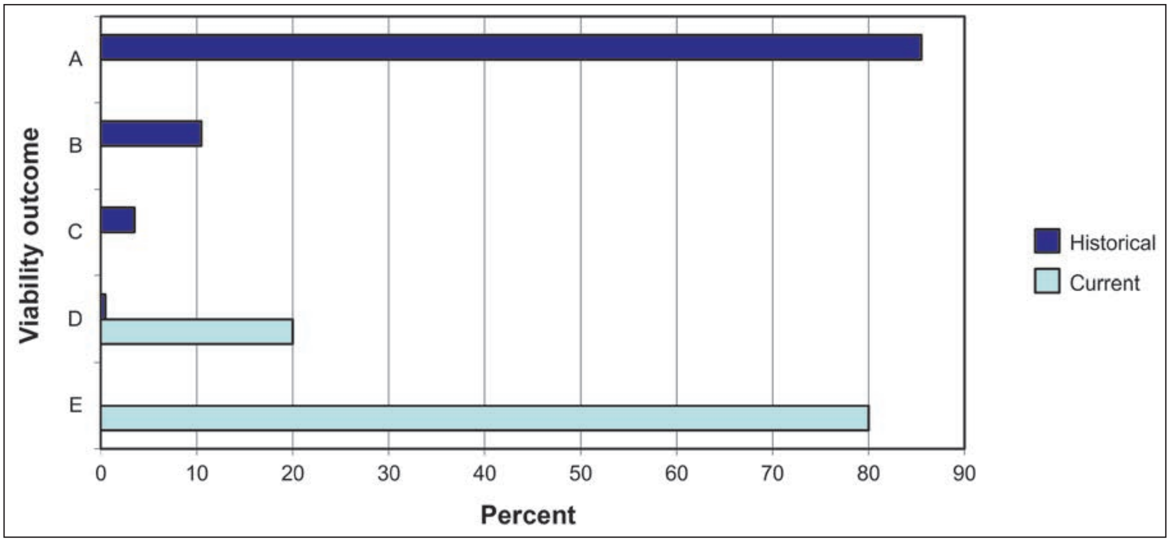

Figure 29—Current and historical viability outcomes for fox sparrows in the northeast Washington assessment area.

In summary, under historical conditions, fox sparrows and other species in the early-successional group of the open forest family were likely well distributed throughout the assessment area; currently they were likely not well distributed and at risk of extirpation.

Our results for this species were similar to those reported in the broad-scale habitat analysis by Wisdom et al. (2000) in ICBEMP. According to the ICBEMP, terrestrial vertebrate habitat analyses, historical source habitats for Lazuli bunting, which was associated with source habitats similar to those used by fox sparrows, included portions of the Northern Cascades and the Northern Glaciated Mountains ERUs, which overlap our assessment area (Wisdom et al. 2000). Within this historical habitat, declines in source habitats for this species have been extensive, -100 percent in the Northern Cascades and -66 percent in the Northern Glaciated Mountains according to Wisdom et al. (2000).

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Reduction of early-seral habitats, primarily those resulting from fire (Simon et al. 2002).
2. Negative effects of grazing in source habitats.

Golden Eagle

Introduction

Golden eagles were chosen as a surrogate species to represent species of concern associated with the woodland/grass/shrub group in the woodland/grass/shrub family. This species reflected the risk of human disturbance that affected other species in these habitats. It was also associated with cliff structures that were not normally used by other surrogate species within this group and family. Golden eagle nests are readily monitored and are often surveyed by other public agencies and non-governmental groups, so trend data may be readily available. Golden eagles range throughout the assessment area except for the eastern portion (Smith et al. 1997). Golden eagles are year-round residents of the assessment area (Kochert et al. 2002); this assessment was for nesting and rearing habitat.

Model Description

Source habitat—

The fundamental requirements of suitable habitat for golden eagles included (1) sources of food, (2) adequate nesting sites, and (3) limited human intrusion (Beecham and Kochert 1975, Thelander 1974). Golden eagle habitats with the highest population density were characterized by availability of diverse and numerous prey, and abundant nest sites (Phillips et al. 1984). Areas with low population densities had few nest sites available for use and were fragmented by cropland.

Nesting habitat—

Availability of adequate nest sites may limit distribution of golden eagles, especially in sagebrush and grassland habitats (Beecham and Kochert 1975, Carrete et al. 2000, Phillips and Beske 1990). Throughout North America, golden eagles nest primarily on cliffs (McGahan 1968, Mosher and White 1976, Smith and Murphy 1982). Generally, trees were used infrequently as nest substrates but may be important in local areas (Menkens and Anderson 1983).

Forests—

Nests in trees have been reported in northeast and north-central Wyoming (Menkens and Anderson 1987, Phillips and Beske 1990, Phillips et al. 1990), the central and north Coast Range in California (Chinnici et al. 2007, Hunt et al. 1999), and coastal Washington (Bruce et al. 1982, Eaton 1976). Wide varieties of trees were used as nest sites throughout their range (Kochert et al. 2002). Tree species throughout eastern Oregon and Washington that were most likely to provide nest sites include ponderosa pine (*Pinus ponderosa*) (MacLaren 1986, Phillips and

Beske 1990, Phillips et al. 1990), Douglas-fir (*Pseudotsuga menziesii*) (McGahan 1968), and cottonwood (*Populus* spp.) (Bates and Moretti 1994). However, a preference has been reported for large ponderosa pines over cottonwoods (Phillips and Beske 1990). The nest tree was usually the largest or one of the largest trees in a stand (Menkens and Anderson 1987), was isolated or on the fringe of a small stand of trees (Baglien 1975), and was <1,640 ft from open areas (Bruce et al. 1982). Dense forest stands were avoided as nest sites (Phillips and Beske 1990, Phillips et al. 1984, Whitfield et al. 2004). Large trees may have been selected to ensure nest stability and longevity, and placement in the upper portion of tall trees may have improved accessibility (Menkens and Anderson 1987).

Forested source habitat for nesting golden eagles for this analysis was assumed to be large trees (>20 QMD), single- and multistory, open (<50 percent canopy closure) ponderosa pine or Douglas-fir, <1,640 ft from an edge with low-elevation shrub or grassland cover types with an elevation of <3,600 ft. The amount of habitat in each watershed was compared to the current median value across watersheds with habitat (fig. 30).

Cliffs—

Many nests located on cliffs had a wide view of the surrounding area (Beecham 1970) or were on prominent escarpments (Bates and Moretti 1994). Proximity to hunting grounds was an important factor in nest-site selection (Camenzind 1969, McIntyre et al. 2006). In northern areas, weather conditions at the beginning of nesting season were a critical factor in choice of nest site location (Morneau et al. 1994). Average annual snowfall may have limited distribution of nest sites; in southwest Montana, nests were usually built in areas receiving <200 in of snow (Baglien 1975). Cliff nests were built on several rock substrates including sandstone, shale, granite gneiss, limestone, basalt, and granite (Schmalzried 1976). Loosely cemented materials such as breccias, conglomerates, or agglomerate sluff were avoided (Baglien 1975). At four study areas, the mean height of cliffs with nests was 116.5 ft; mean height of nests on the cliff was 67.9 ft (Kochert et al. 2002).

Cliff source habitat for nesting golden eagles for this analysis was assumed to be cliffs >50 ft high at <3,500-ft elevation (to eliminate areas with persistent spring snowpacks). To model the availability of cliff source habitat, a digital elevation model was used to identify cliff structures (similar to López-López et al. 2007) that were >5 ac to distinguish the prominent cliffs structures from the smaller cliffs that were unlikely to provide nesting habitat. The following classes were used to characterize watersheds for cliff nesting habitat (fig. 30):

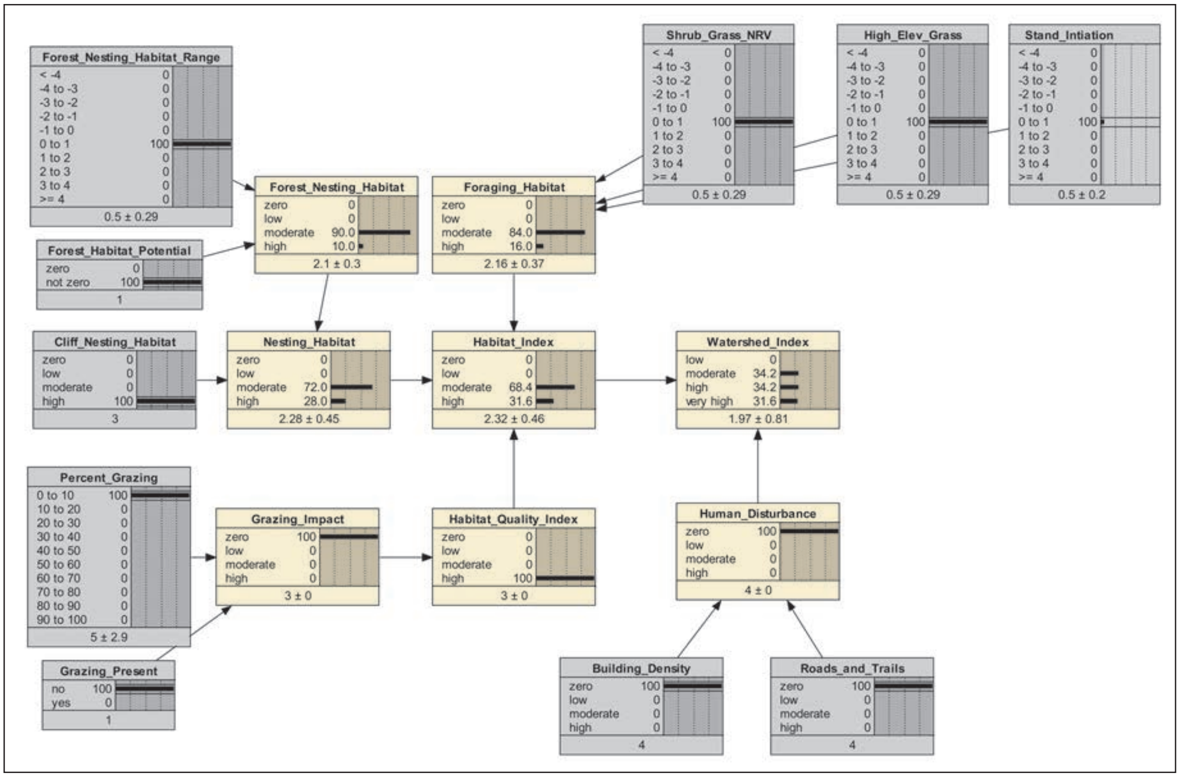

Figure 30—Surrogate species assessment model for golden eagles.

- Zero: Potential nesting habitat does not occur within a watershed
- Low: <10 ac of potential nesting habitat within a watershed
- Moderate: Less than median amount across all watersheds of potential nesting habitat within a watershed
- High: Less than median amount across all watersheds of potential nesting habitat within a watershed

Foraging habitat—

Amount and density of prey had a direct effect on distribution, reproductive rates, and population size of golden eagles (Bates and Moretti 1994, Martin et al. 2009, Pedrini and Sergio 2002, Sergio et al. 2006, Smith and Murphy 1979, Steenhof et al. 1997). Golden eagles fed primarily on mammals (80 to 90 percent of prey items), secondarily on birds, and occasionally on reptiles and fish (Olendorff 1976). Preferred mammal prey were leporids (hares [*Lepus* spp.] and rabbits [*Sylvilagus* spp.]), sciurids (ground squirrels [*Spermophilus* spp.], prairie dogs [*Cynomys* spp.], and marmots [*Marmota* spp.]) (Kochert et al. 2002). Golden eagles typically foraged in open grassland, sagebrush (*Artemisia* spp.), and other native shrub communities that provided habitat for these preferred prey species (Collopy and Edwards

1989, Smith and Nydegger 1985) and avoided agricultural land and burned areas (Beecham and Kochert 1975, Carrette et al. 2000, López-López et al. 2007, Marzluff et al. 1997, Phillips et al. 1984, Sergio et al. 2006). In central California, they were reported to forage in open grassland habitats (Hunt et al. 1999). Similar patterns were reported elsewhere for winter-habitat-use patterns (Craig et al. 1986, Fischer et al. 1984).

Primary foraging areas for golden eagles were located ≤1.9 mi from nesting sites (i.e., ≤0.62 mi during the breeding season, ≤1.9 mi during the nonbreeding season) (Baglien 1975, Chinnici et al. 2007, Kochert et al. 1999, McGrady et al. 2002, McLeod et al. 2002). For the spatial scale of this analysis (i.e., watershed), it was assumed that all foraging areas within watersheds were equally available to all golden eagles nesting in the watersheds.

Fires enhanced by the presence of cheatgrass (*Bromus tectorum*) have caused large-scale losses of foraging habitat in areas used by golden eagles throughout the intermountain West (Brooks 1999). Wildfires that burned >98,000 ac of shrublands between 1981 and 1987 in the Snake River Birds of Prey National Conservation Area adversely affected nesting populations of golden eagles (Kochert et al. 1999). Nesting success in burned territories in the Snake River Canyon declined after major fires. Abandoned burned territories were generally subsumed by neighboring pairs, resulting in a decreased number of nesting pairs. In response to these findings, all potential foraging habitat in the shrub-steppe land cover type that burned recently (i.e., since 1987) was removed from consideration as habitat in the model.

Foraging source habitat for golden eagles in this analysis was assumed to be low-elevation, native grassland cover type; shrub-steppe cover type that has not recently burned; high-elevation, native grassland cover type; and stand initiation size/structure within ponderosa pine or Douglas-fir cover types resulting from timber harvest or fire. The size of patches that was considered foraging source habitat was ≥5 ac to eliminate small, isolated patches that would not be used for foraging.

To evaluate the relative amount of low-elevation, native grassland; shrub-steppe; and stand initiation habitat within watersheds, we compared the current amount of source habitat in the watersheds to the historical median across all watersheds with habitat (fig. 30). This historical median was used to develop classes to classify degree of departure from the median. To evaluate the relative amount of high-elevation, native grassland, we compared the amount in each watershed to the median across all watersheds with this habitat. These processes allowed a relative comparison of the quantity of source habitat across the watersheds for the entire assessment area.

Grazing—

Management of cattle (*Bos taurus*) and domestic sheep (*Ovis aries*) grazing on golden eagle foraging habitat can influence prey density, diversity, and availability (Andersen 1991). Prey species generally decreased with reduced herbaceous cover and foliage height diversity (Kochert 1989). Bock et al. (1993) suggested that raptors would respond negatively to grazing in shrub-steppe habitats, based on the ground-cover requirements of their prey. Jackrabbits and ground squirrels may be moderately tolerant to grazing, but they disappeared where their habitat was overgrazed (i.e., repeated grazing that exceeds the recovery capacity of the vegetation and creates or perpetuates a deteriorated plant community). However, in California, Hunt et al. (1995) suggested that ground squirrels were attracted to areas grazed by cattle because of the reduced grass height, and that, because ground squirrels were a primary prey of golden eagles in the area, golden eagles used grazed grasslands for foraging.

The impact of grazing on source habitat within each watershed was characterized by the percentage of source habitat within an active grazing allotment. The amount of source habitat in an active grazing allotment was categorized using 10 percent increments from 0 to 100 percent, with increasing poorer habitat outcomes as the proportion of source habitat in an active allotment increased (fig. 30).

Human disturbance—

Urbanization and human-population growth have made areas historically used by golden eagles unsuitable, particularly in southern California (Scott 1985, Thelander 1974) and the Colorado Front Range (Boeker 1974, Boeker and Ray 1971). Extensive agricultural development reduced jackrabbit populations and made areas less suitable for nesting and wintering eagles (Beecham and Kochert 1975, Craig et al. 1986, Kimsey and Conley 1988, USDI 1979). Increasing tourism was found to affect territory occupancy and breeding success of golden eagles in Finland (Kaisanlahti-Jokimäki et al. 2008). Human disturbance factors were included in models of habitat suitability developed for golden eagles in the European alps (Brendel et al. 2002). Evaluation and application of these models led to a recommendation of a 980-ft buffer zone on nest sites for paragliders, climbers, and hikers and a 1,640-ft buffer zone on nest sites for helicopters. Holmes et al. (1993) recommended placement of a 980-ft buffer to reduce disturbance of golden eagles on winter foraging areas. The effects of human disturbance were addressed through building density, roads, and trails as described below.

Building density—
Abandoned territories of golden eagles had more dwellings within 1.0 mi and higher human populations within 3.0 mi than territories that continued to be occupied (Scott 1985). Golden eagles were observed almost exclusively in undeveloped areas in central Utah (Fischer et al. 1984). Human impacts may have caused high rates of golden eagle nest failure, direct mortality, and territory abandonment in southwestern Idaho (Steenhof et al. 1983) and in Caucasia (Abuladze and Shergalinn 2002). Nest sites selected in northern Spain tended to be farther away from villages than random sites (Fernandez 1993).

Densities of buildings were calculated within source habitat across the range of golden eagles in the assessment area. Singleton and Lehmkuhl's (2000) characterization of building densities was used to create the following relationships, which were used to estimate their effect on habitat quality for golden eagles in this analysis (fig. 30):

- Zero: 0 residences/mi^2
- Low: >0 to <1.5 residences/mi^2
- Moderate: 1.5 to 7.7 residences/mi^2
- High: >7.7 residences/mi^2

Habitat effectiveness—
Recreation and other human activity near golden eagle nests can disrupt breeding dynamics, but most evidence was equivocal (e.g., Martin et al. 2009). In southwestern Idaho, nest sites were located in areas with fewer roads (Steenhof et al. 1993), and proximity of nests to roads may have been related to high rates of nest failure, direct mortality, and territory abandonment (Steenhof et al. 1983). Nesting success in Scotland was related inversely to human disturbance around golden eagle nests (Watson 1997). Nest sites selected in northern Spain tended to be farther away from roads and trails than random sites (Fernandez 1993). Adults spent less time at nests and fed young less food less frequently when observers camped 1,300 ft versus 2,600 ft from nests in Alaska (Steidl et al. 1993). Mean distance of nest sites to roads was 1,500 ft in southeast Wyoming (MacLaren 1986). Flush distance of golden eagles increased as distance to roads increased (Holmes et al. 1993). Baglien (1975) recommended that roads and other developments be out of sight of nests to reduce risk of disturbance.

We estimated the potential for human disturbance to affect nesting habitat of golden eagles with an adaptation of the habitat disturbance index described in Gaines et al. (2003a). We buffered open roads and trails by 1,640 ft on each side and then intersected this with our map of source habitat. We then used the following categories to characterize the potential effects of human disturbance on golden eagles for each watershed (fig. 30):

- Low: >50 percent of the source habitat outside road and trail buffer within a watershed.
- Moderate: 25 to 50 percent of the source habitat outside road and trail buffer within a watershed.
- High: <25 percent of the source habitat outside road and trail buffer within a watershed.

Variables considered but not included—
Size of patches of sagebrush has been demonstrated to be related to use of those habitats by the golden eagle's principle prey species (e.g., leporids) (Kochert et al. 2002). Mean patch size for jackrabbit use of this habitat was 12,360 ac, with increased likelihood of jackrabbit use with increasing patch size and number of patches (Knick and Dyer 1997). Also, Carrette et al. (2000) reported a negative relationship between increasing number of habitat patches and golden eagle densities. However, this variable was not included in this model because of the difficulty in accurately describing size of patches of source habitat with data sets that were available to us.

Calculation of historical conditions—
Values of the model variables were set with the following values to estimate historical habitat conditions:

- Nesting habitat
- Forests: 0.5
- Cliffs: Same as current condition
- Foraging
- Low-elevation, native grassland cover type; shrub-steppe: 0.5
- High-elevation, native grassland: Same as current condition
- Stand initiation size/structure within ponderosa pine or Douglas-fir cover types: 0.5
- Grazing: None
- Human disturbance
- Building density: Class zero
- Roads and trails: Class low

The relative sensitivity of WI values to variables in the model for golden eagles is shown in table 28.

Table 28—Relative sensitivity of watershed index values to variables in the model for golden eagles

Variable	Sensitivity rank
Cliff nesting habitat	1
Shrub and grass departure	2
High-elevation grassland	3
Grazing impact	3
Roads and trails	4
Building density	4
Forest nesting habitat	5
Stand initiation departure	6

Model evaluation—

We used 296 documented occurrence points for golden eagles compared to 296 random points to evaluate the surrogate species assessment model for this species. The mean WI value for the occurrence points (1.251) was significantly higher (t = -8.827, P = <0.001) than for the random points (0.905), indicating that our model identified habitat conditions favorable to the occurrence of golden eagles.

Assessment Results

Watershed scores—

Historically, 69 of 72 watersheds within the assessment area provided an adequate combination of nesting and foraging habitat for golden eagles. Currently, 86 percent (n = 62) of the watersheds contain habitat and were within the present extent of the range of golden eagles in the northeast Washington assessment area (Smith et al. 1997) (fig. 31). Six percent (n = 4) of the 72 watersheds did not contain any cliff-nesting source habitat, 47 percent (n = 34) of the watersheds provided low to moderate amounts, and 47 percent (n = 34) of the watersheds provided a high amount. Watersheds with a high amount of cliff nesting habitat were generally located in the northeast, northwest, and central portions of the assessment area. Thirty-six percent (n = 26) of the watersheds were at or above the median amount of forested nesting habitat calculated across all watersheds; 65 percent (n = 46) of the watersheds were below the median amount (fig. 31). Watersheds with high amounts of forested nesting habitat were primarily located in the northwest and central portions of the assessment area.

Currently, 53 percent (n = 33) of the watersheds with habitat had moderate WI scores (>1.0 to <2.0) (fig. 31). These watersheds were concentrated in the southwestern portion of the assessment area. Forty-seven percent (n = 29) of the watersheds

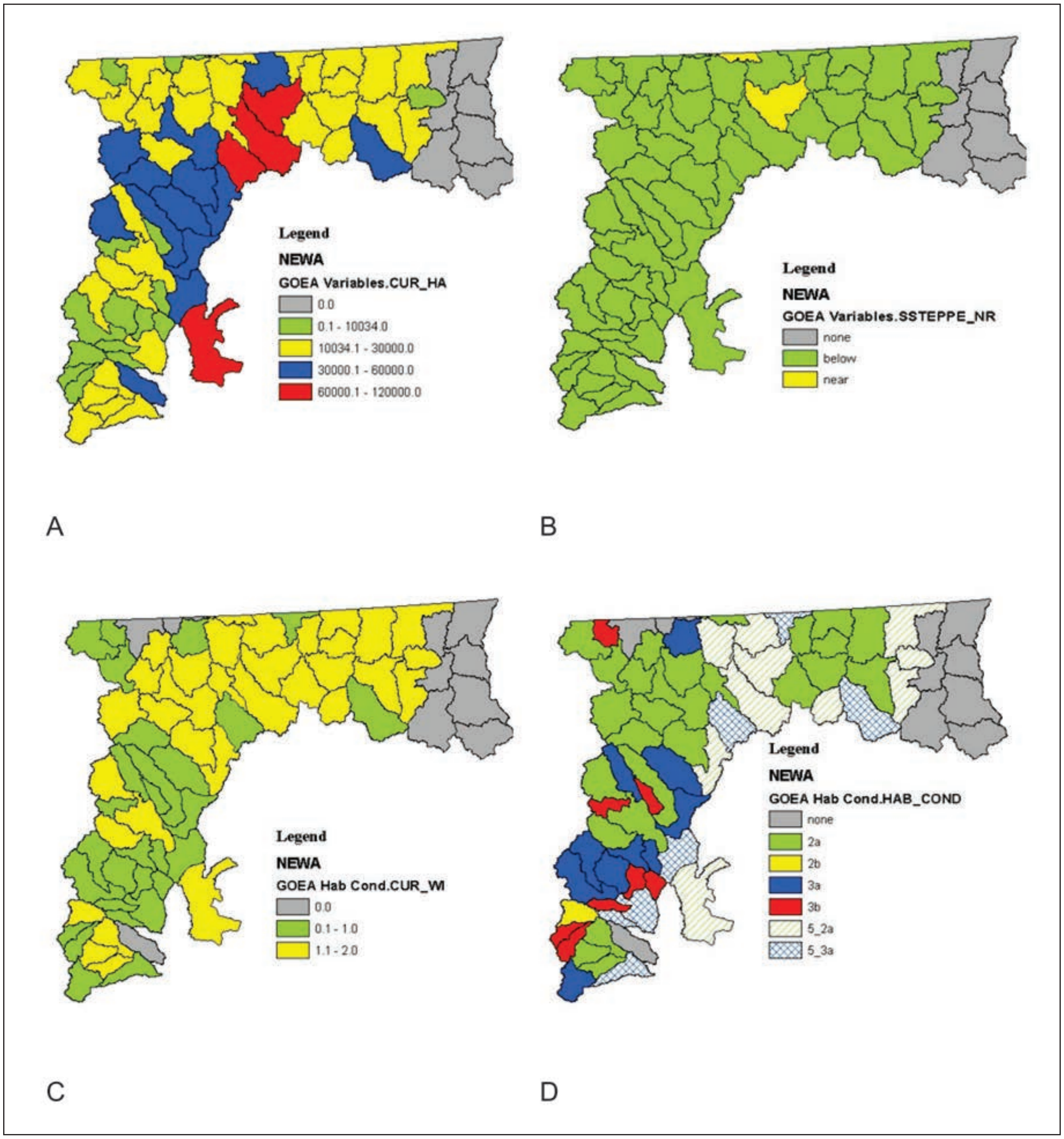

Figure 31—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D) habitat condition class for golden eagles (GOEA) by watershed in the northeast Washington assessment area (NEWA).

had low WI scores (>0.0 to ≤1.0). These were generally distributed across the northern and central portions of the assessment area.

Watersheds with the least loss of low-elevation grassland and shrub-steppe foraging habitat were located in the central portion of the assessment area (fig. 31). Foraging habitat associated with forest stand initiation was limited but occurred throughout the western portion of the assessment area. Grassland foraging habitat at high elevation was located across the western and northeastern portions of the assessment area.

Factors that influenced the WI scores included the amount of nesting source habitat (i.e., cliff and forest) compared to the median amount across watersheds and foraging source habitat (i.e., low-elevation grassland and shrub-steppe, forest initiation following timber harvest and fire, high-elevation grassland) compared to levels historically available in the watersheds. The effect of grazing on foraging habitat was assessed by the amount of habitat in an active grazing allotment. Watersheds with >25 percent of source habitats for golden eagles in an active grazing allotment (34 percent, n = 21) were concentrated in the central portion of the assessment area.

Twenty-seven percent (n = 17) of the watersheds in the assessment area had low influence from open roads on golden eagle source habitat (i.e., >50 percent of the source habitat was outside a 1,640-ft buffer on roads and trails). These watersheds were primarily located in the northwest portion of the assessment area. Across the assessment area, a large majority of the watersheds (79 percent, n = 49) had a low density of buildings in >50 percent of the source habitat. Watersheds with higher building densities primarily occurred in the central part of the assessment area (fig. 31).

Viability outcome scores—

The VOI model incorporated the WWI scores (described earlier) and a habitat distribution index. The WWI scores indicated that the current habitat capability for golden eagles within the assessment area was 67 percent of the historical capability. Dispersal across the assessment area was not considered an issue for this species. All ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Seventy-four percent (n = 53) of watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 32.0 percent probability that the current viability outcome for golden eagles is A and 56 percent probability of outcome B, indicating habitats are broadly distributed and of high abundance, but there are gaps where suitable environments are absent or only present in low abundance (fig. 32). Likely, other species associated with the woodland/grass/shrub group in the woodland/grass/shrub family have similar outcomes.

Historically, dispersal across the assessment area was not considered an issue for this species. All ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across

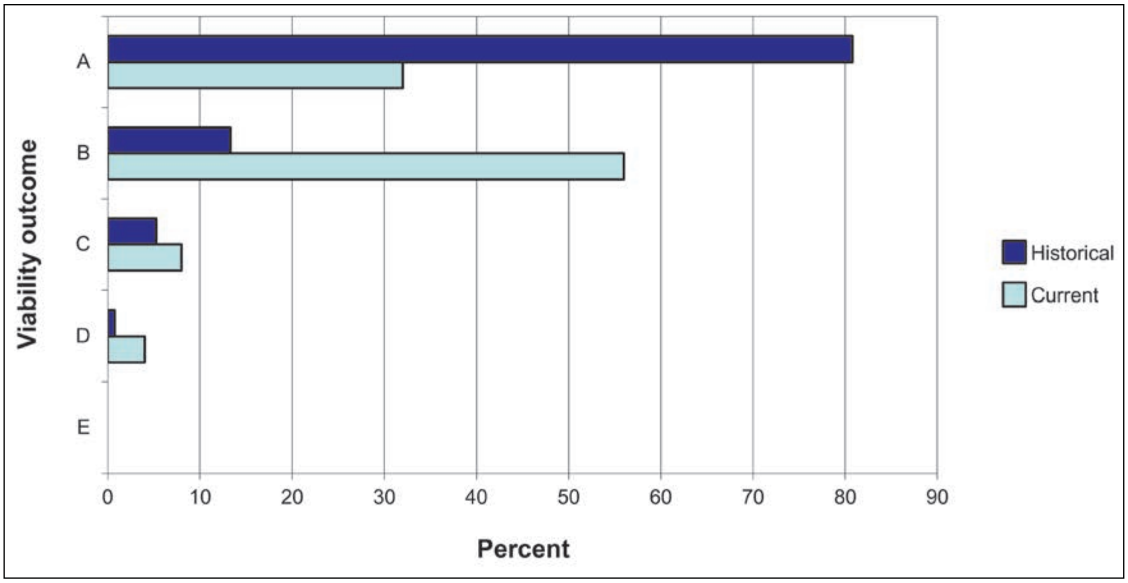

Figure 32—Current and historical viability outcomes for golden eagles in the northeast Washington assessment area.

all watersheds with source habitat). Eighty-six percent (n = 62) of watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was an 80.8 percent probability that the historical viability outcome for golden eagle was A, indicating habitats were broadly distributed and highly abundant (fig. 32).

In summary, under historical conditions, golden eagles and other species associated with the woodland/grass/shrub group in the woodland/grass/shrub family were likely well distributed throughout the assessment area. Currently, although they are likely well distributed throughout most of the assessment area, their distribution has been somewhat reduced from historical conditions.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Reduction and fragmentation of foraging source habitat.
2. Effects of grazing on foraging source habitat.
3. Negative effects of roads and building use in nesting source habitats.
4. The sustainability of dry forests as nesting source habitat for golden eagles (Townsley et al. 2004).

Harlequin Duck

Introduction

Harlequin ducks were selected as a surrogate species to represent the forested riparian group, specifically at mid-low elevations. Harlequin ducks breed and use summer habitats in mountain streams on the east and west side of the Cascade Range, in the Selkirk Mountains in northeastern Washington, and in the Blue Mountains (Jewett et al. 1953, Schirato 1994). Their presence in the Blue Mountains is now in question (Schirato 1994); however, in northeastern Washington, they are still a good surrogate species for this group because of their association with smaller mid-elevation streams and because human disturbance is a risk factor. Other species in this group, including several duck species, have similar habitat associations and risk factors.

Model Description

Source habitat—

Breeding habitat for Harlequin ducks occurs on streams as reaches with average gradients between 1 and 7 percent, with some areas of shallow water (riffles), clear water, rocky, gravel to boulder-size substrate, and forested bank vegetation (Bengtson and Ulfstrand 1971, Lewis and Kraege 1999).

Streams usually have substrate that ranges from cobble to boulder, with adjacent vegetated banks. Harlequins often nest on the ground (Bengtson 1972); however, cavities in trees and cliff faces also provide nest sites (Cassier and Groves 1989, Cassier et al. 1993). Midstream loafing sites are an important part of suitable habitat (Cassier and Groves 1989, Cassier et al. 1993). Broods remain near nesting areas for the first few weeks after hatching, then move downstream during the summer (Cassier and Groves 1989, Cassier et al. 1993, Kuchel 1977, Wallen and Groves 1989). Broods prefer low-gradient streams with adequate macro invertebrate food sources (Bengtson and Ulfstrand 1971). Aquatic insect larvae make up the bulk of their diet during the breeding season (Cassier and Groves 1989, Cassier et al. 1993).

We modeled source habitat for harlequin ducks using a stream-order layer and a 330-ft distance buffer from stream orders 6 and 7 (fig. 33).

Late-successional habitat—

Cassier and Groves (1989) found that harlequins preferred to nest in areas where mature and old-growth forests occurred adjacent to suitable streams. Therefore, we used the amount of late-successional forest within source habitat as a variable to describe habitat quality. We mapped late-successional forest using the following GIS data layers (fig. 33):

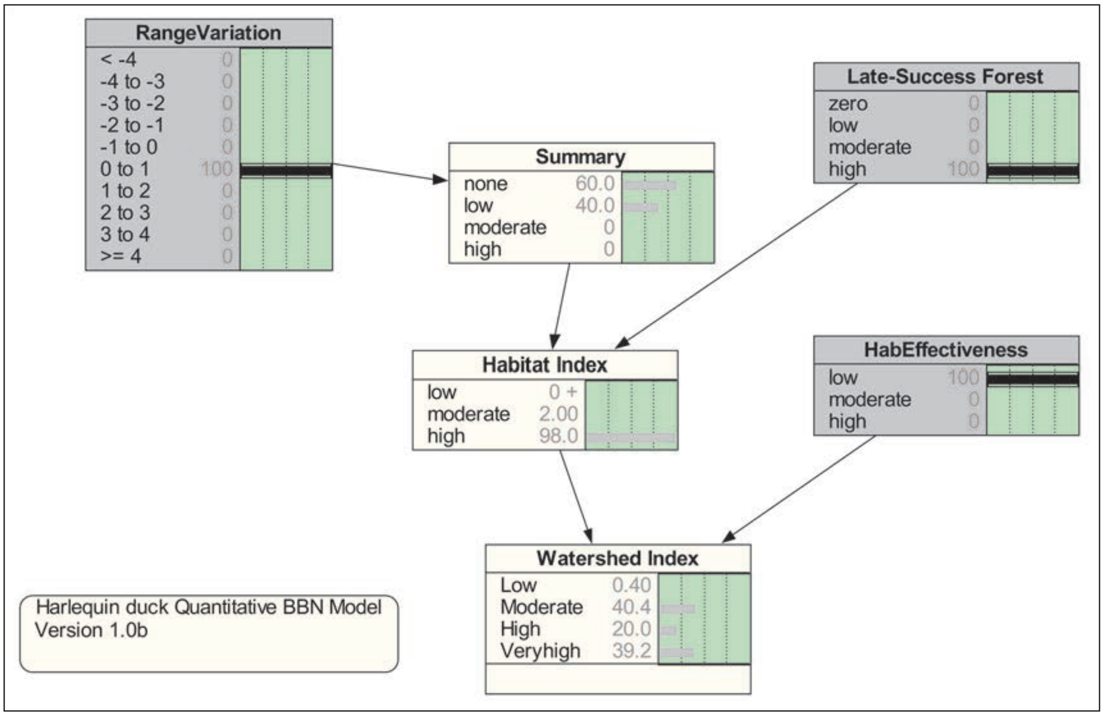

Figure 33—Surrogate species assessment model for harlequin ducks.

- Tree structure and size: Single/multilayer, >15 in QMD
- Canopy closure: >50 percent

We then intersected the source habitat and late-successional forest layers and used the following categories to assess the influence of late-successional forest on harlequin duck source habitat within each watershed (fig. 33):

- Zero: No source habitat composed of late-successional forest
- Low: >0 to 20 percent of the source habitat composed of late-successional forest
- Moderate: >20 to 50 percent of the source habitat composed of late-successional forest
- High: >50 percent of the source habitat composed of late-successional forest

Habitat effectiveness—
Studies have shown that harlequin ducks are sensitive to human disturbances during the breeding season (Cassier and Groves 1989, Wallen and Groves 1989). Ashley (1994) found that harlequin ducks use stream habitats inaccessible to humans more than expected. Wallen and Groves (1989) reported that fishing along trails seemed more disruptive to harlequin ducks than hiking. Harlequins avoided humans on

the bank or in the streambed and would typically swim or dive downstream past people, remaining partially submerged and watchful while moving out of the area. Fishing also can directly affect harlequin ducks as birds have been found entangled in fishing line (Ashley 1994, Clarkson 1992). Cassier and Groves (1989) recommended that trails and roads be located at least 160 ft from streams used by harlequin ducks.

To evaluate the potential effects of human activities on harlequin duck source habitat, we used the harlequin duck nesting habitat disturbance index in Gaines et al. (2003a) in which roads and trails are buffered by 160 ft on each side. We then intersected this data layer with our source habitat map and developed the following categories to assess habitat effectiveness within each watershed (fig. 33):

- Low habitat effectiveness: >50 percent of the source habitat in a zone of influence of a road or trail
- Moderate habitat effectiveness: 30 to 50 percent of the source habitat in a zone of influence of a road or trail
- High habitat effectiveness: <30 percent of the source habitat in a zone of influence of a road or trail

Calculation of historical conditions—

- Departure of source habitat: Departure class 1
- Late-successional forest: Moderate
- Habitat effectiveness: High

The relative sensitivity of watershed values to variables used in the model for harlequin ducks is shown in table 29.

Table 29—Relative sensitivity of watershed index values to variables in the model for harlequin ducks

Model variables	Order of variable weighting
Source habitat	1
Late-successional forest	2
Habitat effectiveness	3

Assessment Results

Watershed scores—

Twenty-seven (45 percent) of the watersheds had high WI scores (>2.0), and 31 (52 percent) had moderate scores (1.0 to 2.0). The WI variables for those watersheds with a WI of <2.0 indicate that the quality of habitat is likely affected by a low amount of late-successional habitat, or habitat effectiveness may be low.

Watersheds with the most source habitat (>2,500 ac) included North Lake Roosevelt, Lower Tieton, Boulder/Deadman, Wenas Creek, Naches River, Upper Pend Orielle, Stensgar/Stranger, Upper Yakima, Upper Columbia-Swamp Creek, Middle Methow, Cowiche, and the Wenatchee Rivers. The median amount of habitat across all watersheds with at least some source habitat (59 watersheds) was 1,600 ac. Six watersheds had <250 ac of source habitat (Bumping River, Ruby Creek, Upper Tieton River, Mad River, Columbia Tribs). Watersheds that have a high proportion of source habitat on federal lands include Boulder/Deadman, Chiwawa, Cle Elum, Icicle, Little Naches, Lower Chewuch Lower Pend Oreille, Lower Tieton, Naches, Nason, West Fork San Poil, and White-Little Wenatchee.

The proportion of the source habitat that was in late-successional forest, an indicator of habitat quality, was low overall. Fifty-three percent of the watersheds (n = 31) had no late-successional habitat within the source habitat, 44 percent (n = 26) had a low level (>0 to 20 percent) of late-successional forest in source habitat, 3 percent (n = 2) had a moderate (>20 to 50 percent) level of late-successional forest in source habitat, and no watersheds had a high level (>50 percent).

Fifty-nine percent of the watersheds (n = 35) had a low level of human disturbance (<30 percent of the source habitat in a disturbance buffer), 37 percent (n = 22) had a moderate level of human disturbance (30 to 50 percent of the source habitat in a disturbance buffer), and 3 percent (n = 2) had a high level of human disturbance (>50 percent of the source habitat in the disturbance buffer).

Viability outcome scores—

The VOI model incorporated the WWI scores (described earlier) and a habitat distribution index. The WWI scores indicated that the current habitat capability for harlequin ducks across the assessment area was 68 percent of the historical capability. This score was largely influenced by the amount of late-successional forest that is in the source habitat and the level of human disturbance that is currently

A reduction in the availability of suitable environments for harlequin ducks may have occurred in the assessment area compared to the historical distribution and condition of their habitats, but their source habitats are still relatively widely distributed.

occurring within the source habitat. Many of the roads, trails, and recreation facilities occur within the valley bottoms and adjacent to harlequin duck habitat.

Forty percent of the historical median amount of source habitat across all watersheds with at least some source habitat was 657 ac. Historically and currently, 49 of the watersheds (83 percent) in the assessment area met this habitat minimum. The watersheds with >40 percent were distributed across all of the five ecoregions.

The VOI for the assessment area had a 34 percent probability of outcome A and a 57 percent probability of outcome B (fig. 34), which indicates that suitable environments for the harlequin duck are broadly distributed and of relatively high abundance, but there are gaps where suitable environments are absent or only present in low abundance. These gaps are typically not large enough to prevent species from interacting as a metapopulation. Historically, there was an 80.8 percent probability of outcome A and a 13.3 percent probability of outcome B where suitable environments were more broadly distributed or of high abundance (fig. 34). In addition, the suitable environments were better connected, allowing for interspecific interactions. A reduction in the availability of suitable environments for harlequin ducks may have occurred in the assessment area compared to the historical distribution and condition of their habitats, but their source habitats are still relatively widely

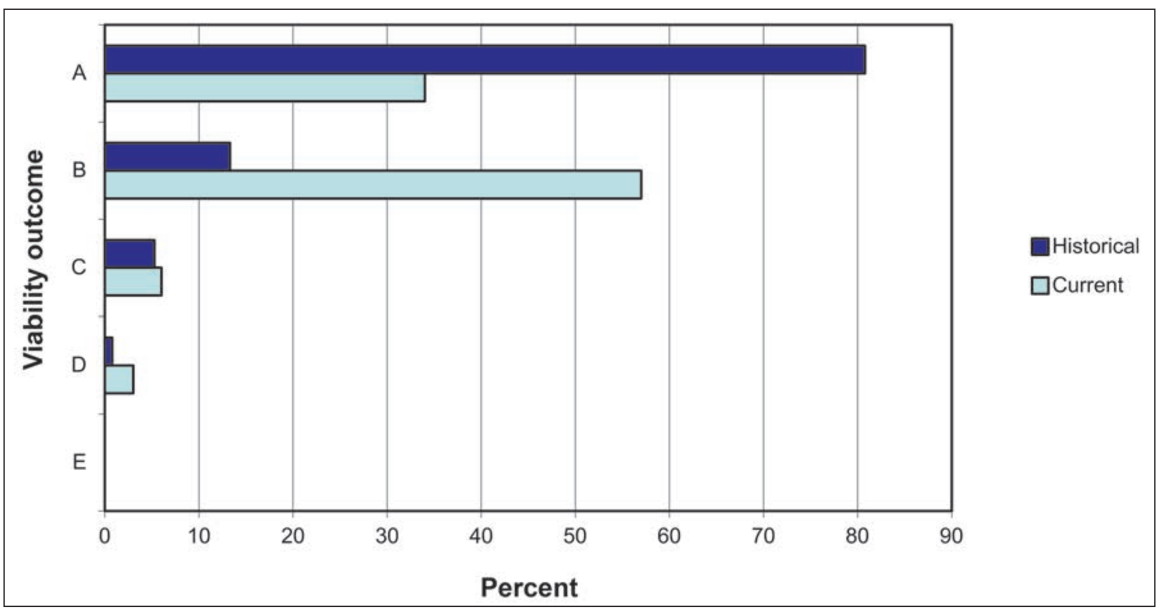

Figure 34—Current and historical viability outcomes for the harlequin duck in the northeast Washington assessment area.

distributed across the assessment area. Similar outcomes are expected for other species associated with forested riparian habitats.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. The amount of late-successional forest adjacent to streams that provide source habitat for harlequin ducks is low in many watersheds.
2. The level of human activities that occurred within harlequin duck source habitat reduced the effectiveness of their habitat, especially because many of the roads, trails, and recreation facilities occur within the valley bottoms and thus are adjacent to harlequin duck habitat.

Larch Mountain Salamander

Introduction

The larch mountain salamander is a surrogate species for the cool-moist forest with medium to large trees group. In addition, this species is closely associated with talus, a fine-scale habitat feature. The distribution of the larch mountain salamander in Washington is disjunct, with most known sites located in southern Washington, north of the Columbia River Gorge. However, two isolated populations have been found near Snoqualmie Pass (Crisafulli 1999, Nordstrom and Milner 1997). Within the assessment area, they are currently known to occur only in the Upper Yakima River watershed.

Because of the limited distribution of this species within the assessment area, and its unique habitat that we did not have spatial data to evaluate, a surrogate species assessment model was not developed for this species; rather a qualitative assessment of its habitat relationships and general management considerations were completed.

Source habitat—

Larch mountain salamanders depend on the availability of undisturbed, shaded talus slopes with stable, moist microclimates (Herrington and Larsen 1985, Nordstrom and Milner 1997). In addition, they have been discovered in moist forests that possess late-seral features such as complex stand structure and moderate to high levels of woody debris (Crisafulli 1999).

Risk factors—

The risk factors that were identified for the larch mountain salamander include the effects of road construction, timber harvest, and high-intensity fire on key habitat elements (talus, woody debris) and on changes to microclimate conditions (Crisafulli 1999, Herrington and Larsen 1985, Nordstrom and Milner 1997).

Management Considerations

The following issues were identified for consideration by managers:

1. Predisturbance surveys may be implemented where suitable habitat conditions occur following standardized protocols (Crisafulli 1999).
2. Microclimates and habitat features such as woody debris and talus can be influenced by management activities.

Lark Sparrow

Introduction

Lark sparrows were chosen as a surrogate species to represent species of conservation concern in the grassland group of the woodland/grass/shrub family. Lark sparrows and other species in the grassland group are of conservation concern because grassland habitats throughout the United States are being lost to woody plant invasion and development (Grant et al. 2004). Lark sparrows were associated with dry, open grasslands and respond positively to well-managed grazing of domestic livestock, although they are highly susceptible to nest parasitism by brown-headed cowbirds (*Molothrus ater*). They have a distinctive song, making this species easy to survey and monitor. Lark sparrows range across the central portion of the assessment area and in part of the eastern portion (Smith et al. 1997). Lark sparrows are breeding-season residents of the assessment area (Martin and Parrish 2000); this assessment is for nesting and rearing habitat.

Model Description

Source habitat—
Lark sparrow habitat included shrub-steppe, and mixed-grass and shortgrass uplands with a shrub component and sparse litter (Bock et al. 1995, Walcheck 1970, Wiens and Rotenberry 1981). Martin and Parrish (2000) reported that lark sparrows prefer structurally open herbaceous ground cover containing scattered trees or shrubs with <24 percent canopy cover. In northeastern Colorado, lark sparrows were found in grazed prairies with widely spaced cottonwoods (Fitzgerald 1978, Jacobson 1972). In piñon-juniper woodlands, lark sparrow abundance increased with decreasing tree density (Tazik 1991). Studies in the Eastern United States indicated that habitat patches with >15 percent tree cover were avoided by nesting lark sparrows (Coulter 2008). Also, lark sparrows were significantly more abundant in

native-grass-dominated areas than in areas dominated by exotic grasses (Flanders et al. 2006). Lark sparrow abundance has been reported to be negatively correlated with sagebrush density (McAdoo et al. 1989). Lark sparrow habitat in Arizona had mean values of 38 percent bare ground, 54 percent grass cover, 7 percent forb cover, <2 percent canopy cover, 5-in grass height, and 0.73 shrubs/ft² (Bock and Webb 1984). For this analysis, source habitat was defined as structurally open habitats with grass or herbaceous ground cover with scattered shrubs or trees (fig. 35).

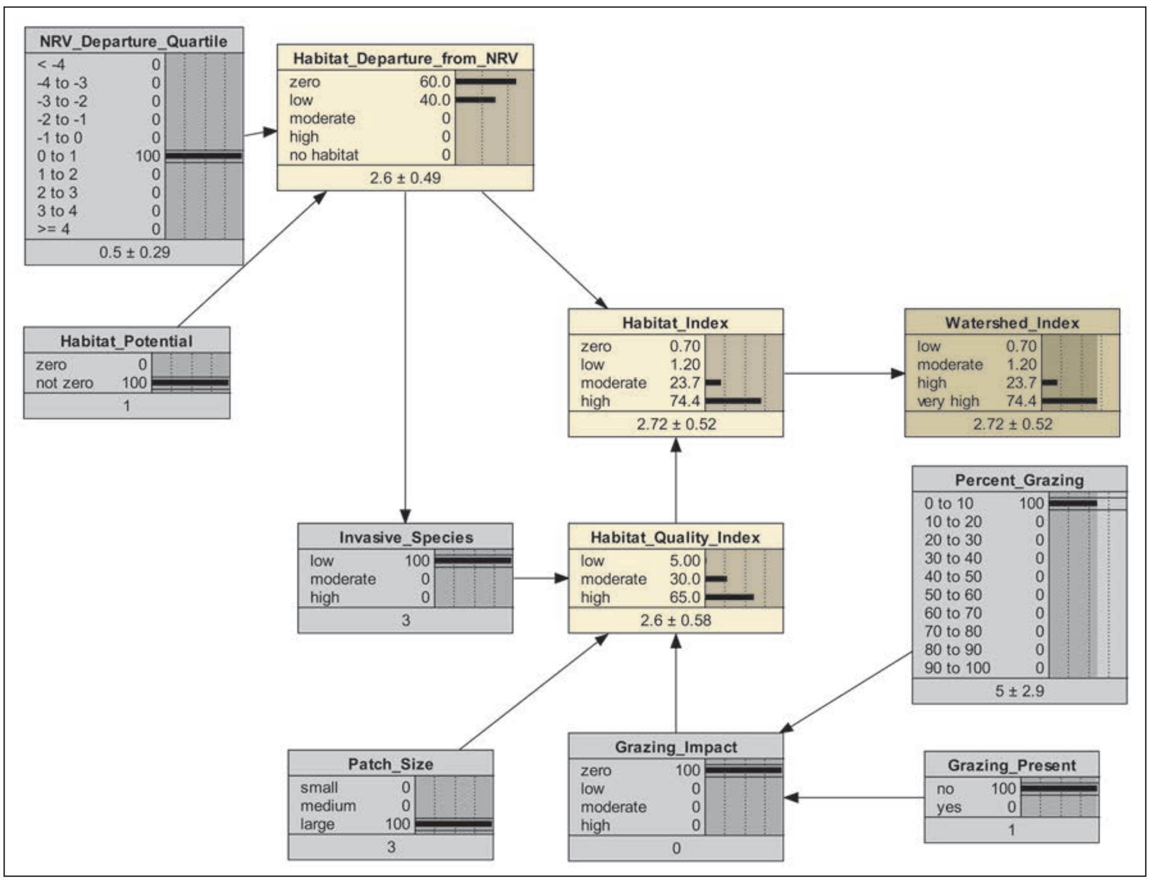

Figure 35—Surrogate species assessment model for lark sparrows.

Invasive animals—

Lark sparrows were vulnerable to parasitism by brown-headed cowbirds (Hill 1976, Newman 1970, Shaffer et al. 2003). Proximity to agricultural areas increased the potential of parasitism (Goguen and Mathews 1999, 2000; Tewksbury et al. 1999; Young and Hutto 1999). The following classes were used to estimate the potential effect of brown-headed cowbirds on lark sparrows (fig. 35):

- Low: <30 percent of source habitat within 1.2 mi of agricultural areas within a watershed
- Moderate: 30 to 50 percent of source habitat within 1.2 mi of agricultural areas within a watershed
- High: >50 percent of source habitat within 1.2 mi of agricultural areas within a watershed

Patch size—

In the core of their range, lark sparrows often inhabit large, unbroken prairies or fields (Martin and Parrish 2000). At the landscape scale, lark sparrows used large habitat patches with low edge to interior ratios (Coulter 2008). Proximity of habitat patches and amount of edge were reported to be important predictors of grassland bird richness (including lark sparrows) (Hamer et al. 2006). Lark sparrows were more frequently found in interior survey plots >650 ft from an edge in a habitat patch than in survey plots closer to an edge (Bock et al. 1999). They were edge sensitive with reduced abundance near edges (Bolger et al. 1997). This suggests that patches increasingly >32 ac provide progressively better habitat. They also exhibited a negative response to urban development (Jones and Bock 2002). Lark sparrows were strongly negatively affected by habitat fragmentation and preferred patches >250 ac (Bolger 2002). Occurrence of grassland species may be negatively affected by larger amounts of edge because of increased risk of predation and brood parasitism near wooded edges (Johnson and Temple 1990, Winter et al. 2000). The following classes were used to estimate the potential effect of patch size on lark sparrows (fig. 35):

- Small: <50 ac mean size for source habitat patches within a watershed
- Medium: 50 to 250 ac mean size for source habitat patches within a watershed
- Large: >250 ac mean size for source habitat patches within a watershed

Grazing—

Results reported in the literature on the effects of grazing on lark sparrows were unequivocal. Numerous sources reported a positive response from lark sparrows associated with livestock grazing (Bock and Webb 1984, Bock and Bock 1988, Bock et al. 1984, Lusk et al. 2003, Martin and Parrish 2000). However, timing and intensity of grazing may affect the magnitude of the response of lark sparrows (Goguen and Mathews 1998).

Impact of grazing on source habitat within a watershed was based on the percentage of source habitat with an active grazing allotment. The amount of source habitat in an active grazing allotment was categorized using 10 percent increments from 0 to 100 percent, with increasing habitat outcomes as the proportion of source habitat in an active allotment increased (fig. 35).

Calculation of historical conditions—

Values of the model variables were set with the following values to estimate historical habitat conditions:

- Departure of source habitat from HRV: 0.5
- Invasive animals: Class low
- Patch size: Class large
- Grazing: None

The relative sensitivity of WI values to variables used in the model for lark sparrow are shown in table 30.

Assessment Results

Watershed scores—

Historically, 71 of 72 watersheds within the assessment area provided habitat for lark sparrows (i.e., >50 ac of habitat within the watershed). However, most of those watersheds either no longer support habitat for lark sparrows or historical amounts

Table 30—Relative sensitivity of watershed index values to variables in the model for lark sparrow

Variable	Sensitivity rank
Habitat departure	1
Patch size	2
Grazing impact	3
Invasive species	4

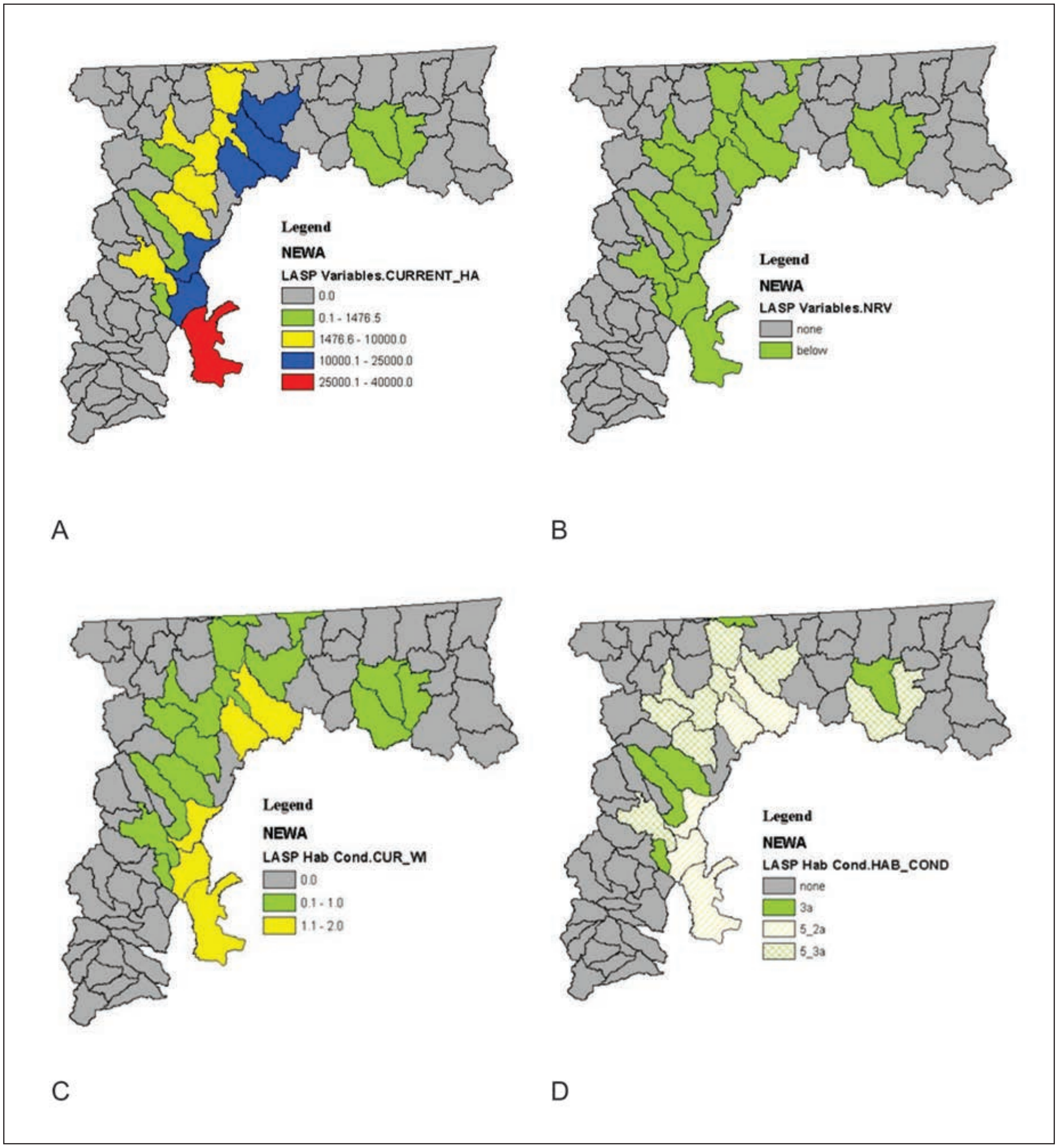

Figure 36—Current amount of (A) source habitat (acres), (B) current habitat depature, (C) watershed index, and (D) habitat condition class for lark sparrow (LASP) by watershed in the northeast Washington assessment area (NEWA).

have been significantly reduced (fig. 36). Forty-five percent (n = 32) of watersheds that had habitat historically for lark sparrows no longer have habitat. However, those watersheds historically had minimal amounts that were likely highly fragmented. Watersheds with habitat remaining were concentrated in the central and south-central portions of the planning area. Seven percent (n = 5) of those watersheds supported >2,470 ac of source habitat (fig. 36) and had WI scores that were moderate or higher (≥1.0) (fig. 36). The remaining 27 percent (n = 19) of watersheds that were within the current range of lark sparrows had low WI scores (>0.0 and <1.0).

A major factor that influenced the WI scores was the amount of source habitat compared to levels historically available in the watersheds. Watersheds that currently had the greatest amount of source habitat included Columbia River–Lynch Coulee, Columbia Tributaries, Lake Entiat, Lower Okanogan River; and Okanogan River–Bonaparte Creek, and Okanogan River–Omak Creek (fig. 36). However, none of those watersheds had >25 percent of the source habitat managed by federal agencies. The watersheds with the least amount of source habitat were located across the eastern and western portions of the assessment area.

Lark sparrows were strongly negatively affected by habitat fragmentation and preferred patches >250 ac (Bolger 2002). However, essentially all remaining source habitat for lark sparrows has been highly fragmented. Lark sparrows are also vulnerable to brood parasitism by brown-headed cowbirds (Newman 1970). Proximity to agricultural areas increases the potential of parasitism (Goguen and Mathews 2000). Basically, all remaining source habitat for lark sparrows was at high risk to brood parasitism. Grazing by livestock may have a positive effect on lark sparrows and their habitat depending on the intensity and season of grazing (Bock and Webb 1984). The percentage of source habitat by watershed that was grazed was generally low to moderate across the assessment area (fig. 36).

Viability outcome scores—

The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for lark sparrows within the assessment area was 44 percent of the historical capability. Dispersal across the assessment area was not considered an issue for this species. Four of five ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Eighteen percent (n = 13) of watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there is a 45 percent probability that the current viability outcome for lark sparrows is C, and a 52 percent probability that the current viability outcome for lark sparrows is D, indicating that habitat was patchily distributed or isolated and in low abundance (fig. 37). It is likely that other species associated with the grassland group of the woodland/grass/shrub family had similar outcomes.

Historically, dispersal across the assessment area was not considered an issue for this species. Four of five ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Sixty-two percent (n = 45) of watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 71.2 percent probability that the historical viability outcome for lark sparrows was A, and a 18.8 percent probability that the historical viability outcome for these species was B, indicating habitat was broadly distributed with a high abundance of quality habitat (fig. 37).

In summary, under historical conditions, lark sparrows and other species associated with the grassland group of the woodland/grass/shrub family were likely numerous and well distributed throughout the assessment area. However, under current conditions, populations of these species are likely not well distributed across the assessment area with substantial potential for extirpation of individual populations.

Our results for this species were similar to those reported in the broad-scale habitat analysis by Wisdom et al. (2000) in ICBEMP. According to the ICBEMP, terrestrial vertebrate habitat analyses, historical source habitats for lark sparrows included portions of the Northern Cascades and the Northern Glaciated Mountains ERUs that overlap our assessment area (Wisdom et al. 2000). Within this historical habitat, declines in source habitats have been extensive, -61 percent in the Northern Cascades and -84 percent in the Northern Glaciated Mountains according to Wisdom et al. (2000).

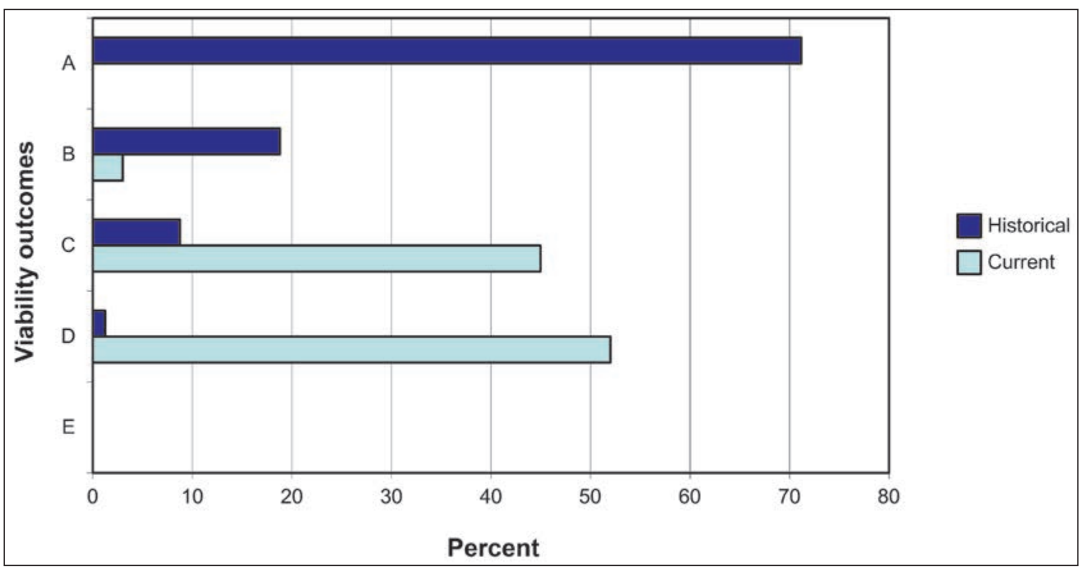

Figure 37—Current and historical viability outcomes for lark sparrows in the northeast Washington assessment area.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Reduction and fragmentation of suitable grassland source habitats.
2. Negative effects of agricultural practices adjacent to source habitats that promote nest parasitism by brown-headed cowbirds.

Lewis's Woodpecker

Introduction

Lewis's woodpecker was chosen as a surrogate species for the postfire group to represent postfire habitat with lower densities of large snags and trees present as compared to other species in the group that prefer postfire habitat with a high density of fire-killed trees. This species was selected as a surrogate species because it is closely tied to postfire habitats, is widespread across the Western United States, and occurs in suitable habitat across the assessment area. This woodpecker is also associated with unburned ponderosa pine forests with open canopies and large trees as well as cottonwood/willow habitat. However, it generally is at lower abundance in these habitats than in postfire habitat.

Model Description

Source habitat—

Lewis's woodpeckers breed in wooded areas with an open canopy, often with a dense shrub cover, and generally avoid dense forest. Three main habitats used throughout its range are burned or logged areas, open ponderosa pine savanna at high elevations, and riparian woodland dominated by large cottonwoods at low elevations (Abele et al. 2004, Bock 1970, Saab and Dudley 1998, Saab and Vierling 2001, Tobalske 1997). Suitability of burned areas as habitat for Lewis's woodpeckers may vary with size of burn, time since burn, intensity of burn, and geographic region (Russell et al. 2007, Saab and Dudley 1998, Saab and Vierling 2001, Tobalske 1997). Research by Russell et al. (2007) found that the best predictors of nest location for Lewis's woodpeckers after a wildfire in Idaho were burn severity, patch area, and snag diameter. In a Wyoming study, nests were preferentially located within or adjacent to burned ponderosa pine forests, and in sites with greater ground cover, more down logs, and greater amounts of open sky than random sites (Linder and Anderson 1998). Linder and Anderson (1998) found that use was declining in an area that burned 20 years earlier.

Optimal canopy closure for nest sites was ≤30 percent (Linder and Anderson 1998, Sousa and Farmer 1983). Some studies have suggested that Lewis's woodpeckers require a shrubby understory (Bock 1970, Sousa and Farmer 1983), while others have shown that preferred habitat included a relatively sparse shrub layer (<18 percent) (Block and Brennan 1987, Linder and Anderson 1998). In winter, this species occupies a variety of habitat types that offer proximity to mast, fruit, or corn. Typically, these are oak woodlands or orchards. In portions of the Southwest, this species may winter in areas without mast (Bock 1970).

Saab and Vierling (2001) found that some cottonwood riparian forests, primarily near agricultural development, may be acting as sink habitat. More research on the productivity of Lewis's woodpeckers in different habitat types is needed.

We identified both primary and secondary source habitat for the Lewis's woodpecker (fig. 38). Primary source habitat for this analysis was characterized as forested habitat in the dry potential vegetation types that was burned in the past 5 years (year >1999–2003) and was salvage harvested. We also included areas that were burned in the previous 5 to 15 years (1985–1999) regardless of salvage history but without any regeneration harvest. Secondary habitat was characterized as any forested areas in the dry potential vegetation type and Oregon white oak with a canopy closure <50 percent and tree d.b.h. >15 in QMD. We also included cottonwood/willow habitat that was primarily located in riparian areas as secondary source habitat. Cottonwood/willow habitats were mapped using the National Wetlands Inventory data.

We identified primary source habitat as:

- Potential vegetation types: Dry forests
- Postfire habitat 1999–2003, salvage harvested in all forested cover-types
- Postfire habitat 1985–1999 in all forested cover types

We identified secondary source habitat as:

- Potential vegetation types: Dry forests
- Cover types: Ponderosa pine, riparian and deciduous
- Tree size: ≥15 in QMD
- Canopy closure: >50 percent
- National Wetlands Inventory: Palustrine forested wetlands

Snag habitat—

Unlike other woodpeckers, Lewis's woodpecker is not morphologically well adapted to excavate cavities in hardwood (Spring 1965). Lewis's woodpeckers tend to nest in a natural cavity, reuse preexisting cavities, or excavate a new cavity in a soft

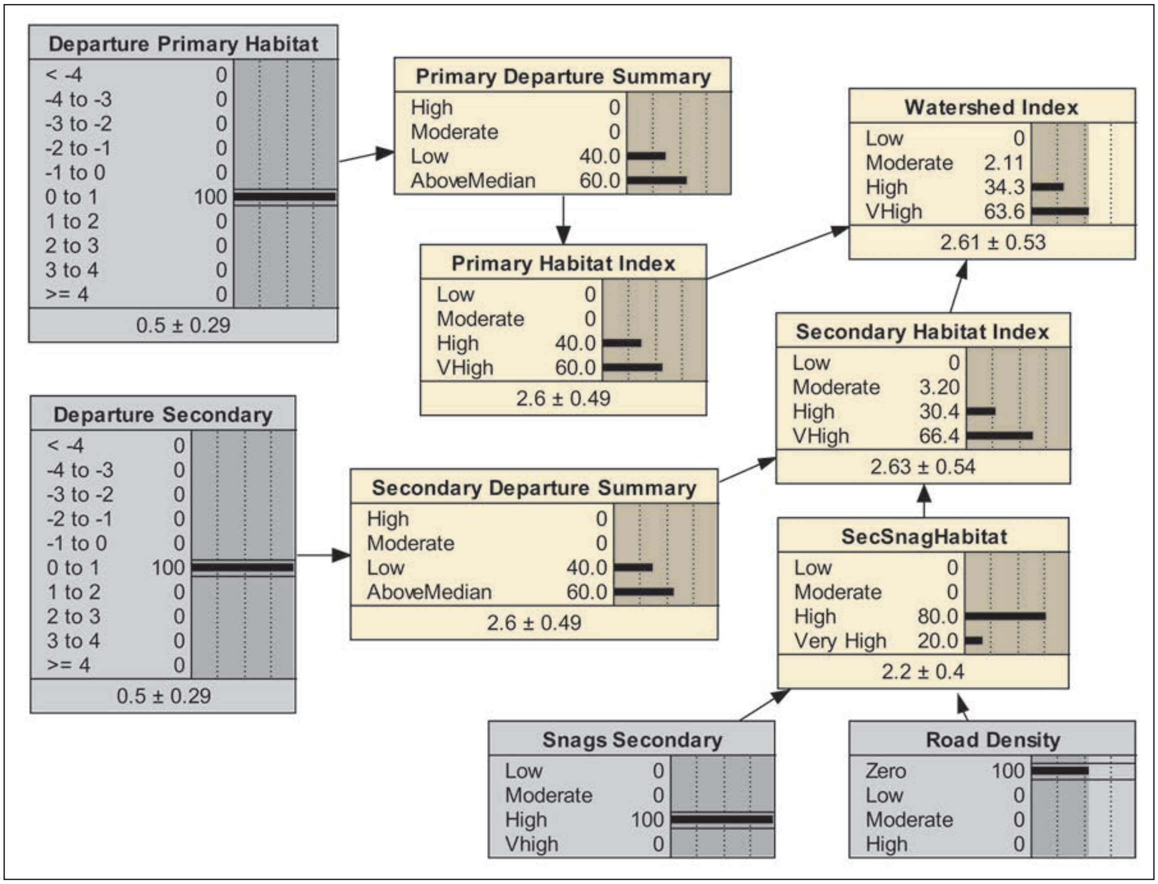

Figure 38—Surrogate species assessment model for Lewis's woodpecker.

snag (Harrison 1979, Raphael and White 1984, Saab and Dudley 1998, Tobalske 1997). Mated pairs may return to the same nest site in successive years. On partially logged burns with high nesting densities in Idaho, nest sites were characterized by the presence of large, soft snags and an average of 150 snags/acre, >9 in d.b.h. (Saab and Dudley 1998). Galen (1989) in eastern Oregon found that in unburned ponderosa pine/Oregon white oak habitat the mean d.b.h. of nest trees was 26 in with a range of 12.5 to 43 in. Haggard and Gaines (2001) in northeast Washington found Lewis's woodpeckers in postfire habitat were more abundant in areas with <12 snags/acre ≥10 in d.b.h. and were not found in areas with ≥91 snags/acre ≥10 in d.b.h.) following salvage logging of the burn. Saab et al. (2009) also found Lewis's woodpecker nest sites were primarily associated with partially logged burns.

In primary habitat (postfire), we assumed snag density was adequate for this species. In secondary habitat, we calculated the percentage of source habitat within each watershed that had densities of snags >20 in d.b.h. in the following classes based on data from Harrod et al. (1998): low ≤5/ac, moderate 7/ac, high 10/ac, and very high ≥12/ac (fig. 38).

Road density—

Bate et al. (2007) found that snag numbers were lower adjacent to roads owing to safety considerations, firewood cutters, and other management activities. Other literature has also indicated the potential for reduced snag abundance along roads (Gaines et al. 2003a, Wisdom and Bate 2008, Wisdom et al. 2000). To account for reduced snag density along roads, we calculated the percentage of forests in the dry potential vegetation types in the following road density classes by watershed (fig. 38):

- Zero: <0.1 mi/mi^2 open roads in watershed
- Low: 0.1 to 1.0 mi/mi^2 open roads in watershed
- Moderate: 1.1 to 2.0 mi/mi^2 open roads in watershed
- High: >2.0 mi/mi^2 open roads in watershed

Historical inputs for surrogate species assessment model—

- Departure of primary source habitat: Class 1
- Departure of secondary source habitat: Class 1
- Secondary habitat snag density: High
- Road density: Zero

The relative sensitivity of WI values to variables used in the model for Lewis's woodpecker are shown in table 31.

Assessment Results

Watershed scores—

Primary habitat was below the historical median in most watersheds (n = 65, 92 percent) (fig. 39). The remaining six watersheds contained most of the existing primary source habitat with all having >3,700 ac of source habitat (fig. 39). All of these watersheds have experienced recent wildfires. The amount of primary source habitat in the Entiat drainage (43,000 ac) makes up nearly 40 percent of the total amount of primary source habitat in the assessment area. Overall, nine watersheds currently have >2,300 ac of primary source habitat, the amount calculated as 40 percent of the historical median amount of source habitat. These watersheds are Curlew, Icicle Creek, Okanogan River-Bonaparte, Peshastin Creek, Wenatchee River, Mad River, Lower Lake Chelan, Lake Entiat, and Entiat River.

In addition to the large reduction in primary source habitat, the amount of secondary source habitat is also far below the historical median (fig. 39). The three watersheds that are near or above the historical median for amount of secondary habitat are the Okanogan River-Omak Creek, Upper Okanogan River, and Okanogan River-Bonaparte Creek.

Table 31—Relative sensitivity of watershed index values to variables in the model for Lewis's woodpecker

Variable	Sensitivity rank
Primary habitat departure	1
Secondary habitat departure	2
Snag density	3
Road density	4

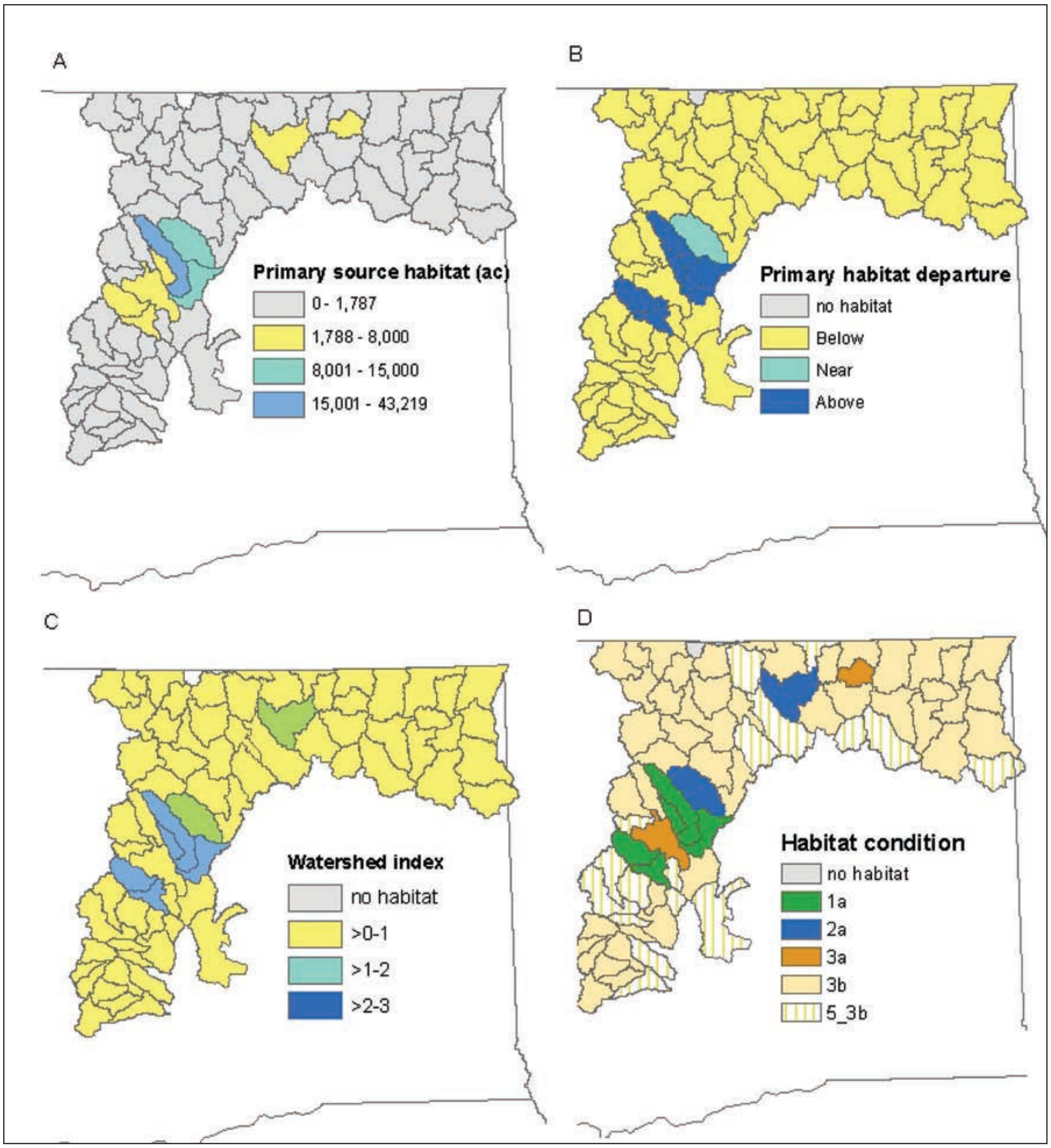

Figure 39—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watrshed index, and (D) habitat condition class for Lewis's woodpecker by watershed in the northeast Washington assessment area.

Of the 71 watersheds assessed in the assessment area, 90 percent (n = 64) had a WI of low current (<1.0) (fig. 39). Two watersheds were moderate (>1.0 and <2.0) and five watersheds had a high score (> = 2.0). The five watersheds in the high class have all experienced recent wildfire activity and include Entiat River, Peshastin Creek, Icicle Creek, Mad River, and Lake Entiat.

Although snag densities in secondary habitat were primarily in the low class (<5/ac), the low WI scores were primarily indicative of the overall low amount of both primary and secondary source habitats.

Our results for declining habitats for this species are similar to other broad-scale habitat analysis by Wisdom et al. (2000) in the ICBEMP. According to the ICBEMP terrestrial vertebrate habitat analyses, historical source habitats for Lewis's woodpeckers included only portions of the Northern Cascades and the Northern Glaciated Mountains ERUs (Wisdom et al. 2000). The analysis by Wisdom et al. (2000) would be the most equivalent of our analysis of the trend in secondary habitat. Within this historical habitat, declines in source habitats have been extensive—80 percent in the Northern Cascades and 95 percent in the Northern Glaciated Mountains (Wisdom et al. 2000). Within the entire interior Columbia Basin, there have been widespread declines in source habitats (83 percent)—the greatest of any species analyzed (Wisdom et al. 2000).

Viability outcome—
Currently, the viability outcome is a 60 percent C and a 40 percent D, indicating habitats are patchily distributed or isolated and are in low abundance. Historically, the outcome was primarily an A (76 percent), indicating habitats were broadly distributed and in high abundance (fig. 40)

The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for Lewis's woodpecker within the assessment area is 67 percent of the historical capability. This index is a bit misleading because current primary habitat in two watersheds is about five times as great as their historical median (Lake Entiat and Entiat River) and is largely the cause for this relatively high value currently. One-half of the current total WWI is a result of the sum of these two watersheds that have experienced recent wildfire activity.

The main factor leading to a lower current viability outcome compared to the estimated historical outcome was the reduction in percentage of watersheds with recent postfire habitat. Historically, 76 percent (n = 54) of the watersheds contained 40 percent of the median historical amount of postfire habitat, while currently, primary habitat is not well distributed across the assessment area, occurring in

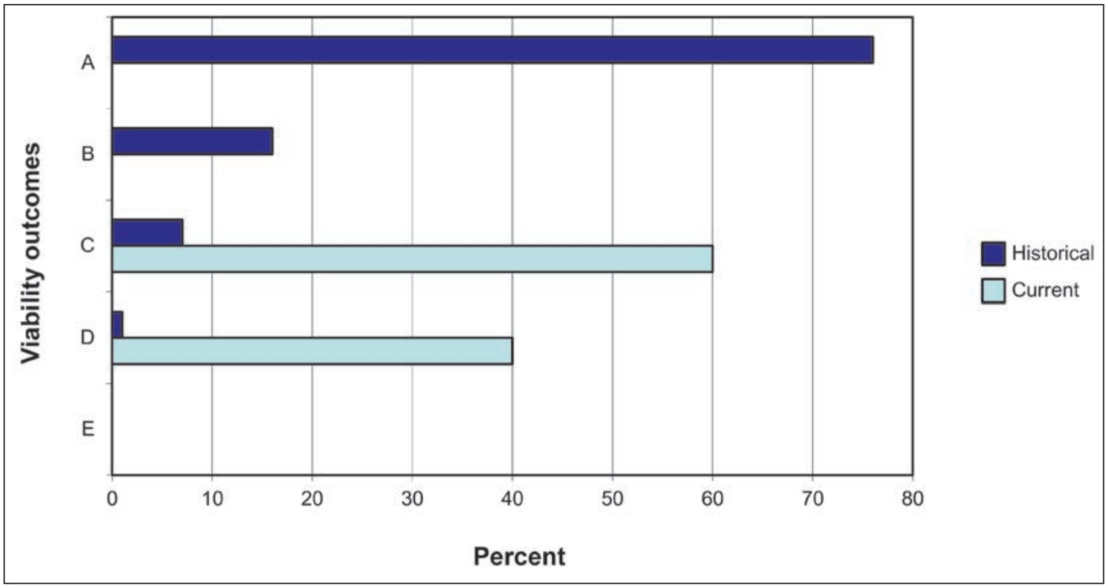

Figure 40—Current and historical viability outcomes for the Lewis's woodpecker in the northeast Washington assessment area.

this quantity in only 13 percent (n = 9). Three of five ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat).

Under historical conditions, the Lewis's woodpecker and other species associated with the postfire group were likely well distributed across the assessment area. Currently, we estimated that both the abundance and distribution of suitable environments for these species has declined and led to a decline in viability from a projected A outcome to a D outcome.

Management Considerations

The following issues were identified during this assessment and from the published literature regarding the Lewis's woodpecker, and other species associated with the postfire group for consideration by managers:

1. Timber harvest, firewood collection, and postfire salvage harvest may affect the availability of large snags used for nesting and foraging (Wisdom et al. 2000). There has been a decline in the amount of both primary (postfire) and secondary (large, open forest and cottonwood/willow) source habitat throughout the dry forests across the assessment area.
2. Wildfire intensity, postfire salvage logging and firewood collection influence the distribution of snags suitable for nesting and perching sites (Abele et al. 2005, Saab et al. 2009).

Fire-suppression efforts in eastern Washington dry forests have resulted in forest structure that is apparently not suitable as a breeding habitat owing to reductions in insect populations and limited space for foraging activity.

3. Alterations to water regimes have been shown to negatively affect cottonwood recruitment along many western streams and rivers (Johnson and Haight 1984).
4. Fire-suppression efforts in eastern Washington dry forests have resulted in stands with increased stem densities (often of more shade-tolerant species such as Douglas-fir and grand fir), reduced shrub and grass understories, and increased canopy closure (Morgan 1994). The resulting forest structure is apparently not suitable as a breeding habitat owing to reductions in insect populations and limited space for foraging activity (Abele et al. 2005).

MacGillivray's Warbler

Introduction

MacGillivray's warbler was selected as a surrogate species to represent shrubby-deciduous habitats within the deciduous riparian group. This warbler's distribution is large and widespread across the assessment area during the breeding season. The primary risk factor of grazing for this species applies to several other species in the group.

Model Description

Source habitat—

This species prefers canyons and draws, dense willows along streams, second-growth woodland habitat that can be created by fire or logging, including dead or fallen trees, brushy areas near low moist ground, and brushy dry hillsides not far from water (Terres 1980). It requires dense undergrowth and moderate cover for breeding (Morrison and Meslow 1983). Morrison (1981) described breeding habitat in coniferous or deciduous forests as having 74.2 and 60.1 percent total cover, composed of 63.8 and 44.8 percent shrubs, 3.7 and 7.7 percent coniferous species, and 6.7 and 7.6 percent deciduous species, respectively. In eastern Oregon, MacGillivray's warblers breed in dense willow thickets around springs and stream bottoms (Gabrielson and Jewett 1940). This warbler does not nest in sagebrush habitats (Gilligan et al. 1994). In the Cascade Range of Washington, Lehmkuhl et al. (2007) reported this species having a strong association with riparian habitats in dry forest types.

Source habitat is defined in this analysis as areas with a 330-ft buffer on perennial streams (i.e., stream orders 3 through 8) that have ≥70 percent shrub cover using gradient nearest neighbor (GNN) vegetation data set. In addition, we included meadow habitat from the cover-type map and palustrine, scrub-shrub (PSS) and palustrine forested wetlands (PFO) from the National Wetland Inventory (Cowardin et al. 1979).

Human developments (e.g., those from dams, diversions, agriculture conversion, stream channelization, road construction) have permanently altered millions of acres of wetland habitat. Based on these findings, we made a conservative estimate that source habitat for MacGillivray's warbler in the assessment area was about 70 percent of the historical amount. Applying these assumptions, we considered the current departure of wetland habitat (-30 percent) to be at the -2 class.

Grazing—

MacGillivray's warbler is a Neotropical migrant known to be negatively affected by livestock grazing. In three separate studies, this species was absent from heavily grazed or browsed areas but was found on nearby ungrazed or lightly grazed comparison plots (Berger et al. 2001, Medin and Clary 1991, Mosconi and Hutto 1982). The negative impact was considered to be a result of alteration of important vegetation structure and composition, as well as negative impacts on water quality or water regimes that affect vegetation (Zwartjes et al. 2005).

The presence of domestic grazing was used to assess the quantity and quality of shrub habitat considered important for MacGillivray's warbler. We categorized the amount of source habitat in an active grazing allotment using 10 percent increments from 0 to 100 percent, with increasing poorer shrub habitat as the proportion of source habitat in an active allotment increased (fig. 41).

Invasive species—

MacGillivray's warblers are reported to be occasionally parasitized by brown-headed cowbird (*Molothrus ater*), but extent and vulnerability are unknown (Pitocchelli 1995). Other research found that these warblers may be heavily parasitized by cowbirds in areas near agriculture, but have also been found breeding in smaller riparian areas far from agriculture (Tewksbury et al. 1999). Though breeding success in these areas has not been sufficiently studied, smaller deciduous riparian areas far from agriculture likely provide nesting sites free from cowbird parasitism (Tewksbury et al. 1999).

To assess the effects of nest parasitism by cowbirds, we categorized the percentage (per watershed) of source habitat within 0.62-mi buffer of agricultural lands using 10 percent increments from 0 to 100 percent, with increasing poorer habitat outcomes as the proportion of source habitat in the buffer increased (fig. 41).

Calculation of historical conditions—

- Departure of source habitat: Class 1
- Livestock grazing: 0 percent
- Nest parasitism: 0 percent

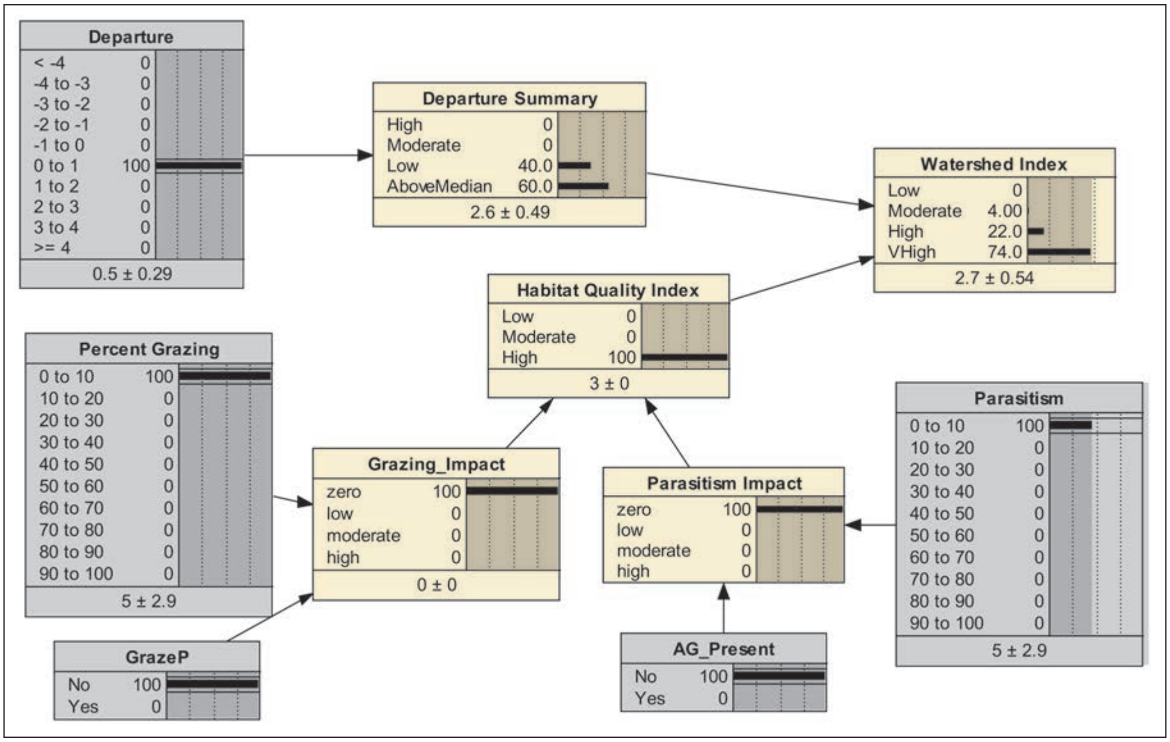

Figure 41—Surrogate species assessment model for MacGillivray's warbler.

The relative sensitivity of WI values to the variables used in the model for MacGillivray's warbler are shown in table 32.

Surrogate Species Assessment Results

Watershed scores—

Owing to presumed habitat loss in all watersheds, the WI values in all watersheds was currently moderate or low (≤2.0, fig. 42). Watersheds that had the most source habitat both currently and historically were the Upper Pend Oreille, Stehekin, Ross Lake, and White-Little Wenatchee (all >12,360 ac). Except for the Upper Pend Oreille, nearly all the habitat for MacGillivray's warbler is on Forest Service-managed lands.

Although nearly 50 percent of the watersheds (n = 33, 46 percent) had ≤10 percent of the source habitat in an active grazing allotment, 8 percent had >50 percent (n = 16) of the source habitat in an active grazing allotment. These watersheds with >50 percent habitat in active grazing allotments had low WI values (<1). As described earlier, livestock grazing has been shown to be a negative impact on the quality of MacGillivray's warbler habitat. The negative effect of nest parasitism was less of an influence on watershed scores.

Table 32—Relative sensitivity of watershed index values to variables in the model for MacGillivray's warbler

Variable	Sensitivity rank
Habitat departure	1
Livestock grazing	2
Nest parasitism	3

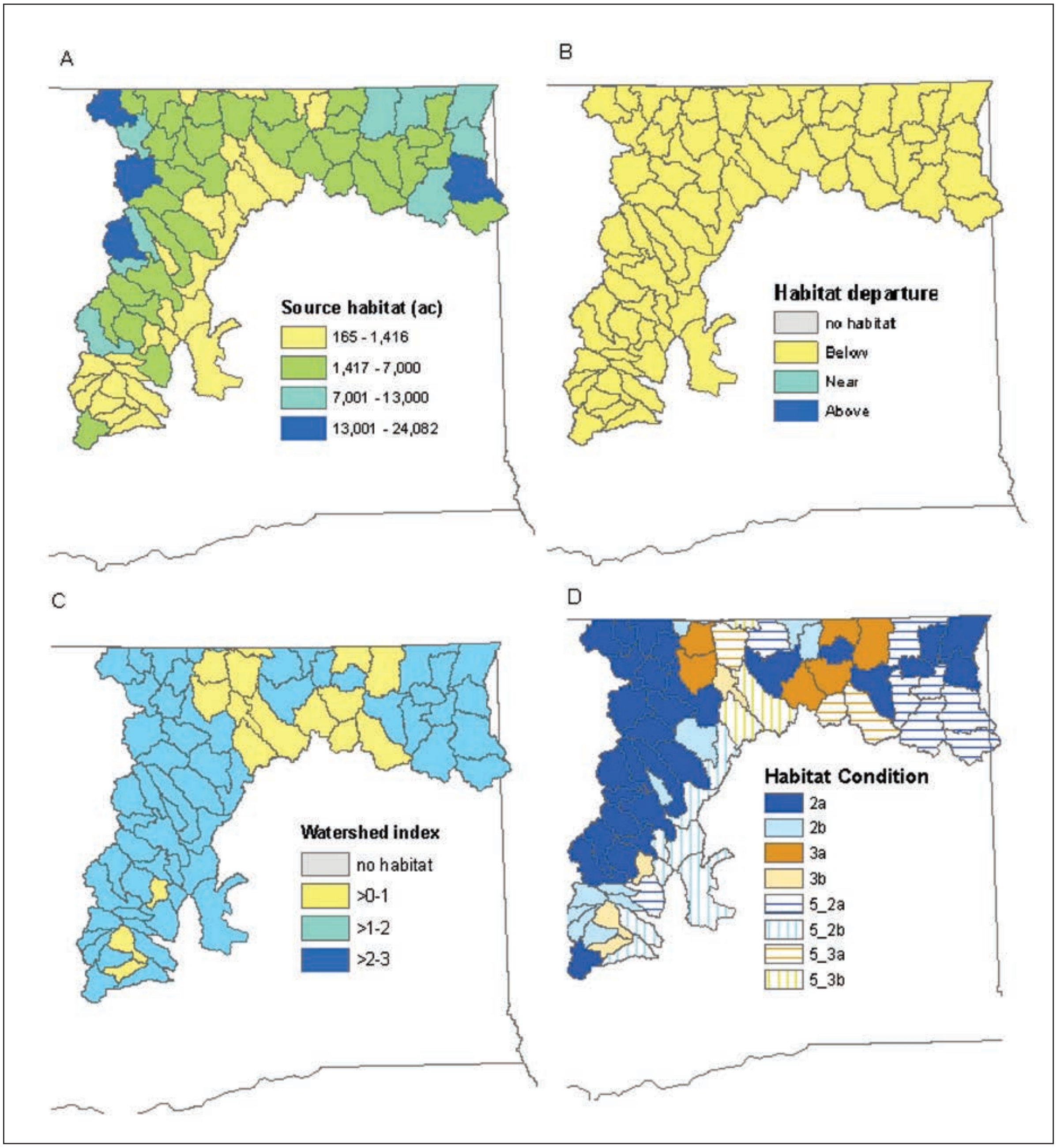

Figure 42—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D) habitat conditions for McGillivary's warbler by watershed in the northeast Washington assessment area.

Although our index of potential negative impacts of nest parasitism (percentage of source habitat within 0.62 mi of agriculture) ranged from 0 to 99 percent, the median value for all watersheds was about 10 percent. Fifty percent (n = 36) of the watersheds had <10 percent, while 13 percent (n = 9) had >50 percent of their habitat near agriculture.

Viability outcome score—

The VOI model incorporated the weighted WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for MacGillivray's warbler within the assessment area is 51 percent of the historical capability. Sixty-four percent (n = 46) of the watersheds had >40 percent of the median amount of historical source habitat across all ecoregions. The watersheds with >40 percent were distributed across all five ecoregions. Dispersal across the assessment area was not considered an issue for this species owing to its high mobility. The resulting viability outcome for MacGillivray's warbler is primarily C (fig. 43), indicating that suitable environments are distributed frequently as patches or exist at low abundance.

Historically, we estimated that 75 percent (n = 54) of the watersheds contained >40 percent of the median amount of source habitat and were distributed across all five ecoregions, which led to primarily A viability outcome (fig. 43). MacGillivray's warbler habitat amount and distribution has declined as a result of loss and modification of source habitats.

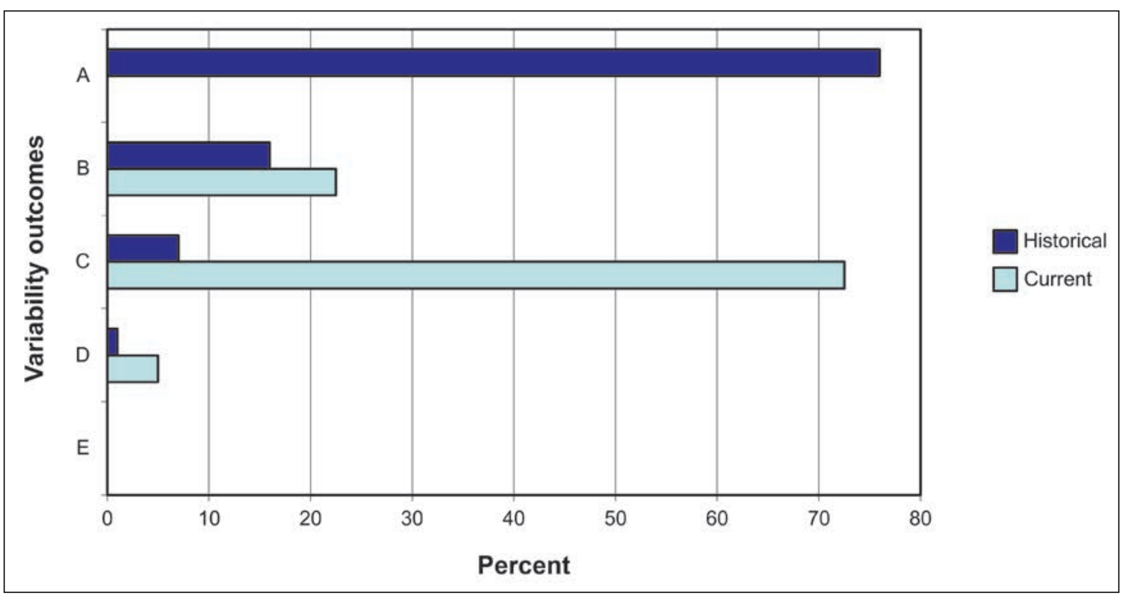

Figure 43—Current and historical viability outcomes for the MacGillivray's warbler in the northeast Washington assessment area.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Loss of riparian shrub habitat owing to the encroachment of conifers as a result of fire suppression.
2. Loss of riparian habitat owing to road construction and other human developments.
3. Reduced quality and quantity of shrub habitat owing to the effects of livestock grazing.

Marsh Wren

Introduction

Marsh wrens were chosen as a surrogate species to represent species associated with the marsh group of the wetland family. They have been shown to be sensitive to hydrologic change in wetland habitats (Steen et al. 2006, Timmermans et al. 2008). Water-level changes and associated reductions in the amount or extent of standing water in emergent vegetation affected habitat quality for marsh wrens (Meyer 2003, Timmermans et al. 2008, Tozer 2002). Shallow-water species, such as marsh wrens, may be more sensitive to habitat suitability changes caused by hydrological dynamics than other wetland species (Steen et al. 2006, Timmermans et al. 2008). The main risk factors for all species associated with marsh habitat were draining, filling, and degradation of marshes; environmental contaminants; and predators at nest sites. Marsh wrens were chosen as the surrogate species for this group because they have widespread distribution in eastern Washington and their risk factors include those of the other species in this group. Marsh wrens range across the central portion of the assessment area (Smith et al. 1997). Marsh wrens were year-round residents of the assessment area (Kroodsma and Verner 1997); this assessment was for nesting habitat.

Model Description

Source habitat—

Presence and depth of standing water within emergent vegetation was an important habitat feature for many marsh birds, including marsh wrens, because it facilitated foraging activities, cover for predator avoidance, and often dictated food or nest site availability (Kroodsma and Verner 1997, Picman et al. 1993). Cattail marshes with interspersed open water >3.3 ft deep were preferred nesting sites for marsh wrens (Linz et al. 1996, Manci and Rusch 1988, Ozesmi and Ozesmi 1999, Picman et al.

1993, Verner and Engelsen 1970). Leonard and Picman (1986) reported that nests of marsh wrens in dense vegetation with deep water were more successful than those in shallower water (i.e., means of 36 vs. 52 in). Banner and Schaller (2001) suggested that palustrine, emergent wetlands PEM (Cowardin et al. 1979) were preferred habitat for nesting marsh wrens. For this analysis, PEMs, as described and mapped through the National Wetlands Inventory (Cowardin et al. 1979), were considered source habitat for marsh wrens (fig. 44).

Invasive species—

Marshes invaded with purple loosestrife have been reported to be less suitable as habitat for marsh wrens than marshes with cattails or other natural vegetation (Rawinski and Malecki 1984, Whitt et al. 1999). Although it has been suggested that the conclusions reached by these studies were equivocal (Anderson 1995, Hagar and McCoy 1998), a more recent review (Blossey et al. 2001) confirmed the threat of habitat degradation in marshes and other wetlands as a result of invasion by purple loosestrife.

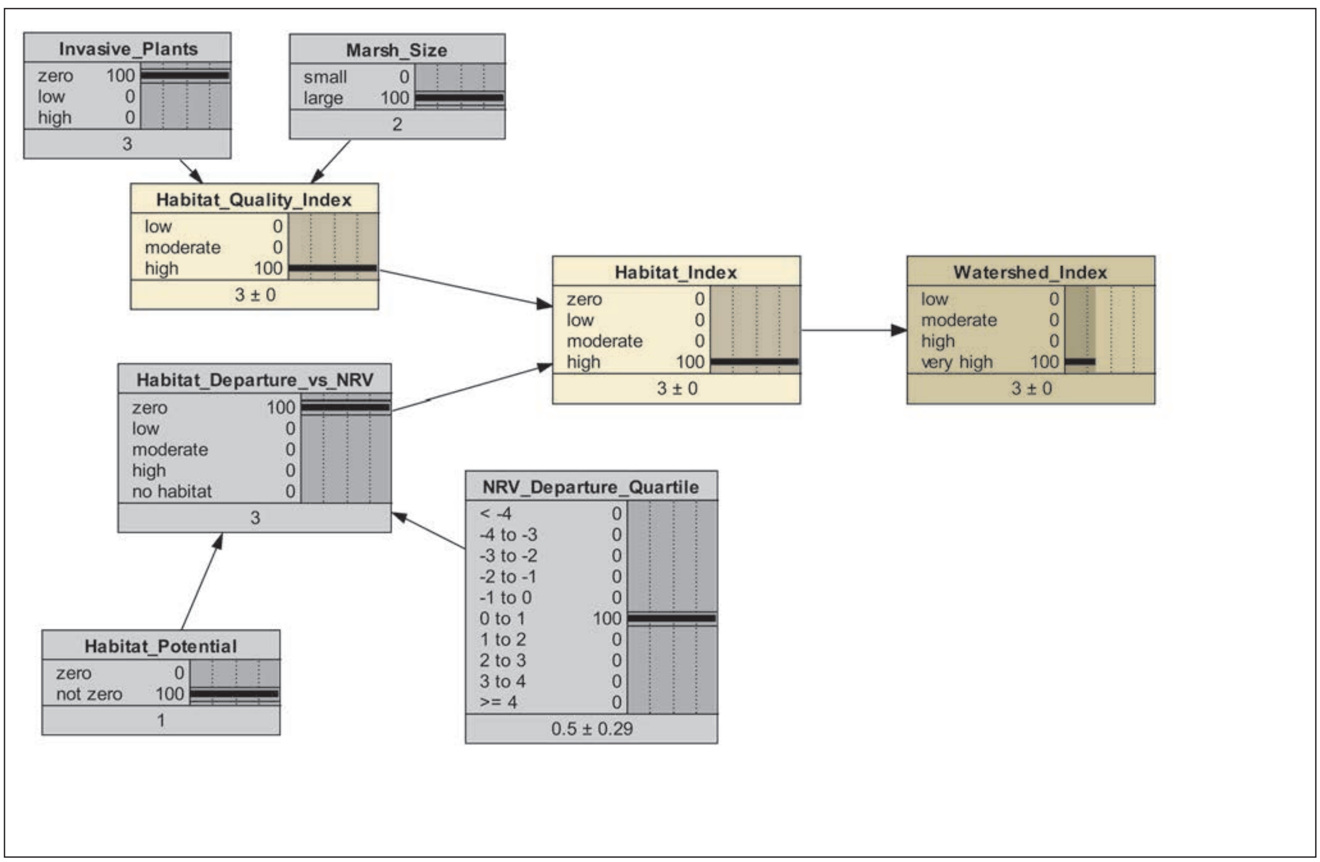

Figure 44—Surrogate species assessment model for marsh wren.

The following classes of purple loosestrife presence in wetlands were used to evaluate the effect of the invasion of habitats within the assessment area (fig. 44):

- Zero: Purple loosestrife not present within a watershed
- Low: Purple loosestrife present in <30 percent of wetlands within a watershed
- High: Purple loosestrife present in ≥30 percent of wetlands within a watershed

Marsh size—

Birds nesting in the interior of marshes have been reported to be more secure from predation (Picman et al. 1993, Richter 1984), indicating that marshes in large patches provide more productive habitat than small patches. This was also supported by the finding that marsh wrens suffered more predation when nesting at dry sites at the edge of marshes than at sites in the center of marshes (Leonard and Picman 1986). Gibbs and Melvin (1990) and Brown and Dinsmore (1986) found that marsh wrens preferred larger to small marshes. Although their statistical power was low (i.e., 0.73), Benoit and Askins (2002) showed a tendency for marsh wrens to prefer large patches (i.e., >250 ac) for nesting. Banner and Schaller (2001) suggested that marshes >40 ac were more valuable as habitat for marsh wrens than smaller marshes. Sites >460 ft from the edge in cattail marshes were preferred for nesting (Ozesmi and Ozesmi 1999). This finding suggests that marshes >40 ac provide progressively more nesting habitat.

The following classes of size of source habitat for marsh wrens were used to evaluate the effect of marsh size within the assessment area (fig. 44):

- Small: <40 ac mean size of PEM wetlands within a watershed
- Large: ≥40 ac mean size of PEM wetlands within a watershed

Calculation of historical conditions—

Values of the model variables were set with the following values to estimate historical habitat conditions:

- Departure of source habitat from HRV: 0.5
- Invasive species: Class zero
- Marsh size: Same as current condition
- Current amount of habitat in each watershed was increased by 30 percent.

The relative sensitivity of the WI values to the variables used in the model for marsh wren are shown in table 33.

Table 33—Relative sensitivity of watershed index values to variables in the model for marsh wren

Variable	Sensitivity rank
Habitat departure	1
Marsh size	2
Invasive plants	3

Assessment Results

Watershed scores—

Historically, 72 percent (n = 52) of the watersheds within the assessment area provided habitat for marsh wrens and other species associated with the marsh group of the wetland family. Currently, the same watersheds contained some habitat for this species group, although several watersheds have minimal amounts (i.e., <50 ac) (fig. 45). Watersheds with the largest amounts of habitat were located in the central, eastern, and southern portions of the assessment area. All watersheds with habitat had WI scores that were moderate (>1.0 and < 2.0) (fig. 45).

Marsh wrens have been reported to be sensitive to invasion of source habitats by purple loosestrife (Rawinski and Malecki 1984, Whitt et al. 1999). All counties within the assessment area, except Ferry County, have recorded occurrences of purple loosestrife (USDA NRCS 2004). Twelve percent (n = 6) of the watersheds had habitat wholly or mostly within Ferry County and were considered free from the effects of purple loosestrife for this analysis. Twelve percent (n = 6) of the watersheds had habitat immediately adjacent to Ferry County and were considered to be in the low invasion category. The remaining 76 percent (n = 40) of the watersheds with habitat were considered to be in the high invasion category.

The size of marshes was thought to be directly related to habitat quality for marsh wrens (Banner and Schaller 2001). All watersheds with habitat had >90 percent of marshes in the small size category.

Viability outcome scores—

The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for marsh wren within the assessment area was 48 percent of the historical capability. Dispersal across the assessment area was not considered an issue for this species owing to their high mobility. All ecoregions currently contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Forty-four percent

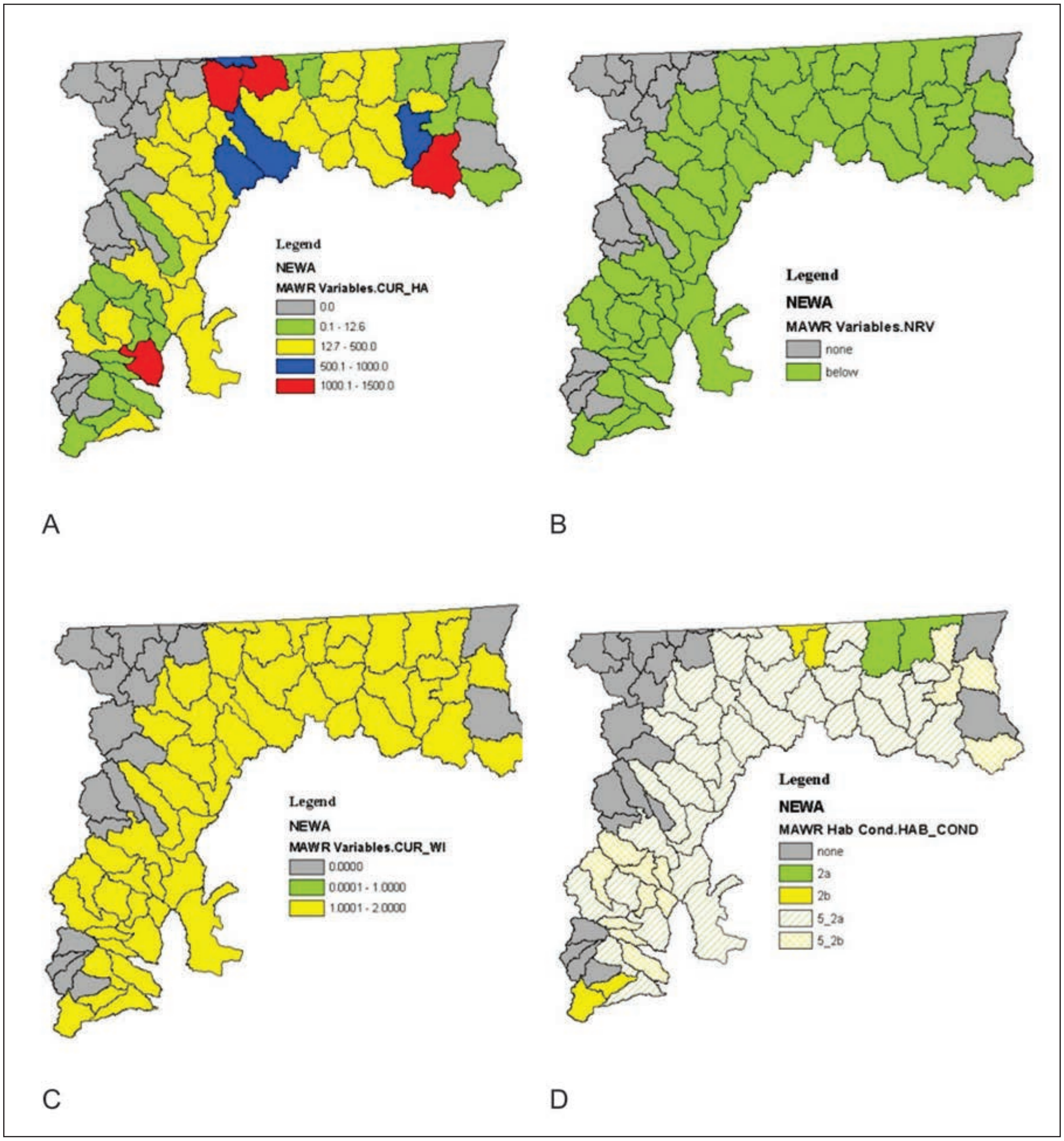

Figure 45—Current amount of (A) source habitat (acre), (B) current habitat departure, (C) watershed index, and (D) habitat condition class for marsh wrens (MAWR) by watrshed in the northeast Washington assessment area (NEWA).

(n = 32) of the watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there is a 71 percent probability that the current viability outcome for marsh wrens was C, indicating that habitat is patchily distributed and in low abundance (fig. 46). Other species associated with the marsh group of the wetland family literally have similar outcomes.

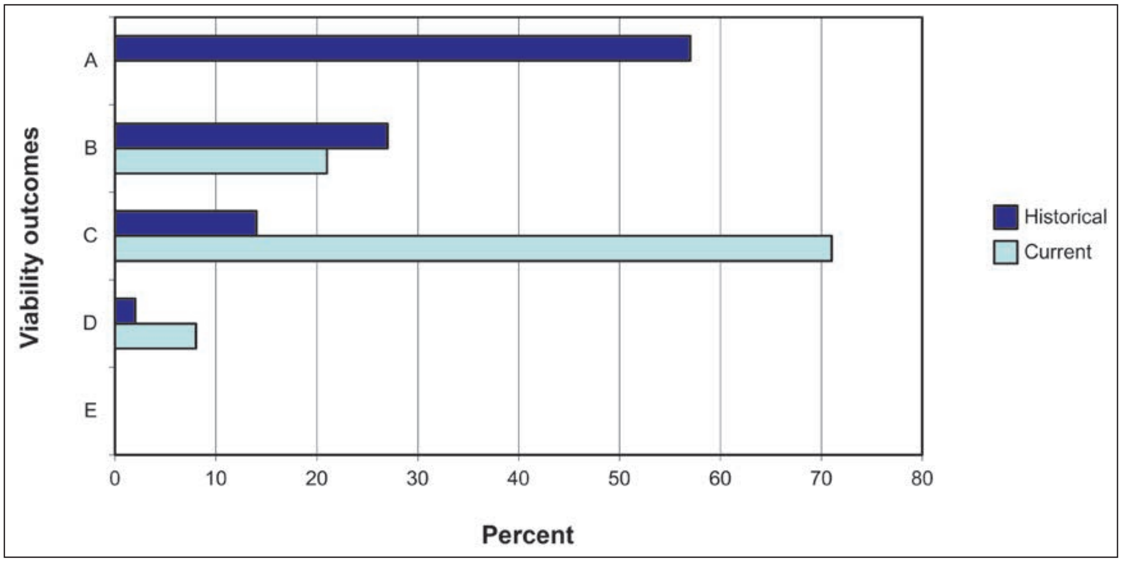

Figure 46—Current and historical viability outcomes for marsh wrens in the northeast Washington assessment area.

Marsh wrens and other species associated with the marsh group of the wetland family were likely well distributed throughout the assessment area. Currently, there were likely fewer populations occupying lower quality habitat throughout the assessment area.

Historically, dispersal across the assessment area was not considered an issue for this species owing to their high mobility. Five of five ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Forty-nine percent (n = 35) of the watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 57 percent probability that the historical viability outcome for marsh wrens was A, and a 27 percent probability that the historical viability outcome for these species was B, indicating that habitat was broadly distributed with an abundance of high-quality habitat (fig. 46).

In summary, under historical conditions, marsh wrens and other species associated with the marsh group of the wetland family were likely well distributed throughout the assessment area. Currently, there were likely fewer populations occupying lower quality habitat throughout the assessment area.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Loss and degradation of wetland habitats.
2. Negative effects of purple loosestrife invasion in source habitats.

Northern Bog Lemming

Introduction

The northern bog lemming was selected as a surrogate species in the boreal forest group. The northern bog lemming is limited to the cold, wet bogs or grass/forb meadows within or on the edges of the boreal coniferous forest (Groves and Yenson 1989, Reichel and Beckstrom 1994, Sallabanks et al. 2001). The watersheds that contain known records of the northern bog lemming include the Lower Pend Oreille, Middle Pend Orielle, Upper Methow, Lost River, Upper Chewuch, and Sinlahekin Creek. Very little is known about the ecological relationships of this species (Johnson and Cassidy 1997).

Owing to the limited distribution of this species within the assessment area, a surrogate species assessment model was not developed. However, a qualitative assessment of its habitat relationships and general management considerations is provided.

Source habitat—

Bog lemmings are found in sphagnum bogs, wet meadows, moist mixed and coniferous forests; alpine sedge meadows, krummholz spruce-fir forest with dense herbaceous and mossy understory, and mossy streamsides (Clough and Albright 1987, Groves and Yenson 1989, Reichel and Beckstrom 1994, Sallabanks et al. 2001).

Risk factors—

The risk factors that were identified for this species include the fragmentation or loss of habitat as a result of road construction and mortality associated with winter recreational activities causing snow compaction. Snow compaction has been cited to cause mortality and to present barriers to small mammals that move in subnivean spaces, such as bog lemmings (Layser and Burke 1973, Schmid 1972). Layser and Burke (1973) have also identified heavy grazing and loss of habitats owing to impoundments as additional risk factors.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Wetlands and alpine meadows provide source habitat for bog lemmings.
2. Areas of sphagnum or other fen/bog moss mats or associated riparian areas could provide corridors for interpatch movements.

3. Domestic livestock grazing in drainages with unsurveyed moss mats present or known lemming populations may reduce habitat quality for bog lemmings.
4. Snow compaction may reduce subnivean habitats and movements during winter.

Northern goshawk

Introduction

The northern goshawk was selected as a surrogate species to represent the forest mosaic and all forest communities medium- and large-tree family group. Risk factors that the goshawk represents include the potential for human disturbance to disrupt breeding activities (Reynolds et al. 1992) and the effects of forest roads in habitat loss and fragmentation (Wisdom et al. 2000). Goshawks are widely distributed across the forested portions of the assessment area (Smith et al. 1997), and this assessment considered year-round habitats.

Model Description

Source habitat—

The northern goshawk uses a complex mosaic of landscape conditions to meet various life history requirements for nesting, postfledgling, and foraging (Desimone and Hays 2003, Reynolds et al. 1992). Goshawk nesting habitat in eastern Washington and Oregon was generally composed of mature and older forests (McGrath et al. 2003). Nest stands were typically composed of a relatively high number of large trees, high canopy closure (>50 percent), multiple canopy layers, and a relatively high number of snags and down wood (McGrath et al. 2003).

Postfledgling areas contain the nest area(s) and are areas of concentrated use by adult females and developing juveniles after fledgling and prior to natal dispersal (Kennedy et al. 1994, Reynolds et al. 1992). Postfledgling areas surround and include the nesting area and provide foraging opportunities for adult females and fledgling goshawks, as well as cover for fledglings (Kennedy et al. 1994, Reynolds et al. 1992). Postfledgling areas in eastern Washington and Oregon were composed largely of structurally complex late-successional forests (McGrath et al. 2003).

Changes in forest structure owing to fire exclusion within the dry forest cover types may seem to increase the availability of source habitat for the goshawk. However, they may not be as valuable as the more open habitats they replaced because the ingrowth of small trees may obstruct flight during foraging, suppress growth of large trees needed for nesting, and reduce the growth of herbaceous understory that provides habitat for prey (Reynolds et al. 1992).

We modeled goshawk source habitat using the following variables that were available in our GIS data layers (fig. 47):

- Forest types: Dry forest, mesic forest, cold-moist forest
- Tree size: >15 in QMD
- Layers: Single/multistory
- Canopy closure: >50 percent

Late-successional forest—

Goshawks forage in a variety of forest types; however, several studies have shown the importance of mid- to late-successional forests as foraging habitat for goshawks (Austin 1993, Beier and Drennen 1997, Bright-Smith and Mannan 1994, Daw and DeStefano 2001, Desimone and DeStefano 2005, Drennan and Beier 2003, Finn et al. 2002a, 2002b; Hargis et al. 1994, Patla 1997). Results from Beier and Drennen (1997) supported the hypothesis that goshawk morphology and behavior are adapted for hunting in moderately dense, mature forests, and that prey availability (as determined by the occurrence of favorable vegetation structure) is more important than prey density in habitat selection. Salafsky and Reynolds (2005) showed that

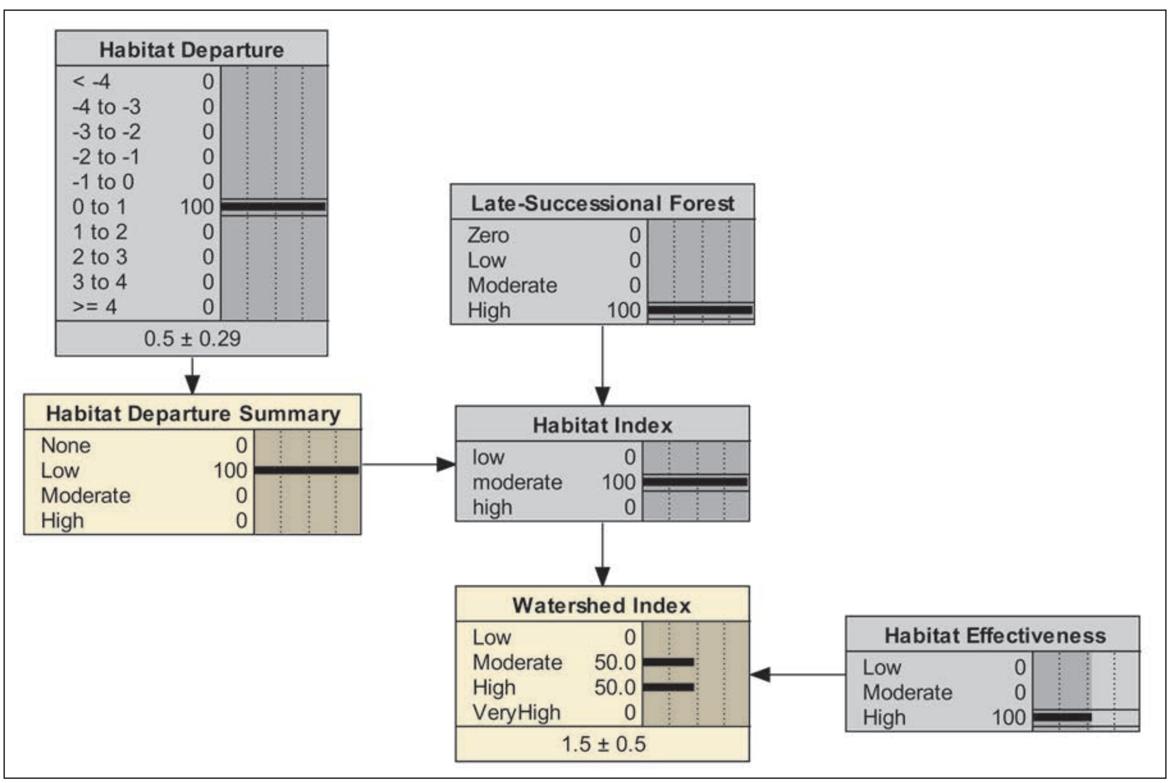

Figure 47—Surrogate species assessment model for northern goshawk.

goshawk productivity was related to prey availability, especially critical prey species. Taken together, these studies show the importance of habitat structure to goshawk foraging behavior and productivity.

Because of the importance of late-successional forests in many of the life history stages of the goshawk, we chose to map late-successional forests as a factor that influenced the quality of source habitat (fig. 47). We modeled late-successional forest habitats using the following variables that were available in our GIS data layers:

- Forest types: Dry forest, mesic forest, cold-moist forest
- Tree size: >20 in QMD
- Layers: Single/multistory
- Canopy closure: >50 percent

We then categorized the amount of source habitat composed of late-successional forest as follows (fig. 47):

- Zero: Late-successional forest in source habitat
- Low: >0 to 20 percent of the source habitat in late-successional forest
- Moderate: >20 to 50 percent of the source habitat in late-successional forest
- High: >50 percent of the source habitat in late-successional forest

Habitat effectiveness—
Human disturbances at goshawk nest sites have been suspected as a cause of nest abandonment (Reynolds et al. 1992). In addition, roads and trails may facilitate access for falconers to remove young from nests (Erdman et al. 1998). Wisdom et al. (2000) identified habitat fragmentation or habitat loss as a forest road-associated factor for goshawks. In addition, roads may increase the likelihood of the removal of snags for safety and firewood collection, which could have negative effects on the prey base for goshawks (Wisdom et al. 2000). However, Grubb et al. (1998) reported that vehicle traffic with a noise level of <54 decibels on roads >1,320 ft from nest sites did not result in discernible behavioral response by goshawks in forested habitats.

Because of these potential influences of forest roads on goshawk source habitat, we used the late-successional forest habitat disturbance index described in Gaines et al. (2003). This index buffers open roads and motorized trails that occur within source habitat by 660 ft on each side, and nonmotorized trails that occur within source habitat by 330 ft on each side. The amount of source habitat that was influenced by human activities was then categorized as follows for each watershed (fig. 47):

- Low habitat effectiveness: <50 percent of the source habitat outside a zone of influence
- Moderate habitat effectiveness: 50 to 70 percent of the source habitat outside a zone of influence
- High habitat effectiveness: >70 percent of the source habitat outside a zone of influence

Calculation of historical conditions—

- Departure of source habitat: Departure class 1
- Late-successional habitat: High
- Habitat effectiveness: High

The relative sensitivity of the WI values to the variables used in the model for northern goshawk are shown in table 34.

Surrogate species model evaluation—
We compared the mean WI value derived from 674 points with documented occurrences of northern goshawks to the mean WI value derived from 674 random points. The mean WI for the occurrence points (1.72) was significantly higher (t = 1.96, P <0.0001) than the mean derived from the random points (1.56) indicating that our model successfully identified watersheds with suitable environments for northern goshawks.

Assessment Results

Watershed scores—
Thirty-one (43 percent) of the watersheds had WI scores that were high (>2.0). Eight (11 percent) of the watersheds had moderate scores (1.0 to 2.0). Our assessment showed that the departure in the amount of source habitat from the expected historical median amount was the variable with the most influence on the northern goshawk watershed scores.

Table 34—Relative sensitivity of watershed index values to variables in the model for northern goshawk

Model variables	Order of variable weighting
Source habitat	1
Late-successional forest	2
Habitat effectiveness	3

We found that the amount of source habitat in 15 percent (n = 11) of the watersheds was above the historical median of source habitat, 35 percent (n = 25) were near the historical median, and 50 percent (n = 36) were below the historical median. The lack of big tree structure, particularly in the watersheds located on the eastern portion of the Okanogan-Wenatchee National Forest and most of the Colville National Forest, limited the availability of source habitat for northern goshawks in these areas. This finding is similar to that reported in Wisdom et al. (2000) where strongly negative trends in the amount of source habitat for goshawk occurred in the northern portion of the Northern Cascade Range and throughout the Northern Glaciated Mountain ERUs.

Watersheds with the most source habitat (>49,420 ac) included the Stehekin, Ross Lake, White-Little Wenatchee River, Upper Tieton River, and Little Naches Rivers. Watersheds with the least amount of source habitat (<1,230 ac) included the Middle Yakima River, Upper Columbia-Swamp Creek, and Upper Little Spokane River.

The amount of source habitat that was in a late-successional stage was low overall. Currently, 21 percent (n = 15) of the watersheds were rated as having a high (>50 percent of the source habitat) amount of late-successional forest habitat, 12 percent (n = 9) had a moderate amount (30 to 50 percent), 64 percent (n = 46) low (>0 to <30 percent), and 3 percent (n = 2) no late-successional forest habitat.

Habitat effectiveness was indexed by buffering the amount of source habitat adjacent to roads (Gaines et al. 2003). Habitat effectiveness for goshawks was considered to be high within 54 percent (n = 39) of the watersheds, moderate in 42 percent (n = 30), and low in 4 percent (n = 3).

Viability outcome scores—
The VOI model incorporated the WWI scores and a habitat distribution index. The WWI provides a relative measure across watersheds of the potential capability of the watershed to contribute to the viability of the surrogate species. The current WWI across the assessment area was 68 percent of the historical WWI.

Currently, 54 percent (n = 39) of the watersheds are above the 40 percent threshold of the historical median amount of source habitat, whereas historically, 88 percent (n = 63) were above this minimum habitat amount. Currently, the watersheds with >40 percent of the historical median amount of source habitat were distributed across all of the five ecoregions, as was the case historically. Dispersal across the assessment area was not considered an issue for this species (fig. 48).

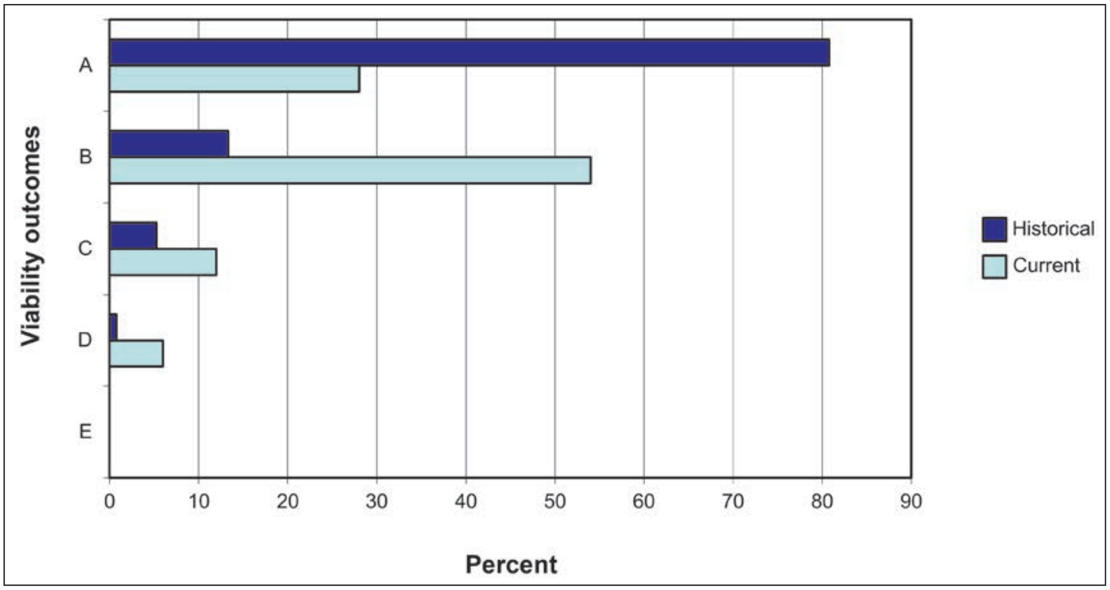

Figure 48—Current and historical viability outcomes for the northern goshawk in the northeast Washington assessment area.

Under current conditions, there is a 28 percent probability that the viability outcome for goshawks across the assessment area is A, and a 54 percent probability of outcome B. This indicates that suitable environments for the northern goshawk are broadly distributed and of relatively high abundance, but there are gaps where suitable environments are absent or only present in low abundance (fig. 48). These gaps are typically not large enough to prevent species from interacting as a metapopulation. Historically, the viability outcome had an 80.8 percent probability of A, where suitable environments were more broadly distributed or of high abundance. In addition, the suitable environments were better connected, allowing for interspecific interactions. A reduction in the availability of suitable environments for the northern goshawk may have occurred in the assessment area compared to the historical distribution and condition of their habitats.

Historically, northern goshawks and other species in the forest mosaic/all forest communities medium-to-large old-tree group/family were likely well distributed with viable populations across the assessment area. Wisdom et al. (2000) assessed viability for the northern goshawk across the Columbia Basin, at a broader scale and coarser resolution, and reported similar results.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Reduction in the amount of source habitat, particularly within the eastern portion of the Okanogan-Wenatchee National Forest and across much of the Colville National Forest.
2. Declines in the densities of large-diameter (>21 in) trees and snags from historical to current levels (Hann et al. 1997, Harrod et al. 1999, Hessburg et al. 1999a), which are important components of habitat for the species in this group.
3. Potential loss of snag and down log habitat as a result of high open road densities.
4. Fire exclusion within much of the dry forest cover types may have reduced the sustainability of these habitats and has resulted in increased susceptibility to stand-replacing fires (Everett et al. 2000, Hessburg et al. 1999a, McGrath et al. 2003, Townsley et al. 2004).
5. Limited information on the effects of dry forest restoration treatments on goshawk habitat use and productivity.

Northern Harrier

Introduction

The northern harrier was selected as a surrogate species for the grasslands group because it is a widely distributed species across grasslands in the assessment area. In addition, this species will also be found in wetter grassy and marsh areas, similar to the short-eared owl, another member of the group. All species in this group share human disturbance as a risk factor. Though some harriers may remain in the area during the winter, we primarily evaluated breeding habitat.

Model Description

Source habitat—

Northern harriers prefer relatively open grassland habitats characterized by tall, dense vegetation and abundant residual vegetation (Apfelbaum and Seelbach 1983, Duebbert and Lokemoen 1977, Hamerstrom and Kopeny 1981, Kantrud and Higgins 1992). They are associated with wet or dry grasslands, fresh to alkali wetlands, lightly grazed pastures, croplands, fallow fields, old fields, and shrubby areas (Apfelbaum and Seelbach 1983, Evans 1982, Faanes 1983, Kantrud and Higgins 1992, Linner 1980, MacWhirter and Bildstein 1996, Prescott et al. 1995, Prescott 1997, Stewart 1975, Stewart and Kantrud 1965). Although cropland and fallow fields were used for nesting, most nests were found in undisturbed wetlands or grasslands dominated by dense vegetation (Apfelbaum and Seelbach 1983, Duebbert and Lokemoen 1977, Kantrud and Higgins 1992). Nest success may have been lower in cropland and fallow fields than in undisturbed areas (Kibbe 1975).

Northern harriers nested on the ground or over water on platforms of vegetation in stands of cattail (*Typha* spp.) or other emergent vegetation (Bent 1961, Clark 1972, MacWhirter and Bildstein 1996, Saunders 1913, Sealy 1967, Stewart 1975). Ground nests were well concealed by tall, dense vegetation, including living and residual grasses and forbs, or low shrubs, and are located in undisturbed areas with much residual cover (Duebbert and Lokemoen 1977, Hamerstrom and Kopeny 1981, Hecht 1951, Herkert et al. 1999, Kantrud and Higgins 1992).

Nests in wet sites may have an advantage in that fewer predators have access to them (Sealy 1967, Simmons and Smith 1985). In Alberta, northern harriers were more abundant in large (>20 ac) fresh wetlands than in small (<2.5 ac) fresh wetlands (Prescott et al. 1995).

Northern harriers had large territories; in Idaho, home ranges averaged 3,880 ac for males and 280 ac for females (Martin 1987). In North Dakota, breeding harriers were found only in grassland patches ≥250 ac and were encountered in large patches more than expected (Johnson and Igle 2001). All occupied patches exceeded 250 ac. In contrast, Herkert et al. (1999) suggested that harriers may respond more strongly to total amount of grassland within the landscape rather than to sizes of individual grassland tracts.

For this assessment, we identified grassland, and meadows (wet and dry meadow) cover types in the shrub-steppe potential vegetation type as source habitat for this species. We included PEM and PSS habitats as identified in the National Wetlands Inventory data set (Cowardin et al. 1979). In addition, we described source habitat as areas with <20 percent slope and patches of habitat >1 ac. Only watersheds with >120 ac of habitat historically were included in the analysis. Source habitat was identified as (fig. 49):

- Potential vegetation types: Shrub-steppe
- Cover types: Grassland, wet meadows, dry meadows
- National Wetlands Inventory: PEM and PSS
- Slope: <20 percent

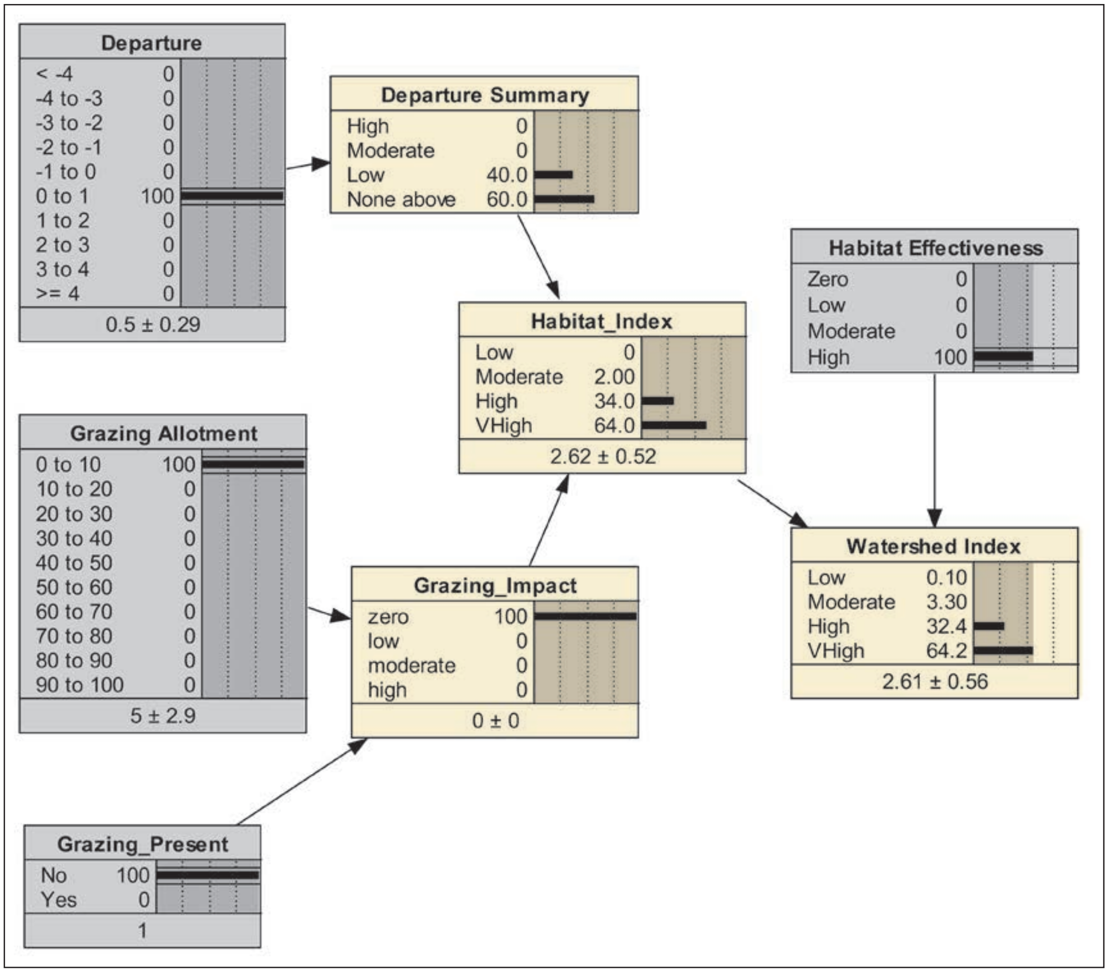

Figure 49—Surrogate species assessment model for northern harriers.

Grazing—

Overgrazing, the advent of larger crop fields, and fewer fence rows, together with the widespread use of insecticides and rodenticides, have reduced the availability of prey for northern harriers and thus the amount of suitable habitat for this species (Duebbert and Lokemoen 1977, Hamerstrom 1986). In the Great Plains, Southwest, and U.S. intermountain West, northern harriers have been found to use livestock-grazed grasslands less than ungrazed areas (Bildstein and Gollop 1988, Bock et al. 1993, Linner 1980). Northern harriers preferred idle areas to grazed areas in North Dakota (Sedivec 1994). Northern harriers do not use heavily grazed habitats (Berkey et al. 1993, Bock et al. 1993, Stewart 1975) but may use lightly to moderately grazed grasslands (Bock et al. 1993, Kantrud and Kologiski 1982). In North Dakota, northern harriers had significantly higher nesting density on ungrazed areas than areas grazed season-long or under a twice-over grazing rotation schedule (Messmer 1990, Sedivec 1994). In aspen parkland of Alberta, northern harriers were most abundant in deferred grazed (grazed after 15 July) mixed grass, but were absent from continuously grazed mixed grass and deferred or continuously grazed tame pasture (Prescott et al. 1995).

To account for possible impacts of livestock grazing on habitat, we categorized the amount of source habitat in an active grazing allotment using 10 percent increments from 0 to 100 percent, with increasing poorer habitat outcomes as the proportion of source habitat in an active allotment increased (fig. 49).

Habitat effectiveness—

Nesting harriers are sensitive to human disturbance especially from the prelaying and egg-laying stages up to hatching (Fyfe and Olendorff 1976, Hamerstrom 1969). Predation of harrier young has occurred when predators followed humans to nests (Toland 1985, Watson 1977). Harriers will leave wintering areas with potentially suitable nesting habitat presumably, in part, owing to heavy use by humans (Serrentino 1992).

Because of the potential effects of humans on harriers, we mapped 660-ft buffers on each side of open roads and motorized trails that occurred within source habitat. We also mapped 330-ft buffers on each side of nonmotorized trails that occurred within source habitat. The amount of source habitat that was influenced by human activities (within the buffers) was then categorized as follows for each watershed (fig. 49):

- Zero habitat effectiveness: 100 percent of the source habitat inside the zone of influence
- Low habitat effectiveness: <50 percent of the source habitat outside a zone of influence
- Moderate habitat effectiveness: 50 to 70 percent of the source habitat outside a zone of influence
- High habitat effectiveness: >70 percent of the source habitat outside a zone of influence

Historical inputs for surrogate species assessment model—

- Departure of source habitat: Class 1
- Grazing: 0 percent
- Habitat effectiveness: Zero

The relative sensitivity of WI values to the variables used in the model for the northern harrier are shown in table 35.

Table 35—Relative sensitivity of watershed index values to variables in the model for northern harriers

Variable	Sensitivity rank
Habitat departure	1
Grazing	2
Habitat effectiveness	3

Assessment Results

Watershed scores—

Thirty-six watersheds were estimated to have >120 ac of source habitat historically and were included in our analysis. Three watersheds contained >24,700 ac of source habitat: Columbia River-Lynch Coulee, Lower Okanogan River, and Okanogan and River-Omak Creek (fig. 50). Very little source habitat for northern harriers existed on National Forest System lands. While the Columbia River-Lynch Coulee watershed contains more than 8,650 ac of source habitat on National Forest System lands, all other watersheds contained less than 300 ac of source habitat on National Forest System lands.

Although seven watersheds (18 percent) had <10 percent departure from historical habitat conditions, the remaining watersheds lost >25 percent of historical source habitat for harriers (fig. 50). Ten watersheds (28 percent) were found to have >60 percent losses in sources habitat. Other research has shown that extensive draining of wetlands, monotypic farming, and reforestation of farmlands have led to a decline in habitat and population sizes of northern harriers (MacWhirter and Bildstein 1996, Serrentino 1992, USDI FWS 1987).

The overall loss of habitat led to overall low WI scores (fig. 50). Sixty-four percent (n = 23) of watersheds had WI scores of low (<1.0); 28 percent (n = 10) had a score of moderate (>1.0 to <2.0); and the remaining 8 percent (n = 3) had a high score (≥ 2.0). The watersheds that had a high WI score were Salmon Creek and Sinlahekin.

The watersheds that had a WI between 1.0 and 2.0 had generally either a <15 percent reduction in habitat and higher levels (>40 percent) of grazing or 15 to 30 percent loss in habitat and lower levels of grazing (<10 percent). Watersheds with the highest WI values generally had <15 percent loss in habitat and lower levels of grazing (<15 percent). Habitat effectiveness was low in most watersheds (55 percent, n = 20), which also contributed to lower WI values.

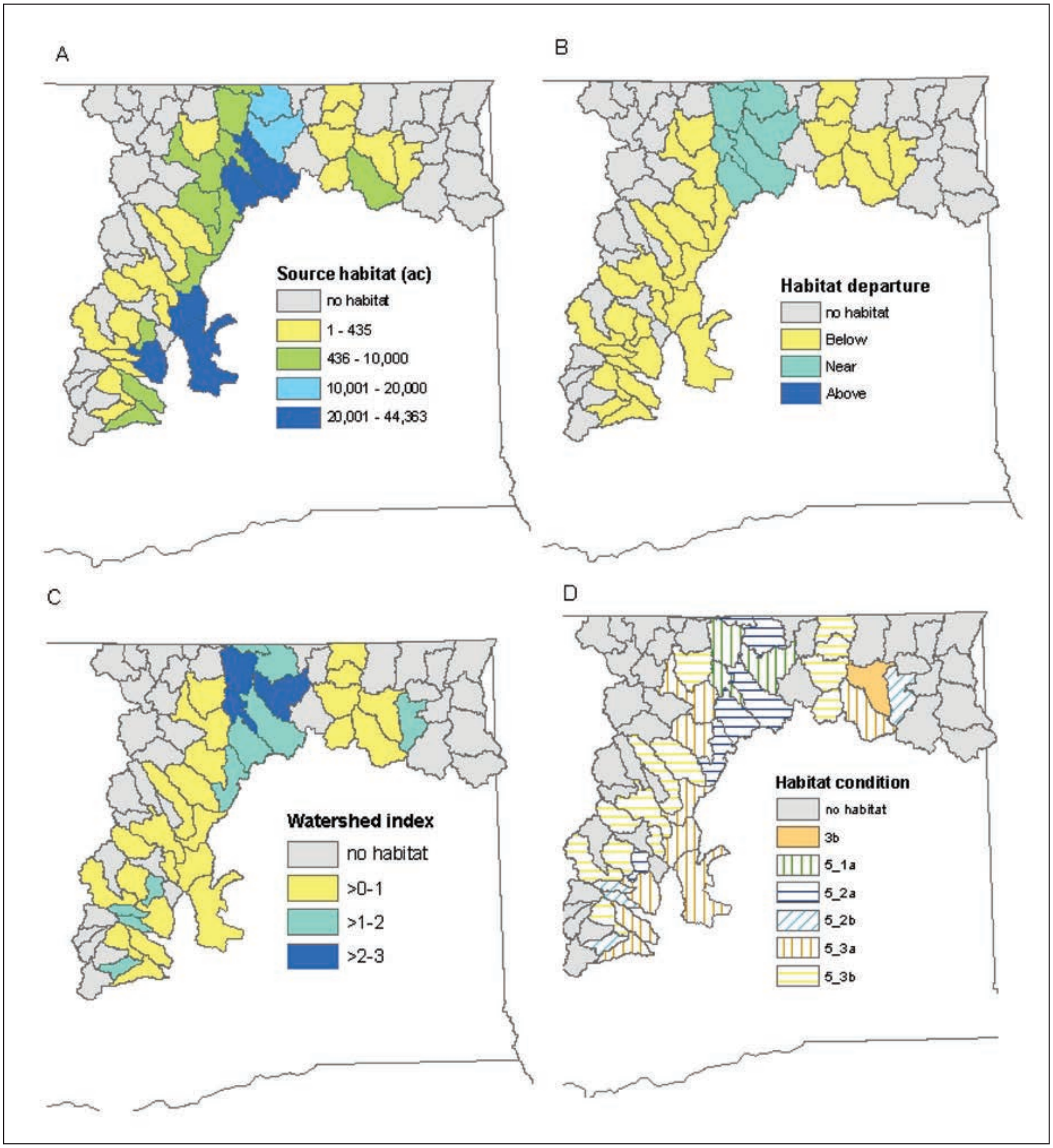

Figure 50—Current amount of (A) source habitat (acres), (B) habitat departure, (C) watershed index, and (D) habitat condition class for northern harriers by watershed in the northeast Washington assessment area.

Viability outcome—
The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The comparison of the current WWI to the historical WWI showed that current habitat was at 50 percent of historical capability. Currently, 50 percent of the watersheds have source habitats >40 percent of the historical median, which is a decline from 67 percent historically. Currently, the viability outcome for harriers across the assessment area is primarily a C outcome indicating that suitable environments are distributed frequently as patches and/or exist at low abundance (fig. 51).

In summary, the northern harrier and other species associated with the grassland group likely have experienced a loss in the abundance and distribution of suitable environments across the assessment area.

The current outcome is a decline from an estimated A outcome historically (fig. 51). Historically, we estimated that 67 percent (n = 24) dispersed across all five ecoregions contained >40 percent of the historical median amount of source habitat leading to primarily an A outcome where habitats were abundant and well distributed. In summary, the northern harrier and other species associated with the grassland group likely have experienced a loss in the abundance and distribution of suitable environments across the assessment area.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Loss of grassland and wetland habitat because of conversion to agricultural and other human developments.
2. Degradation of grassland and wetland habitat through extensive livestock grazing.
3. Adverse effects of human disturbance.

Peregrine Falcon

Introduction

The peregrine falcon was selected as a surrogate species for the habitat generalist/cliff group because of their association with large cliff habitats as compared to the other species in the group. Within the assessment area, the known peregrine falcon nest sites occur on the Naches, Wenatchee River, and Methow Valley Ranger Districts. However, suitable nesting habitat occurs on both the Colville and Okanogan-Wenatchee National Forests. The availability of nesting habitat (e.g., suitable cliff structures) may have changed little from what was available historically (Hayes and Buchanan 2002); however, other factors, such as availability of foraging habitat and habitat effectiveness, have changed. The occurrence of peregrine falcons within the assessment area during the winter is considered rare (Hayes and Buchanan 2002), thus this assessment addresses breeding habitats.

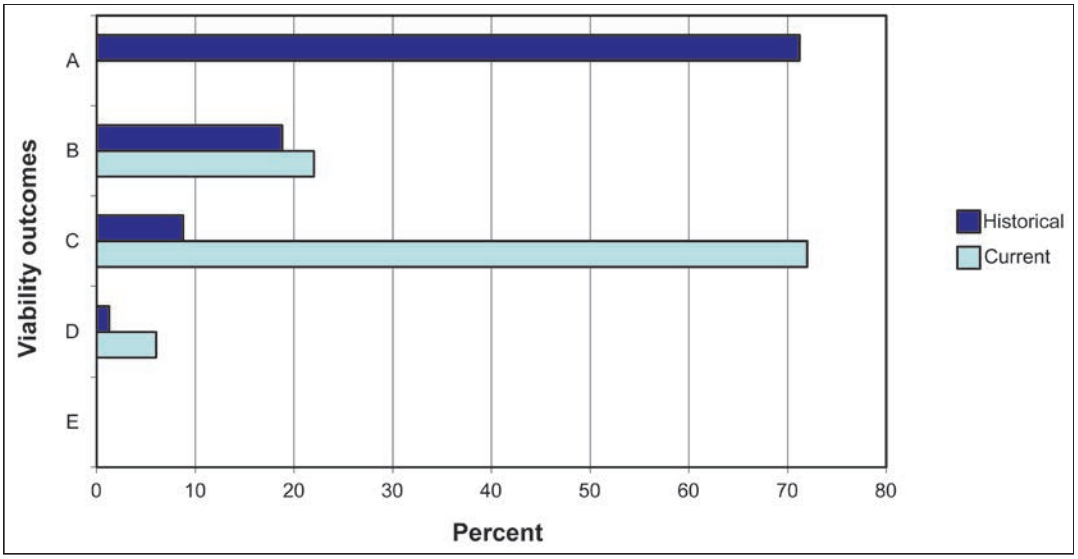

Figure 51—Current and historical viability outcomes for northern harriers in the northeast Washington assessment area.

Model Description

Source habitat—

The presence of prominent cliffs is the most common habitat characteristic of peregrine falcon nesting territories (Hayes and Buchanan 2002, Hays and Milner 1999). Prominent cliffs function as both nesting and perching sites, and provide unobstructed views of the surrounding landscape (Hayes and Buchanan 2002, Ratcliffe 1993). Nest site suitability requires the presence of ledges that are essentially inaccessible to mammalian predators, that provide protection from the elements, and that are dry (Campbell et al. 1990, Johnsgard 1990). A source of water, such as a river, lake, marsh, or marine waters is typically in proximity to the nest site and likely is associated with an adequate prey base of small- to medium-sized birds (Cade 1982, Johnsgard 1990).

On average, peregrine falcon eyries were about 200 ft from a fresh water source in Washington (Hayes and Buchanan 2002). This study reported only a few sites more than 1,000 ft from a creek or a body of water >3 ac (Hayes and Buchanan 2002).

To model the availability of source habitat for the peregrine falcon, we identified both nesting and foraging habitats. To identify nesting habitat, we used a digital elevation model to identify areas that were >38 degrees, which corresponded well with cliff structures. To identify the most prominent features, a minimum size of ≥5 ac was used. This allowed us to distinguish the prominent cliff structures from the smaller cliffs that were unlikely to provide nesting habitat (Hayes and

Buchanan 2002). Finally, we used an elevation cutoff of <3,300-ft elevation in order to screen out high-elevation cliff structures that would likely be unavailable to peregrine falcon for nesting owing to the presence of persistent spring snow. Peregrine eyries in eastern Washington occur between 666 to 1,860 ft in elevation (Hayes and Buchanan 2002).

We assumed that the amount of available nesting habitat had not changed from historical to current, thus we used the amount of nesting habitat within the watershed as a measure of habitat quality (e.g., the more the better). We categorized the amount of nesting habitat as follows (fig. 52):

- Zero: <10 ac of nesting habitat
- Low source habitat: >10 ac but less than the median of nesting habitat across all watersheds
- High source habitat: Less than the median of nesting habitat across all watersheds

We also assessed the amount of foraging habitat within each watershed (fig. 52). Foraging habitat was defined as any water body ≥3 ac (Hayes and Buchanan 2002). We did not assess the proximity of nesting and foraging habitat as described in Hayes and Buchanan (2002) because we assumed each watershed was small enough and peregrine falcons mobile enough that they could forage anywhere in the watershed. We used the following categories to assess the amount of foraging habitat for each watershed (fig. 52):

- Low: <10 ac of foraging habitat
- Moderate: 10 ac to median across all watersheds
- High: Greater than median of all watersheds

Habitat effectiveness—

Human activities have been documented to cause disturbance to nesting peregrine falcons (Holthuijzen et al. 1990, Lanier and Joseph 1989, Windsor 1975). Several authors have recommended 2,625-ft buffers on nest sites to reduce the potential effects of human disturbances on nesting peregrine falcons (Hays and Milner 1999, Richardson and Miller 1997). We assessed the potential for human disturbance to affect nesting habitat using the peregrine falcon nesting habitat disturbance index described in Gaines et al. (2003). We mapped 2,625-ft buffers on each side of open roads and trails to delineate zones of influence and then overlaid this with our map of source habitat.

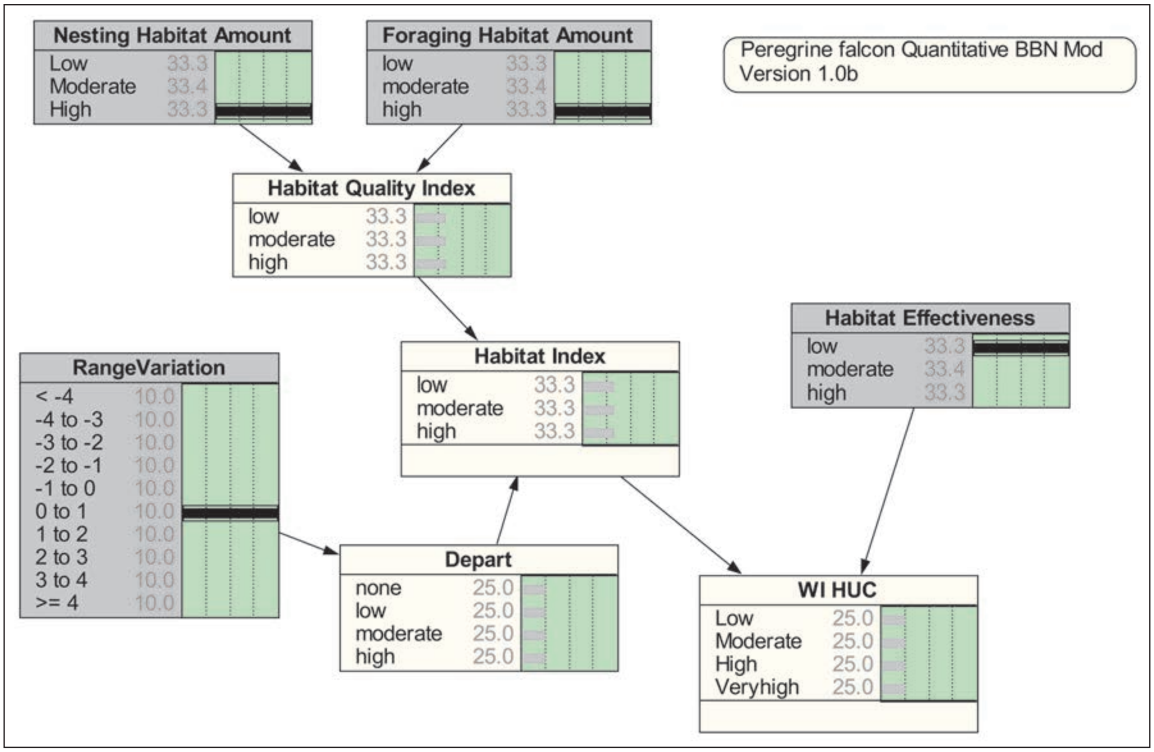

Figure 52—Surrogate species assessment model for the peregrine falcon.

We used the following categories to assess the potential effects of human disturbance on peregrine falcon habitat effectiveness for each watershed (fig. 52):

- Low habitat effectiveness: <25 percent of the source habitat outside a zone of influence
- Moderate habitat effectiveness: 25 to 50 percent of the source habitat outside a zone of influence
- High habitat effectiveness: >50 percent of the source habitat outside a zone of influence

Calculation of historical conditions—

Source habitat: Current habitat amount—habitat departure class 1

Nesting habitat amount: Based on the current amount

Foraging habitat amount: Based on the current amount

Habitat effectiveness: High

The relative sensitivity of WI values to variables used in the peregrine falcon model are shown in table 36.

Table 36—Relative sensitivity of watershed index values to variables in the model for the peregrine falcon

Model variables	Order of variable weighting
Source habitat	1
Nesting habitat amount	2
Foraging habitat amount	3
Habitat effectiveness	4

Model Evaluation

We compared the mean WI value from 33 points of known occurrences of peregrine falcon nests with an equal number of random points. The mean WI for the occurrence points (1.89) was significantly higher (t = 2.00, P = 0.004) than the mean from the random points (1.33), indicating that our model successfully identified suitable environments for peregrine falcons.

Assessment Results

Watershed scores—

Of the 50 watersheds that had source habitat for the peregrine falcon, 64 percent (n = 32) had high (>2.0) WI scores, and 28 percent (n = 14) had moderate (1.5 to 2.0) scores (fig. 53). Watersheds with the greatest amount of source habitat (>370 ac) included Ross Lake, Wenatchee River, Upper Lake Chelan, Columbia River-Lynch Coulee, and Lower Lake Chelan (fig. 53). Because we assumed that there was no departure in the amount of historical source habitat, changes in WI from the historical median were largely related to changes in habitat effectiveness.

The distribution of scores for nesting habitat included 37 percent (n = 19) of the watersheds with a low score, none with a moderate score, and 63 percent (n = 32) with a high score. Fifty-three percent (n = 27) of the watersheds had a low score for foraging habitat, 41 percent (n = 21) with a moderate score, and 6 percent (n = 3) with a high score.

Of the watersheds that contained source habitat for peregrine falcons, 57 percent (n = 29) had a low level of habitat effectiveness based on the proximity of source habitats to potential human activities. In addition, 33 percent (n = 33) of the watersheds had a moderate level of habitat effectiveness, and only 10 percent (n = 5) had a high level of habitat effectiveness.

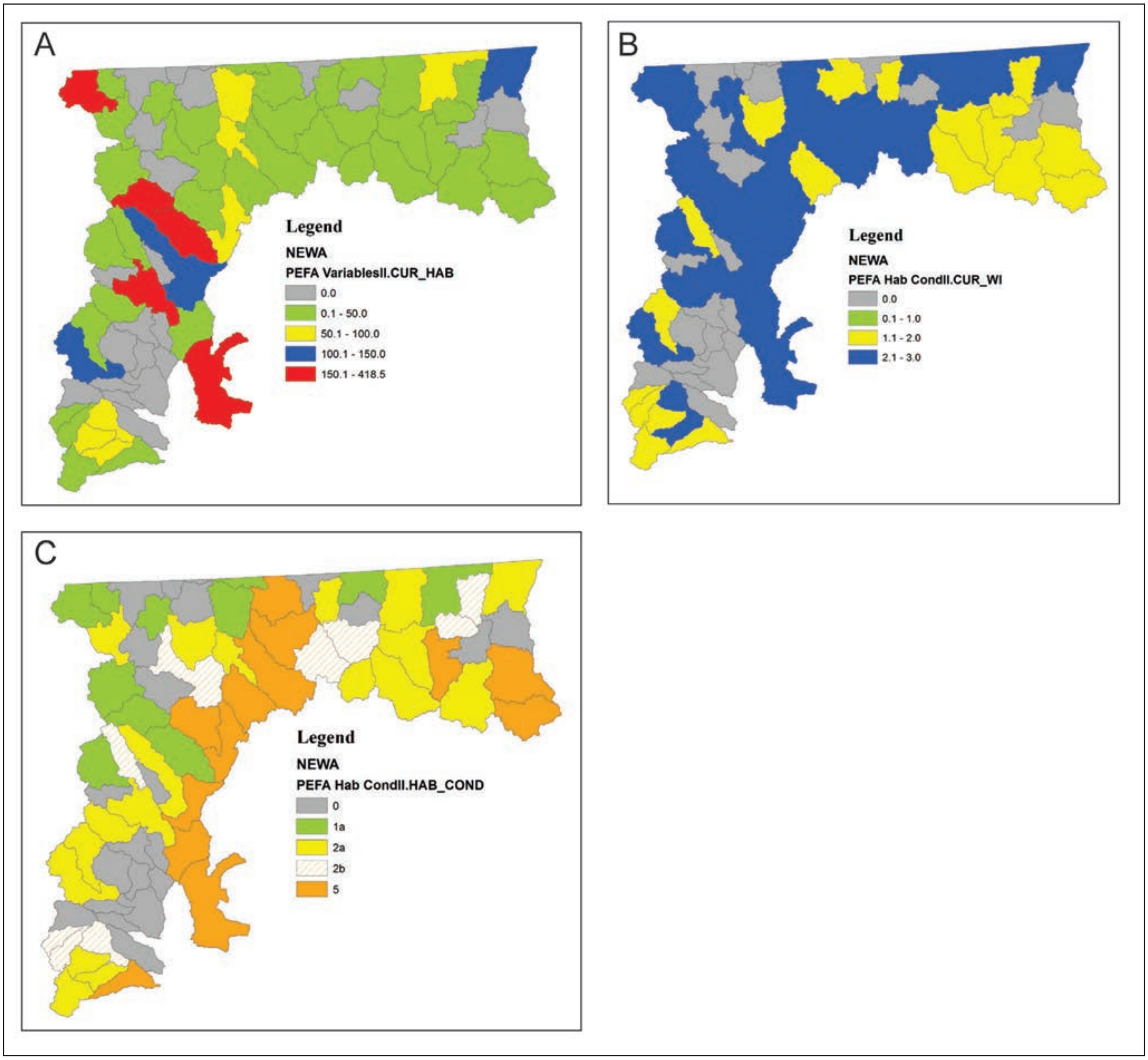

Figure 53—Current amount of (A) source habitat (acres), (B) watershed index, and (C) habitat condition class for peregrine falcon (PEFA) by watershed in the northeast Washington assessment area (NEWA).

Viability outcome score—

The VOI model incorporated the WWI scores (described earlier) and a habitat distribution index. The WWI scores indicated that the current habitat capability across the assessment area was about 79 percent of the historical capability. This largely had to do with the effects of human activities in source habitat for peregrine falcons. The median amount of nesting habitat was 51.4 ac (across all watersheds with at least 2.5 ac of nesting habitat). Currently, 76 percent of the watersheds had habitat amounts that were >40 percent of the historical median, and these watersheds were distributed across all of the ecoregions.

Currently, there is a 32 percent probability that the viability outcome for the peregrine falcon is A, and a 56 percent probability of B, which indicates that suitable environments for peregrine falcons are broadly distributed and of relatively high abundance, but there are gaps where suitable environments are absent or only present in low abundance (fig. 54). These gaps are typically not large enough to prevent the species from interacting as a metapopulation. Historically, there was a 76 percent probability of a viability outcome of A where suitable environments were more broadly distributed or of high abundance (fig. 54). In addition, the suitable environments were better connected, allowing for interspecific interactions. A reduction in the availability of high-quality habitats for peregrine falcons, and likely other species in the group, appears to have occurred in the assessment area compared to the historical distribution and condition of their habitats.

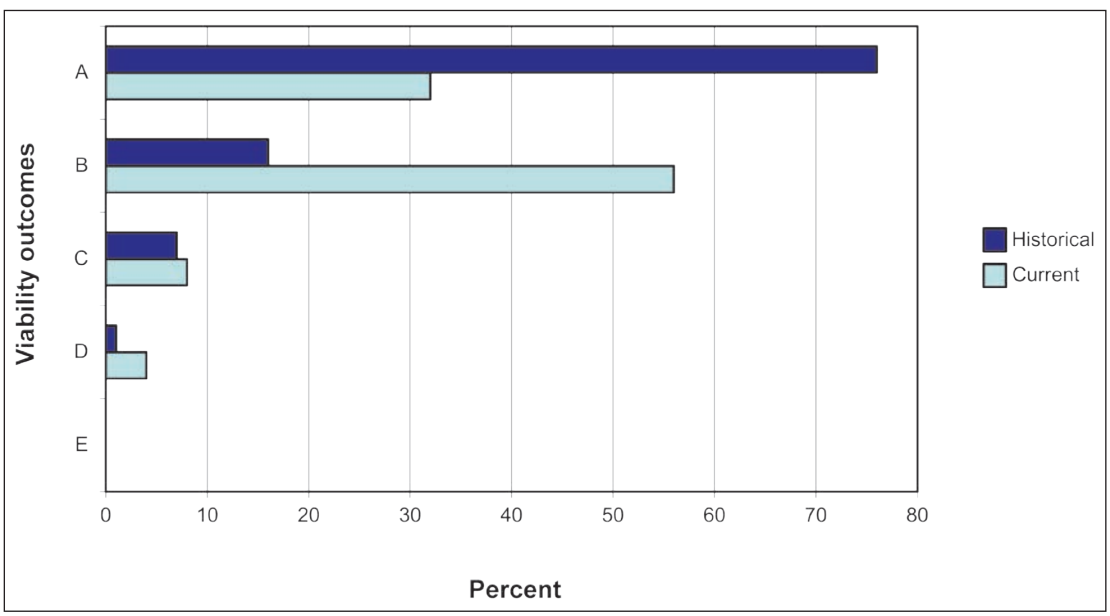

Figure 54—Current and historical viability outcomes for the peregrine falcon in the northeast Washington assessment area.

Management Considerations

The following issue was identified during this assessment and from the published literature for consideration by managers:

1. The effects of disturbance from human activities within source habitats.

Pileated Woodpecker

Introduction

The pileated woodpecker was chosen as a surrogate species to represent species of conservation concern associated with medium-large trees/cool/moist forests group. This species also prefers areas with high densities of large snags and logs for foraging, roosting and nesting, as do many of the other species in this group and family. This species is well distributed across the assessment area year round.

Model Description

Source habitat—

Pileated woodpeckers prefer late-successional stages of coniferous or deciduous forest, but also use younger forests that have scattered, large, dead trees (Bull and Jackson 1995, Bull et al. 2007). In northeastern Oregon, pileated woodpeckers selected unlogged stands of old-growth grand fir (*Abies grandis*) with closed canopies (Bull and Holthausen 1993) and in some cases open stands with high densities of large snags and logs (Bull et al. 2007). These woodpeckers are rarely found in stands of pure ponderosa pine (Bull and Holthausen 1993). They will use Engelmann spruce at high elevation if big trees are present. In western Oregon, pileated woodpecker densities are greater in forests >80 years old than in younger forests (Nelson 1988). Their association with late-seral stages stems from their use of large-diameter snags or living trees with decay for nest and roost sites, large-diameter trees and logs for foraging on ants and other arthropods, and a dense canopy to provide cover from predators (Bull 2003b).

In the Coast Range, mature stands (>70 years) were selected by pileated woodpeckers, and younger stands were avoided for foraging (Mellen 1987). Mannan (1984) reported 44 percent of the foraging occurred in dead trees, 36 percent on downed logs, and the remainder in other substrates. Results of foraging location were similar in northeastern Oregon (Bull and Holthausen 1993).

We described source habitat for this species as (fig. 55):

- Potential vegetation: Dry, mesic or cold-moist
- Cover types: All forested except lodgepole pine and ponderosa pine
- Tree size and structure: >15 in QMD, ≥40 percent canopy closure, single- and multstory

Snag density—

Pileated woodpecker nest cavities are quite large (mean diameter of 8 in and depth of 22 in) and are excavated at an average height of 50 ft above the ground, so nest trees must have a girth large enough to contain nest cavities at this height (Bull 1987). Of 105 nest trees located in northeastern Oregon, 75 percent were in ponderosa pine, 25 percent in western larch, and 2 percent in grand fir; the mean d.b.h. was 33 in (Bull 1987). In western Oregon, 73 percent of nest trees were Douglas-fir (*Pseudosuga menziesii*), and nest trees averaged 27 in d.b.h. (Mellen 1987). In northwest Montana, most of 54 nest trees were large western larch (*Larix occidenatalis*), and nest trees averaged 29.5 in d.b.h. (McClelland 1979).

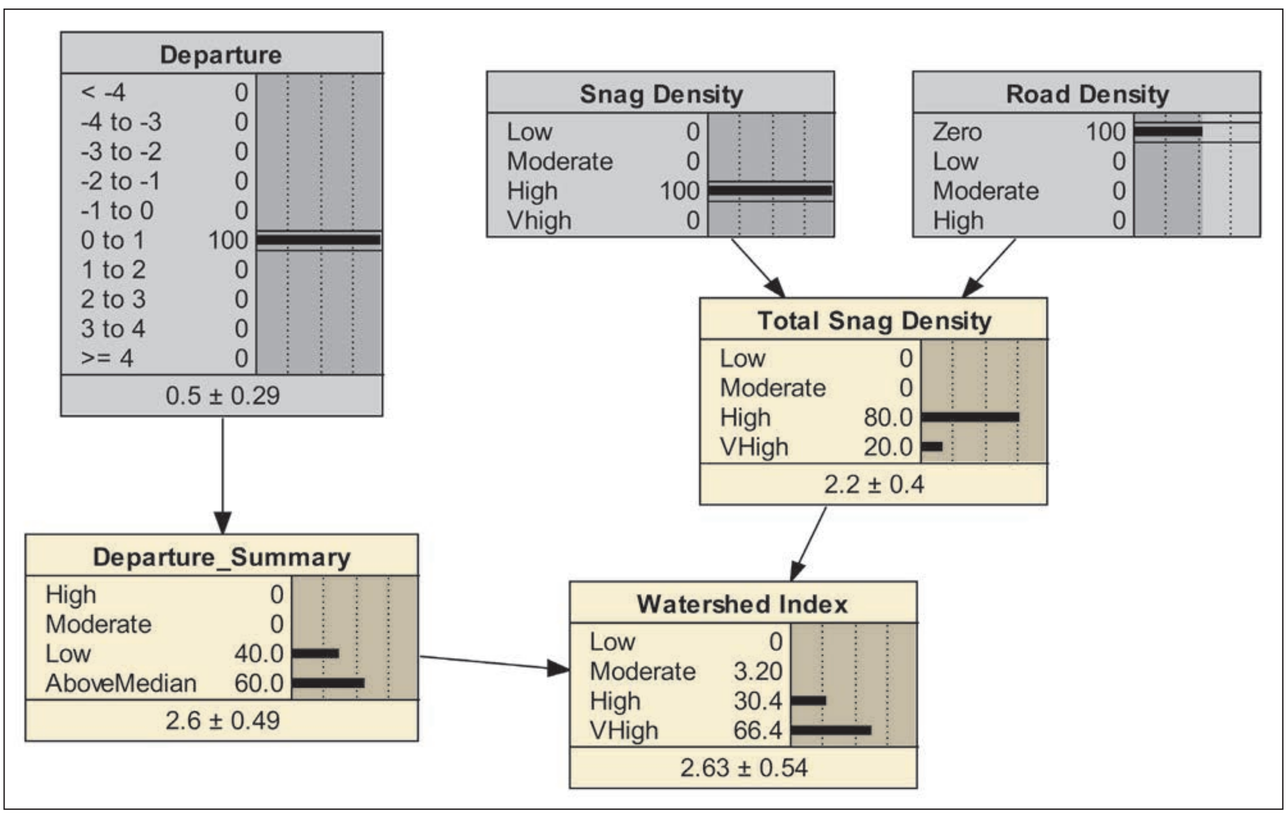

Figure 55—Surrogate species assessment model for pileated woodpecker.

In northeastern Oregon, pileated woodpecker roosts were typically located in a live or dead grand fir with a mean d.b.h. of 28 in (Bull et al. 1992). In the Coast Range, Douglas-fir, red alder, western redcedar, and big-leaf maple contained roosts (Mellen 1987).

Timber harvest has had a negative effect on habitat for this woodpecker (Bull 2003b, Bull et al. 2007). Removal of large-diameter, live and dead trees, down woody material, and forest canopy eliminates nest and roost sites, foraging habitat, and protective cover. In addition, prescribed fire may eliminate or reduce the number of snags, logs, and cover (Bull 2003b).

We calculated the percentage of source habitat within a watershed that had snag densities (>20 in d.b.h.) in the following classes (fig. 55):

- Low: <2.5/ac
- Moderate: 2.6 to 8.9/ac
- High: 9.0 to 39.3/ac
- Very high: >39.3/ac

These density classes were taken from DecAID, as described in the historical range of variability (HRV) methods sections to correspond to the different tolerance levels of the east-side mixed-conifer forest type (Mellen et al. 2006).

Road density—

We included a road density variable to account for likely reduced snag densities along roads. Bate et al. (2007) found that snag numbers were lower adjacent to roads owing to removal for safety considerations, removal as firewood, and other management activities. Other literature has also indicated the potential for reduced snag abundance along roads (Wisdom et al. 2000).

We calculated the percentage of source habitat in the following road density classes (fig. 55):

- Zero: <0.1 mi/mi^2 open roads in watershed
- Low: 0.1 to 1.0 mi/mi^2 open roads in a watershed
- Moderate: 1.1 to 2.0 mi/mi^2 open roads in a watershed
- High: >2.0 mi/mi^2 open roads in a watershed

Calculation of historical conditions—

- Habitat departure: Class 1
- Snag density: High
- Road density: Zero

The relative sensitivity of WI values to variables used in the pileated woodpecker model are shown in table 37.

Table 37—Relative sensitivity of watershed index values to variables in the model for pileated woodpeckers

Variable	Sensitivity rank
Habitat departure	1
Snag density	2
Road density	3

Assessment Results

Watershed scores—

The abundance of closed-canopied late-successional forests in the dry, mesic, and cold-moist forest types has declined from the historical condition. Currently, 76 percent (n = 55) of the watersheds in the assessment area have less habitat than the historical median while the remainder of the watersheds (n = 17) showed little departure from the historical median (fig. 56). These conditions reflected the reduction in late-successional forests that have occurred in the assessment area and are consistent with the findings that Wisdom et al. (2000) reported for the North Cascades and Northern Glaciated Mountains ERUs.

Watersheds containing <740 ac of source habitat currently included the Ashnola, Lower Silkameen River, Upper Chewuch River, Lost River, and the Middle Yakima (fig. 56). The watersheds with the greatest amount of source habitat currently (>24,700 ac) are the Teanaway River, Upper Yakima River, West Fork Sanpoil, White-Little Wenatchee, Little Naches River, Wenatchee River, Stehekin, Chiwawa, and Ross Lake (fig. 56), about 39,500 ac. Historically, the Lower Pend Oreille watershed had the highest estimated amount of source habitat, 74,100 ac, yet we modeled less than 9,900 ac currently.

The availability of snag habitat was an important habitat feature within source habitat for pileated woodpeckers (Bull et al. 1986, 1992; Raphael and White 1984). We assessed the density of large-diameter snags (>20 in d.b.h.) within source habitat for each watershed. In 53 watersheds, ≥50 percent of the current source habitat had snag densities of <2.5 snags/acre (low category). Six watersheds (Icicle River, Pasayten River, White-Little Wenatchee River, Little Naches River, American River, and Bumping River) had >50 percent of the current source habitat with snag densities in the high category (9 to 39 snags/acre).

Road densities were calculated to assess the effects roads have on snag densities. Road densities were variable. Ten watersheds had >50 percent of the source habitat in a road density of high while source habitat in 15 watersheds had snag densities primarily in the “zero” category.

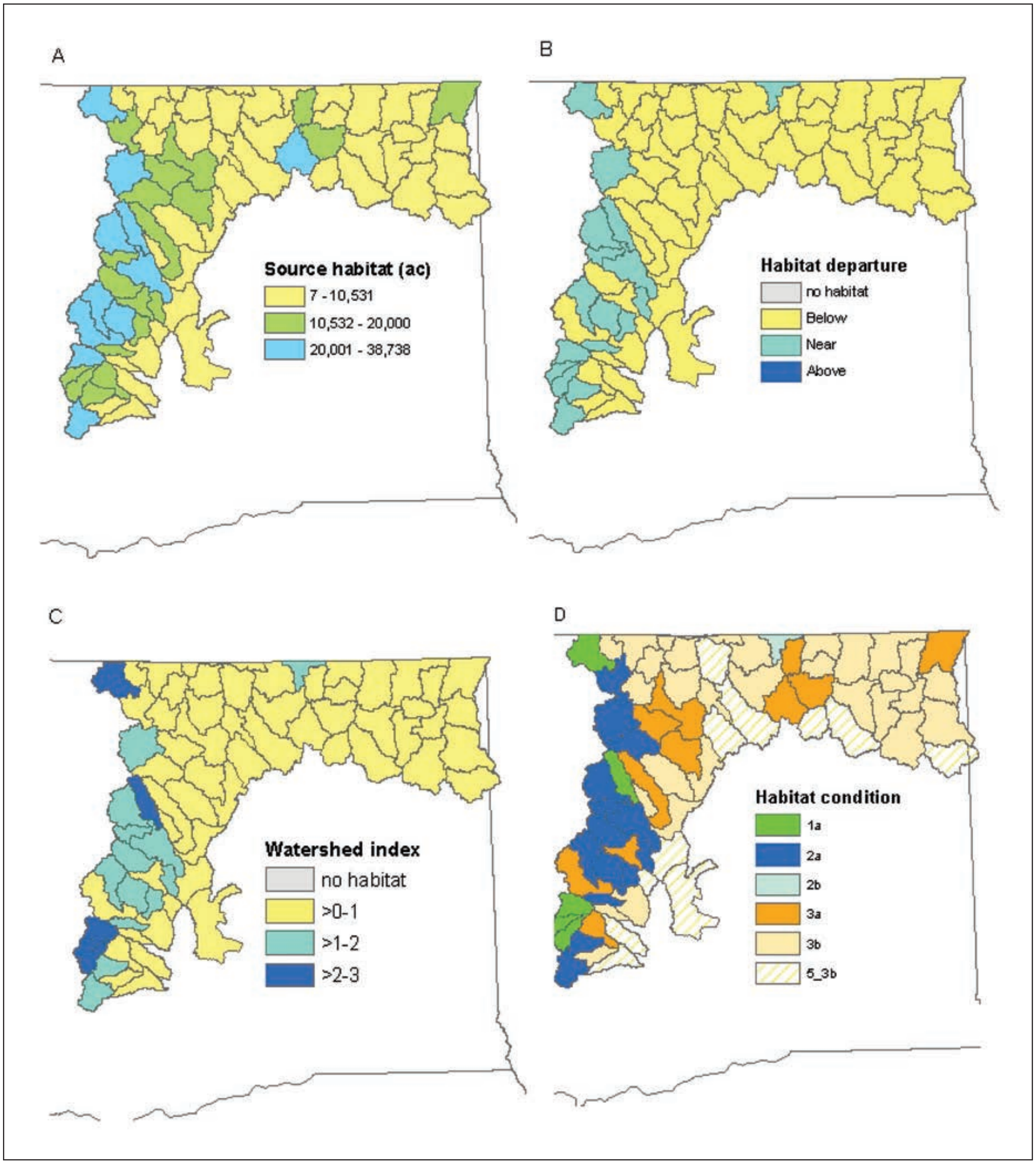

Figure 56—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D) habitat condition class for pileated woodpeckers by watershed in the northeast Washington assessment area.

Because of the overall reduction in amount of source habitat, and lower snag densities when compared to historical conditions, the WI variables are generally low for this species (fig. 56). Seventy-five percent (n = 54) of the watersheds had a current WI value of low (≤1). Eighteen percent (n = 13) had a WI value of moderate (>1 and <2), while 7 percent (n = 5) had high WIs (>2.0). We estimated the WI was historically about 2.6.

Viability outcome scores—
The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for pileated woodpeckers within the assessment area is 45 percent of the historical capability. Dispersal across the assessment area was not considered an issue for this species because of their relatively high mobility. All five ecoregions currently contained at least one watershed with >40 percent of the median amount of historical source habitat. Thirty watersheds had >40 percent of the median amount of historical source habitat (42 percent). The current viability outcome is a 21-percent probability of an outcome B, 71 percent probability of a C outcome, indicating that habitat is likely patchily distributed or isolated and in low abundance (fig. 57).

We estimated that 89 percent (n = 64) of the watersheds contained greater than 40 percent of the median amount of habitat historically and were distributed throughout all five ecoregions. The viability outcome historically was primarily an A outcome indicating habitat was broadly distributed and highly abundant (fig. 57).

In summary, under historical conditions, pileated woodpeckers were likely well distributed throughout the assessment area; currently they are likely not as well distributed, and source habitat is less abundant. Likely, other species in the medium-large trees/cool/moist forests group have experienced similar declines in suitable environments.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Decline in the availability of late-successional source habitats, including large-diameter trees.
2. Loss of large snags in some of the source habitat and in areas that could be managed to provide future source habitat.
3. The future sustainability of dense dry forests with larger trees that have high fuel loads (Hessburg et al. 1999, 2007; Townsley et al. 2004).

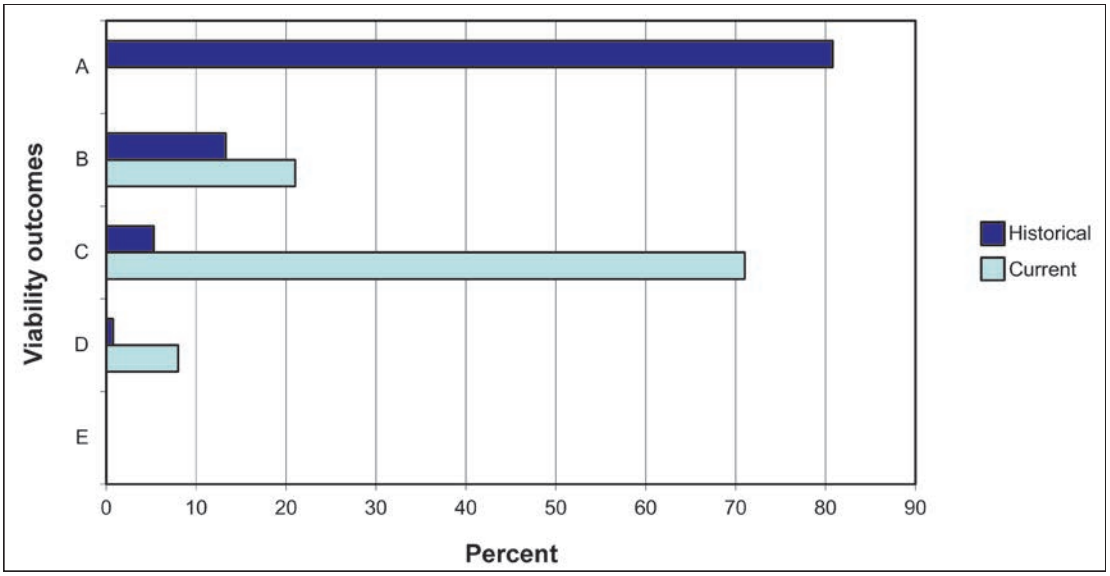

Figure 57—Current and historical viability outcomes for the pileated woodpecker in the northeastern Washington assessment area.

Sage Thrasher

Introduction

Sage thrashers were selected as a surrogate species to represent species of conservation concern associated with the shrub-steppe group in the woodland/grass/shrub family. This species represents the full range of habitats and risks associated with that group, including loss, fragmentation, and degradation of sagebrush (*Artemisia* spp.) habitats. Sage thrashers were distributed throughout the central portion of the assessment area (Smith et al. 1997). Sage thrashers are easily surveyed using standard point count protocols. Sage thrashers were breeding-season residents of the assessment area (Reynolds et al. 1999); this assessment was for nesting habitat.

Model Description

Source habitat—

Probability of occurrence of sage thrashers in shrub-steppe habitats was most directly related to sagebrush cover, total shrub cover, shrub patch size, decreased disturbance, and similarity of habitat within a 0.62-mi radius (Knick and Rotenberry 1995). Sage thrashers were almost entirely dependent on sagebrush habitats during the breeding season (Braun et al. 1976, Dobler et al. 1996, Knick and Rotenberry 1995, McAdoo et al. 1989, Reinkensmeyer et al. 2007). An abundance of breeding individuals has been positively correlated with sagebrush cover and negatively correlated with the cover of annual grasses (Kerley and Anderson 1995, Reynolds et al. 1999, Wiens and Rotenberry 1981). The primary limiting factor for sage thrashers was the loss, alteration, or degradation of sagebrush habitats (Braun et al. 1976, Cannings 2000, Weber 1980). Where complete replacement of native sagebrush habitat with crested wheatgrass (*Agropyron cristatum*) occurred, this species was eliminated (Reynolds and Trost 1980, 1981). Even removal of only large sagebrush in breeding habitats can limit use by thrashers (Castrale 1982). Sage thrashers were least abundant on sagebrush sites in poor condition, suggesting that they were more productive in less disturbed communities (Vander Hagen et al. 2000).

The spread of cheatgrass (*Bromus tectorum*) has had a negative effect on sage thrasher populations through its influence on fire regimes in western grasslands (Knick and Rotenberry 1997). Fires pose a threat to sage thrashers in terms of habitat loss as sagebrush does not resprout after burning (Castrale 1982). Kerley and Anderson (1995) found that sage thrashers were not present on burned areas 9 years after a fire, and areas treated with herbicide had low sage thrasher populations 22 years after treatment. Although Petersen and Best (1987, 1999) found that sage thrasher abundance was unaffected by prescribed burning, which resulted in a mosaic of burned and unburned areas in southeastern Idaho, Welch (2002), McIntyre (2002), and Holmes (2007) reported that sage thrasher presence was reduced or did not occur on burned sagebrush sites.

For this analysis, source habitat for sage thrashers was considered to be the shrub-steppe vegetation zone (fig. 58). All potential habitat that burned since 1998 was removed from consideration as source habitat in the model.

- Vegetation zone: Shrub-steppe
- Cover type: Shrub-steppe

Habitat effectiveness—

Density of sagebrush obligate birds (including sage thrashers) was reported to decrease 39 to 60 percent within a 330-ft buffer of roads with low traffic volumes (Ingelfinger and Anderson 2004). As a result, we assumed that roads have a negative effect on the effectiveness of source habitat for sage thrashers. We assessed the potential for human disturbance to affect source habitat of sage thrashers with an adaptation of the habitat disturbance index described in Gaines et al. (2003). We buffered open roads by 330 ft on each side and then intersected this with our map of

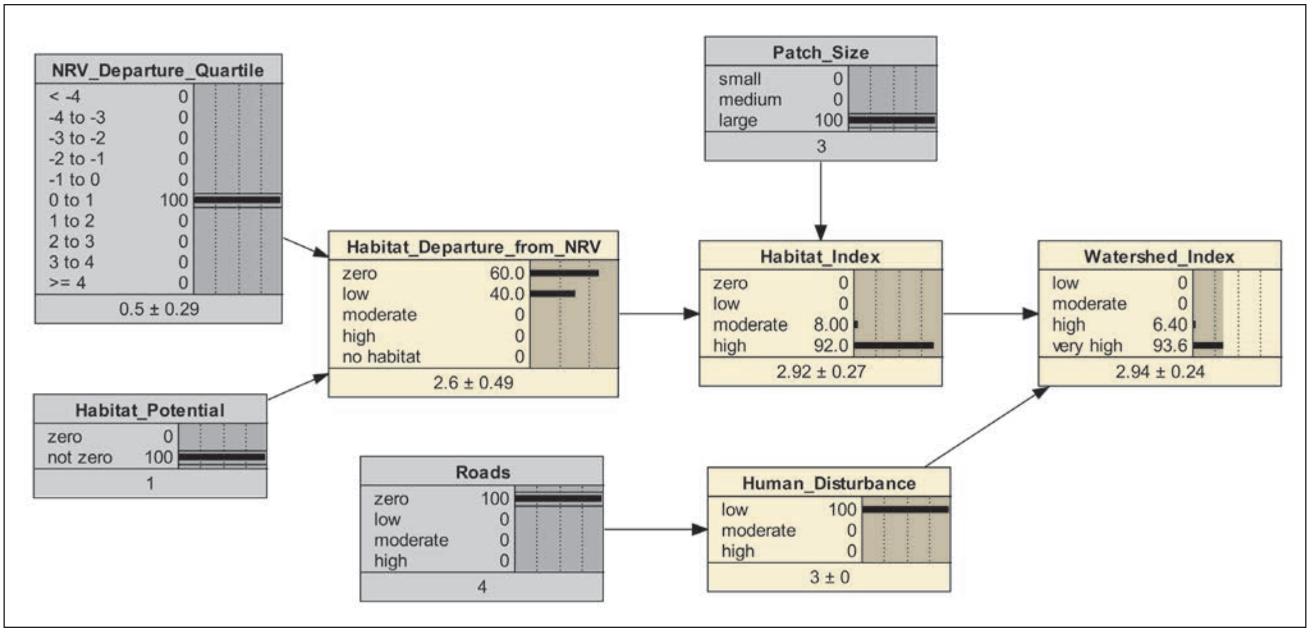

Figure 58—Surrogate species assessment model for sage thrashers.

source habitat. We then used the following categories to estimate the potential effects of human disturbance on sage thrashers for each watershed (fig. 58):

- Low: >75 percent of the source habitat outside road and trail buffer within a watershed
- Moderate: 50 to 75 percent of the source habitat outside road and trail buffer within a watershed
- High: <50 percent of the source habitat outside road and trail buffer within a watershed

Patch size—

Knick and Rotenberry (1995, 2002) reported that sage thrashers were highly sensitive to fragmentation of shrublands in southeast Idaho. Also, Vander Haegen et al. (2002) found higher predation rates on nests of sage thrashers in small patches of sagebrush (median 360 ac) compared to large patches (median 285,100 ac). Although Vander Haegen et al. (2000) reported that sage thrashers were not area-limited in eastern Washington state and were often found nesting in small habitat patches (<25 ac), subsequent analyses indicated that birds nesting in small patches experienced reduced nest success when compared to birds nesting in large habitat patches (Vander Haegen 2007). This lower reproductive success was manifested in lower rates of nest survival, largely as a result of increased predation on nests. Thus, small patches of sagebrush were reproductive sinks for this species. The following classes were used to describe the effect of patch size on habitat quality (fig. 58):

- Small: 0 to <1,240 ac mean patch size of sagebrush habitat within a watershed
- Moderate: 1,240 to 2,470 ac mean patch size of sagebrush habitat within a watershed
- Large: >2,470 ac mean patch size of sagebrush habitat within a watershed

Variables considered but not included—

Grazing—Heavy grazing pressure has been reported to affect sage thrasher populations negatively (Bradford et al. 1998, Kerley and Anderson 1995), but they may be less sensitive to intensive grazing than other birds associated with shrub-steppe habitats (Kantrud and Kologiski 1982, Reynolds and Trost 1981). Saab et al. (1995) further reviewed several studies where heavy grazing resulted in a positive response in sage thrasher abundance. Because of the equivocal nature of the reported effects of grazing on sage thrashers, this variable was not included in the model.

Calculation of historical conditions—

- Departure of source habitat from HRV: 0.5
- Roads: Class low
- Patch size: Class large

The relative sensitivity of WI values to the variables used in the model for sage thrasher are shown in table 38.

Assessment Results

Watershed scores—

Historically, 67 of 72 watersheds within the assessment area provided habitat for sage thrashers (i.e., >50 ac of habitat within the watershed). This analysis indicated

Table 38—Relative sensitivity of watershed index values to variables in the model for sage thrashers

Variable	Sensitivity rank
Habitat departure	1
Patch size	2
Road density	3

that 31 percent (n = 22) of watersheds within the assessment area currently provided habitat for sage thrashers. Watersheds with the greatest amounts of habitat were concentrated in the central portion of the assessment areas, including Columbia River-Lynch Coulee, Okanogan River-Omak Creek, and Upper Columbia-Swamp Creek (fig. 59). However, within Okanogan River-Omak Creek and Upper Columbia-Swamp Creek <25 percent of the source habitat was managed by federal agencies. The watersheds with the least amount of source habitat were located across the eastern and western portions of the assessment area. All watersheds with habitat had low WI scores (>0.0 but <1.0) (fig. 59). A major factor that influenced the WI scores was the amount of source habitat compared to levels historically available in the watersheds (fig. 59).

Road density also affected suitability of watersheds as habitat for sage thrashers (Ingelfinger and Anderson 2004). However, road density in source habitat was generally low across the assessment area.

Viability outcome scores—

The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for sage thrashers within the assessment area is 32 percent of the historical capability. Dispersal across the assessment area was not considered an issue for these species owing to their relatively high mobility. Four of five ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Eighteen percent (n = 13) of the watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there is a 50 percent probability that the current viability outcome for sage thrashers is D and a 40 percent probability that the current viability outcome for these species is E, indicating a patchy to isolated distribution and low abundance of source habitat (fig. 60). Other species associated with the shrub-steppe group in the woodland/grass/shrub family likely have similar outcomes.

Historically, dispersal across the assessment area was not considered an issue for this species. Four of five ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Fifty-eight percent (n = 42) of the watersheds had >40 percent of the median amount of historical source habitat. Under those circumstances, there was a 66.5 percent probability that the historical viability outcome for sage thrashers was A and a 21.5 percent probability that the historical viability outcome for these species was B, indicating habitat was broadly distributed and highly abundant (fig. 60).

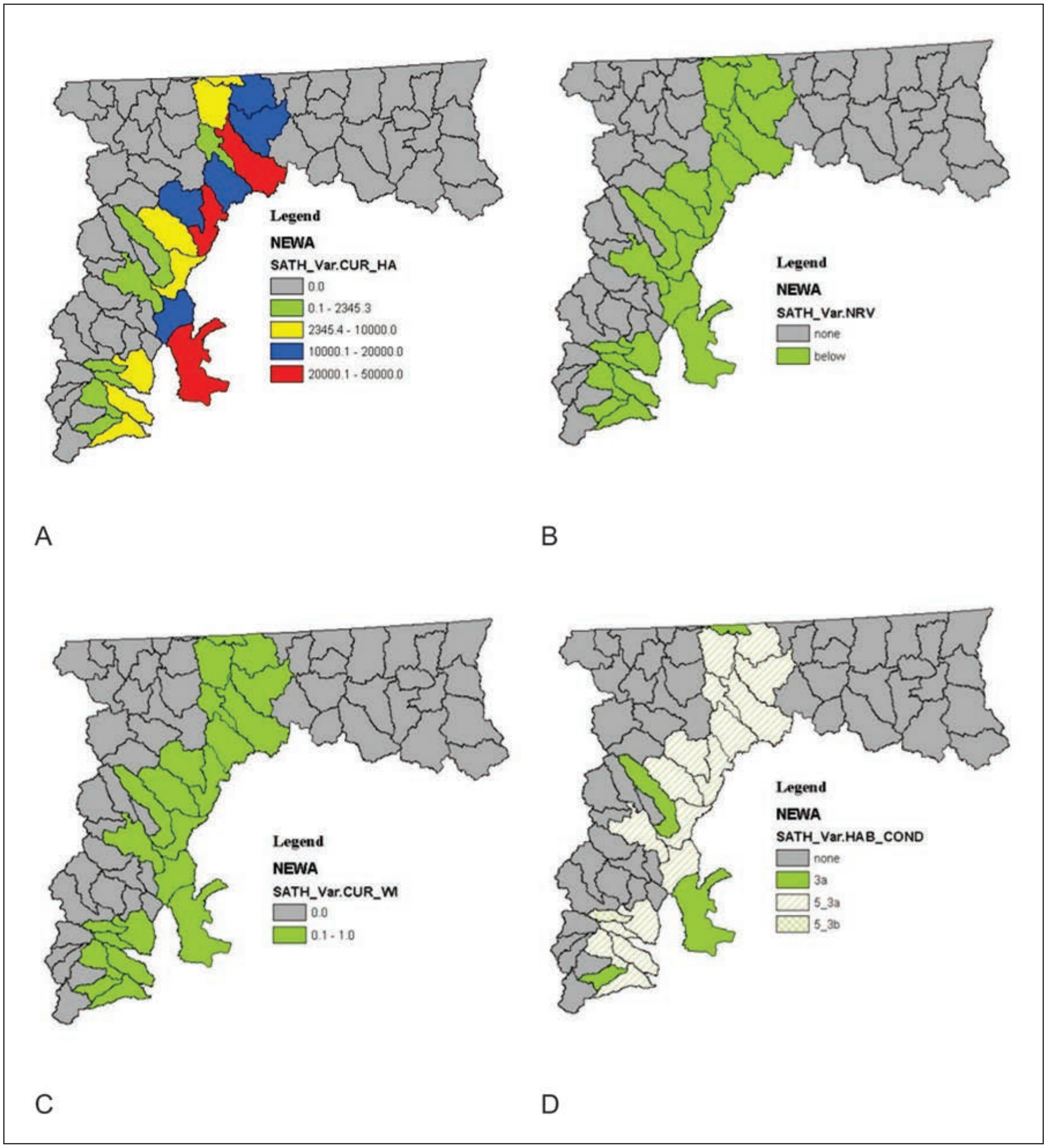

Figure 59—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D) habitat condition class for sage thrashers (SATH) by watershed in the northeast Washington assessment area (NEWA).

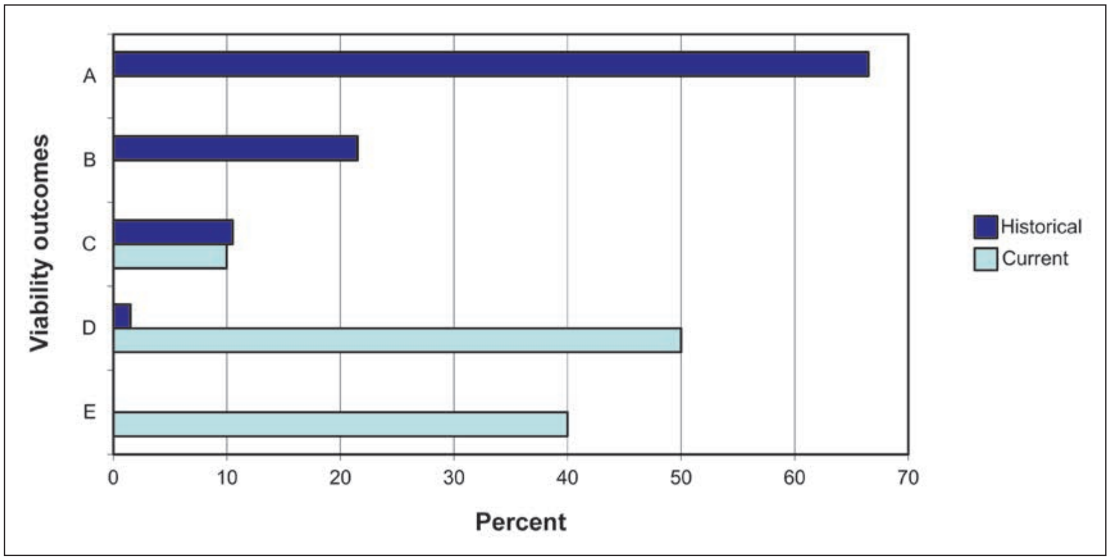

Figure 60—Current and historical viability outcomes for sage thrashers in the northeast Washington assessment area.

In summary, under historical conditions, sage thrashers and other species associated with the shrub-steppe group in the woodland/grass/shrub family were likely well distributed throughout the assessment area; currently, they are not well distributed, have limited opportunity for interactions among populations, and are likely to be extirpated.

Our results for this species were similar to those reported in the broad-scale habitat analysis by Wisdom et al. (2000) in the ICBEMP. According to the ICBEMP terrestrial vertebrate habitat analyses, historical source habitats for sage thrashers included portions of the northern Cascade Range and the Northern Glaciated Mountains ERUs that overlap our assessment area (Wisdom et al. 2000). Within this historical habitat, declines in source habitats have been extensive—71 percent in the Northern Cascades and -84 percent in the Northern Glaciated Mountains according to Wisdom et al. (2000).

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Reduction and fragmentation of suitable shrub-steppe source habitats from historical levels.
2. Negative effects of roads and motorized trails within source habitats.

Tailed Frog

Introduction

The tailed frog was selected as a surrogate species to represent the conifer riparian group, specifically habitats associated with moderate-elevation streams. Only one other species, the black swift, occurs in this group, and the swift is associated with steep cliffs near waterfalls. Locations of the black swift in the assessment area are few. Tailed frogs occur in mountainous streams on the west and east side of the Cascade Range and in the Coast Ranges of western Oregon and Washington (Leonard et al. 1993). In addition, they have a disjunct distribution that includes the Blue Mountains (Leonard et al. 1993). Tailed frog distribution within the assessment area is limited to the east side of the north Cascade Range, and they do not occur on the portion of the Okanogan-Wenatchee National Forest east of the Okanogan River or on the Colville National Forest (Dvornich et al. 1997a).

Model Description

Source habitat—

Tailed frogs reside in and next to perennial mountain streams (Dupuis et al. 2000). Mating, egg-laying, and larval development occur in streams. Adult female frogs deposit egg masses beneath large relatively stable cobbles or boulders in the summer, and hatchlings emerge the following spring. At northern latitudes, it takes up to four additional summers for tadpoles to metamorphose and begin a life of both lotic and terrestrial activity (Brown 1990, Daugherty and Sheldon 1982). Thus, the larval life stages are particularly vulnerable to land uses that alter channel conditions (Aubry 2000, Bull and Carter 1996, Bury 1983, Corn and Bury 1989, Dupuis and Steventon 1999, Welsh and Ollivier 1998).

Amphibian populations, and specifically tailed frogs, are believed to be at risk from the effects of timber harvesting in both upland and riparian zones (Aubry 2000, Bull and Carter 1996, Bury and Corn 1988, Bury 1994, Dupuis 1997). Considerable efforts have been made to understand these effects on the west-side forests of Oregon, Washington, and British Columbia (Aubry 2000, Biek et al. 2002, Dupuis and Steventon 1999, Stoddard and Hayes 2005, Vesely and McComb 2002). These studies have reported differences in amphibian community composition depending on stand age (Aubry 2000) and width of riparian buffers (Veseley and McComb 2002), positive associations with the presence of amphibians and old forests adjacent to streams (Stoddard and Hayes 2005), and amphibian population declines following clearcut harvest (Dupuis and Steventon 1999). In the drier interior forests east of the Cascade Crest in Washington, Piper (1996) conducted monitoring of tailed frogs on the Wenatchee National Forest in areas with and without regeneration timber harvest adjacent to the streams. She found that the number of tailed frog captures were considerably less where timber harvest had occurred. However, since the implementation of the Northwest Forest Plan, watersheds, including those that provide tailed frog source habitats, have received considerable protection from the negative effects associated with timber harvest and road building, improving conditions for riparian-associated species (Gallo et al. 2005).

Owing to our limited ability to map riparian habitats, we assumed that the amount of habitat currently available was about the same as the amount of habitat that was historically available. Therefore, our assessment for the tailed frog focused on factors that influenced habitat quality and not factors that may have caused habitat loss. We modeled source habitat for tailed frogs using a combination of stream order, cover type, and tree structure. Our model included the following GIS layers (fig. 61).

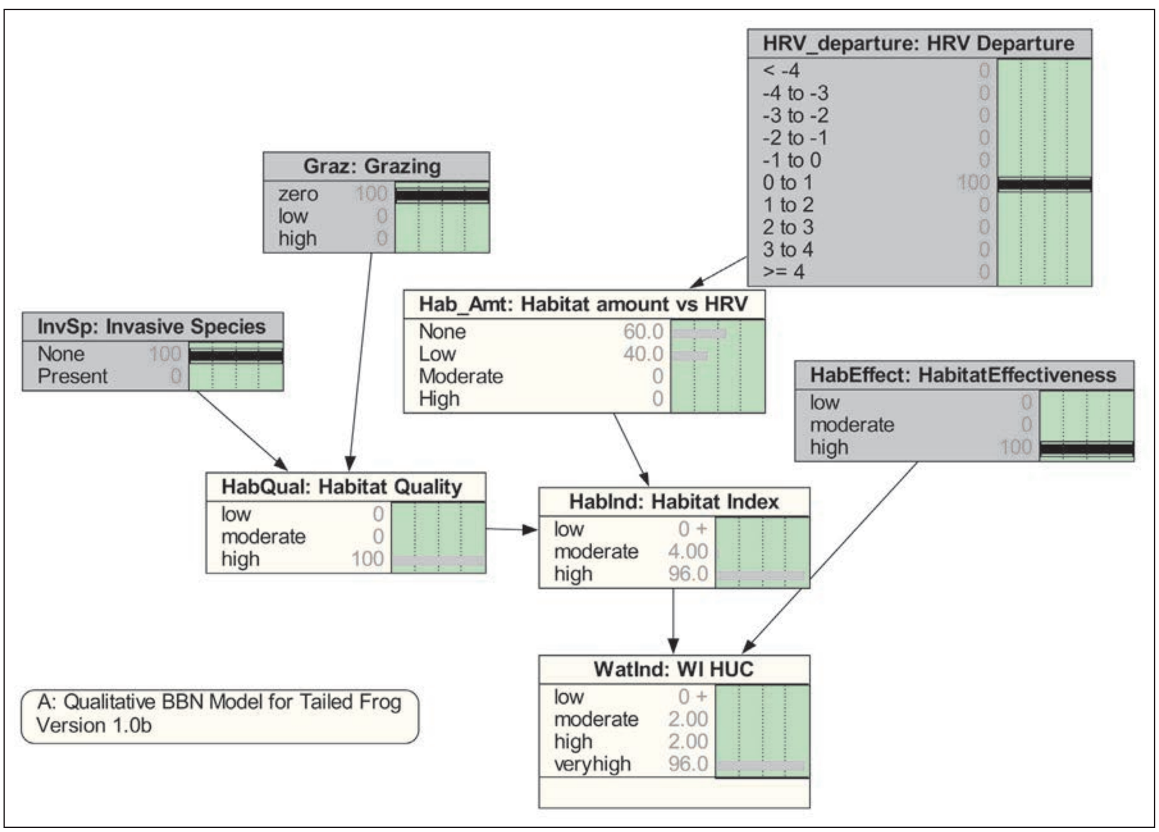

Figure 61—Surrogate species assessment model for the tailed frog.

Forest cover types: Douglas-fir, grand fir, western hemlock, Pacific silver fir, Engelmann spruce, western redcedar, mountain hemlock, subalpine fir, riparian deciduous

Tree structure and size: Single/multistory, >38 cm d.b.h. QMD

Canopy closure: >50 percent

Stream orders: 3 to 5 with 328-ft buffer on each side

Grazing—

We found no studies on the effects of grazing on tailed frogs; several studies have shown that livestock grazing can change the composition and quality of riparian habitats, cause soil compaction, and streambank trampling (see Krausman 1996 and Wales 2001 for reviews). Of particular importance is the potential for grazing to contribute sedimentation to a stream providing tailed frog habitat (Waters 1995, Welsh and Ollivier 1998). Thus, we accounted for the potential effects of grazing on tailed frogs by mapping cattle grazing allotments (with attributes to identify active allotments) and overlaying these onto maps of tailed frog source habitat (fig. 61). We used the following categories to assess these potential impacts within each watershed:

- Zero: No source habitat within an active cattle grazing allotment
- Low: <25 percent of the source habitat within an active cattle grazing allotment
- High: >25 percent of the source habitat within an active cattle grazing allotment

Habitat effectiveness—

Roads can influence riparian habitats for amphibians by removing habitat, limiting the ability of amphibians to disperse across roads, creating a source of mortality, and as a source of fine sediment deposited in amphibian habitats (Demaynadier and Hunter 2000, Dupuis and Steventon 1999, Fahrig et al. 1995, Welsh and Ollivier 1998, Yanes et al. 1995). Roads can contribute sediment to streams and reduce the densities of tailed frogs (Welsh and Ollivier 1998). In addition, where roads intersect with riparian habitats, removal of hazard trees for safety and snags for woodcutting can alter the structure of riparian habitats. This is most likely to occur within 200 ft on each side of a road (Hamman et al. 1999, Gaines et al. 2003, Wisdom and Bate 2008). We assessed the potential impacts of roads on tailed frogs using road density within source habitat as an indicator of the effects of roads on habitat effectiveness (fig. 61). To estimate road density, a moving-windows routine with a 0.9-km (0.6-mi) radius circular window was used. We used the riparian route density index (described in Gaines et al. 2003) to assess the amount of source habitat within different road density classes and assigned each watershed to a level of habitat effectiveness:

- Low habitat effectiveness: >25 percent of the source habitat with road densities >2 mi/mi^2
- Moderate habitat effectiveness: >25 percent of the source habitat with road densities >1 mi/mi^2
- High habitat effectiveness: <25 percent of the source habitat with road densities >1 mi/mi^2

Invasive species—
Studies have shown the negative effects of nonnative trout on amphibian communities (Dunham et al. 2004, Hecnar and M'Closkey 1997) and specifically on tailed frog occurrence (Feminella and Hawkins 1994). We used fish distribution data collected during stream surveys to determine if the presence of nonnative fish were likely present within source habitats in each watershed (fig. 61).

Calculation of historical conditions—

- Source habitat: Depature class 1
- Grazing: Zero
- Habitat effectiveness: High
- Invasive species: Not present

The relative sensitivity of WI values to variables used in the model for tailed frog are shown in table 39.

Model evaluation—
We compared mean WI values derived from 279 points where tailed frogs were known to occur to 146 random points. The mean WI values for the occurrence points (1.89) were significantly higher (t = 1.97, P = 0.0008) than the mean derived from the random points (1.71) indicating that our model was effective in identifying watersheds with suitable environments for tailed frogs.

Assessment Results

Watershed scores—
Forty-eight watersheds on the Okanogan-Wenatchee National Forest were evaluated for tailed frogs; they are not known to occur on the Colville National Forest. Eleven of the watersheds (23 percent) had WI scores that were high (>2.0), 12 watersheds

Table 39—Relative sensitivity of watershed index values to variables in the model for tailed frogs

Model variables	Order of variable weighting
Source habitat	1
Grazing	2
Habitat effectiveness	4
Invasive species	3

(25 percent) had moderate scores (1.0 to 2.0), and 26 watersheds (54 percent) had low scores (<1) (fig. 62). Watersheds that had some of the greatest abundance of habitat (>3,700 ac), and high WI scores were Ross Lake, Icicle Creek, and Stehekin (fig. 62). Other watersheds that had large amounts (>2,500 ac) of tailed frog source habitat, but moderate or low WI scores, included Sinlahekin Creek, Little Naches River, Peshastin Creek, White-Little Wenatchee, Cle Elum River, Chiwawa River, Teanaway River, Upper Yakima River, and Wenatchee River (fig. 62).

Road densities in tailed frog source habitat were high, leading to overall low habitat effectiveness. Within 19 percent (n = 9) of the watersheds, there was a high level of habitat effectiveness for tailed frog source habitat, in 23 percent (n = 11) of the watersheds there was a moderate level of habitat effectiveness, and in 58 percent (n = 28) of the watersheds habitat effectiveness was low.

The assessment of the amount of source habitat for tailed frogs in active grazing allotments had mixed results. Our analysis showed that in 27 percent (n = 13) of the watersheds, cattle were not grazed in source habitat; in 35 percent (n = 17) of the watersheds, >0 to 25 percent of the source habitat was in an active cattle grazing allotment; and in 38 percent (n = 18) of the watersheds, >25 percent of the source habitat was in an active cattle grazing allotment.

Although 90 percent (n = 43) of the watersheds have nonnative trout present, the true impact of this on tailed frog populations is not known, and it is a risk factor that is in need of further investigation and monitoring. We calibrated the overall negative effect of this risk factor to be relatively small owing to uncertainty in the effects of this risk factor.

Viability outcome scores—

The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for the tailed frog within the assessment area is 43 percent of the historical capability.

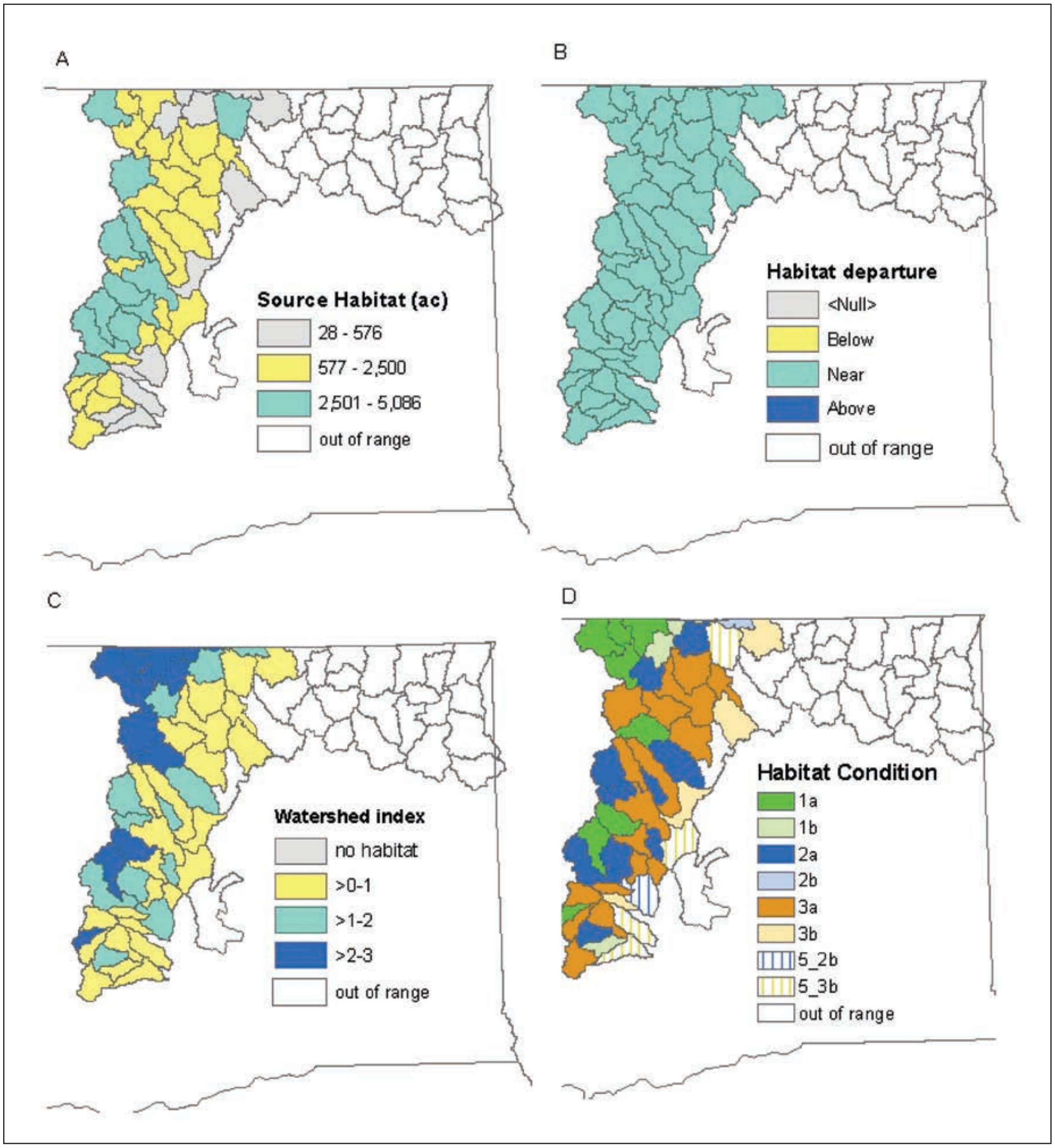

Figure 62—Current amount of (A) source (acres), (B) current habitat departure, (C) watershed index, and (D) habitat condition class for tailed frogs by watershed in the northeast Washington assessment area.

Currently and historically, 75 percent (n = 36) of the watersheds contain source habitats that were estimated to be above 40 percent of the historical median. The watersheds with >40 percent were distributed across all of the three ecoregions in which the tailed frog is distributed.

The current viability outcome for tailed frogs in the assessment area is a 73 percent probability of C and a 23 percent probability of B, which indicates that suitable environments are distributed frequently as patches or existed at low abundance (fig. 63). Gaps in suitable environments are likely large enough such that some subpopulations are isolated, limiting opportunity for interspecific interactions. Historically, the viability outcome for tailed frogs was estimated to have a 76 percent probability of A, and a 16 percent probability of B with some gaps in suitable environments occurring naturally (fig. 63).

In summary, the abundance and distribution of suitable environments for the tailed frog, and likely other species associated with the conifer riparian group, have been reduced. The viability of species associated with this habitat could be enhanced by habitat restoration.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Grazing has a negative influence on quality of tailed frog habitat in some watersheds.

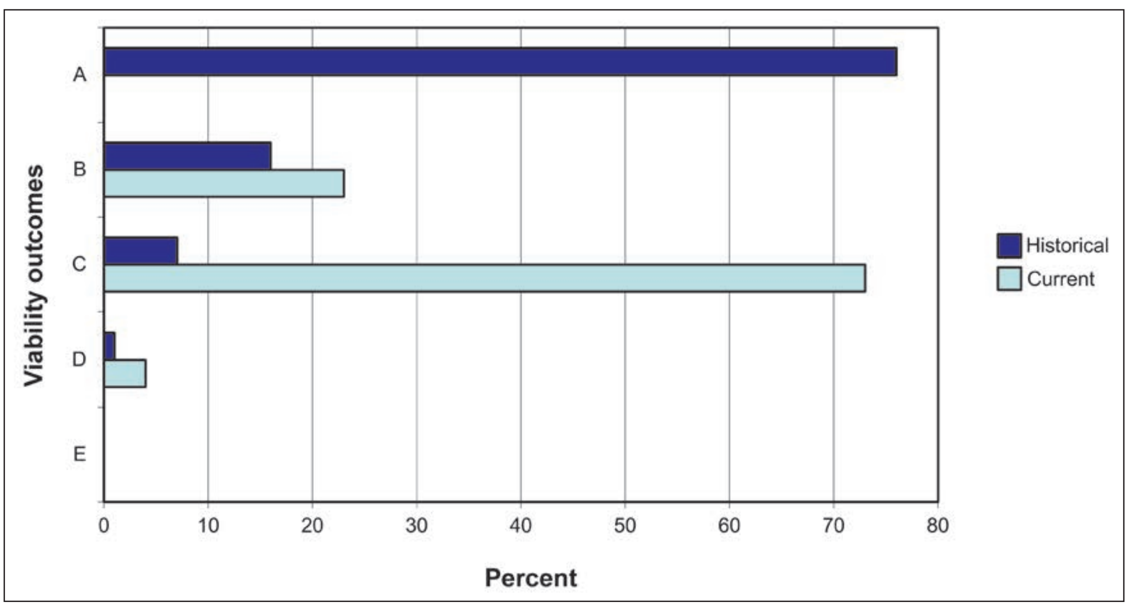

Figure 63—Current and historical viability outcomes for the tailed frogs in the northeast Washington assessment area.

2. Currently, roads are having widespread impacts on tailed frog source habitats in a high proportion of watersheds.
3. Vegetation management within source habitat may reduce habitat availability, increase sedimentation, and affect stream temperatures.

Tiger Salamander

Introduction

The tiger salamander was selected as a surrogate species for the grass/shrub group owing to their specific association with wetland and ponds that occur within dry forest and shrub-steppe habitats not represented by other surrogate species in the family or group. In Washington, the tiger salamander occurs in portions of the Columbia Basin and northeastern Washington, and they range in elevation from 670 to 3,000 ft (Leonard et al. 1993, Nussbaum et al. 1983). Habitat occurs on the Colville and Okanogan-Wenatchee National Forests with the exception of Kittitas and Yakima Counties; the Columbia River is considered a barrier to their expansion into these counties (Dvornich et al. 1997).

Currently, roads are having widespread impacts on tailed frog source habitats in a high proportion of watersheds.

Model Description

Source habitat—

Tiger salamanders use a variety of seasonal and permanent water bodies, including lakes, reservoirs, and farm ponds (Leonard et al. 1993). Previous efforts to model tiger salamander habitats used open water and wetlands that occurred within shrub-steppe and open dry forests (Dvornich et al. 1997). Important features of breeding sites include persistence of water until larval development is complete (from mid-March to mid-August), shallow (generally <3 ft) water depths along at least portions of the water body, soft bottom substrate, abundant emergent vegetation, suitable cover for metamorphs (amphibians that have recently transformed to the adult stage) along the shoreline, and absence of introduced fish (COSEWIC 2001, Sarell 2004). Outside the breeding period, terrestrial tiger salamanders use grassland, shrub-steppe, and open forest habitats (COSEWIC 2001). Important habitat features include friable soils that permit burrowing, rodent burrows for shelter, and availability of food (COSEWIC 2001, Sarell 2004, Semlitsch 1998).

For this assessment, to identify source habitat for the tiger salamander, we used the National Wetlands Inventory and the cover type GIS data layers (fig. 64).

- National Wetlands Inventory: All lacustrine and palustrine wetlands
- Cover types/vegetation zones: Shrub-steppe, ponderosa pine

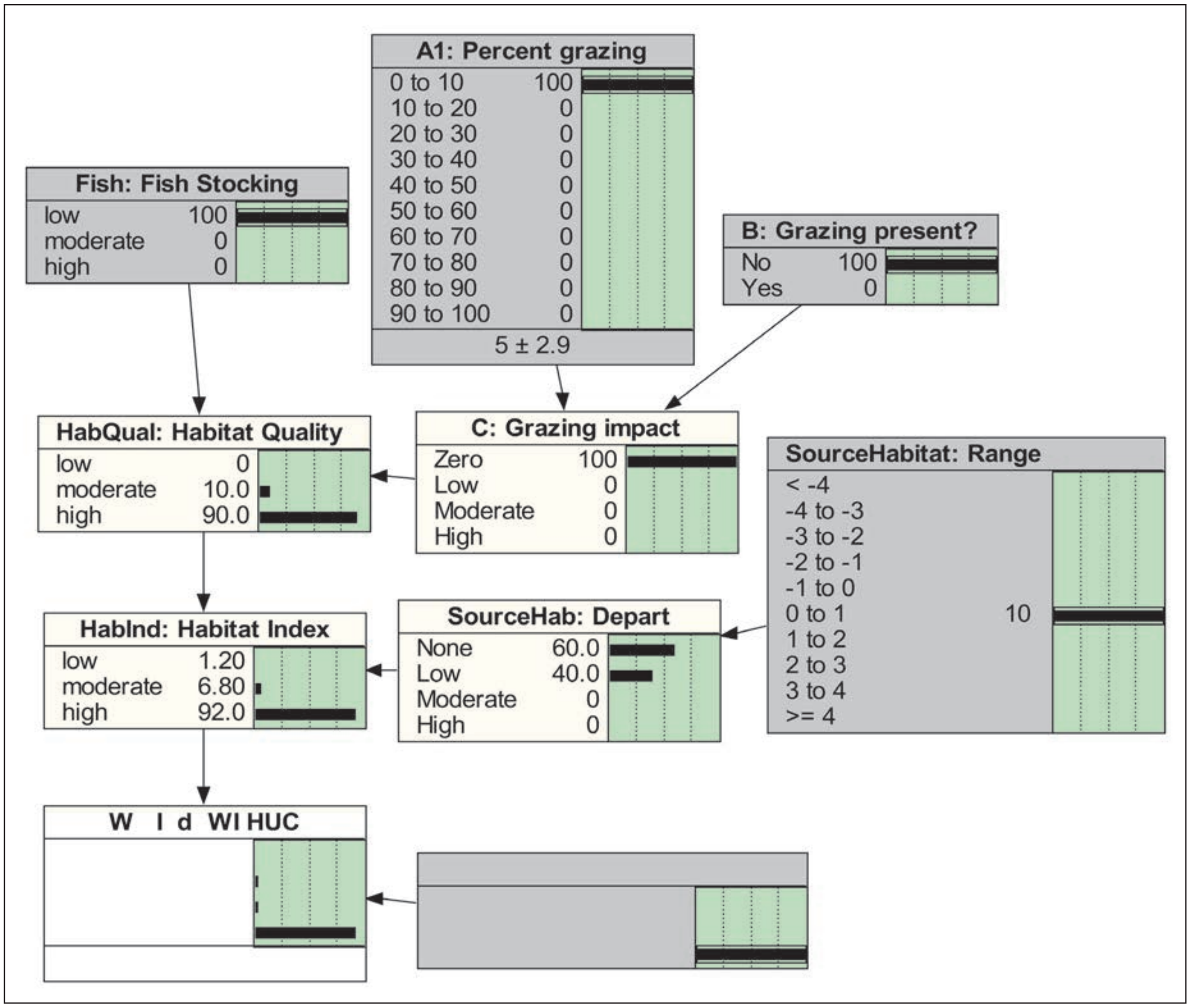

Figure 64—Surrogate species assessment model for the tiger salamander.

Grazing—

While studies were not found on the effects of grazing on tiger salamanders, several studies have shown that livestock grazing can change the composition and quality of riparian habitats, cause soil compaction, streambank trampling, and increased nutrient input to water (see Krausman 1996, Sarell 2004 , Wales 2001 for reviews). Leege et al. (1981) found litter to be an important habitat component for many small mammals, reptiles, and amphibians, with litter biomass twice as high in livestock exclosures as compared to grazed areas. Heavy livestock use near the ponds can cause the collapse of small mammal burrow entrances needed for aestivation (Harvey et al. 2000). Thus, we accounted for the potential effects of grazing on tiger salamanders by mapping grazing allotments (with attributes to identify active allotments) and overlaying these onto maps of source habitats. We then categorized the amount of source habitat in an active grazing allotment using 10 percent increments from 0 to 100 percent, with increasing poorer habitat outcomes as the proportion of source habitat in an active allotment increased (fig. 64).

Invasive animals—
Several studies have documented the negative effects of nonnative fishes on amphibians (see Dunham et al. 2004 for a review), and some studies have specifically addressed effects to tiger salamanders (Collins et al. 1988, Corn et al. 1997, Fisher and Shaffer 1996). The effects that nonnative fishes had on amphibians include direct mortality from predation, competition for food resources, and displacement from important habitats (Dunham et al. 2004, Fisher and Shaffer 1996, Hecnar and M'Closkey 1997).

To assess the potential impacts of nonnative fishes and fish stocking, we reviewed the fish stocking records (available from Washington Department of Fish and Wildlife and digitized by U.S. Forest Service) for areas that comprised source habitat for tiger salamanders. We then categorized the amount of source habitat that received fish stocking for each 5^{th}-field watershed as follows (fig. 64):

- Low: <25 percent of the source habitats had fish stocking
- Moderate: 25 to 50 percent of the source habitats had fish stocking
- High: >50 percent of the source habitats had fish stocking

Habitat effectiveness—
Studies of the effects of roads on amphibians have documented road-related mortalities (Ashley and Robinson 1996), reduced permeability to amphibian movements as a result of edge effects (Gibbs 1998), and roads as partial barriers to movements (DeMaynadier and Hunter 2000, Marsh et al. 2005). To account for these potentially negative effects in our tiger salamander model, we compiled roads information from the national forests and Washington Department of Natural Resources. We used a moving windows analysis to estimate road density within source habitat for each watershed. We then used the following categories to assess habitat effectiveness (fig. 64):

- Low habitat effectiveness: >25 percent of the source habitat with road densities >2.0 mi/mi^2
- Moderate habitat effectiveness: >25 percent of the source habitat with road densities >1.0 mi/mi^2
- High habitat effectiveness: <25 percent of the source habitat with road densities >1.0 mi/mi^2

Calculation of historical conditions—

- Source habitat: Habitat departure class -2
- Grazing: None
- Invasive animals: Low
- Habitat effectiveness: High

The relative sensitivity of WI values to the variables used in the model for tiger salamander are shown in table 40.

Assessment Results

Watershed scores—

We evaluated 58 watersheds as having potential habitat for tiger salamanders in the assessment area. Seventy-four percent (n = 43) of the watersheds had WI scores that were high (>2.0) and they were widely distributed across the assessment area. In addition, 26 percent (n = 15) of the watersheds had moderate (1.0 to 2.0) WI scores (fig. 65).

Watersheds that currently have the greatest amount of source habitat include the Lower Okanogan River, Okanogan River-Omak Creek, Upper Okanogan River, Middle Methow River, and Okanogan River-Bonaparte Creek (fig. 65). The median amount of source habitat across all of the watersheds with at least some source habitat was 754.4 ac.

Fish stocking was estimated to be at a moderate level in all of the watersheds. Twelve percent of the watersheds (n = 7) had >50 percent of the source habitat within an active grazing allotment, 7 percent (n = 4) had 25 to 50 percent of the source habitat in an active grazing allotment, and 81 percent (n = 47) had <25 percent of the source habitat in an active grazing allotment. Most (71 percent, n = 41) of the watersheds had a high level of habitat effectiveness, 26 percent (n = 15) had a moderate level of habitat effectiveness, and 3 percent (n = 2) had a low level.

Viability outcome scores—

The VOI model incorporated the WWI scores (described earlier) and a habitat distribution index. The WWI portion of the viability outcome model showed that the current habitat capability is 48 percent of the historical habitat capability. This score is largely influenced by a reduction in the availability of source habitat from historical conditions. Thirty-five (48 percent) of the watersheds have source habitat that was >40 percent of the historical median. These watersheds are distributed across all of the ecoregions within the distribution of the tiger salamander.

Table 40—Relative sensitivity of watershed index values to variables in the model for tiger salamander

Model variables	Order of variable weighting
Source habitat	1
Grazing	2
Fish stocking	3
Habitat effectiveness	4

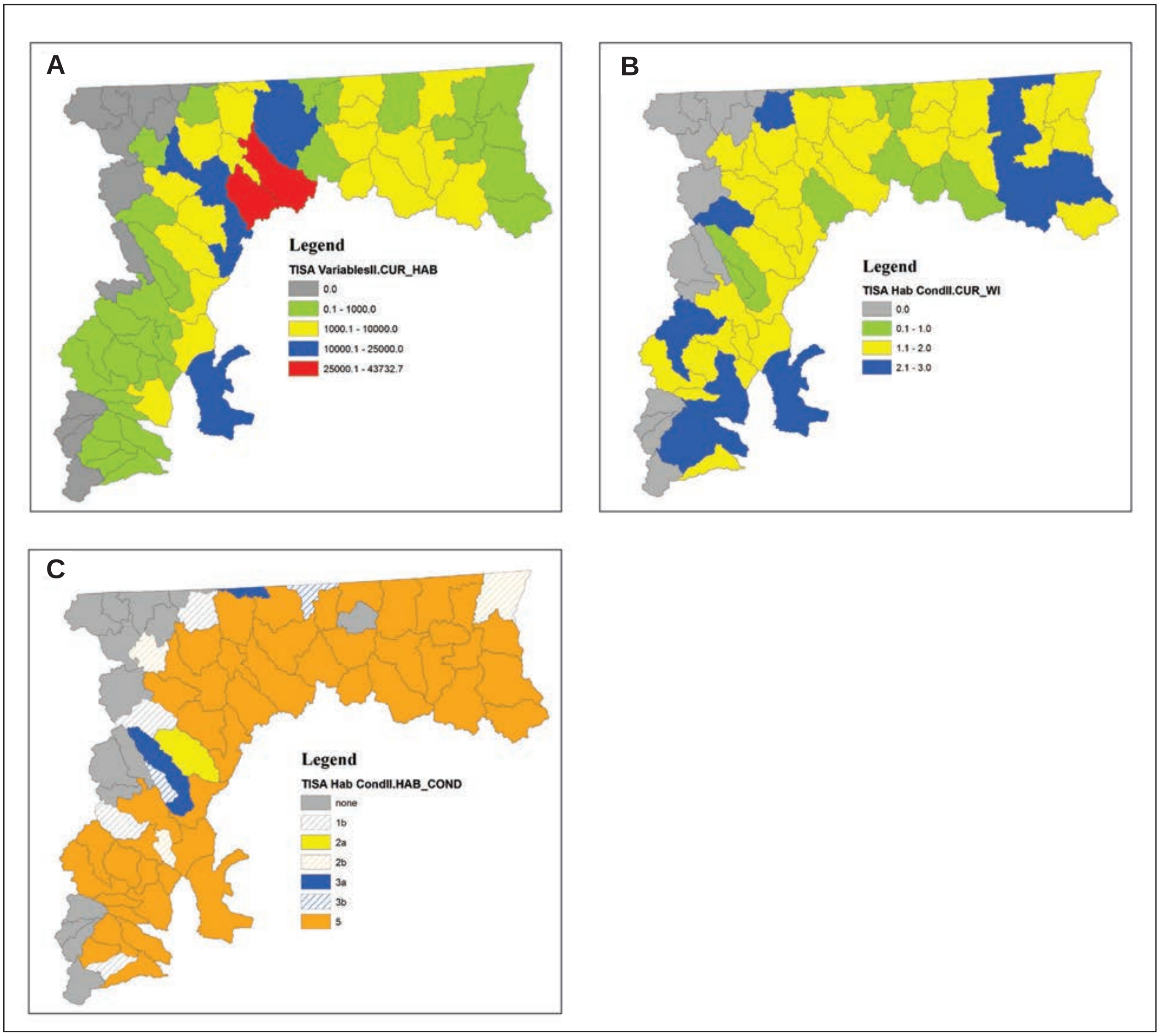

Figure 65—Current amount of (A) source habitat (acres), (B) watershed index, and (C) habitat condition class for tiger salamanders (TISA) by watershed in the northeast Washington assessment area.

Historically, 50 percent of the watersheds had source habitat amounts >40 percent of the historical median of all watersheds. The watersheds with >40 percent were distributed across all four of the ecoregions that occur within the range of the tiger salamander.

Currently, the viability outcome for the tiger salamander is estimated to be a 71 percent probability of outcome C and a 21 percent probability of outcome B, which suggests that suitable environments are distributed frequently as patches or exist in low abundance (fig. 66). Species with this outcome are likely well distributed in only a portion of the assessment area. Historically, viability outcomes were a 66.5 percent probability of outcome A and a 21.5 percent probability of B where suitable environments were more broadly distributed or of high abundance (fig. 66). In addition, the suitable environments were better connected, allowing for relatively more interspecific interactions. A reduction in the availability of suitable environments for the tiger salamander may have occurred in the assessment area compared to the historical distribution and condition of their habitats. However, only 12 watersheds have source habitat amounts in which >25 percent of habitat is located in federal ownership. This greatly limits the contribution that federal lands can make to the viability of this species.

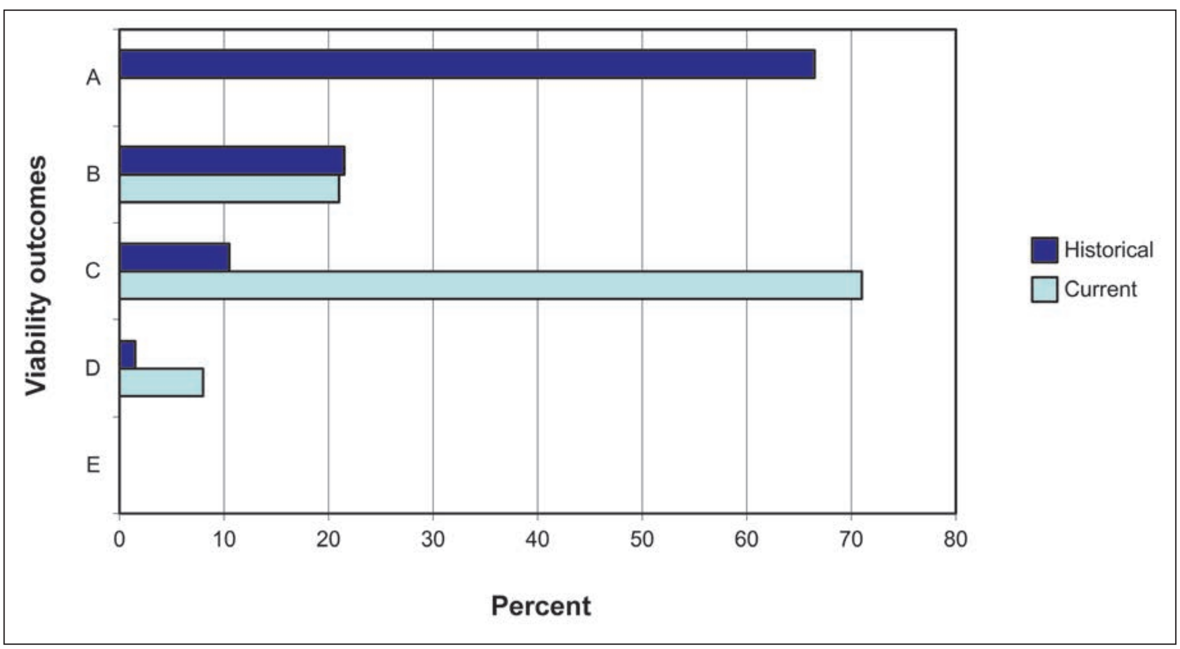

Figure 66—Current and historical viability outcomes for the tiger salamander in the northeast Washington assessment area.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Fish stocking can influence the survival of juvenile amphibians.
2. Grazing has reduced the quality of source habitat in some watersheds.
3. Roads can influence the survival of amphibians when roads occur in proximity to source habitats.
4. The amount of source habitat that is on federal lands is limited across the assessment area and limits the contribution that federal lands can make to the viability of this species.

Townsend's Big-Eared Bat

Introduction

The Townsend's big-eared bat was selected as a surrogate species to represent the unique habitat of the chambers and caves group (Marcot 1984, Nagorsen and Brigham 1993) along with other bat species. In addition, these bats use large trees and snags (Feller and Pierson 2002, Mazurek 2004). Townsend's big-eared bats are moth specialists but also consume a variety of other arthropods when available (Nagorsen and Brigham 1993, Ross 1967, Whitaker et al. 1977). In Washington, Townsend's bats are found in west-side lowland conifer-hardwood forest, ponderosa pine forest and woodlands, mixed highland conifer forest, east-side mixed-conifer forest, shrub-steppe, and both east- and west-side riparian wetlands (Johnson and Cassidy 1997, WDFW 2005).

Though Townsend's big-eared bats are relatively widespread across the assessment area, with at least one site occurring on or adjacent to each of the national forests, they are rare owing to their restrictive roosting requirements (Johnson and Cassidy 1997, Woodruff and Ferguson 2005). Because of the limited number of known locations for this species, we did not develop a surrogate species assessment model but rather provide a qualitative assessment of Townsend's habitat relationships and general management considerations.

In Washington, old buildings, silos, concrete bunkers, barns, caves, and mines are common roost structures (WDFW 2005). In northwestern California, both individuals and nursery colonies have been located in very large trees (Feller and Pierson 2002, Heady and Frick 2001, Mazurek 2004).

Risk factors—

The risk factors that were identified for this species include loss of roost sites because of cave and mine closures and destruction of abandoned buildings, disturbance of roosting bats from human activities, mortality of roosting bats or total loss of colonies because of vandalism and shooting, and reduction in prey base (moths) through use of insecticides.

The loss of roosts is a large concern because new mines are not being created at the rate they are being lost, and abandoned buildings are becoming much less common (Woodruff and Ferguson 2005). In addition, loss of large hollow trees that might serve as valuable roosts may occur during either wild or prescribed fire.

Because insecticides reduce insects that are a potential source of prey, insecticide use near hibernacula and nursery roosts may limit populations, especially if the bats leaving the hibernacula are nursing (Humphrey and Kunz 1976, Sample 1991, Wackenhut 1990).

Disturbance of roost by humans (e.g., recreation, mining, bat research, vandalism) is noted as a concern by many researchers (Ellison et al. 2005, Pierson and Rainey 1998, Woodruff and Ferguson 2005).

Management Considerations

The following issues were identified during this assessment from published literature for consideration by managers:

1. Caves and mines may provide summer or winter roosting habitat or both.
2. Big-eared bats may be sensitive to human disturbance during critical periods (for hibernacula 15 May to 15 September, for nursery sites 15 September to 1 April). It is especially important to consider the potential for the spread of white-nosed syndrome that is facilitated by human visitation.
3. Application of pesticides near bat roosts may cause disturbance and reduce prey populations.
4. Research and monitoring may cause disturbance of roosts.
5. Large trees and snags are key habitats for bats that are known to be below historical levels (Hann et al. 1997, Harrod et al. 1999, Hessburg et al. 1999a).
6. Buildings may be used by Townsend's bats as roosting habitat.

Various Bat Species

Introduction

We identified 11 species of bats as species of conservation concern. We placed these species among four of the family groups described in chapter 1. They were placed in the medium to large tree forests, open forests, woodland/grass/shrub, and chambers/caves groups (see table 41). The general habitats of these species are described by the Western Bat Working Group; their known roosting sites, and a list of desired conservation actions by species are identified in the Oregon Conservation Strategy (ODFW 2005). The fringed myotis, pallid bat, and Townsend's big-eared bat were chosen as surrogate species for their particular groups, largely owing to their high dependence on unique and not necessarily widespread roosting sites.

However, we did not develop WI models for any of these species. We felt we did not have the knowledge to adequately map habitat and develop a model at this scale for these species. We have described habitat variables researchers have found important for all bats in general.

Source habitat—

Bats use resources at the landscape scale. Land management considers the juxtaposition of all habitat components: roosting, foraging areas, and water resources. It is suspected that the closer the essential components are to each other (e.g., less than several miles: Keinath 2004 fringed myotis assessment), the higher the likelihood of persistence. Hayes and Loeb (2007) added clutter vegetation that has the potential to impede bat echolocation and flight to this list of habitat attributes that play a critical role in defining niches for bats.

Roost sites—

Suitable characteristics of roost sites differ among species and sex (Broders and Forbes 2004), and optimal thermal conditions at roost likely differ with species, reproductive status, weather, age, and time of year (Hayes and Loeb 2007). A recent meta-analysis of tree roost selection of North American forest bats showed that roost trees of bats were tall with large d.b.h. and in stands with open canopy and high snag density (Kalcounis-Ruppell et al. 2005). However, Hayes (2003) suggested that an overreliance on one habitat type or topographic setting for retaining roost habitat is unlikely to provide the conditions necessary to meet the habitat needs for bats across seasons. For example, thermal characteristics of riparian areas often differ from upslope forests so the exclusive retention of snags and wildlife trees in riparian areas is not likely to be in the best interest of bat conservation.

In addition, the ephemeral nature of snag roosts and the movement by colonies of bats among several snags within seasons indicate that tree-roosting bats require areas of high snag density, perhaps more so than cavity-nesting birds (Baker and Lacki 2006, Rabe et al. 1998). Baker and Lacki (2006) suggested forest management practices target and set aside large-diameter (e.g., >24 in d.b.h.) snags surrounded by snag densities of ≥100/ac in snag management efforts directed toward

Table 41—Summary of the conservation status, habitat relations, and conservation actions for various bat species within the northeastern Washington assessment area

Family	Species	Focal[a]	Status[b]	Western Bat Working Group generalized habitat description	Roosting sites	Potential conservation actions (from Oregon Conservation Strategy)
Medium- to large-tree forests	Hoary bat (*Lasiurus cinereus*)		I	Solitary and roosts primarily in foliage of both coniferous and deciduous trees, near the ends of branches, 3 to 12 m above the ground. Roosts are usually at the edge of a clearing. Some unusual roosting situations have been reported in caves, beneath a rock ledge, in a woodpecker hole, in a grey squirrel nest, under a driftwood plank, and clinging to the side of a building.	Trees, rocks	Investigate data gaps and use results to guide management actions.
Medium- to large-tree forests	Silver-haired bat (*Lasionycteris noctivagans*)		I	Maternity roosts appear to be almost exclusively in trees—inside natural hollows and bird-excavated cavities or under loose bark of large-diameter snags. Roosting sites are generally at least 15 m above the ground. Males and females change roosts frequently, and use multiple roosts within a limited area throughout the summer, indicating that clusters of large trees are necessary.	Trees	Maintain late-successional conifer habitats; maintain and create large-diameter hollow trees, and large-diameter, tall, and newly dead snags during forest management activities.
Medium- to large-tree forests	Long-legged myotis (*Myotis volans*)		I	A bat primarily of coniferous forests. Also occurs seasonally in riparian and desert habitats. Uses abandoned buildings, cliff crevices, exfoliating tree bark, and hollows within snags as summer day roosts; caves and mine tunnels as hibernacula.	Trees, snags, rocks, cliffs, caves, mines	Maintain and create large-diameter hollow trees and large-diameter tall, newly dead snags in riparian and upland habitat; maintain and restore diverse riparian areas; complete bridge replacement and maintenance when bats are absent.

Table 41—Summary of the conservation status, habitat relations, and conservation actions for various bat species within the northeastern Washington assessment area (continued)

Family	Species	Focal[a]	Status[b]	Western Bat Working Group generalized habitat description	Roosting sites	Potential conservation actions (from Oregon Conservation Strategy)
Open forest	California myotis (*Myotis californicus*)		I	While typical of deserts and interior basins in the Western United States, *M. californicus* also occurs in forested and montane regions. During summer, roost alone or in small groups in caves, mines, rocky hillsides, under tree bark, and in buildings. Recent studies in Canada have documented maternity colonies of up to 52 individuals roosting under sloughing bark, and in cracks and hollows of large-diameter, intermediate-stage snags (preferably ponderosa pine).	Trees, caves, mines, buildings	Maintain and create large snags during forest management activities; complete bridge replacement and maintenance when bats are absent.
Open forest	Fringed myotis (*Myotis thysanodes*)	F	I	Appears to be most common in drier woodlands (oak, ponderosa pine) but is found in a wide variety of habitats including desert scrub, mesic coniferous forest, grassland, and sage-grass steppe. Roosts in crevices in buildings, underground mines, rocks, cliff faces, and bridges. Roosting in decadent trees and snags, particularly large ones, is common throughout its range in the Western United States and Canada.	Trees, snags, cliffs, rocks, mines, buildings	Use gates and seasonal closures to protect known hibernacula; maintain and create large-diameter hollow trees and large-diameter tall, newly dead snags during forest management activities.
Open forest	Long-eared myotis (*Myotis evotis*)		I	Occurs in semiarid shrublands, sage, chaparral, and agricultural areas, but is usually associated with coniferous forests. Individuals roost under exfoliating tree bark, and in hollow trees, caves, mines, cliff crevices, sinkholes, and rocky outcrops on the ground.	Trees, snags, cliffs, rocks, mines, buildings	

Table 41—Summary of the conservation status, habitat relations, and conservation actions for various bat species within the northeastern Washington assessment area (continued)

Family	Species	Focal[a]	Status[b]	Western Bat Working Group generalized habitat description	Roosting sites	Potential conservation actions (from Oregon Conservation Strategy)
Woodland/ grass/shrub	Western small-footed myotis (*Myotis ciliolabrum*)		I	Occurs in deserts, chaparral, riparian zones, and western coniferous forest; it is most common above piñon-juniper forest. Individuals are known to roost singly or in small groups in cliff and rock crevices, buildings, concrete overpasses, caves, and mines	Trees, cliffs, rocks, caves, mines, buildings	
Woodland/ grass/shrub	Yuma myotis *(Myotis yumanensis)*		I	Occurs in a variety of habitats including riparian, arid scrublands and deserts, and forests. Roosts in bridges, buildings, cliff crevices, caves, mines, and trees. Individuals become active and forage just after sunset, feeding primarily on aquatic emergent insects.	Trees, cliffs, rocks, caves, mines, buildings	
Woodland/ grass/shrub	Spotted bat *(Euderma maculatum)*		I	Has been found in vegetation types that range from desert to subalpine meadows, including desert-scrub, pinyon-juniper woodland, ponderosa pine, mixed-conifer forest, canyon bottoms, rims of cliffs, riparian areas, fields, and open pasture. During summer, bats may travel from roosts in desert-scrub to forage in high-elevation meadows, returning to roosts within an hour of dawn.	Cliffs, caves	Maintain open water sources in desert landscapes. Manage rock features such as cliffs to avoid conflict with recreational use and rock removal. Maintain and restore native shrub-steppe habitat.

Table 41—Summary of the conservation status, habitat relations, and conservation actions for various bat species within the northeastern Washington assessment area (continued)

Family	Species	Focal[a]	Status[b]	Western Bat Working Group generalized habitat description	Roosting sites	Potential conservation actions (from Oregon Conservation Strategy)
Woodland/ grass/shrub	Pallid bat (*Antrozous pallidus*)	F	I	Day and night roosts include crevices in rocky outcrops and cliffs, caves, mines, tree boles, cavities in oaks, exfoliating ponderosa pine and valley oak bark, deciduous trees in riparian areas, and fruit trees in orchards), and various human structures such as bridges (especially wooden and concrete girder designs), barns, porches, bat boxes, and human-occupied as well as vacant buildings. They forage over open shrub-steppe grasslands, oak savannah grasslands, open ponderosa pine forests, talus slopes, gravel roads, lava flows, fruit orchards, and vineyards.	Trees, cliffs, rocks, caves, mines, buildings	Use gates and seasonal closures to protect known roost sites during sensitive times (raising young and hibernation). Maintain open water sources in dry landscapes. Manage rock features such as cliffs to avoid conflict with recreational use and rock removal. Complete bridge replacement and maintenance when bats are absent. Maintain large pine snags in shrub-steppe/forest ecotones. Maintain and restore native grassland, shrub-steppe and open ponderosa pine habitats.
Chambers/ caves	Townsend's big-eared bat (*Corynorhinus townsendii*)	F	C	Reported in a wide variety of habitat types ranging from sea level to 2.1 mi. Habitat associations include coniferous forests, mixed meso-phytic forests, deserts, native prairies, riparian communities, active agricultural areas, and coastal habitat types. Distribution is strongly correlated with the availability of caves and cavelike roosting habitat, including abandoned mines. Foraging associations include edge habitats along streams, adjacent to and within a variety of wooded habitats.	Caves	Use gates and seasonal closures to protect known roost sites during sensitive times (raising young and hibernation). Maintain buildings used as roosts. Maintain and create large-diameter hollow trees during forest management activities. Monitor roosts.

a = F = Surrogate species, b = I = species of conservation concern.; C = USFWS canadiates species conservation of bat-roosting habitat. A study in northern California on *Myotis thysanoides* found that regular pockets containing over 200 large (>24 in d.b.h.) snags/ac may be necessary to support populations of this species (Weller and Zabel 2001). Also, because of the short time that bark remains on snags (sloughing bark is used for roosting by bats), bats require higher early decay snag densities than birds (Ellison et al. 2005, Rabe et al. 1998).

Maintaining roost trees and replacements across the landscape in a variety of topographic settings is a logical and conservative approach that should provide the broad spectrum of conditions necessary to meet the varying needs of bats.

Efforts to restore ponderosa pine forest with reintroduction of fire could result in the loss of large-diameter trees, dead tops, and snags (Rancourt et al. 2008). Management strategies should be implemented to protect these large defective trees and snags during forest restoration. Selective thinning of areas with dense ponderosa pine surrounding potential roost trees and removal of excess duff and debris around the base of the tree(s), dead-top or snag prior to burning may help protect these potential roost sites.

Recreational rock climbing is increasing in popularity in Washington. The cracks and crevices in rock faces that provide attractive sites for climbers also provide sites for bat roosting. High climbing activity may displace roosting bats and increase threats to species of concern.

Limited research suggests that vegetative structure and habitat that surrounds caves may have an influence on use of caves as roosts by some species or in some situations, but not on others (Raesly and Gates 1987, Wethington et al. 1997).

Foraging—

When foraging, bats often move along forest edges more than within the forest interior (Black 1974, Crampton and Barclay 1996, de Jong 1994, Kunz and Martin 1982). This may facilitate orientation but may also maximize contact with insect prey. When comparing bat foraging activity among forests, clearcuts, and water bodies, activity was found to be higher around water bodies (Lunde and Harestad 1986). Other researchers have also found that foraging areas usually encompassed a body of open water or riparian corridor (Grindal et al. 1999, Waldien and Hayes 2001). Forested corridors connecting forested patches have been shown to provide valuable foraging habitat as well as travel corridors for bats between roosting and foraging sites (van Zyll de Jong 1995).

Bat activity has been found to be higher in thinned stands than in unthinned stands (Humes et al 1999, Loeb and Waldrop 2007); however, this effect may vary by forest type (Patriquin and Barclay 2003, Tibbels and Kurta 2003). Bat activity is highly variable in space and time (Broders 2003, Ellison et al. 2005, Hayes 1997) owing to variation in prey availability, weather conditions, and proximity to roosts (Loeb and Waldrop 2007).

Prescribed fire, wildfire, fire suppression, and fire management all influence insect populations and thus may affect bat populations. However, the influences of fire on insects depend on the timing of the fire with, respect to the life history of insects, the intensity of the fire, the fire's rate of spread, and the area affected by the fire. As a result, the impact of fire and fire management on prey availability for bats and on the ecology of bats is generally poorly understood (Carter et al. 2002, Hayes and Loeb 2007).

Use of insecticides and herbicides likely influences prey availability. The influence of the chemicals applied, the ecological context, and bat-prey relationships have not been well studied. Insecticides can have a direct effect on prey availability; herbicides can have an indirect effect on insect populations by changing the abundance and composition of the plant communities (Guynn et al. 2004); however, no data are available on the effects of chemical treatments and bat-prey relationships (Hayes and Loeb 2007).

Large-diameter or tall snags and wildlife trees within forest stands with the following characteristics loose bark, dead or broken tops, natural cavities, or woodpecker cavities provide important habitats for bats.

Water resources—

Daily water loss in bats is extreme compared to other mammals, largely owing to the respiratory demands imposed by flight (Studier and O'Farrell 1980). Land management activities that alter bodies of water, water regimes, or water quality may affect bats and should be carefully evaluated. Management activities such as livestock grazing of mountain meadows, springs, and riparian zones should be managed to retain native vegetation, natural hydrological regimes, and water quality sources in order to retain habitat of prey species and quality sources of open water for drinking.

Management Considerations

The following issues were derived from the published illustrator for consideration by managers:

1. Large-diameter or tall snags and wildlife trees within forest stands with the following characteristics loose bark, dead or broken tops, natural cavities, or woodpecker cavities provide important habitats for bats.
2. Snags in clumped or clustered patterns across the landscape better address frequent roost switching that occurs with many forest-dwelling bats.
3. Snags, live cavity trees, and trees with evidence of heart rot within intact habitat patches provide important habitat for bats.

4. Restoration of fire to forest stands can meet management objectives. Periodic low-intensity burning in some forest systems could help maintain a more open understory and reduce small-tree density that impedes bat flight.
5. Livestock grazing of mountain meadows, areas near springs, and riparian zones can influence native vegetation, natural hydrological regimes, and water quality sources that affect habitat of prey species and quality sources of open water for drinking.
6. An area within a 0.21-mi radius of the roost (Keinath 2004) may be most suceptible to human disturbances.

Western Bluebird

Introduction

The western bluebird was identified as the surrogate species for the open-forest/all forest group because it is widely distributed in open, low-elevation forests and is limited by the availability of snags with existing cavities. The bluebird represents the array of risk factors of snags and grazing common to other members of the group. Some species in the group use down wood; it is assumed that if snags are present for the bluebird, down wood will be available as snags fall.

Model Description

Source habitat—

Western bluebirds are found in open coniferous and deciduous woodlands; wooded riparian areas; grasslands; farmlands; and burned, moderately logged, and edge areas with scattered trees, snags, or other suitable nest and perch sites (Guinan et al. 2000). This species is common in Douglas-fir and open pine forests east of the Cascade Range. In ponderosa pine and pine-oak forests, abundance was inversely related to canopy cover, and highest where canopy cover was <20 percent (Rosenstock 1996). In the western Cascade Range, this species breeds in snags in clearcuts and in and around the Willamette Valley, in open country with scattered trees and in orchards (Gilligan et al. 1994). In western Oregon, Hansen et al. (1995) estimated mean bluebird densities were greatest at about 10 trees/ac and declined to zero at about 50 trees/ac (for all stems >4 in d.b.h.) in the western Cascade Range. These bluebirds have shown a preference for areas with an open overstory and are abundant in moderately disturbed areas, including moderately logged forests (Franzreb 1977, Szaro 1976), and burned areas (Haggard and Gaines 2001, Johnson and Wauer 1996, Saab and Dudley 1998), where sufficient nest sites and foraging perches are available. Studies on effects of fire and salvage logging in burned forests to western bluebirds and other cavity nesters (Guinan et al. 2000) have varied results. In Washington, there was a higher abundance of western bluebirds in areas of low snag density, but more nests in areas of medium to high snag density (Haggard and Gaines 2001). In Idaho, there were more western bluebird nests in areas of low to medium snag density than in higher snag density areas (Saab and Dudley 1998). In Arizona, forests with no salvage logging, western bluebird abundance was higher in severely burned than in unburned areas (Dwyer and Block 2000). Restoration of ponderosa pine forests by thinning of dense stands, followed by control burns, increased western bluebird abundance, nest and fledgling success, and decreased predation (Gaines et al. 2007, Germaine and Germaine 2002).

We identified source habitat as follows (fig. 67):

- Potential vegetation: Dry forests
- Cover type: Conifer mix, Douglas-fir, grand fir, ponderosa pine, western larch, white oak
- Tree size: All
- Canopy cover: <60 percent (includes all postfire areas as canopy closure is <60 percent)

Snag density—

Nests of western bluebirds are usually found in rotted or previously excavated cavities in trees and snags, or between trunk and bark (Guinan et al. 2000). In northern Arizona, western bluebirds preferred snags over live trees for nesting; 70 percent of nests (n = 33) were found in snags. In areas where snag density was low, they found the birds switched to live trees for nests (Cunningham et al. 1980).

There is often a high degree of inter- and intraspecific competition among cavity nesters for nest sites. Competition for nest sites has increased with the invasion of European starlings, house sparrows, and tree swallows (Gillis 1989, Hedges 1994, Herlugson 1980). On a burned site in southwestern Idaho, Lewis' woodpeckers (*Melanerpes lewis*) frequently usurped western bluebird nests, sometimes ejecting nestlings (Saab and Dudley 1995). We calculated the percentage of source habitat within a watershed that had densities of snags >15.0 in d.b.h. in the following classes (per acre). These density classes were derived from Harrod et al. (1998) for open dry forests for eastern Washington (fig. 67).

- Low: <2.1/ac
- Moderate: 2.1 to 2.8/ac
- High: >2.8 to 3.5/ac
- Very high: ≥3.5/ac

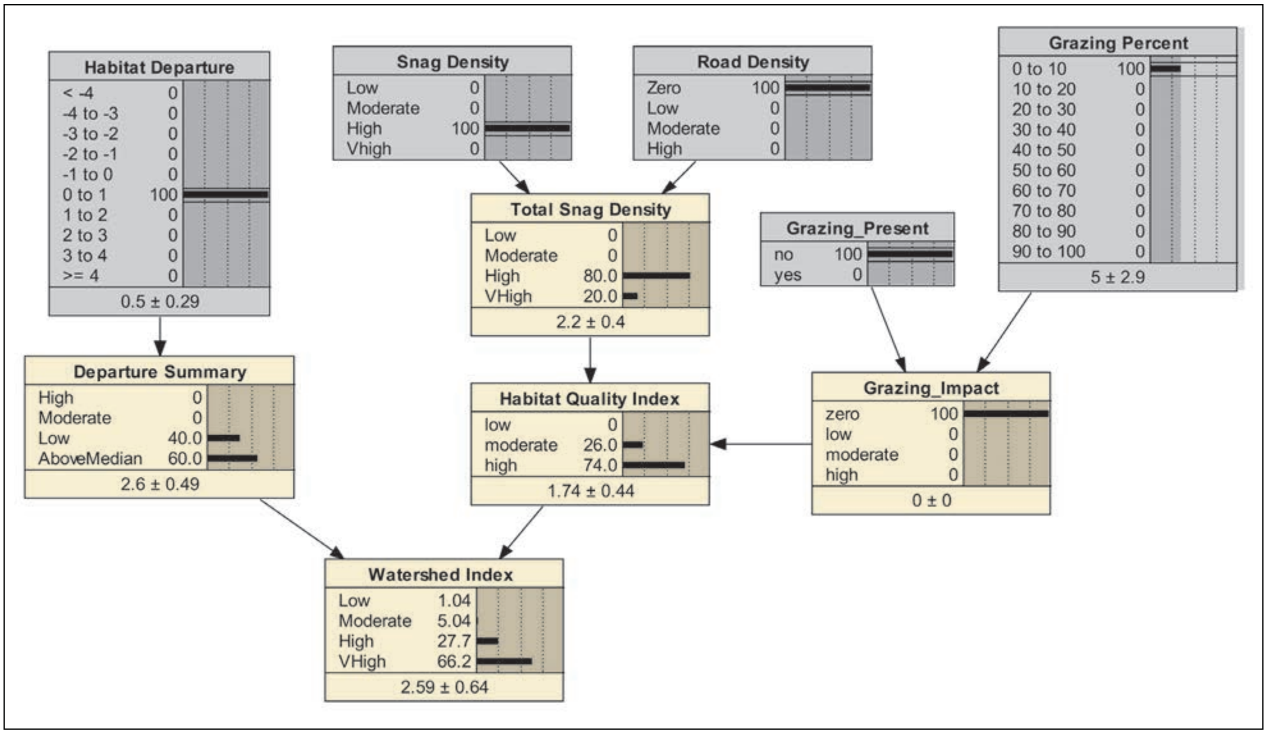

Figure 67—Surrogate species assessment model for western bluebirds.

Road density—

We analyzed road density to evaluate the potential of reduced snag densities along roads (Bate et al. 2007, Wisdom and Bate 2008). Our snag density data are from a modeled dataset that did not account for road-associated factors.

We calculated the percentage of source habitat within each watershed in the following road density classes (fig. 67):

- Zero: <0.1 mi/mi^2
- Low: 0.1 to 1.0 mi/mi^2
- Moderate: 1.1 to 2.0 mi/mi^2
- High: >2.0 mi/mi^2

Grazing—

Livestock grazing may contribute to reduced fire frequency in ponderosa pine forests by reducing the amount of grass that facilitated the spreading of low-intensity fires (Zwartjes et al. 2005). The depletion of competing grasses and lack of fire encouraged the growth of shrubs and dense stands of young conifers (Chambers and Holthausen 2000, Touchan et al. 1996). Dense ponderosa pine forests that resulted from reduced frequency of low-intensity fires are at a greater risk of stand-replacing fires (Chambers and Holthausen 2000, Touchan et al. 1996).

A great reduction in grass biomass owing to grazing is likely to negatively affect the prey base for western bluebirds (Zwartjes et al. 2005). In addition, Bull et al. (2001), found western bluebirds to be more abundant at ponds that were protected from livestock grazing than those not protected.

We categorized the amount of source habitat in an active grazing allotment using 10-percent increments from 0 to 100 percent, with increasing poorer habitat outcomes as the proportion of source habitat in an active allotment increased (fig. 67).

Calculations of historical conditions—

- Source habitat: 0 to 1 departure
- Snag density: High
- Road density: 0
- Grazing: 0

The relative sensitivity of WI values to the variables used in the model for western bluebird are shown in table 42.

Watershed scores—

The WI values were primarily (93 percent, n = 66) low (<1.0), while three watersheds had moderate values (≥1 and <2), and two watersheds had high WI values (>2.0). The two watersheds with the greatest WI values were the Upper Chewuch River and the Upper Columbia-Swamp Creek.

Habitat for western bluebirds was well below the historical median of source habitat in nearly all watersheds (n = 66, 92 percent). One watershed, Upper Columbia-Swamp creek, was above the historical median, and four were near the median: Lake Entiat, Okanogan River-Omak Creek, Lower Okanogan River, and Upper Chewuch River.

Ten watersheds had >49,420 ac of habitat currently including the Chuwelah with >93,900 ac. The Wenatchee River watershed likely had the greatest decline in abundance of western bluebird habitat, with reduction of nearly 74,130 ac.

Table 42—Relative sensitivity of watershed index values to variables in the model for western bluebird

Variable	Sensitivity rank
Habitat departure	1
Snag density	2
Grazing	3
Road density	4

Although loss of habitat had the greatest effect on the WI evaluation, snag densities were also relatively low, and road densities were relatively high. Greater than 50 percent of the source habitat in 92 percent of the watersheds (n = 65) had low snag densities (<1/ac), while four watersheds had high snag densities in >50 percent of the source habitats: Pasayten River, Ruby Creek, Lightning Creek, and Ross Lake. Nearly all source habitats in these watersheds are managed by the USDA Forest Service. Road densities were in the high class (>2.0 mi/mi^2) for >50 percent of the source habitat in 42 percent (n = 31) of the watersheds. About 20 percent of the watersheds (n = 19) had very little or none (<5 percent) of the source habitat in an active grazing allotment while another 25 percent had >50 percent of the source habitat in an active grazing allotment.

These habitat declines were similar to the findings of Wisdom et al. (2000) who listed declines of western bluebird habitat in the North Cascades and Northern Glaciated Mountains as -65 percent and -82 percent, respectively.

Viability outcome—

The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. Currently, the likelihood of viability of western bluebirds is reduced compared to historical conditions (fig. 68). The WWI scores indicated that the current habitat capability for the western bluebird within the assessment area is 18 percent of the historical capability. Dispersal across the assessment area was not considered an issue for this species. All five ecoregions currently contained at least one watershed with >40 percent of the median amount of historical source habitat. Thirty-two watersheds (45 percent) had >40 percent of the median amount of historical source habitat.

The current viability outcome is a 66 percent probability of D, and a 28 percent probability of E. An outcome of D indicates suitable environments are low to moderately distributed across the historical range of the species (fig. 68). Suitable environments exist at low abundance relative to their historical conditions. While some of the subpopulations associated with these environments may be self-sustaining, there is limited opportunity for population interactions among many of the suitable environmental patches for species with limited dispersal ability. For species for which this is not the historical condition, reduction in species' range in the assessment area may have resulted. These species may not be well distributed across the assessment area.

Historically, we estimated that 79 percent (n = 56) of the watersheds had >40 percent of the median amount of historical source habitat, and the distribution was throughout all ecoregions. Therefore, historically, western bluebirds likely would

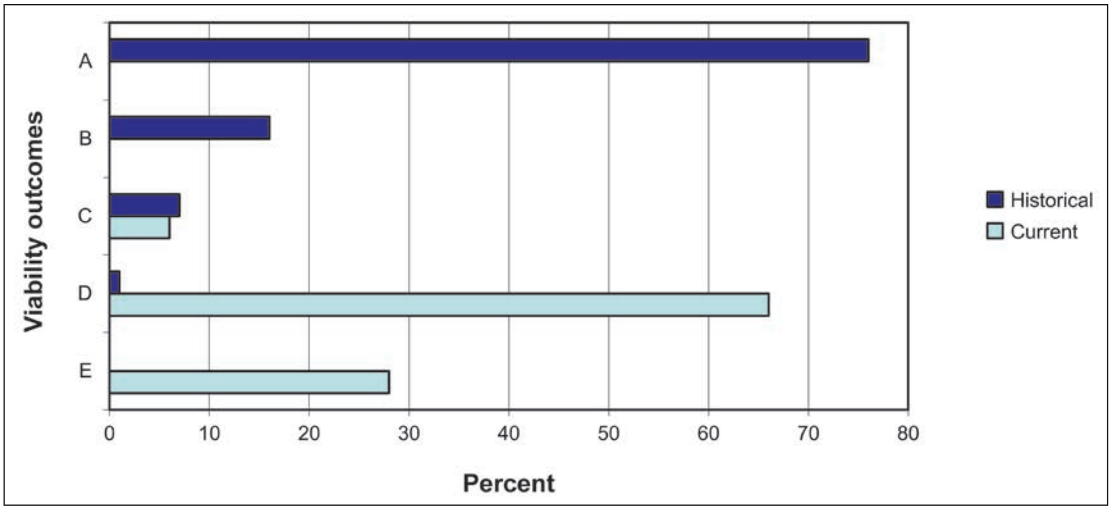

Figure 68—Current and historical viability outcomes for the western bluebird in the northeast Washington assessment area.

have had primarily an A outcome with habitat broadly distributed and abundant (fig. 68). Raphael et al. (2001) also found a decline from historical conditions in their evaluation of habitats across the entire interior Columbia Basin for western bluebirds; however, though measured differently, not as great of a decline was measured.

Decline in the amount of source habitat of open canopy forests for western bluebirds throughout the dry forests across the assessment area.

Management Considerations

The following issues were identified during this assessment and from the published literature for managers to consider:

1. Decline in the amount of source habitat of open canopy forests for western bluebirds throughout the dry forests across the assessment area.
2. Loss of snags (>15 in d.b.h.) used by bluebirds for nesting and roosting.
3. Livestock grazing may reduce the quality of the foraging habitat.

Western Gray Squirrel

Introduction

The western gray squirrel is a surrogate species for the open forest/pine/oak group. They are highly associated with ponderosa pine and oak forests with medium to large trees (Ryan and Carey 1995). Owing to the limited distribution of this species within the assessment area, we did not develop a surrogate species assessment model but provide a qualitative assessment of its habitat relationships and general management considerations.

The western gray squirrel was once one of the most commonly encountered mammals in the Northwest (Bowles 1921). Wisdom et al. (2000) reported that source habitat for western gray squirrels showed declines in 65 percent of the watersheds in the North Cascade Range. Their range in Washington now consists of small, scattered populations that generally follow the range of Oregon white oak (fig. 69). There are three major subpopulations in Washington (Rodrick 1986): one in Klickitat County along the southern Columbia River (Linders et al. 2004), another in Okanogan and Chelan Counties along the northern Columbia River basin, and a third in Thurston and Pierce Counties in the Puget Trough (Linders and Stinson 2007). The population within the assessment area (Okanogan and Chelan Counties) occurs from the western tip of Lake Chelan near Stehekin, on the northern shore of Lake Chelan, and in the Black Canyon, McFarland, and Squaw Creek drainages within the Methow subbasin (Linders and Stinson 2007).

Source habitat—

The population of gray squirrels within the assessment area is associated with groves of English walnut and black walnut planted by early settlers (Barnum 1975)

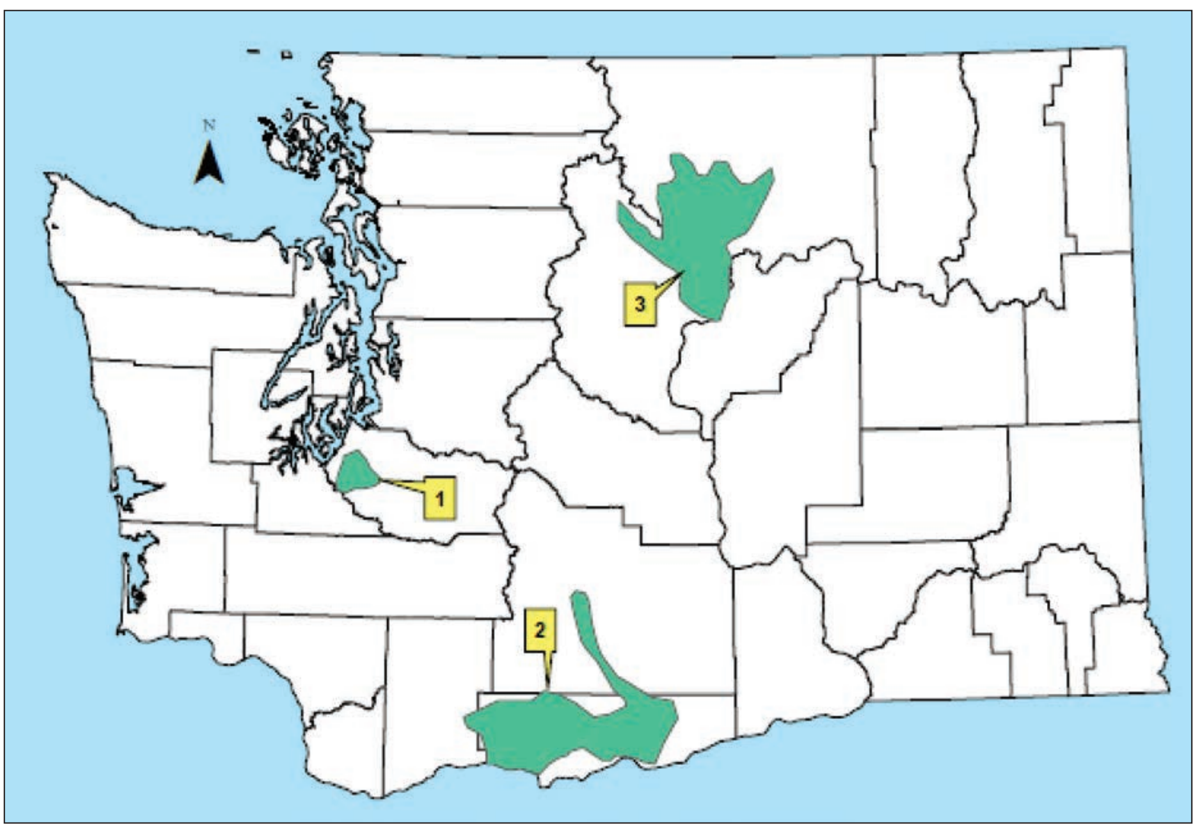

Figure 69—Current distribution of western gray squirrel populations in Washington: (1) Puget Trough, (2) Klickitat County, and (3) Okanogan and Chelan Counties.

and with dry forests composed primarily of ponderosa pine and Douglas-fir. This population is small and isolated (WDFW 1993). There have been anecdotal reports that western gray squirrels were introduced in the Okanogan area (Linders and Stinson 2007). However, a recent assessment suggests that they were present historically and became isolated as populations and habitat have contracted (Linders and Stinson 2007).

Risk factors—
Risk factors for the western gray squirrel include loss of habitat from private land development, road-related mortality along State Highway 153 and forest roads, fire exclusion, improperly designed fuels-reduction projects, potential loss of habitat from high-severity fire, genetic effects of a small population, and disease outbreak (Cornish et al. 2001). Potential for competition from nonnative eastern gray squirrels and fox squirrels (Linders and Stinson 2007) has also been identified as a threat. The risk factors addressed in this assessment that are relevant to management of federal lands include road-related mortality on forest roads, fire exclusion and improperly designed fuels-reduction projects, and potential habitat loss from high-severity fire.

Management Considerations

The following management considerations address risk factors associated with western gray squirrels on National Forest System lands.

1. Road-related mortality is an important risk factor for gray squirrels.
2. Restoration of ponderosa pine forests to enhance seed production, restore stand structure and composition, promote the development of large tree structure and snags, and to reduce the risk of habitat loss from high-severity fire. The following characteristics can be used to help guide silvicultural prescriptions (based on Linders and Stinson 2007):
    a. Multiaged ponderosa pine stands.
    b. Low to moderate stem density with a clumped distribution of trees providing nest sites and canopy connections for arboreal travel.
    c. Large (>20 in d.b.h.) ponderosa pine trees habitats.
    d. Ground cover mostly of litter and grass with sparse understory of scattered shrubs.
    e. Large-cavity trees and snags.
    f. Presence of additional food species such as bigleaf maple, vine maple, serviceberry, or aspen.

White-Headed Woodpecker

Introduction

The white-headed woodpecker was chosen as a surrogate species to represent the medium-large trees/dry forest group. This woodpecker is associated with open-canopied ponderosa pine forests, specifically with large trees and snags that are important habitat components for other species in the group and family. White-headed woodpeckers range across the entire Pacific Northwest in dry forests east of the Cascade Range of Oregon and Washington and are year-round residents.

Model Description

Source habitat—

White-headed woodpeckers occur in open ponderosa pine or mixed-conifer forests dominated by ponderosa pine (Bull et al. 1986; Dixon 1995a,1995b). Dixon (1995a, 1995b) found that population density increased with increasing volumes of old-growth ponderosa pine in both contiguous and fragmented sites. In addition, these woodpeckers may use areas that have undergone various silvicultural treatments, including postfire areas, if large-diameter ponderosa pines and other old-growth components remain (Dixon 1995a,1995b; Raphael 1981, Raphael and White 1984, Raphael et al. 1987). Average canopy closure at 66 nest sites was 12 percent.

Throughout the range, habitat components include an abundance of mature pines (with large cones and abundant seed production), relatively open canopy (50 to 70 percent open), and availability of snags and stumps for nest cavities (Garrett et al. 1996). Understory vegetation is generally sparse within preferred habitat (Garrett et al. 1996).

For the period 1997–2004, Frenzel (2004) found nesting success was 39 percent at sites with low densities of big trees, as opposed to 61 percent for nests in uncut stands. Uncut sites had big-tree (>21 in d.b.h.) densities ≥12 trees per acre. White-headed woodpeckers foraged predominantly on large-diameter live ponderosa pine trees (Dixon 1995b). Ponderosa pine seeds are the most important vegetable food item for this species in Oregon (Bull et al. 1986, Dixon 1995b), especially in winter. Source habitat was defined in this analysis as (fig. 70):

- Potential vegetation: Dry
- Cover type: Ponderosa pine is the dominant cover type in the dry natural vegetation group (PVG) conditions, though we included other species: western larch, conifer mix, Douglas-fir, and grand fir; as in the dry PVG, these types usually contain a large proportion of ponderosa pine types.
- Forest structure and size: Single- and multilayered stands with >15 in QMD
- Canopy closure: <50 percent canopy

Snag density—

Several studies have documented the importance of large-diameter ponderosa pine snags for white-headed woodpeckers (Dixon 1995a, 1995b; Milne and Hejl 1989, Raphael and White 1984). Of 43 white-headed woodpecker nests in central Oregon (Dixon 1995b), 36 were in ponderosa pine snags, 2 in ponderosa pine stumps, 2 in quaking aspen snags, and 1 each in live quaking aspen, white-fir snag, and the dead top of a live ponderosa pine tree. Most nest snags were moderately decayed. Nest tree size averaged 26 in d.b.h., and nest tree height averaged 46 ft; excluding one nest 105 ft high in a dead-topped live ponderosa pine, nest-cavity height averaged 14.4 ft In south-central Oregon, all 16 nests studied by Dixon (1995a) were in completely dead substrates (37 percent in snags, 56 percent in stumps, and 6 percent in leaning logs). Mean size of nest trees was 31.5 in d.b.h., and nest tree height averaged 10 ft.

Frenzel (2004) found that of 405 nests of white-headed woodpeckers, all but 12 were in completely dead trees. Mean size of nest trees was 27 in d.b.h. (n = 405), mean canopy closure at nest sites was 11.0 percent, and density of large trees >21 in d.b.h. was 61 trees/ac.

We calculated the percentage of area of source habitat within each watershed that had snag (>20 in d.b.h.) densities in the following classes based on data from Harrod et al. (1998) of (fig. 70):

- Low: <1.0/ac
- Moderate: 1.0 to 1.3/ac
- High: 1.3 to 1.5/ac
- Very high: >1.5/ac

Road density—

We included a road density variable to account for likely reduced snag densities along roads (Bate et al. 2007, Wisdom and Bate 2008). We calculated road densities in four classes (fig. 70):

- Zero: - <0.1 mi/mi^2 open roads in a watershed
- Low: 0.1 to 1.0 mi/mi^2 open roads in a watershed
- Moderate: 0.1 to 2.0 mi/mi^2 open roads in a watershed
- High: >2.0 mi/mi^2 open roads in a watershed

Shrub cover—

Frenzel (2004) found that shrub cover was a significant variable in predicting nest success. Nest sites with <5 percent shrub cover had the highest mean nesting success of 61 percent. Nest success with shrub cover >5 percent had a mean nest success of 42 percent.

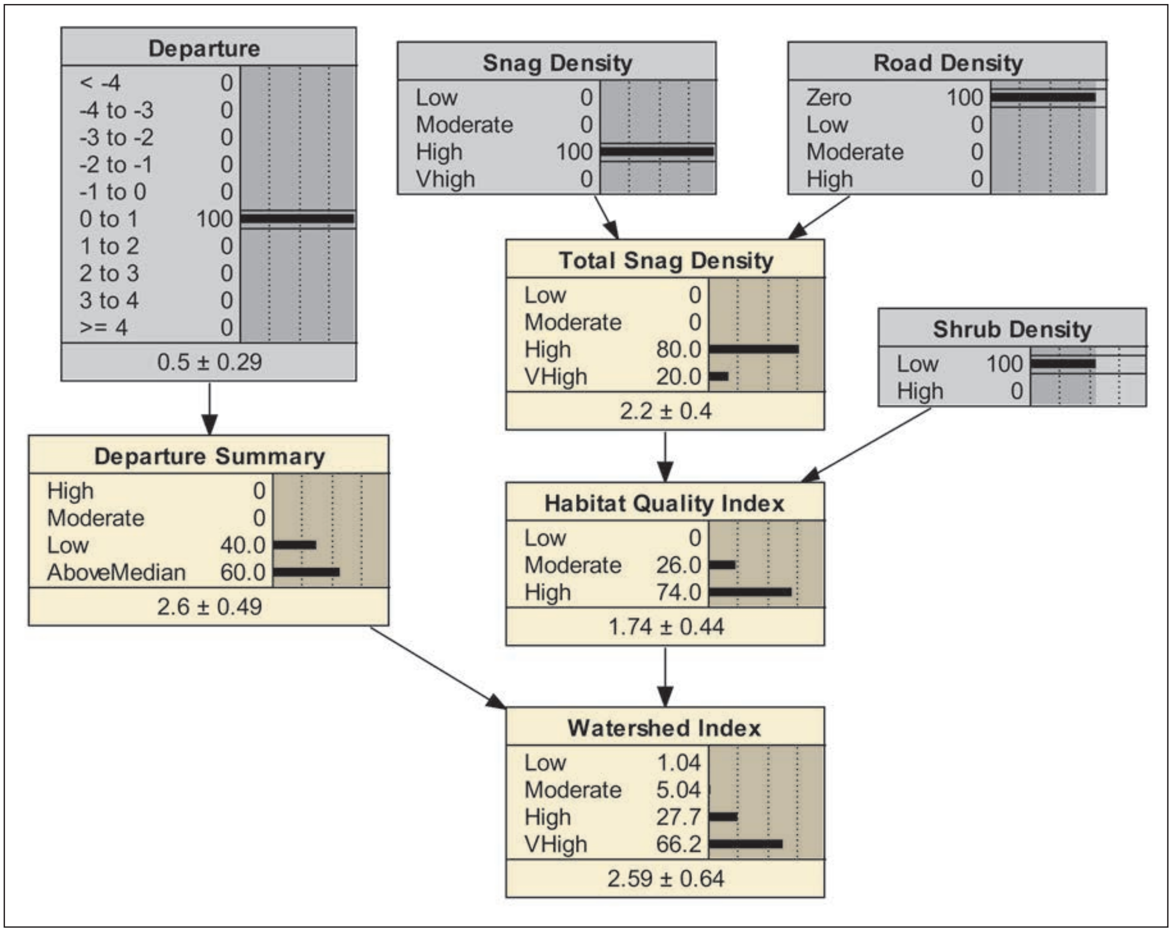

Figure 70—Surrogate species assessment model for white-headed woodpeckers.

Smith (2002) reported that densities of chipmunks in ponderosa pine habitat in central Oregon increased with shrub cover, and densities of golden-mantled ground squirrels increased with amounts of down wood. Both of these species are nest predators of white-headed woodpeckers (Mellen-McLean et al. 2013), suggesting that higher levels of shrubs and woody debris may lead to increased levels of predation.

Using gradient nearest-neighbor shrub density data, we calculated the percentage of source habitat with high (>15 percent) and low (<15 percent) shrub density per watershed. Although the research suggest 5 percent cover (Mellen-McLean et al. 2013), after reviewing the data we had on shrub density, we felt the 15 percent shrub density from the data-set was likely representing areas we knew as having closer to 5 percent shrub density.

Calculation of historical conditions—

- Source habitat: 0 to 1 departure
- Snag density: High
- Road density: 0
- Shrub density: Low

The relative sensitivity of the WI values to variables used in the model for white-headed woodpecker are shown in table 43.

Model evaluation—

We evaluated the model for white-headed woodpeckers using 88 documented occurrences compiled by Mellen-McLean et al. (2013) and an equal number of random points. We did not find a statistical difference ($t = 1.97$, $P = 0.78$) between the mean WI derived from points of the occurrences (1.6) and random points (1.7). We believe that the WI values were so low for all watersheds that there was an insufficient distribution of values to make our statistical approach meaningful.

Assessment Results

Watershed scores—

Historically, all but one watershed (Ashnola) contained source habitat for the white-headed woodpecker. Currently, 97 percent (n = 69) had a high departure from the historical median amount of habitat across all watersheds (fig. 71). Okanogan River-Bonaparte Creek, and Okanogan River-Omak Creek were currently at or above the historical median amount of habitat.

We assumed in this analysis that if the amount of source habitat in a watershed was reduced to <40 percent of the historical amount, the ecological function of the remaining source habitat to provide for the viability of the surrogate species was greatly diminished. For the white-headed woodpecker, this value is 3,330 ac.

Table 43—Relative sensitivity of watershed index values to variables in the model for white-headed woodpeckers

Variable	Sensitivity rank
Habitat departure	1
Snag density	2
Road density	3
Shrub density	4

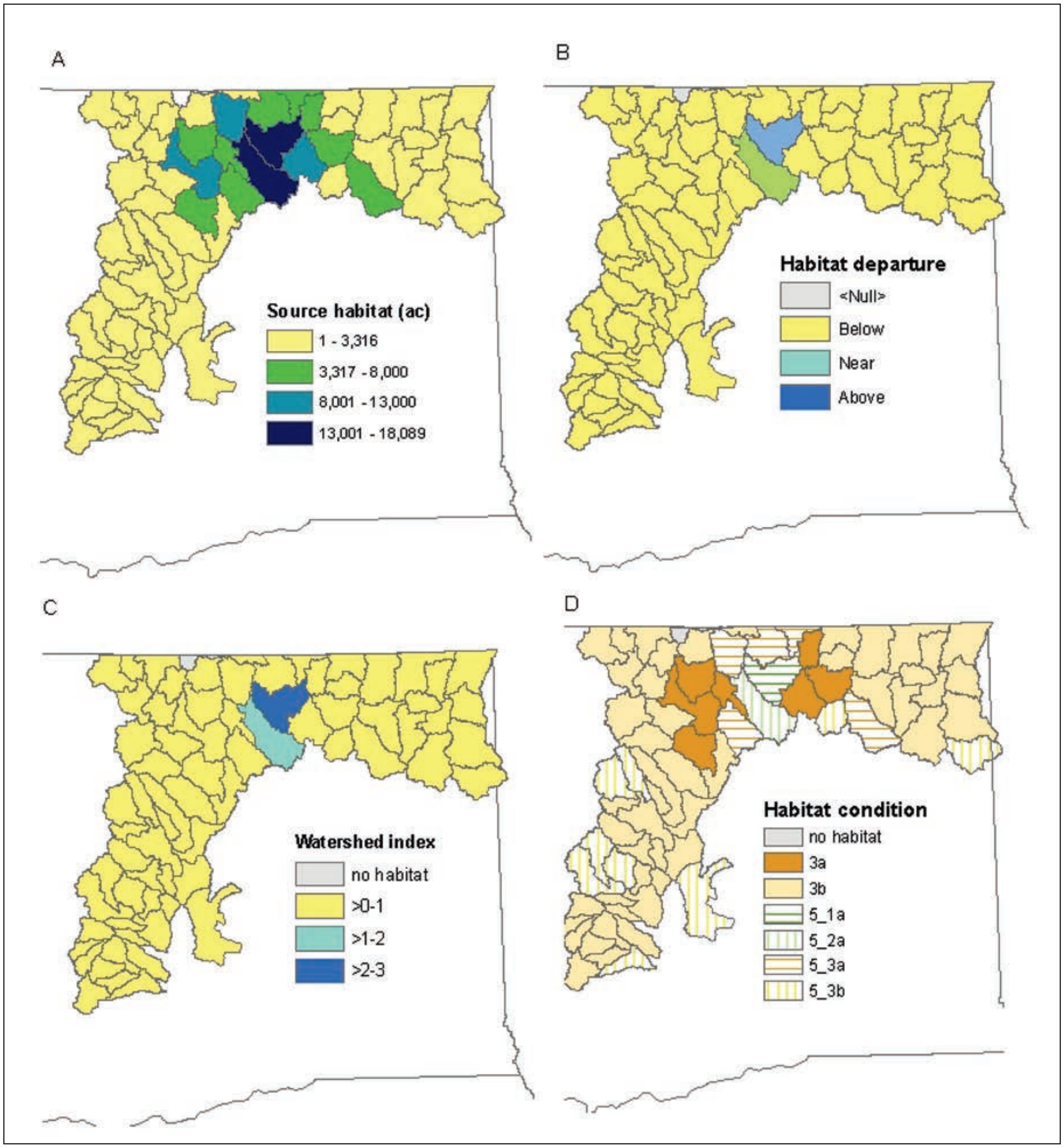

Figure 71—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watershed index, and (D) habitat condition class for white-headed woodpeckers by watershed in the northeast Washington assessment area.

Historically, 55 watersheds met this condition. Currently, only 20 percent of the (n = 14) watersheds contained >40 percent of the historical median amount of habitat. Watersheds with the most source habitat currently were the Okanogan River-Bonaparte Creek (18,090 ac), Okanogan River-Omak Creek (15,400 ac), West Fork Sanpoil (12,130 ac), and the Middle Methow River (10,030 ac) (fig. 71).

The watersheds with the greatest potential habitat on National Forest System lands are the Lower Chewuch River, Lower Lake Chelan, Lower Methow, Middle Methow, and Boulder/Deadman. Boulder/Deadman has ≥64,250 ac of potential habitat on National Forest System lands, but <740 ac currently that was suitable. The watersheds with the greatest amount of current source habitat managed by the Forest Service were the Middle Methow River (8,900 ac) and the West Fork Sanpoil (5,095 ac).

The majority of watersheds (n = 63, 86 percent) had ≥50 percent of the source habitat with low (≤1.0 snags/acre) snag densities. About on-third (n = 24) of the watersheds had high road densities in more than 50 percent of the dry forests, while 10 (14 percent) watersheds had ≥50 percent of the dry forests having zero road densities. Most watersheds had high shrub densities in source habitat (n = 65, 90 percent), which along with the propensity of high road densities and low snags contribute to lower WI values (poorer habitat quality).

Only two watersheds had a WI score that was not low (≥1): Okanogan River-Bonaparte Creek (2.0) and Okanogan River-Omak Creek (1.5) (fig. 71). All other watersheds had low WI values (<1.0) primarily owing to loss of habitat.

Viability outcome—

The VOI model incorporated the WWI scores (described earlier) and a habitat distribution index. The WWI scores indicated that the current habitat capability for the white-headed woodpecker within the assessment area is 21 percent of the historical capability. Dispersal across the assessment area was not considered an issue for this species. Two of five ecoregions currently contained at least one watershed with >40 percent of the median amount of historical source habitat. Fourteen watersheds (20 percent) had >40 percent of the median amount of historical source habitat currently. We estimated the current viability outcome is a 50 percent probability of D and 47.5 percent probability of E, indicating habitat was isolated and occurs at very low abundance (fig. 72).

We estimated that historical conditions were much different for this woodpecker. Dispersal across the assessment area was not considered an issue for this species. Five of five ecoregions and 55 (77 percent) of the watersheds contained >40 percent of the median amount of habitat historically.

Decline in the amount of source habitat for white-headed woodpeckers throughout dry forests across the assessment area.

Historically, we estimated that 77 percent of the watersheds (n = 55) contained source habitat >40 percent of the historical median and was widespread throughout the assessment area. The viability outcome historically was estimated to be 76 percent A and 16 percent B, indicating a broad distribution of abundant habitat (fig. 72).

In summary, under historical conditions, white-headed woodpeckers were likely well distributed throughout the assessment area; currently, they are likely not well distributed and at risk of extirpation.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Decline in the amount of source habitat for white-headed woodpeckers throughout dry forests across the assessment area.
2. Low levels of large-diameter (>21 in d.b.h.) snags within the majority of watersheds assessed within the planning area.
3. High road densities in dry forest in 32 percent of the watersheds that may contribute to loss of large snags.
4. High shrub densities in existing habitat that may lead to higher predation.
5. The sustainability of dry forest habitats that have experienced several decades of fire exclusion and are susceptible to stand-replacing fire events.

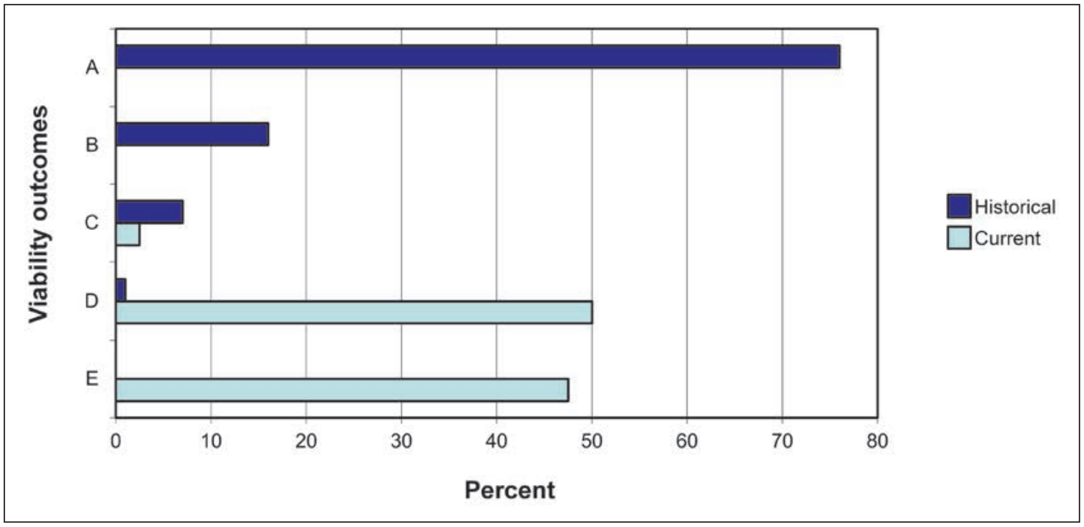

Figure 72—Current and historical viability outcomes for the white-headed woodpecker in the northeast Washington assessment area.

Wilson's Snipe

Introduction

The wilson's snipe was chosen as a surrogate species to represent habitats in the marsh/wet meadow group of the wetland family. Wilson's snipes have one of the widest distributions for species in these habitats. However, habitats for species in this group were not abundant on National Forest System lands in eastern Washington, and they were patchily distributed across the assessment area with concentrations in the central and eastern portions (Smith et al. 1997). Wilson's snipes generally forage in shallow water and mudflats; major risks to the species are draining, filling, and degradation of marshes; and environmental contaminants. Although grazing by domestic livestock was considered a risk for several species associated with these habitats, these risks varied by species and by intensity and season of grazing, and so grazing may not have always had a negative impact. Wilson's snipes are year-round residents of the assessment area (Mueller 1999); this assessment was for nesting habitat.

Model Description

Source habitat—

Breeding habitat of Wilson's snipes has been characterized as sedge bogs, fens, and alder or willow wetlands (McKibben and Hofmann 1985, Tuck 1972). Banner and Schaller (2001) interpreted these associations to equate to palustrine emergent (PEM) and palustrine scrub shrub (PSS) wetlands as described by Cowardin et al. (1979). We used those definitions and maps in the National Wetlands Inventory to describe and delineate source habitat for this analysis where they occurred in vegetation zones shrub-steppe, ponderosa pine, Douglas-fir, and grand fir (fig. 73).

Wetland size—

Gibbs et al. (1991) reported a positive relationship between the presence of snipes during the breeding season and size of wetland (i.e., <1 to >50 ac). Banner and Schaller (2001) suggested that wetlands <7 ac had limited value as habitat for snipe. Based on those findings, we characterized wetland size with the following classes (fig. 73):

- Small: <25 ac mean size of PEM or PSS wetlands within a watershed.
- Large: ≥25 ac mean size of PEM or PSS wetlands within a watershed.

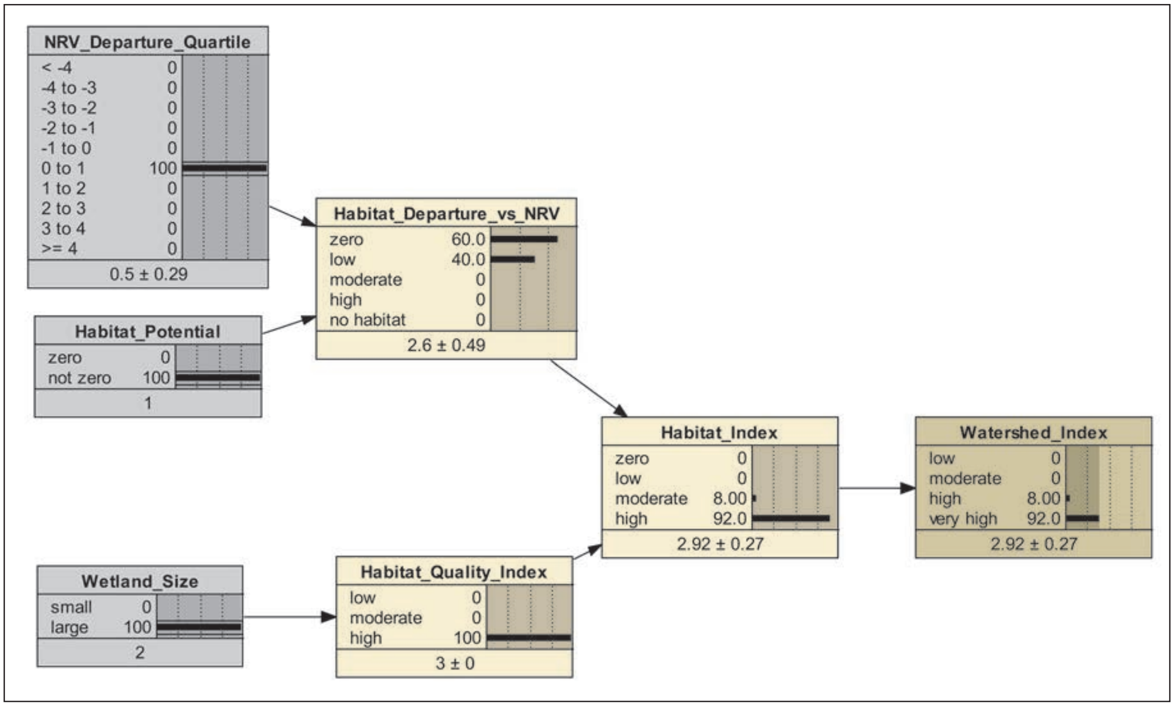

Figure 73—Surrogate species assessment model for Wilson's snipes.

GIS databases used—

- National Wetlands Inventory
- Vegetation Zone

Calculation of historical conditions—

- Departure of source habitat from HRV: 0.5
- Wetland size: Class large
- Current amount of habitat in each watershed was increased by 30 percent.

The relative sensitivity of WI values to the variables used in the model for Wilson's snipes are shown in table 44.

Assessment Results

Watershed scores—

Sixty-two of 72 watersheds within the assessment area provided habitat for Wilson's snipes (fig. 74). All watersheds with habitat have experienced habitat loss from historical conditions (fig. 74). Thirty-four percent (n = 21) of watersheds had high WI scores (>2.0); 66 percent (n = 41) of watersheds had moderate WI scores (>1.0 to <2.0) (fig. 74). Factors that influenced the WI scores included the amount of source habitat (fig. 74). Gibbs et al. (1991) and Banner and Schaller (2001) reported positive

Table 44—Relative sensitivity of watershed index values to variables in the model for Wilson's snipes

Variable	Sensitivity rank
Habitat departure	1
Wetland size	2

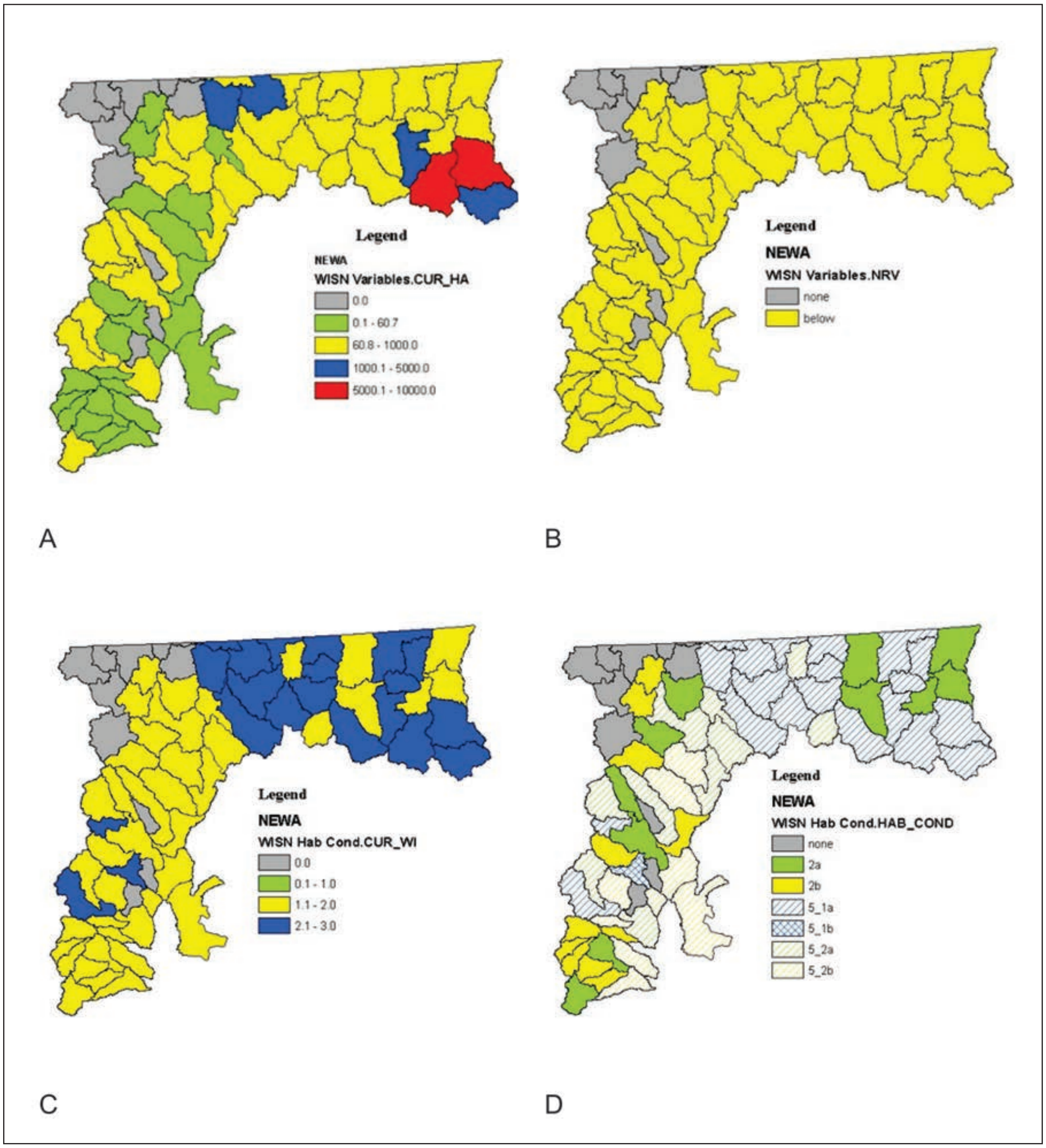

Figure 74—Current amount of (A) source habitat (acres), (B) current habitat departure, (C) watersed index, and (D) habitat condition for Wilson's snipes (WISN) by watershed in the northeast Washington assessment area (NEWA).

relationships between the presence of snipe and size of wetland. Thirty-four percent (n = 21) of the watersheds had small mean wetland size (<25 ac); 66 percent (n = 41) had large mean wetland size (≥25 ac).

Viability outcome scores—
The VOI model incorporated the WWI scores (described earlier), and a habitat distribution index. The WWI scores indicated that the current habitat capability for Wilson's snipe within the assessment area is 66 percent of the historical capability. Dispersal across the assessment area is not considered an issue for this species owing to their relatively high mobility. All five ecoregions contain at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Fifty-eight percent (n = 42) of the watersheds currently have >40 percent of the median amount of historical source habitat. Under those circumstances, there is a 28 percent probability that the current viability outcome for Wilson's snipes is A and a 54 percent probability of outcome B, indicating habitat is broadly distributed and abundant, but there are gaps where suitable environments are absent or only present in low abundance (fig. 75). Other species associated with habitats in the marsh/wet meadow group of the wetland family likely have similar outcomes.

Historically, dispersal across the assessment area was not considered an issue for this species. All ecoregions contained at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Forty-two percent (n = 30) of watersheds had

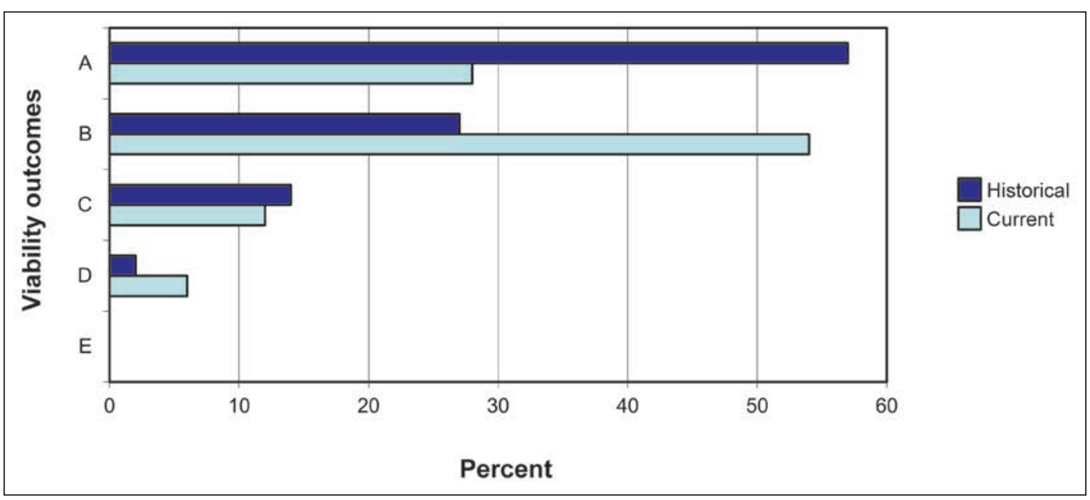

Figure 75—Current and historical viability outcomes for Wilson's snipes in the northeast Washington assessment area.

>40 percent of the median amount of historical source habitat. Under those circumstances, there was a 57 percent probability that the historical viability outcome for Wilson's snipes was A, and a 27 percent probability that the historical viability outcome was B, indicating habitat was broadly distributed and abundant (fig. 75).

In summary, under historical conditions, Wilson's snipes and other species associated with habitats in the marsh/wet meadow group of the wetland family were likely well distributed throughout the assessment area. Currently, they continue to be well distributed, but there are gaps where suitable environments are absent or only present in low abundance (e.g., high-quality habitats are clustered in the eastern and central portions of the assessment area). However, these habitats are estimated to be large enough and close enough together to permit dispersal among subpopulations and to allow the species to potentially interact as a metapopulation in those areas. However, some subpopulations are so disjunct or of such low density that they are essentially isolated from other populations (e.g., southwestern portion of the assessment area).

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. Loss and degradation of wetland habitats.

Wolverine

Introduction

The wolverine was selected as a surrogate species for the habitat generalist group. It is sensitive to risk factors that can cause disturbance (Copeland et al. 2007, Krebs et al. 2007) as are the other species in this group. Reports of wolverines within the assessment area have been steadily increasing since the 1960s (Aubry et al. 2007, Edelmann and Copeland 1999, Johnson 1977). Currently, their distribution appears to include the Cascade Range, Kettle Range and Selkirk Mountains, although their density is likely low (Aubry et al. 2007, Edelmann and Copeland 1999). Wolverines are year-round residents within the assessment area, and this assessment represents their year-round habitat use.

Model Description

Source habitat—

Montane coniferous forests, suitable for winter foraging and summer kit rearing, may only be useful if connected with subalpine cirque habitats required for natal denning, security areas, and summer foraging (Copeland 1996, Copeland et al. 2010). Similar to other large mammalian carnivores (e.g., *Ursus arctos*, *Canis lupus*), the current distribution of wolverines is likely determined by the intensity of human settlement, and in addition, the persistence of spring snow cover (Aubry et al. 2007, Copeland et al. 2010) rather than by vegetation type or topography (Banci 1994, Carroll et al. 2001, Kelsall 1981).

Several researchers have documented the effects of roads on wolverines and their habitat and have included roads in models of source habitat (Carroll et al. 2001, Copeland et al. 2007, Krebs et al. 2007, Raphael et al. 2001, Rowland et al. 2003, Wisdom et al. 2000). Carroll et al. (2001) found areas with road densities <1 mi/mi^2 to be strongly correlated with the presence of wolverine. Rowland et al. (2003) in a test of the Raphael et al. (2001) model found that road density was a better predictor than habitat amount of wolverine abundance when applied at the watershed scale (such as our WI model). Thus, we incorporated road densities into our definition of source habitat. To identify source habitat for this species, we limited the analysis of current source habitat to those areas with road densities of <1.0 mi/mi^2. This road density classification was developed from Gaines et al. (2003).

We included most cover types and structural stages in montane forest, subalpine forest, alpine tundra, as did Wisdom et al. (2000) and Raphael et al. (2001). These cover types also coincide with areas where there is a higher likelihood of persistent spring snow cover (Copeland et al. 2010). We used the following variables to identify wolverine source habitat within the assessment area (fig. 76):

- Road density: Areas with road densities <1.0 mi/mi^2
- Cover types: Alpine, parkland, subalpine fir, Pacific silver fir, Engelmann spruce, western hemlock, western redcedar, mountain hemlock, lodgepole pine, western larch, mixed conifer, Douglas-fir, and grand fir. We did not include low-elevation cover types such as ponderosa pine and shrub-steppe.

Mean patch size—

Banci (1994) identified the need for large areas of the appropriate vegetation types and with low human use to provide for the conservation of wolverine. We evaluated the relative size of the areas of source habitat within a watershed by computing a mean patch size and classified the data into three classes, representing high, medium, and low. Our assumption was that the greater the mean patch size, the more conservation value the watershed would have for wolverine. We categorized the mean patch size as follows (fig. 76):

- Low mean patch size: <2 mi^2
- Moderate mean patch size: 2 to 4 mi^2
- High mean patch size: >4 mi^2

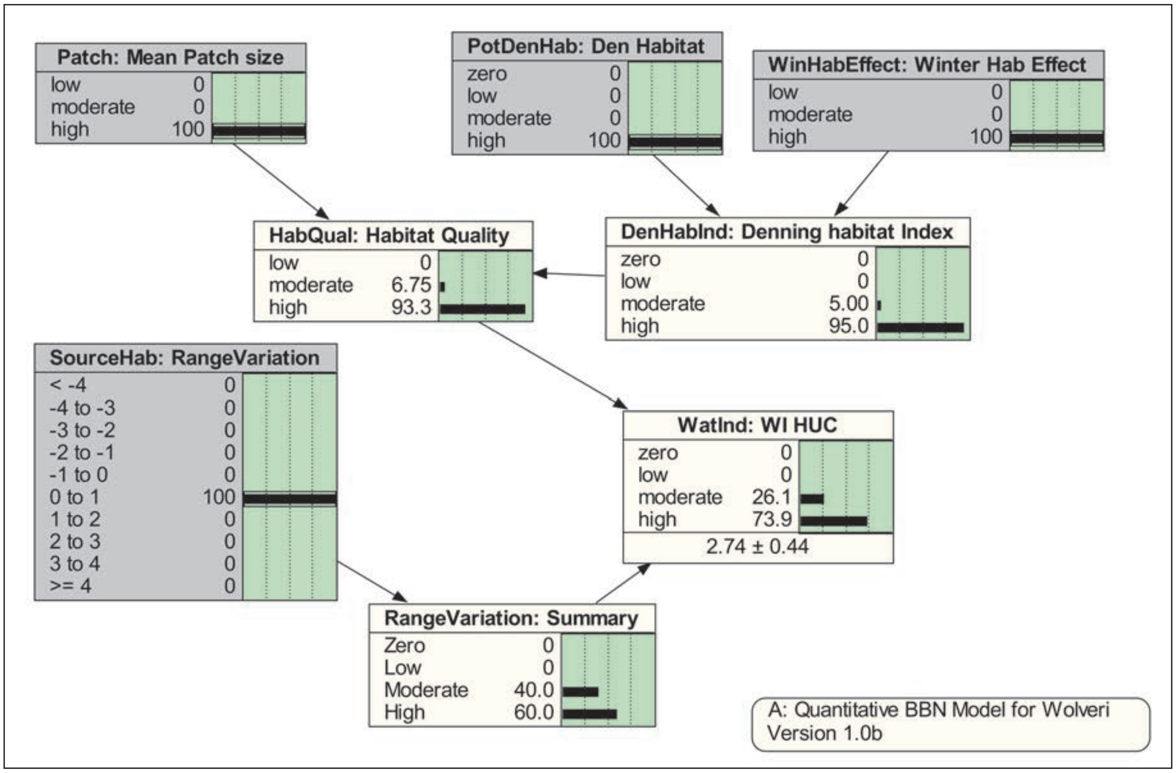

Figure 76—Surrogate species assessment model for the wolverine.

Potential den habitat—

Natal dens are typically above or near tree line, require snow depths of 3 to 10 ft that persist into spring, and are in proximity to rocky areas such as talus slopes or boulder fields (Copeland 1996, Copeland et al. 2010). The predictive habitat model for wolverine developed by Carroll et al. (2001) was improved when alpine cirque habitat was added as a variable as a surrogate to denning habitat. We modeled potential den habitat by using land type associations (Ha7, Ha8, Hb9, Hi9) that represented alpine and subalpine boulder fields and talus slopes (USFS 2000) (fig. 76). The amount of potential wolverine den habitat was categorized as follows:

- Zero: 0 ac of potential den habitat
- Low: >0 to 3,700 ac of potential den habitat
- Moderate: >3,700 to 8,650 ac of potential den habitat
- High: >8,650 ac of potential den habitat

Winter habitat effectiveness—

Copeland (1996) and Krebs and Lewis (1999) documented the potential for disturbance to wolverine natal dens because of late winter to spring snowmobile and other winter recreation activities. We assessed the potential effects of winter recreation on the effectiveness of wolverine habitat by overlaying winter recreation routes onto wolverine habitat and calculating the density of these routes (fig. 76). This was an underestimate of the impacts of winter activities as other recreation routes were present in the assessment area but not in our digital inventory. We categorized the effects of winter recreation activities on wolverine habitat as follows (fig. 76):

- Low habitat effectiveness: >25 percent of habitat with winter route densities >2.0 mi/mi^2
- Moderate habitat effectiveness: >25 percent of habitat with winter route densities >1.0 mi/mi^2
- High habitat effectiveness: <25 percent of habitat with winter route densities <1.0 mi/mi^2

Calculations of historical conditions—
Source habitat: Area of alpine, parkland, subalpine fir, pacific silver fir, Engelmann spruce, western hemlock, western redcedar, mountain hemlock, lodgepole pine, western larch, mixed-conifer, Douglas-fir, and grand fir cover types.

- Patch size: Calculated average patch size without the influence of roads
- Den habitat: Same as the current amount
- Winter habitat effectiveness: High

The relative sensitivity of WI values to variables used in the model for wolverine are shown in table 45.

Model evaluation—
We derived the mean WI value from 64 points where occurrences of wolverines were confirmed and compared it with the mean WI value from 63 random points. The mean WI for the occurrence points (2.01) was significantly higher ($t = 1.98$, $P = 0.0001$) than the mean derived from the random points (1.58) indicating that our model identified suitable environments within watersheds for wolverines.

Table 45—Relative sensitivity of watershed index values to variables in the model for the wolverine

Model variables	Order of variable weighting
Source habitat	1
Patch size	2
Den habitat	3
Winter habitat effectiveness	4

Assessment Results

Watershed scores—

Currently, five (7 percent) of the watersheds have high WI scores (≥2.0). The remaining additional 67 (93 percent) watersheds had WI scores with moderate (between 1.0 and 2.0) scores (fig. 77). The lower scores were largely owing to the influence of roads on the loss of source habitat for wolverines. In our model, areas with high road densities reduced the availability of source habitat and the patch size of source habitat. Areas with high road densities have been shown to have lower probabilities of wolverine occurrence (Caroll et al. 2001, Rowland et al. 2003).

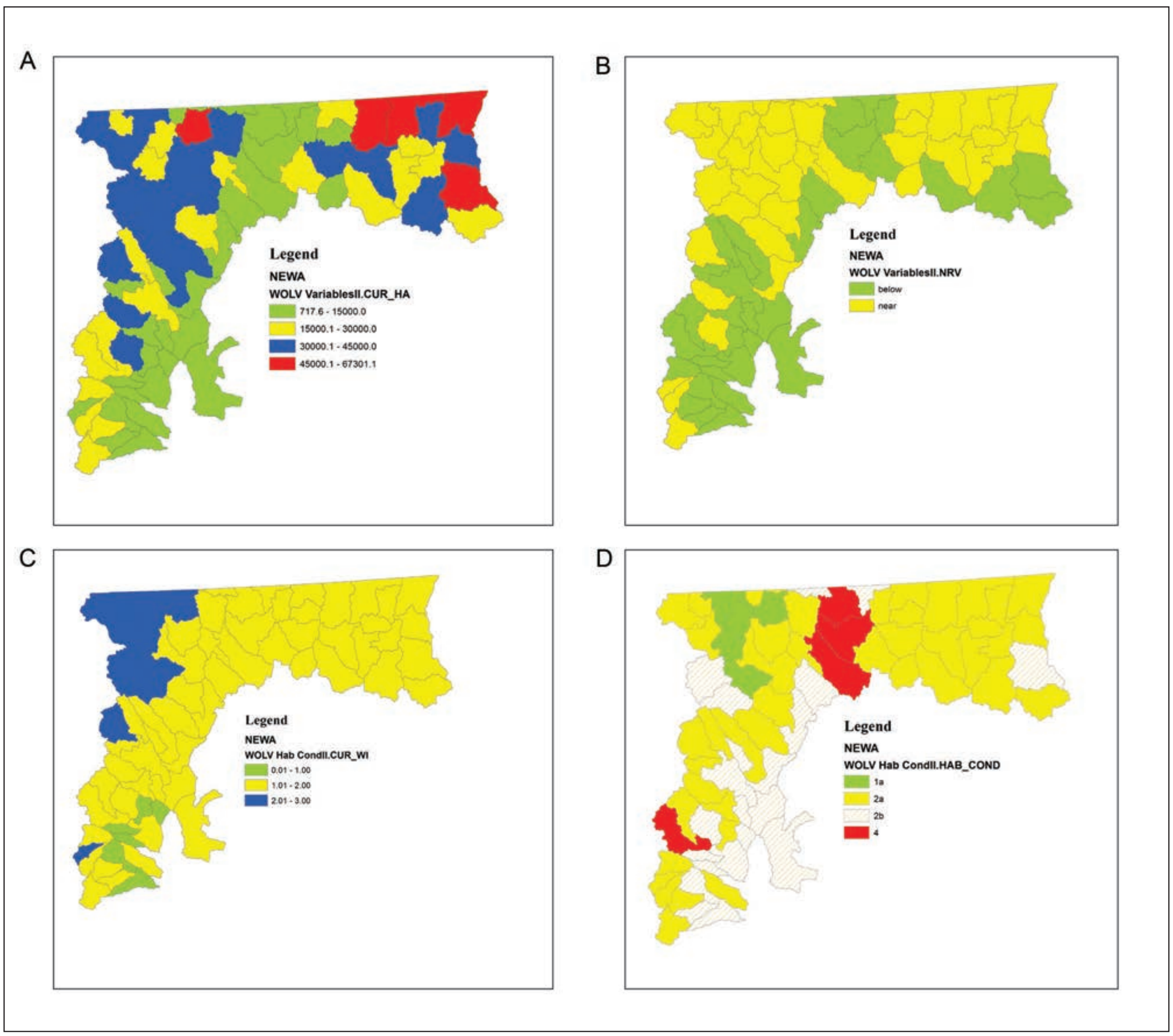

Figure 77—Current amount of (A) source habitat (acres) for, (B) current habitat departure, (C) watershed index, and (D) habitat condition class for wolverine, (WOLV) by watershed in the northeast Washington assessment area (NEWA).

All of the watersheds with high (>2.0) WI scores occurred within the Okanogan-Wenatchee National Forest portion of the assessment area and are largely a result of the presence of wilderness and roadless areas. These results are similar to those reported in other efforts to evaluate wolverine habitat. Raphael et al. (2001) evaluated wolverine habitat across the Columbia Basin and showed that the best habitat occurred along the Cascade Crest within the Okanogan-Wenatchee National Forest, while much lower scores associated with poorer habitat were found on the Colville National Forest. Similarly, Singleton et al. (2002) did not identify any "habitat concentration areas" for wolverine on the Colville National Forest, with the exception of a small area in the northeast portion of the Lower Pend Oreille watershed. However, they did identify important habitat linkages across the Kettle Crest and the Okanogan Highlands (Singleton et al. 2002).

This assessment of the amount of source habitat currently suggests declines from historical conditions. Our analysis found that 54 percent (n = 39) of the watersheds were near the historical median of source habitat, and 46 percent (n = 33) were below the historical median (fig. 77). Watersheds with the most source habitat (>98,800 ac) included Ross Lake, Pasayten River, Stehekin River, Middle Pend Oreille River, Lower Chewuch River, North Lake Roosevelt, Upper Pend Oreille River, Upper Chewuch River, Boulder/Deadman, and Lower Pend Oreille River. Two of the five watersheds with the high WI scores are in this group: Pasayten River and Upper Chewuch River.

Other factors that influenced the WI scores included the availability of alpine cirques used for denning habitat (Copeland 1996). One-half of the watersheds (n = 36) did not contain any potential wolverine denning habitat, while 14 percent (n = 10) included moderate to high levels of potential wolverine denning habitat. Watersheds with the greatest amount of potential denning habitat included Lost River, Pasayten, Twisp River, Upper Chewuch, and Upper Methow.

Currently, the influence of winter recreation routes has little effect on the potential wolverine denning habitat that we modeled because much of the denning habitat we identified occurred in wilderness areas or in remote areas that are difficult to access. However, there are winter recreation routes not in our inventory that may influence denning habitat. For example, helicopter skiing occurs within the Upper Methow watershed, which currently has a high amount of potential denning habitat.

Viability outcome scores—
The VOI model incorporated the WWI scores (described earlier), a habitat distribution index, and a habitat connectivity or permeability index. The WWI provides a

relative measure across watersheds of the potential capability of the watershed to contribute to the viability of the surrogate species. The WWI scores indicated that the current habitat capability for wolverine within the assessment area is 74 percent of the historical capability. This is largely due to the influence of human activities (roads) on wolverine habitats.

Currently, 72 percent (n = 52) of the watersheds contain source habitat amounts above 40 percent of the historical median, whereas historically, 86 percent (n = 62) were above this minimum habitat amount. The watersheds with >40 percent were distributed across all of the five ecoregions both currently and historically.

Because wolverines are highly mobile, we evaluated the contribution of dispersal habitat in the viability outcome model. Currently, dispersal habitat suitability of the assessment area for wolverine was rated as moderate to high. Across the assessment area, 8 percent of the watersheds rated as low dispersal habitat suitability, 48 percent rated as moderate, and 44 percent rated as high. Historically, dispersal habitat was projected to be high across the majority of the assessment area. These results are similar to other efforts to evaluate the dispersal habitat suitability for wolverine in the same general area (Singleton et al. 2002, WWHCWG 2010). Singleton et al. (2002) identified “fracture zones” that occur within the assessment area and warrant careful management attention. Fracture zones were defined as areas with considerable disruption of suitable habitat conditions for wolverine dispersal. Fracture zones included Stevens Pass, Snoqualmie Pass, and the portion of the assessment area from the Okanogan-Kettle-Selkirk Mountains where areas in public ownership are more limited and disjunct.

The current viability outcome for the assessment area is a 68 percent probability of B, which indicated that suitable environments for the wolverine were broadly distributed and of high abundance, but there were gaps where suitable environments are absent or only present in low abundance (fig. 78). However, the disjunct areas of suitable environments are typically large enough and close enough together to permit dispersal among subpopulations and to allow the species to potentially interact as a metapopulation. Historically, the viability outcome for wolverine had a probability of 79 percent A where suitable environments were more broadly distributed or of high abundance (fig. 78). In addition, the suitable environments were better connected, allowing for interspecific interactions. Our analysis indicated some reduction in the availability of suitable environments for the wolverine, and likely other species in the human disturbance group, occurred in the assessment area compared to the historical distribution and condition of their habitats.

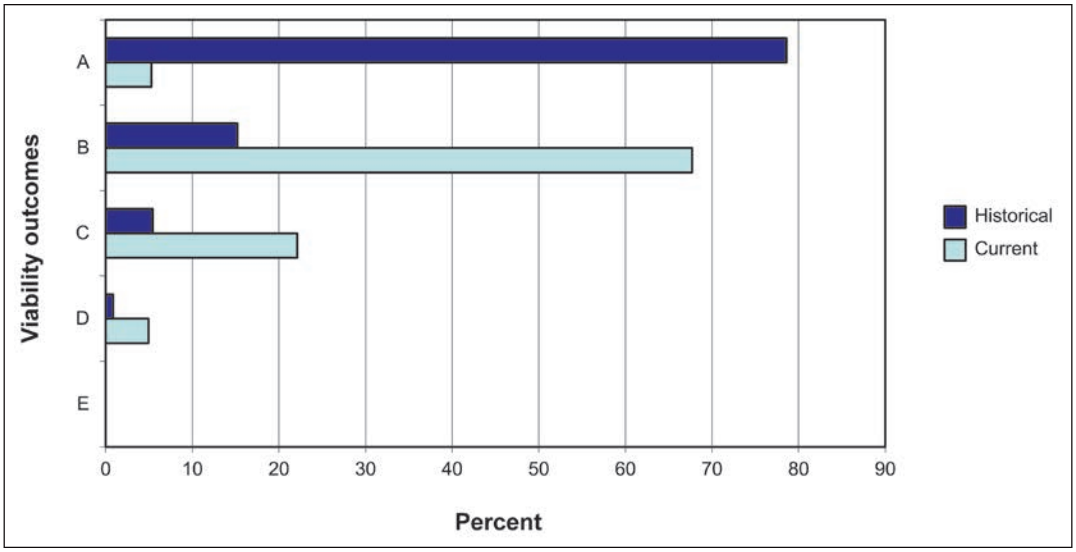

Figure 78—Current and historical viability outcomes for the wolverine in the northeast Washington assessment area.

High road densities have reduced the amount of source habitat in the assessment area.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. High road densities have reduced the amount of source habitat (Raphael et al. 2001, Wisdom et al. 2000) in the assessment area.
2. Habitat connectivity between patches of existing source habitats remains high north and south along the crest of the Cascade Range with exceptions along major highway corridors (Singleton et al. 2002, WWHCWG 2010). More concern about habitat connectivity for wolverines occurs between the North Cascade Range and Selkirk Mountains in the northeast portion of the assessment area where public lands are more limited (Singleton et al. 2002).

Wood Duck

Introduction

The wood duck was selected as a surrogate species for the riparian/large tree or snag/open water group to represent cavity-nesting species associated with forested riparian areas (streams, wetlands, ponds, lakes). The wood duck represents the cavity-nesting ducks in this group; unlike the other ducks in the group, wood ducks are widespread throughout Oregon and Washington. The common merganser is an exception because this species does not always nest in tree cavities. Breeding areas for wood ducks occurs primarily within western Washington; however, known breeding areas are patchily distributed across eastern Washington within the assessment area (Smith et al. 1997). Wood ducks typically winter farther south than Washington; however, significant wintering numbers can be found in the Yakima Valley (Lewis and Kraege 2000).

Model Description

Source habitat—

Wood ducks nest primarily in late-successional forests and riparian areas adjacent to low-gradient rivers, lakes, and wetlands (Lewis and Kraege 2000). At least 10 ac of wetland or other aquatic habitat should be available in a contiguous unit (USGS 2004) for successful nesting. Wood ducks nest almost exclusively in tree cavities, which offer protection from weather and predators (Peterson and Gauthier 1985, Robb and Bookhout 1995, Soulliere 1988). They are secondary cavity nesters, using cavities created by large woodpeckers or by decay or damage to the tree. Cavity use is dependent upon the proximity of nesting habitat and brood habitat (Robb and Bookhout 1995). Shallow wetlands within 0.5 mi of cavities provide optimal brood habitat (Lewis and Kraege 2000).

We modeled source habitat for wood ducks using a combination of forest structure and tree size data along with information from the National Wetlands Inventory. We used the following specific variables to map source habitat within each watershed (fig. 79):

- Cover type: All cover types at all elevations
- Tree structure and size: Single/multistory, >15 in QMD, 0.5 mi from a suitable waterbody or wetland complex as described below
- National Wetlands Inventory: Waterbodies and wetland complexes (PFO, R2, R3) >10 ac

Habitat departure was calculated as estimated as for other wetland-associated species by assuming that source habitat has been reduced from historical levels by 30 percent thus the habitat departure variable to assess current conditions was set at -2.

Snag habitat—

Soulliere (1988) suggested that trees needed to be >12 in d.b.h. to provide a suitable cavity for wood ducks to nest in. The optimal density of potential nest trees described by Sousa and Farmer (1983) was five per acre. We used snag data from forest inventory plots to assess the availability of suitable nesting habitat within source habitat (fig. 79). We used the following categories to assess the size and density of snag habitat on source habitat quality within each watershed.

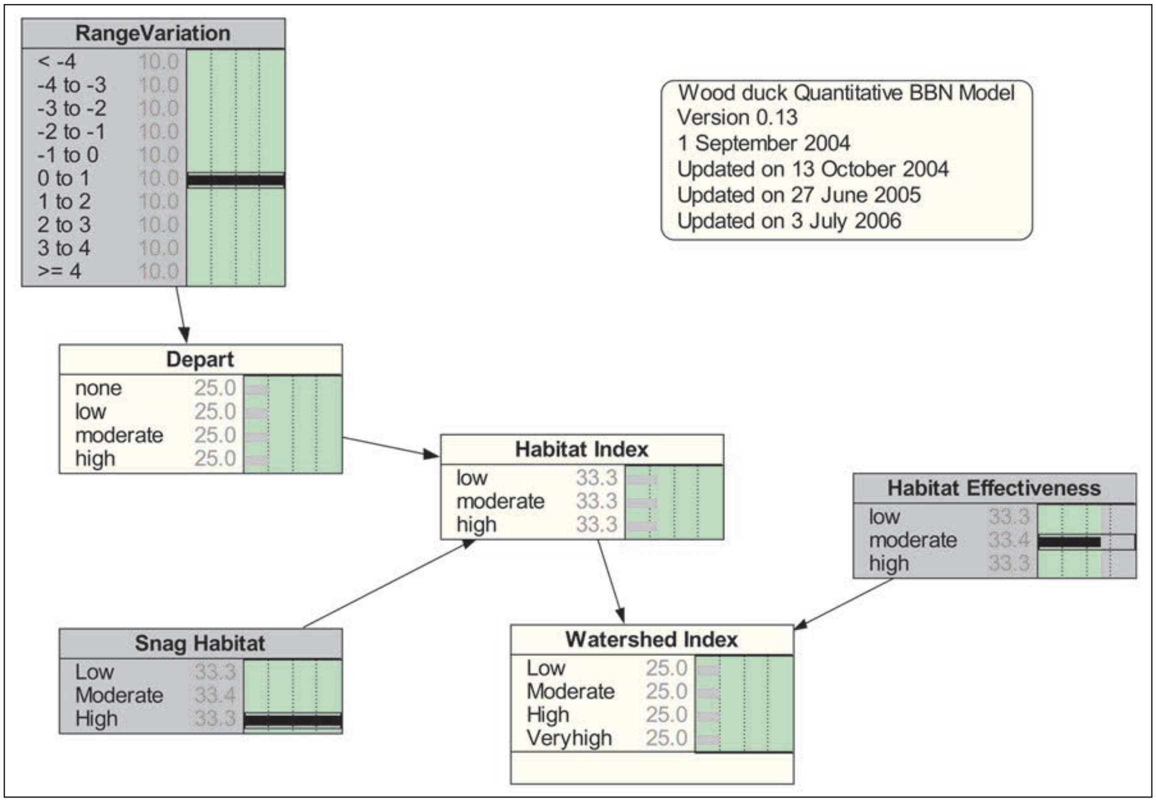

Figure 79—Surrogate species assessment model for the wood duck.

- Low habitat quality: <5.0 snags/ac >12 in d.b.h.
- Moderate habitat quality: 6.0 to 7.0 snags/ac >12 in d.b.h.
- High habitat quality: 7 to 10 snags/ac >12 in d.b.h.
- Very high habitat quality: >10 snags/ac >12 in d.b.h.

Habitat effectiveness—

Human disturbance has been shown to affect productivity of wood ducks by causing nest abandonment, egg mortality from exposure, increased predation of eggs and hatchlings, depressed feeding rates, and avoidance of otherwise suitable habitat (Hamman et al. 1999, Havera et al. 1992, Lewis and Kraege 2000). We used the waterfowl habitat disturbance index (Gaines et al. 2003) to evaluate the potential effects of human disturbance associated with roads and trails on source habitat. Open roads and trails were buffered by 820 ft and then overlaid with maps of source habitat to estimate the proportion of source habitat within a zone of influence in each watershed (fig. 79). We then categorized these potential effects as follows:

- Low habitat effectiveness: >50 percent of the source habitat in a zone of influence

- Moderate habitat effectiveness: 30 to 50 percent of the source habitat in a zone of influence
- High habitat effectiveness: <30 percent of the source habitat in a zone of influence

The relative sensitivity of WI values to the variables used in the model for wood duck are shown in table 46.

Assessment Results

Watershed scores—

There were two (3 percent) watersheds that had WI scores >2.0, 32 (47 percent) with WI scores from 1.0 to 2.0, and 34 (50 percent) were degraded with scores <1.0. Watersheds with the most source habitat (>1,240 ac) included Lower Chewuch, Middle Yakima, Upper Pend Oreille, Entiat, Stehekin, Upper Yakima, Chiwawa, Wenatchee, and Middle Methow Rivers. The median amount of source habitat across all watersheds with at least some habitat (68 watersheds) was 368.2 ac.

The availability of snag habitat (>12 in d.b.h.) within source habitat was very high in 6 (9 percent) of the watersheds, high in 4 (6 percent) of the watersheds, moderate in 15 (22 percent) of the watersheds, and low in 43 (63 percent) of the watersheds. Snag habitat is important for nesting wood ducks, and the majority of the watersheds had low availability of this critical habitat component.

Habitat effectiveness was high in three watersheds (4 percent), moderate in 10 (15 percent) watersheds, and low in 55 (81 percent) of the watersheds. Habitat effectiveness may be restored through management of human access. This would also reduce the loss of snags from roadside hazard tree removal and firewood cutting, as well as reduce the potential of negative effects associated with human disturbance at nest sites.

Table 46–Relative sensitivity of watershed index values to variables in the model for wood ducks

Model variables	Order of variable weighting
Source habitat	1
Snag density	3
Habitat effectiveness	2

Viability outcome scores—

The VOI model incorporated the WWI scores (described earlier) and a habitat distribution index. The WWI score indicated that the current habitat capability for wood ducks within the assessment area is 41 percent of the historical capability. This reduction occurred because of loss of wetland source habitat and low levels of snag habitat within source habitat in many watersheds.

Forty percent of the historical median amount of source habitat across all watersheds with at least some habitat was 156.9 ac. A total of 40 (50 percent) of the watersheds within the assessment area met this habitat minimum. The watersheds with >40 percent were distributed across all five ecoregions.

Historically, all five ecoregions contain at least one watershed with >40 percent of the median amount of historical source habitat (median was calculated across all watersheds with source habitat). Forty-seven (69 percent) watersheds had >40 percent of the median amount of historical source habitat.

Currently, there is a 72 percent probability that the viability outcome for wood duck within the assessment area is C (fig. 80), suggesting that suitable conditions for the wood duck are likely well distributed in only a portion of the assessment area. Gaps exist where suitable environments are either absent or present in low abundance. Historically, there was a 71 percent probability of outcome A and a 19 percent probability of outcome B (fig. 80), where suitable environments were more

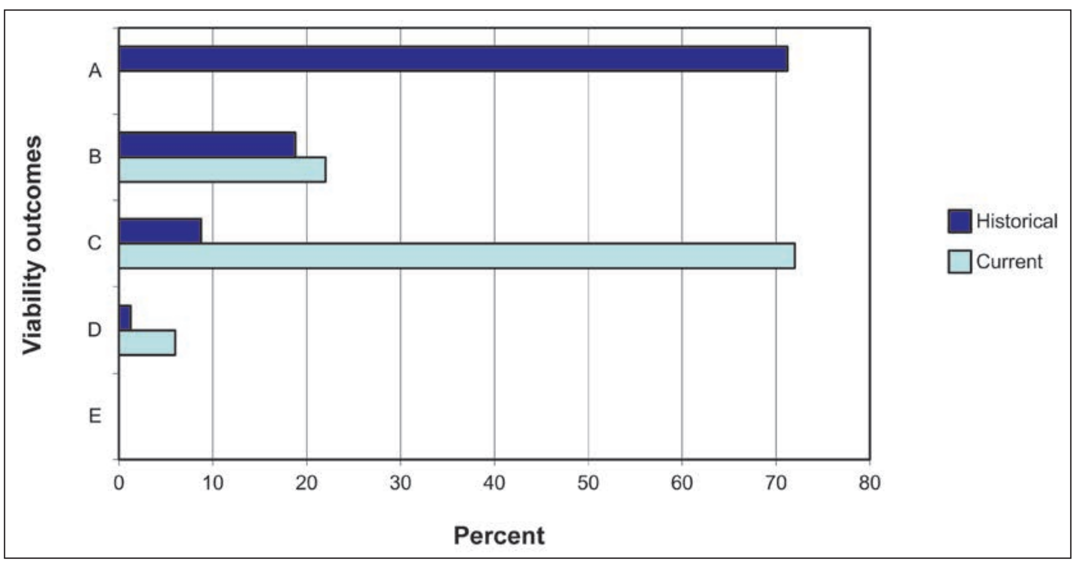

Figure 80—Current and historical viability outcomes for the wood duck in the northeast Washington assessment area.

broadly distributed or at higher abundance. This resulted in suitable environments that were better connected. A reduction in suitable environments for the wood duck, and likely other species in the riparian/large tree or snag/open water group, has occurred in the assessment area compared to historical conditions.

Management Considerations

The following issues were identified during this assessment and from the published literature for consideration by managers:

1. The area of wetland habitats has experienced significant declines across the region and in some portions of the assessment area.
2. The influence of human activities within wood duck source habitat has reduced the availability of nesting habitat (large snags) and habitat effectiveness.

Chapter 3: Multispecies Conservation

Introduction

In this chapter, we present the procedures used to address multiple species using results of our individual species assessments. Wisdom et al. (2002) described a process for assessing the habitat conditions for groups of species in order to identify a habitat network for terrestrial wildlife in the interior Columbia Basin. We modified the approach of Wisdom et al. (2002) to integrate information from individual surrogate species into multispecies assessments. For each surrogate species, we determined the condition of the habitat in each watershed, which led to a conservation emphasis for the surrogate species in each watershed. Conservation emphases consisted of protection, restoration, connectivity, or combinations of each of these. We then created a matrix of all surrogate species and conservation emphasis that addressed their habitat and risk factors for multiple species (table 47). The steps we used to go from individual species to multiple species are described below:

Step 1: Determine habitat conditions for each 5th-field watershed for each surrogate species. The habitat conditions were completed as part of the individual species assessments (see chapter 2) and are based on the watershed index (WI) scores, the current amount of source habitat relative to reference conditions, habitat dispersal suitability (for some species), and the amount of source habitat in federal ownership within a watershed.

Step 2: We grouped surrogate species by whichever conservation emphasis that would best address their habitat and risk factors (table 47). We then used individual surrogate species assessments to identify a habitat condition that best addressed the group for each watershed. We used the most limited (fewest watersheds in good condition) species in the group to identify a habitat condition for each watershed. For example, if species A had more watersheds with habitat condition 1a (which means relative good conditions) than species B, we used species B to identify priority watersheds for protection, restoration, or connectivity.

Step 3: We then identified a single set of priority watersheds and management considerations to address multiple species. By using a combination of management considerations and a set of priority watersheds, managers have important information and tools to contribute to the viability of surrogate species.

Step 4: The final step, which is outside the scope of this assessment, will be to integrate the results of this assessment with other resources through an interdisciplinary planning process.

Table 47—Conservation emphasis areas to improve viability outcomes for surrogate species[a]

	Conservation emphasis areas to improve surrogate species viability							
Surrogate species	**Aquatic and riparian**	**Snag and down wood**	**Moist forest**	**Mesic-dry forest**	**Postfire harvest**	**Human access**	**Domestic grazing**	**Invasive species**
American marten	X	X	X			X		
Bald eagle	X	X	X	X		X		
Bighorn sheep				X		X	X	
Black-backed woodpecker		X			X	X		
Canada lynx		X	X		X	X		
Cassin’s finch			X	X			X	
Columbia spotted frog	X					X	X	X
Eared grebe	X					X		X
Fox sparrow							X	
Fringed myotis	X	X	X	X	X	X		
Golden eagle				X		X	X	X
Harlequin duck	X	X	X			X		
Larch mountain salamander			X					
Lark sparrow							X	
Lewis’s woodpecker		X		X	X	X		
MacGillivray’s warbler	X						X	X
Marsh wren	X							X
Northern bog lemming	X					X	X	
Northern goshawk		X	X	X		X		
Northern harrier	X					X	X	
Peregrine falcon	X					X		
Pileated woodpecker		X	X	X		X		
Sage thrasher						X	X	X
Tailed frog	X		X			X		X
Tiger salamander	X					X	X	X
Townsend’s big-eared bat	X	X				X		

Table 47—Conservation emphasis areas to improve viability outcomes for surrogate species[a] (continued)

	Conservation emphasis areas to improve surrogate species viability							
Surrogate species	**Aquatic and riparian**	**Snag and down wood**	**Moist forest**	**Mesic-dry forest**	**Postfire harvest**	**Human access**	**Domestic grazing**	**Invasive species**
Western bluebird		X		X	X	X		
Western gray squirrel		?		X		X	X	X
Water vole	X						X	
White-headed woodpecker		X		X	X	X		
Wilson’s snipe	X							
Wolverine						X		
Wood duck	X	X				X		

[a] Species were selected and highlighted to guide the development of conservation emphasis and management considerations.

Conservation Emphasis Areas

We used the information from individual species assessments to identify eight broad conservation emphasis areas that address habitat and risk factors for multiple surrogate species. The multispecies emphasis include two parts: management considerations and a prioritized list of watersheds. We prioritized watersheds in order to identify areas with the highest potential to contribute to the viability of the surrogate species. The conservation emphasis areas included aquatic and riparian, snag and down wood, moist forests, mesic-dry forests, and postfire habitats, human access, domestic grazing, and invasive species. The conservation emhpasis areas and associated management considerations address many of the climate change adaptations identified by Gaines et al. (2012), and could be used to inform forest planning through an interdisciplinary planning process to address multiple resource objectives.

Aquatic and Riparian Habitat Conservation Emphasis Area

We identified eight surrogate species whose viability was closely linked to riparian habitats. These species represent a wide range of riparian habitats and included the water vole, inland tailed frog, bald eagle, MacGillivray’s warbler, Columbia spotted frog, Wilson's snipe, eared grebe, and marsh wren. Other surrogate species whose viability would also benefit from management considerations include the northern bog lemming, American marten, fringed myotis, peregrine falcon, tiger salamander, northern harrier, Townsend's big-eared bat, wood duck, and harlequin duck.

The following management considerations address issues we identified regarding the viability of the surrogate species associated with riparian and wetland habitats. The number of watersheds that are priority are shown in table 48.

1. Viability outcomes could be improved by riparian management that considers the needs of fish and wildlife resources and address the effects of roads, campgrounds, grazing, and vegetation management. Riparian management zones could be designated to (a) protect habitat adjacent to streams, wetlands, ponds and lakes; and (b) facilitate the movement/dispersal of wildlife.
2. Restoration of riparian habitats by reducing the negative effects of roads on source habitats within priority watersheds (table 48).
3. Restoration of wetland and wet meadow habitats by reducing the impacts of water-based recreation, invasive species, and conifer encroachment within priority watersheds (table 48).

Snag and Down Wood Conservation Emphasis Area

We reviewed DecAID (Mellen-McLean et al. 2009) to develop reference conditions for snag density distributions (tables 49 and 50) that can be used to develop management considerations for snag and downed wood habitats. We developed estimates (histograms) of the reference conditions using the inventory data for unharvested plots (including plots with no measurable snags) for the structural stages (weighted averages) within the mesic, cold-moist, and cold-dry forest types. For the dry forest, we used reference condition estimates from Harrod et al. (1998).

Forested Habitats Conservation Emphasis Areas

The conservation emphasis areas for forest habitats include three parts: moist forests, mesic and dry forests, and postfire forests.

Moist forests conservation emphasis area—
We used habitat condition information from three surrogate species associated with moist late-successional forests to identify multispecies management considerations: northern goshawk, pileated woodpecker, and American marten. Other surrogate species whose viability is likely influenced by these measures include the Canada

Table 48—Priority watersheds by habitat condition, conservation approach, and conservation emphasis

Habitat condition	Potential management options to consider	Number of priority watersheds by habitat condition and conservation emphasis: Aquatic and riparian	Moist forest habitat	Mesic-dry forest	Postfire habitat	Human access	Domestic grazing	Invasive species
Habitat condition 1	Protection of existing source habitat is a high priority. Restoration to enhance source habitat amount and connectivity would occur as needed.	7	12	22	20	13	11	6
Habitat condition 2	Restoration to enhance source habitat amount and connectivity is a high priority. Protection of existing source habitat is also a priority.	41	19	21	31	45	31	37
Habitat condition 3	A combination of protection and restoration would occur in these watersheds. Restoration would depend on the availability of resources after higher priority (habitat condition 1, 2) watersheds have been restored.	5	26	17	12	1	9	12
Habitat condition 4	The primary emphasis in these watersheds is providing suitable conditions for species dispersal in order to enhance habitat connectivity.	3	7	3	0	5	5	0
Habitat condition 5	The limited amount of source habitat that is in federal ownership limits the contribution of these watersheds to species sustainability. However, depending on their juxtaposition to other watersheds, protecting or restoring source habitat conditions may still be important.	16	10	9	8	13	16	16

Table 49—Snag reference conditions by density distribution classes for small and large snag sizes for dry forests (applied at the watershed scale)

Snag size class	Percent of landscape in snag density classes (number/acre)				
	0-4	**4-12**	**12-20**	**20-28**	**>28**
>10 in d.b.h.	82.2	13.7	2.1	1.4	0.4
	0-2	**2-6**	**6-10**	**10-14**	**>14**
>20 in d.b.h.	89.0	9.6	0.6	0.0	0.0

D.B.H. = diameter at breast height.

Table 50—Snag reference conditions by density distribution classes for small and large snag sizes for mesic forests (applied at the watershed scale)

Snag size class	Percentage of landscape in snag density classes (number/acre)				
	0–6	**6–18**	**18–30**	**30–42**	**>42**
>10 in d.b.h.	70.0	18.0	4.7	4.1	2.8
	0–2	**2–6**	**6–10**	**10–14**	**>14**
>20 in d.b.h.	77.9	12.0	6.0	2.6	1.6

D.B.H. = diameter at breast height.

lynx, Cassin's finch, larch mountain salamander, tailed frog, harlequin duck, fringed myotis, and the bald eagle.

The following management considerations may be used to enhance the viability of the surrogate species associated with moist forests:

Managers could identify and protect large tree and snag habitat (see snag and down wood strategy) within all forest types, including postfire habitats.

1. In moist forests within watersheds with habitat condition 1a and 1b protection or restoration of late-successional forest habitat conditions (Franklin and Johnson 2012), can enhance forest species viability. Reference conditions can be used as a guide to determine sustainable levels of late-successional forest within each subbasin or watershed (Agee 2003, Hessburg et al. 2000, 2013; USFWS 2011; Wimberly et al. 2000). In areas where the management goal is to restore large tree structures, treatments may include thinning young stands that were previously harvested to accelerate the development of old-forest structures and restore patch sizes (Franklin and Johnson 2012).
2. Patch-size distribution measured at the watershed scale could be managed within or toward reference conditions (Hessburg et al. 1999a, Perry et al. 2011).
3. To increase viability outcomes, managers could identify and protect large tree and snag habitat (see snag and down wood strategy) within all forest types, including postfire habitats (Franklin and Johnson 2012). These structures are important for both current and future (legacy structure) habitat for late-successional species.
4. Viability outcomes for goshawk could be improved by maintaining stands with active goshawk nests in old-forest conditions (Wisdom et al. 2000). The Northern Goshawk Scientific Committee recommends three 30-ac nest stands per breeding pair and three additional 30-ac replacement stands within a 6,000-ac area that functions as potential home range (Reynolds et al. 1992).

5. The northern spotted owl could be a high priority for monitoring within its range on the Okanogan-Wenatchee National Forest because of its negative trend in habitat loss owing to fire (Davis and Lint 2005, Davis et al. 2011), negative population trends (Anthony et al. 2006, Forsman et al. 2010), and competition from barred owls (Singleton et al. 2010).

Mesic and dry forests conservation emphasis area—
We used habitat condition information from two surrogate species associated with large-tree structures within mesic and dry forests to identify multispecies management considerations: northern goshawk and white-headed woodpecker. Other surrogate species whose viability is likely influenced include Cassin's finch, pileated woodpecker, western bluebird, western gray squirrel, Lewis's woodpecker, golden eagle, bighorn sheep, and bald eagle.

In mesic and dry forests, restoration of structure, composition, and function using a combination of thinning or prescribed fire could improve habitats.

The following management considerations may be used to enhance the viability of the surrogate species associated with mesic and dry forests. The number of watersheds that are priority management considerations are shown in table 48.

1. Protection of existing old-forest ponderosa pine habitats would enhance viability outcomes (Franklin and Johnson 2012). These forests provide important source habitat for surrogate species associated with dry forests and are currently available at levels well below reference conditions (Hessburg et al. 1999a).
2. In mesic and dry forests, restoration of structure, composition, and function using a combination of thinning or prescribed fire (Agee 2003, Gaines et al. 2007, Harrod et al. 2007) could improve habitats. Reference conditions may be used to guide the development of stand- (Churchill et al. 2013, Harrod et al. 1999, Franklin et al. 2008) and landscape-level desired conditions (Agee 2003; Hessburg et al. 2000, 2005, 2007, 2013).
3. Patch-size distribution measured at the watershed scale could be managed within or toward reference conditions (Hessburg et al. 2007, Perry et al. 2011).
4. Protection of active goshawk nests in old-forest conditions (Wisdom et al. 2000) could contribute to their viability. The Northern Goshawk Scientific Committee recommends three 30-ac nest stands per breeding pair and three additional 30-ac replacement stands within a 6,000-ac area that function as potential home range, (Reynolds et al. 1992). This may require predisturbance surveys for goshawks.

5. Surrogate species associated with mesic and dry forests could be a high priority for monitoring. The white-headed woodpecker and western bluebird were ranked as a high priority (see chapter 4) throughout the assessment area, owing to the strongly negative trends in source habitat availability and the unknown effects of dry forest restoration on their habitat use and productivity.
6. Habitat for the northern spotted owl in dry and mesic forests may be restored using the upper end of the reference conditions to restore mesic and dry forest processes, patterns, and functions (Courtney et al. 2008, Franklin et al. 2008, Gaines et al. 2010b, Hessburg et al. 2013, USFWS 2011). The risk of loss of spotted owl habitat to uncharacteristically high-severity fire may be reduced by strategically locating restoration treatments to reduce landscape fire movement (Agar et al. 2007, Franklin et al. 2008, Gaines et al. 2010b, Lehmkuhl et al. 2007b, USFWS 2011). Where treatments occur, stand conditions favorable for white-headed woodpeckers and other dry forest-associated surrogate species may be created (Gaines et al. 2007, 2010a).

Postfire habitat conservation emphasis area—
The primary surrogate species we used to develop the multispecies management considerations for postfire habitats were the Lewis's woodpecker and black-backed woodpecker. Additional surrogate species that we expect to benefit include Canada lynx, white-headed woodpecker, western bluebird, and fringed myotis.

The following management considerations may be used to enhance the viability of the surrogate species associated with postfire habitats. The number of watersheds that are priority for management considerations are shown in table 48.

1. Reference conditions measured at the landscape scale could be used to evaluate if habitat components are distributed across the landscape in a sustainable fashion (Agee 2000, Hessburg et al. 2007). Increase opportunities to allow wildfire to burn or ignite fires when conditions and opportunities exist within priority watersheds (table 48).
2. Watersheds in habitat conditions 1a and 1b for black-backed and Lewis's woodpeckers could be managed for postfire habitats using reference conditions. Postfire timber harvest could be designed to meet habitat needs for the Lewis's woodpecker (see chapter 2) and evaluated using the Lewis's woodpecker surrogate species assessment model at the watershed and subbasin scales.
3. In watersheds with habitat condition 2 and 3, planned and unplanned ignitions may be used to restore the availability of postfire habitats toward reference conditions.

Human Access Emphasis Area

We used habitat condition information from the Canada lynx, wolverine, bighorn sheep, and harlequin duck to conduct a multispecies assessment that addresses habitat and human-disturbance related risk factors. Other surrogate species whose viability is likely influenced include the golden eagle, sage thrasher, eared grebe, northern bog lemming, northern goshawk, American marten, white-headed woodpecker, western bluebird, fringed myotis, western gray squirrel, Lewis's woodpecker, black-backed woodpecker, peregrine falcon, tiger salamander, northern harrier, Townsend's big-eared bat, and the wood duck.

The following management considerations may be used to enhance the viability of the surrogate species whose source habitats are influenced by roads and trails. The number of watersheds that are priority management considerations are shown in table 48.

1. Winter recreation could be managed using the lynx consideration assessment and strategy guidance for snow compacting activities (Ruediger et al. 2000, ILBT 2013), especially in watersheds identified as habitat condition 1 and 2 (table 48) within the areas identified as core and secondary for the Canada lynx (USFWS 2005, ILBT 2013). A more complete inventory of the existing locations of snowmobile use and other compacted winter routes would aid in the development of a winter recreation strategy.
2. The impacts of human activities (roads, trails, dispersed recreation sites, etc.) on riparian habitats could be reduced by emphasizing watersheds in habitat conditions 1 and 2.
3. The impacts of roads on surrogate species source habitats could be reduced by considering road management that limits overall road density and the amount of area within a zone of influence of a road (Gaines et al. 2003). The number of priority watersheds for addressing road-related effects are identified in table 48. In many cases, these watersheds overlap with grizzly bear recovery areas where access management guidelines will be implemented (USFWS 1997b), providing conservation values for multiple species.
4. Using Singleton et al. (2002), WHCWG (2010), and permeability information from this assessment, unroaded areas or areas with low road densities could be identified to serve as steppingstones to enhance habitat permeability for wildlife, especially within watersheds identified as important for connectivity (Habitat condition 4, table 48). Road crossing structures could facilitate movement across the Interstate 90 corridor, or other similar areas. Human activities adjacent to highways could be managed so that wildlife could access crossing structures.

The impacts of roads on surrogate species source habitats could be reduced by considering road management that limits overall road density and the amount of area within a zone of influence of a road.

Domestic Grazing Emphasis Area

The primary surrogate species we used in our multispecies conservation assessment to address issues associated with domestic grazing were the golden eagle, bighorn sheep, northern harrier, and the MacGillivray's warbler. Additional surrogate species that we expect to benefit include the water vole, western gray squirrel, northern bog lemming, Cassin's finch, fox sparrow, lark sparrow, sage thrasher, tiger salamander, and the Columbia spotted frog. Lehmkuhl et al. (2013) provide a comprehensive assessment of grazing impacts from domestic and wild ungulates on the southern portion of the Okanogan-Wenatchee National Forest.

The following management considerations may be used to enhance the viability of the surrogate species whose source habitats are influenced by domestic grazing. The number of watersheds that are priority are shown in table 48.

1. Grazing management could be developed to (a) restore habitat conditions in riparian and other unique habitats (Beebe et al. 2002, Lehmkuhl et al. 2013); (b) provide forage for ungulates on winter ranges (Lehmkuhl et al. 2013); and foraging habitat for species such as the golden eagle; and (c) maintain known populations of the mardon skipper (U.S. Forest Service Pacific Northwest Region [6] sensitive species).
2. Reducing the potential for disease spread from domestic to bighorn sheep in areas where bighorn sheep are currently present (Schommer and Woolever 2008) would increase the likelihood of persistence.

Invasive Species Emphasis Area

The surrogate species we used to assess impacts from invasive species were the golden eagle, tiger salamander, Columbia spotted frog, eared grebe, and marsh wren. Additional surrogate species that we expect to benefit include sage thrasher, tailed frog, and the MacGillivray's warbler.

The following management considerations may be used to enhance the viability of the surrogate species whose source habitats are influenced by invasive species. The number of watersheds that are priority are shown in table 48.

1. Reducing the impact and spread of invasive plant species into grassland, shrubland, and wetland habitats that provide source habitat for surrogate wildlife species could improve habitat quality.
2. Coordination with the state fisheries agency to reduce the impacts of introduced fish species would increase the likelihood of maintaining viable populations.

Chapter 4: Monitoring and Adaptive Management

Monitoring and adaptive management become vital tools to help resource managers deal with complex management questions and high levels of uncertainty (Busch and Trexler 2003, Christensen et al. 1996, Christensen 1997, Everett et al. 1994, Gaines et al. 2003b, Suring et al. 2011). Assessing the viability of species is complex and involves uncertainties. Key assumptions made in the assessment of species viability in our process include (Suring et al. 2011):

- Surrogate species assessment models provide a conceptual outline of the primary habitat and risk factors that determine the viability of surrogate species.
- The assessment models provide a reasonable and scientifically credible structural approximation of the species niche in the ecosystem that can be used to identify key monitoring elements.
- Surrogate species represent the species group in a manner that provides insights into the capability of the habitat to support other species associated with the group.

These assumptions guide development of specific monitoring and research questions that differ for each surrogate species. The sheer number of surrogate species selected to represent various habitats and risk factors make it impossible to monitor all surrogate species in a rigorous manner owing to cost and impracticality. Therefore, we developed a process to prioritize surrogate species monitoring based on the following:

- The results of the assessment of surrogate species viability.
- Whether the effects of risk factors that influenced species' viability are likely to increase, decrease, or remain the same based on proposed management options.
- The degree of uncertainty associated with our ability to predict the relationship between surrogate species, their source habitat, and associated risk factors.

We anticipate using the surrogate species assessment models in an adaptive management approach where the models provide an initial estimate, based on current science and professional knowledge, of how the surrogate species interacts with source habitat and risk factors.

We anticipate using the surrogate species assessment models in an adaptive management approach where the models provide an initial estimate, based on current science and professional knowledge, of how the surrogate species interacts with source habitat and risk factors (Nyberg et al. 2006). The surrogate species assessment models we developed can serve several important purposes in an adaptive management process, including documenting the current state of knowledge about a species, identifying and clarifying key assumptions, identifying areas of uncertainty, testing sensitivity of outcomes to changes in variable values, and evaluating alternative decisions (Nyberg et al. 2006).

Bayesian belief and decision networks are modeling techniques that are well suited to adaptive management applications (Nyberg et al. 2006). We developed a process to determine the priority of a surrogate species for monitoring and a recommended intensity of monitoring that are based on the information used in the surrogate species assessment models. We based this process largely on the degree of risk posed to the surrogate species.

Surrogate Species Monitoring Priorities Based on Viability Outcomes

We used the viability outcomes based on current conditions to develop an initial priority rating for each surrogate species for monitoring (table 51); those with low viability outcomes were high priority, and those with high viability outcomes were low priority (fig. 81). We defined a species that is low priority for monitoring as one with a ≥60 percent probability of outcome B or better and with <5 percent outcome E. A high-priority species was defined as one with a >10 percent probability of outcome E or >40 percent probability of outcome D. The remainder of the species were moderate priority for monitoring.

Based on this approach, monitoring for species ranked as high would include monitoring habitat, risk factors, and population trend (fig. 81). For those species ranked as moderate, habitat and risk factors would be monitored every 2 years to determine trends in their viability outcomes. For species ranked as low, habitat and risk factors would be monitored every 5 years.

Table 51—Priority of focal species for monitoring based on current condition estimates of their viability outcomes

Monitoring priority	Surrogate species
High	Cassin’s finch, eared grebe, fox sparrow, lark sparrow, sage thrasher, western bluebird, white-headed woodpecker, Wilson’s snipe, bighorn sheep
Moderate	Bald eagle, black-backed woodpecker, spotted frog, Lewis’s woodpecker, MacGillivray’s warbler, marsh wren, northern harrier, pileated woodpecker, tailed frog, tiger salamander, wood duck, American marten
Low	Golden eagle, harlequin duck, goshawk, peregrine falcon, Canada lynx, wolverine

		Risk Factors			
Viability outcome		Increasing	Decreasing/same	Unknown	Degree of uncertainty
	High	Moderate priority Monitor habitat and risk factors every 2 years.	Low priority Monitor habitat and risk factors every 5 years.	Moderate priority Monitor habitat and risk factors every 2 years.	Low
	Low	High priority Monitor habitat, risk factors, and populations.	Moderate priority Monitor habitat and risk factors every 2 years.	High priority Monitor habitat, risk factors, and populations.	High

Figure 81—Relationship between focal species monitoring priorities and the viability outcome, predicted risk factors associated with the management option, and the degree of scientific uncertainty.

The final determination about which species are priority to monitor and the intensity of the monitoring can be based on how well the management guidance in land and resource management plans address habitat and risk factors. The analyses of how well habitats and risk factors are addressed in management guidance could be displayed in the effects analyses for each of the management alternatives considered (e.g., see Lehmkuhl et al. 1997, Raphael et al. 2001).

Metric Equivalents

When you now:	**Multiply by:**	**To find:**
Inches (in)	2.54	Centimeters
Feet (ft)	.305	Meters
Miles (mi)	1.609	Kilometers
Acres (ac)	.405	Hectares

Literature Cited

Abele, S.C.; Saab, V.A.; Garton, E.O. 2004. Lewis's woodpecker (*melanerpes lewis*): a technical conservation assessment. USDA Forest Service, Rocky Mountain Region. http://www.fs.fed.us/r2/projects/scp/assessments/lewisswoodpecker.pdf. (May 2010).

Abuladze, A.; Shergalin, J. 2002. The golden eagle in north Caucasia and Transcaucasia. Journal of Raptor Research. 36 (supplement): 10–17.

Adams, M.J.; Pearl, C.A.; McCreary, B.; Galvan, S.K.; Wessell, S.; Wente, W. H.; Anderson, C.W.; Kuehl, A.B. 2009. Short-term effect of cattle exclosures on Columbia spotted frog populations and habitat in northeastern Oregon. Journal of Herpetology. 43: 132–138.

Agee, J.K. 2000. Disturbance ecology of North American boreal forests and associated northern mixed/subalpine forests. In: Ruggerio, L.F.; Aubry, K.B.; Buskirk, S.W.; Koehler, G.M.; Krebs, C.J.; McKelvey, K.S.; Squires, J.R. Ecology and conservation of lynx in the United States. Gen. Tech. Rep. RMRS-GTR-30WWW. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 39–82.

Agee, J.K. 2003. Historical range of variability in eastern Cascade forests, Washington, USA. Landscape Ecology. 18: 725–740.

Ager, A.A.; Finney, M.A.; Kerns, B.K.; Maffei, H. 2007. Modeling wildfire risk to northern spotted owl (*Strix occidentalis caurina*) habitat in central Oregon, USA. Forest Ecology and Management. 246: 45–56.

Alexander, S.M.; Waters, N.M. 2000. Modeling wildlife movement requisites in the Banff-Bow Valley transportation corridor. International conference on integrating GIS and environmental modeling (GIS/EM4). http://www.colorado.edu/research/cires/banff/pubpapers/24/. (December 7, 2014)

Altman, B. 2000a. Conservation strategy for landbirds in the Northern Rocky Mountains of eastern Oregon and Washington Oregon—Washington Partners in Flight. Version 1. 140 p. http://www.orwapif.org/pdf/northern_rockies.pdf. 1/15/08.

Altman, B. 2000b. Conservation strategy for landbirds of the east-slope of the Cascade Mountains in Oregon and Washington. Version 1.0. June 2000. American Bird Conservancy and Oregon-Washington Partners in Flight. http://www.orwapif.org/sites/default/files/east_slope.pdf. (May 2, 2014).

Andelman, S.J.; Beissinger, S.; Cochrane, J.F.; Gerber, L.; Gomez-Priego, P.; Groves, C.; Haufler, J.; Holthausen, R.; Lee, D.; Maguire, L.; Noon, B.; Ralls, K.; Regan, H. 2001. Scientific standards for conducting viability assessments under the National Forest Management Act: report and recommendations of the NCEAS working group. National Center for Ecological Analysis and Synthesis, University of California, USA. http//www.fs.fed.us/psw/rsl/projects/wild/lee/lee3.pdf. (24 January 2017).

Andelman, S.J.; Groves, C.; Regan, H.M. 2004. A review of protocols for selecting species at risk in the context of U.S. Forest Service viability assessments. Acta Oecologia. 26: 75–83.

Andersen, D.E. 1991. Management of North American grasslands for raptors. In: Pendleton, B.G.; Krahe, M.N.; LeFranc, K., Jr.; Titus, K.; Bednarz, J.C.; Anderson, D.E.; Millsap, B.A., eds. Proceedings of the Midwest raptor management symposium and workshop. Washington, DC: National Wildlife Federation: 203–210.

Anderson, E.M. 2007. Changes in bird communities and willow habitats associated with fed elk. Wilson Journal of Ornithology. 119: 400–409.

Anderson, M. 1995. Interactions between *Lythrum salicaria* and native organisms: a critical review. Environmental Management. 19: 22–231.

Andruskiw, M.; Fryxell, J.M.; Thompson, I.D.; Baker, J.A. 2008. Habitat-mediated variation in predation risk by the American marten. Ecology. 89: 2273–2280.

Anthony, R.G.; Forsman, E.D.; Franklin, A.B.; Anderson, D.R.; Burnham, K.P.; White, G.C.; Schwarz, C.J.; Nichols, J.D.; Hines, J.E.; Olson, G.S.; Ackers, S.H.; Andrews, L.S.; Biswell, B.L.; Carlson, P.C.; Diller, L.V.; Dugger, K.M.; Fehring, K.E.; Fleming, T.L.; Gerhardt, R.P.; Gremel, S.A.; Gutierrez, R.J.; Happe, P.J.; Herter, D.R.; Higley, J.M.; Horn, R.B.; Irwin, L.L.; Loschl, P.J.; Reid, J.A.; Sovern, S.G. 2006. Status and trends in demography of northern spotted owls, 1983-2003. Wildlife Monographs. 163: 1–48.

Anthony, R.G.; Isaacs, F.B. 1989. Characteristics of bald eagle nest sites in Oregon. Journal of Wildlife Management. 53: 148–159.

Anthony, R.G.; O'Connell, W.A.; Pollock, M.M.; Hallett, J.G. 2003. Association of mammals with riparian ecosystems in Pacific Northwest forests. In: Zabel, C.J.; Anthony, R.G., eds. Mammal community dynamics: management and conservation in the coniferous forests of North America. New York: Cambridge University Press: 510–563.

Anthony, R.G.; Steidl, R.J.; McGarigal, K. 1995. Recreation and bald eagles in the Pacific Northwest. In: Knight, R.L.; Gutzwiller, K.J., eds. Wildlife and recreationists: coexistence through management and research. Washington, DC: Island Press: 223–241.

Apfelbaum, S.I.; Seelbach, P. 1983. Nest tree, habitat selection and productivity of seven North American raptor species based on the Cornell University nest record card program. Raptor Research. 17: 97–113.

Ashley, E.P.; Robinson, J.T. 1996. Road mortality of amphibians, reptiles, and other wildlife on the Long Point Causeway, Lake Erie, Ontario. Canadian Field-Naturalist. 110(3): 403–412.

Ashley, J. 1994. 1992-93. Harlequin duck monitoring and inventory in Glacier National Park, Montana. Glacier National Park, MT: U.S. Department of the Interior, National Park Service, Glacier National Park, Division of Resource Management, Montana. 57 p.

Aubry, K.B. 2000. Amphibians in managed, second-growth Douglas-fir forests. Journal of Wildlife Management. 64(4): 1041–1052.

Aubry, K.B.; Koehler, G.; Squires, J.R. 1999. Ecology of Canada lynx in the southern boreal forests. In: Ruggerio, L.F.; Aubry, K.B.; Buskirk, S.W.; Koehler, G.M.; Krebs, C.J.; McKelvey, K.S.; Squires, J.R. Ecology and conservation of lynx in the United States. Gen. Tech. Rep. RMRS-GTR-30WWW. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 373–396.

Aubry, K.B.; McKelvey, K.S.; Copeland, J.P. 2007. Distribution and broadscale habitat relations of the wolverine in the contiguous United States. Journal of Wildlife Management. 71(7): 2147–2159.

Austin, K. 1993. Habitat use and home range size of breeding northern goshawks in the southern Cascades. Corvallis, OR: Oregon State University. 57 p. M.S. thesis.

Baglien, J.W. 1975. Biology and habitat requirements of the nesting golden eagle in southwestern Montana. Bozeman, MT: Montana State University. 53 p. M.S. thesis.

Baker, M.D.; Lacki, M.L. 2006. Day-roosting habitat of female long legged myotis in ponderosa pine forests. Journal of Wildlife Management. 70: 207–215.

Baldwin, R.A.; Bender, L.C. 2008. Distribution, occupancy, and habitat correlates of American martens (*Martes americana*) in Rocky Mountain National Park, Colorado. Journal of Mammalogy. 89: 419–427.

Banci, V. 1994. Wolverine. In: Ruggiero, L.R.; Aubry, K.B.; Buskirk, S.W.; Lyon L.J.; Zielinski, W.J., eds. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the Western United States. Gen. Tech. Rep. RM-GTR-254. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 99–127. Chapter 5.

Banks, R. 1970. The fox sparrow on the west slope of the Oregon Cascades. Condor. 72: 369–370.

Banner, A.; Schaller, S. 2001. Gulf of Maine watershed habitat analysis. Falmouth, ME: U.S. Fish and Wildlife Service, Gulf of Maine Coastal Program. 160 p.

Barnum, D.A. 1975. Aspects of western gray squirrel ecology. Pullman, WA: Washington State University. 55 p. M.S. thesis.

Bate, L.J.; Wisdom, M.J.; Wales, B.C. 2007. Snag densities in relation to human access and associated management factors in forests of Northeastern Oregon, USA. Landscape and Urban Planning. 80: 278–291.

Bates, J.W.; Moretti, M.O. 1994. Golden eagle (*Aquila chrysaetos*) population ecology in eastern Utah. Great Basin Naturalist. 54: 248–255.

Baydeck, R.K.; Campa, H., III; Haufler, J.B. 1999. Practical approaches to the conservation of biological diversity. Covelo, CA: Island Press. 327 p.

Beebe, J.; Everett, R.; Scherer, G.; Davis, C. 2002. Effect of fertilizer applications and grazing exclusion on species composition and biomass in wet meadow restoration in eastern Washington. Res. Pap. PNW-RP-542. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 15 p.

Beecham, J.J., Jr. 1970. Nesting ecology of the golden eagle in southwestern Idaho. Moscow, ID: University of Idaho. 48 p. M.S. thesis.

Beecham, J.J.; Kochert, M.N. 1975. Breeding biology of the golden eagle in southwestern Idaho. Wilson Bulletin. 87: 506–513.

Begley, J. 2007. Modeling Swakane bighorn sheep habitat and dispersal. Ellensburg, WA: Central Washington University. 117 p. M.S. thesis.

Beier, P.; Drennen, J.E. 1997. Forest structure and prey abundance in foraging areas of northern goshawks. Ecological Applications. 7(2): 564–571.

Bengston, S. 1972. Breeding ecology of the harlequin duck, *Histrionicus histrionicus*, in Iceland. Ornis Scandinavica. 3: 1–19.

Bengston, S.; Ulfstrand, S. 1971. Food resources and breeding frequency of the harlequin duck *Histrionicus histrionicus* in Iceland. Oikos. 22: 235–239.

Benoit, L.K.; Askins, R.A. 2002. Relationship between habitat area and the distribution of tidal marsh birds. Wilson Bulletin. 114: 314–323.

Bent, A.C. 1961. Life histories of North American birds of prey. Part 1. New York, NY: Dover Publications Inc. 409 p.

Berger, J.; Stacey, P.B.; Bellis, L.B.; Johnson, M.P. 2001. A mammalian predator-prey imbalance: grizzly bear and wolf extinction affect avian neotropical migrants. Ecological Applications. 11: 947–960.

Berkey, G.; Crawford, R.; Galipeau, S.; Johnson, D.; Lambeth, D.; Kreil, R. 1993. A review of wildlife management practices in North Dakota: effects on nongame bird populations and habitats. Report submitted to Region 6. Denver, CO: U.S. Fish and Wildlife Service. 51 p.

Bestelmeyer, B.T. 2006. Threshold concepts and their use in rangeland management and restoration: the good, the bad and the insidious. Restoration Ecology. 14(3): 325–329.

Bettinger, K.A. 2003. Cassin's finch. In: Marshall, D.B.; Hunter, M.G.; Contreras, A.L., eds. Birds of Oregon: a general reference. Corvallis, OR: Oregon State University Press: 599–601.

Biek, R.; Mills, L.S.; Bury, R.B. 2002. Terrestrial and stream amphibians across clear-cut interfaces in the Siskiyou Mountains, Oregon. Northwest Science. 76(2): 129–139.

Bildstein, K.L.; Gollop, J.B. 1988. Northern harrier. In: Palmer, R.S., ed. Handbook of North American birds: vol. 4. Diurnal raptors. New Haven, CN: Yale University Pres: 251–303.

Bittner, S.L.; Rongstad, O.J. 1982. Snowshoe hare and allies. In: Chapman, J.A.; Feldhamer, G.A. eds. Wild mammals of North America. Baltimore, MD: John Hopkins University Press: 146–163.

Black, H.L. 1974. A north temperate bat community: structure and prey populations. Journal of Mammalogy. 55: 138–157.

Blakesley, J.A.; Reese, K.P. 1988. Avian use of campground and non-campground sites in riparian zones. Journal of Wildlife Management. 52: 399–402.

Block, W.M.; Brennan, L.A. 1987. Characteristics of Lewis's woodpecker on the Modoc Plateau, California. Western Birds. 18: 209–212.

Blossey, B.; Skinner, L.C.; Taylor, J. 2001. Impact and management of purple loosestrife (*Lythrum salicaria*) in North America. Biodiversity and Conservation. 10: 1787–1807.

Bock, C.E. 1970. The ecology and behavior of the Lewis's woodpecker (*Asyndesmus lewis*). University of California Publications in Zoology. 92: 1–100.

Bock, C.E.; Bock, J.H. 1988. Grassland birds in southeastern Arizona: impacts of fire, grazing, and alien vegetation. In: Goriup, P.D., ed. Ecology and conservation of grassland birds. International Council for Bird Preservation Publication. 7: 43–58.

Bock, C.E.; Bock, J.H.; Bennett, B.C. 1995. The avifauna of remnant tallgrass prairie near Boulder, Colorado. Prairie Naturalist. 27: 147–157.

Bock, C.E.; Bock, J.H.; Bennett, B.C. 1999. Songbird abundance in grasslands at a suburban interface on the Colorado High Plains. In: Vickery, P.D.; Herkert, J.R., eds. Ecology and conservation of grassland birds of the Western Hemisphere. Studies in Avian Biology. 19: 131–136.

Bock, C.E.; Bock, J.H.; Kenney, W.R.; Hawthorne, V.M. 1984. Responses of birds, rodents, and vegetation to livestock exclosure in a semidesert grassland site. Journal of Range Management. 37: 239–242.

Bock, C.E.; Saab, V.A.; Rich, T.D.; Dobkin, D.S. 1993. Effects of livestock grazing on Neotropical migratory landbirds in western North America. In: Finch, D.M.; Stangel, P.W., eds. Status and management of Neotropical migratory birds. Gen. Tech. Rep. RM-GTR-229. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Forest and Range Experiment Station: 296–309.

Bock, C.E.; Webb, B. 1984. Birds as grazing indicator species in southeastern Arizona. Journal of Wildlife Management. 48: 1045–1049.

Boe, J.S. 1992. Wetland selection by eared grebes, *Podiceps nigricollis*, in Minnesota. Canadian Field-Naturalist. 106: 480–488.

Boe, J.S. 1993. Colony site selection by eared grebes in Minnesota. Colonial Waterbirds. 16: 28–38.

Boeker, E.L. 1974. Status of golden eagle surveys in the Western States. Wildlife Society Bulletin. 2: 46–49.

Boeker, E.L.; Ray, T.D. 1971. Golden eagle population studies in the Southwest. Condor. 73: 463–467.

Bolger, D.; Scott, T.; Rotenberry, J. 1997. Breeding bird abundance in an urbanizing landscape in coastal southern California. Conservation Biology. 11: 406–421.

Bolger, D.T. 2002. Habitat fragmentation effects on birds in southern California: contrast to the "top-down" paradigm. Studies in Avian Biology. 25: 141–157.

Bonar, R.L. 1999. Availability of pileated woodpecker cavities and use by other species. Journal of Wildlife Management. 64(1): 52–59.

Bonar, S.A.; Bolding, B.; Divens, M. 2002. Effects of triploid grass carp on aquatic plants, water quality, and public satisfaction in Washington state. North American Journal of Fisheries Management. 22: 96–105.

Bosch, J.; Rincón, P.A.; Boyero, L.; Martínez-Solano, I. 2006. Effects of introduced salmonids on a montane population of Iberian frogs. Conservation Biology. 20: 180–189.

Bowles, J.H. 1921. Notes on the California gray squirrel (*Sciurus griseus griseus*) in Pierce County, Washington. Murrelet. 2: 12–13.

Bradford, D.F.; Franson, S.E.; Neale, A.C.; Heggem, D.T.; Miller, G.R.; Canterbury, G.E. 1998. Bird species assemblages as indicators of biological integrity in Great Basin rangeland. Environmental Monitoring and Assessment. 49: 1–22.

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

Breininger, D.R.; Barkaszi, M.J.; Smith, R.B.; Oddy, D.M.; Provancha, J.A. 1998. Prioritizing wildlife taxa for biological diversity conservation at the local scale. Environmental Management. 22: 315–321.

Brendel, U.M.; Eberhardt, R.; Wiesmann, K. 2002. Conservation of the golden eagle (*Aquila chrysaetos*) in the European alps—a combination of education, cooperation, and modern techniques. Journal of Raptor Research. 36(supplement): 20–24.

Bright-Smith, D.J.; Mannan, R.W. 1994. Home range and habitat characteristics of male northern goshawks. Studies in Avian Biology. 16: 58–65.

Brinson, M.M.; Swift, B.L.; Plantico, R.C.; Barclay, J.S. 1981. Riparian ecosystems: their ecology and status. Rep. FWS/OBS-81/17. Kearneysville, WV: U.S. Department of the Interior, Fish and Wildlife Service, Biological Service Program. 155 p.

Broders, H.G. 2003. Another quantitative measure of bat species activity and sampling intensity considerations for the design of ultrasonic monitoring studies. Acta Chiroptera. 5: 235–241.

Broders, H.G.; Forbes, G.J. 2004. Interspecific and intersexual variation in roost-site-selection of northern long-eared and little brown bats in the Greater Fundy National Park Ecosystem. Journal of Wildlife Management. 68: 602–610.

Brooks, M.L. 1999. Alien annual grasses and fire in the Mojave Desert. Madrono. 46: 13–19.

Broquet, T.; Ray, N.; Petit, E.; Fryxell, J.M.; Burel, F. 2006. Genetic isolation by distance and landscape connectivity in the American marten (*Martes americana*). Landscape Ecology. 21: 877–889.

Brown, H.A. 1990. Morphological variation and age-class determination in overwintering tadpoles of the tailed frog. Journal of Zoology. 220: 171–184.

Brown, M.; Dinsmore, J.J. 1986. Implications of marsh size and isolation for marsh bird management. Journal of Wildlife Management. 50: 392–397.

Bruce, A.M.; Anderson, R.J.; Allen, G.T. 1982. Observations of golden eagles nesting in western Washington. Raptor Research. 16: 132–134.

Buehler, D.A.; Mersmann, T.J.; Fraser, J.D.; Seegar, J.K.D. 1991. Non-breeding bald eagle communal and solitary roosting behavior and roost habitat on the northern Chesapeake Bay. Journal of Wildlife Management. 55: 273–281.

Bull, E.L. 1987. Ecology of pileated woodpecker in northeastern Oregon. Journal of Wildlife Management. 51(2): 472–481.

Bull, E.L. 2003a. Diet and prey availability of Columbia spotted frogs in northeastern Oregon. Northwest Science. 77: 349–356.

Bull, E.L. 2003b. Pileated woodpecker. In: Marshall, D.B.; Hunter, M.G.; Contreras, A.L., eds. Birds of Oregon: a general reference. Corvallis, OR: Oregon State University Press: 372–374.

Bull, E.L. 2005. Ecology of the Columbia spotted frog in northeastern Oregon. Gen. Tech. Rep. PNW-GTR-640. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 46 p.

Bull, E.L.; Carter, B.E. 1996. Tailed frogs: distribution, ecology, and association with timber harvest in northeast Oregon. Res. Pap. PNW-RP-497. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 46 p.

Bull, E.L.; Clark, A.A.; Shepherd, J.F. 2005. Short-term effects of fuel reduction on pileated woodpeckers in northeastern Oregon—a pilot study. Res. Pap. PNW-RP-564. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 17 p.

Bull, E.L.; Deal, J.W.; Hohmann, J.E. 2001. Avian and amphibian use of fenced and unfenced stock ponds in northeastern Oregon forests. Res. Pap. PNW-RP-539. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 9 p.

Bull, E.L.; Hayes, M.P. 2000. Livestock effects on reproduction of the Columbia spotted frog. Journal of Range Management. 53: 291–294.

Bull, E.L.; Hayes, M.P. 2001. Post-breeding movements of Columbia spotted frogs (*Rana luteiventris*) in northeastern Oregon. Western North American Naturalist. 61: 119–123.

Bull, E.L.; Hayes, M.P. 2002. Overwintering of Columbia spotted frogs in northeastern Oregon. Northwest Science. 76: 141–147.

Bull, E.L.; Heater, T.W. 2000. Resting and denning sites of American martens in northeastern Oregon. Northwest Science. 74: 179–185.

Bull, E.L.; Heater, T.W.; Shepard, J.F. 2005. Habitat selection by the American marten in northeastern Oregon. Northwest Science. 79: 37–43.

Bull, E.L.; Holthausen, R.S. 1993. Habitat use and management of pileated woodpeckers in northeastern Oregon. Journal of Wildlife Management. 57: 335–345.

Bull, E.L.; Holthausen, R.S.; Henjum, M.G. 1992. Roost trees used by pileated woodpeckers in northeastern Oregon. Journal of Wildlife Management. 56: 786–793.

Bull, E.L.; Jackson, J.E. 1995. Pileated woodpecer (*Dryocopus pileatus*). In: Poole, A.; Gill, F., eds. The birds of North America, No. 148. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union: 1–24.

Bull, E.L.; Marx, D.B. 2002. Influence of fish and habitat on amphibian communities in high-elevation lakes in northeastern Oregon. Northwest Science. 76: 240–248.

Bull, E.L.; Nielsen-Pincus, N.; Wales, B.C.; Hayes, J.L. 2007. The influence of disturbance events on pileated woodpeckers in northeastern Oregon. Forest Ecology and Management. 243: 319–329.

Bull, E.L.; Peterson, S.R.; Thomas, J.W. 1986. Resource partitioning among woodpeckers in northeastern Oregon. Res. Note PNW-444. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 19 p.

Bull, E.L.; Snider, M. 1993. Master carpenter. Wildbird. 7(2): 41–43.

Bunnell, K.D.; Flinders, J.T.; Wolfe, M.L. 2006. Potential impacts of coyotes and snowmobiles on lynx conservation in the intermountain West. Wildlife Society Bulletin. 34(3): 828–838.

Busch, D.E.; Trexler, J.C. 2003. Monitoring ecosystems: interdisciplinary approaches for evaluating ecoregional initiatives. Washington, DC: Island Press. 7(2): 1. http://www.consecol.org/vol7/iss2/art1/. (May 5, 2014).

Bury, R.B. 1983. Differences in amphibian populations in logged and old-growth redwood forest. Northwest Science. 57: 167–178.

Bury, R.B. 1994. Vertebrates in the Pacific Northwest: species richness, endemism and dependency on old-growth forests. In: Majumdar, S.K.; Brenner, F.J.; Lovich, J.E.; Schalles, J.F.; Miller, E.W., eds. Biological diversity: problems and challenges. Philidelphia, PA: Pennsylvania Academy of Sciences: 392–404.

Bury, R.B.; Corn, P.S. 1988. Responses of aquatic and streamside amphibians to timber harvest: a review. In: Raedeke, K.J., ed. Streamside management: riparian wildlife and forestry interactions. Seattle, WA: University of Washington, Institute of Forest Resources, Contribution. 59: 165–181.

Buskirk, S.W.; Forrest, S.C.; Raphael, M.G.; Harlow, H.J. 1989. Winter resting site ecology of marten in the central Rocky Mountains. Journal of Wildlife Management. 53: 191–196.

Buskirk, S.W.; Ruggiero, L.F.; Krebs, C.J. 1999. Habitat fragmentation and interspecific competition: implications for lynx conservation. In: Ruggerio, L.F.; Aubry, K.B.; Buskirk, S.W.; Koehler, G.M.; Krebs, C.J.; McKelvey, K.S.; Squires, J.R., eds. Ecology and conservation of lynx in the United States. Gen. Tech. Rep. RMRS-GTR-30WWW. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 83–100.

Cahall, R.E.; Hayes, J.P. 2009. Influences of postfire salvage logging on forest birds in the eastern Cascades, Oregon, USA. Forest Ecology and Management. 257: 1119–1128.

Camenzind, F.J. 1969. Nesting ecology and behavior of the golden eagle *Aquila chrysaetos* L. Brigham Young University Science Bulletin. 10: 4–15.

Campbell, N.K.; McTaggart-Cowan, I.; Cooper, J.M.; Kaiser, G.W.; McNall, M.C.E. 1990. The birds of British Columbia. Vancouver, BC: University of British Columbia Press.

Campbell, T.M., III. 1979. Short-term effects of timber harvests on pine marten ecology. Fort Collins, CO: Colorado State University. 71 p. M.S. thesis.

Cannings, R.J. 2000. Update COSEWIC status report on sage thrasher (*Oreoscoptes montanus*). Ottawa, ON: Committee on the Status of Endangered Wildlife in Canada. 21 p.

Carrete, M.; Sanchez-Zapata, J.A.; Calvo, J.F. 2000. Breeding densities and habitat attributes of golden eagles in southeastern Spain. Journal of Raptor Research. 34: 48–52.

Carroll, C.; Noss, R.F.; Paquet, P.C. 2001. Carnivores as focal species for conservation planning in the Rocky Mountain region. Ecological Applications. 11(4): 961–980.

Carter, M.F.; Hunter, W.C.; Pashley, D.N.; Rosenberg, K.V. 2000. Setting conservation priorities for landbirds in the United States: the Partners in Flight approach. Auk. 117(2): 541–548.

Carter, T.C.; Ford, W.M.; Menzel, M.A. 2002. Fire and bats in the southeast and mid-Atlantic: more questions than answers? In: Ford, W.M.; Russell, K.R.; Moorman, C.E., eds. Proceedings: the role of fire for nongame wildlife management and community restoration: traditional uses and new directions. Gen. Tech. Rep. NE-288. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station: 139–143.

Cassier, E.F.; Groves, C.R. 1989. Breeding ecology of harlequin ducks of the Kaniksu National Forest. Boise, ID: Idaho Department of Fish and Game.

Cassier, E.F.; Schirato, G.; Sharpe, F.; Groves, C.R.; Anderson, R.N. 1993. Cavity nesting by harlequin ducks in the Pacific Northwest. Wilson Bulletin. 105: 691–694.

Castrale, J.S. 1982. Effects of two sagebrush control methods on non game birds. Journal of Wildlife Management. 46: 945–952.

Chambers, C.L.; Holthausen, R.S. 2000. Montane ecosystems used as rangelands. In: Jemison, R.; Raish, C., eds. Livestock management in the American Southwest: ecology, society, and economics. Amsterdam: Elsevier Science: 213–280.

Chapin, T.G.; Harrison, D.J.; Katnik, D.D. 1998. Influence of landscape pattern on habitat use by American marten in an industrial forest. Conservation Biology. 12: 1327–1337.

Chinnici, S.J.; Bradley, L.C.; Dill, D.R.; Bigger, D. 2007. Using site-specific habitat information on young to late-successional avifauna to guide use and management of coastal redwood and Douglas-fir forest lands. In: Standiford, R.B.; Giusti, G.A.; Valachovic, Y.; Zielinski, W.J.; Furniss, M.J., tech. eds. Proceedings of the redwood region forest science symposium: What does the future hold? Gen. Tech. Rep. PSW-GTR-194. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 147–156.

Christensen, N.L. 1997. Implementing ecosystem management: Where do we go from here? In: Boyce, M.S.; Haney, A. eds. Ecosystem management: applications for sustainable forest and wildlife resources. New Haven, CT: Yale University Press: 325–342.

Christensen, N.L.; Bartuska, A.M.; Brown, J.H.; Carpenter, S.; D'Antonio, C.; Francis, R.; Woodmansee, R.G. 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecological Applications. 6(3): 665–691.

Churchill, D.J.; Larson, A.J.; Dahlgreen, M.C.; Franklin, J.F.; Hessburg, P.F.; Lutz, J.A. 2013. Restoring forest resilience: from reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management. 291: 442–457.

Clark, R.J. 1972. Observations of nesting marsh hawks in Manitoba. Blue Jay. 30: 43–48.

Clarkson, P. 1992. A preliminary investigation into the status and distribution of harlequin ducks in Jasper National Park, Alberta, Canada. Jasper National Park, Natural Resource Conservation, Alberta.

Clough, G.C.; Albright, J.J. 1987. Occurrence of the northern bog lemming, *Synapromys borealis*, in the northeastern United States. Canadian Field-Naturalist. 101: 611–613.

Collins, J.P.; Jones, T.R.; Berna, H.J. 1988. Conserving genetically distinctive populations: the case of the Huachuca tiger salamander. In: Management of amphibians, reptiles, and small mammals in North America. Gen. Tech Rep. GTR-RM-166. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 45–53.

Collopy, M.W.; Edwards, T.C., Jr. 1989. Territory size, activity budget, and role of undulating flight in nesting golden eagles. Journal of Field Ornithology. 60: 43–51.

Committee of Scientists [COS]. 1999. Saving the people's land: Stewardship into the next century. Washington, DC: U.S. Department of Agriculture, Forest Service, Government Printing Office.

Committee on the Status of Endangered Wildlife in Canada [COSEWIC]. 2001. COSEWIC assessment and status report on the tiger salamander *Ambystoma tigrinum* in Canada. Prepared by D.M. Schock for COSEWIC, Hull, QC. [Place of publication unknown]: [Publisher unknown].

Copeland, J.P. 1996. Biology of the wolverine in central Idaho. Moscow, ID: University of Idaho. 138 p. M.S. thesis.

Copeland, J.P.; McKelvey, K.S.; Aubry, K.B.; Landa, A.; Persson, J.; Inman, R.M.; Krebs, J.; Lofroth, E.; Golden, H.; Squires, J.R.; Magoun, A.; Schwartz, M.K.; Wilmot, J.; Copeland, C.L.; Yates, R.E.; Kojola, I.; May, R. 2010. The bioclimatic envelope of the wolverine (*Gulo gulo*): do climatic constraints limit its geographic distribution? Canadian Journal of Zoology. 88: 233–246.

Copeland, J.P.; Peek, J.M.; Groves, C.R.; Melquist, W.E.; McKelvey, K.S.; McDaniel, G.W.; Long, C.D.; Harris, C.E. 2007. Seasonal habitat associations of the wolverine in central Idaho. Journal of Wildlife Management. 71(7): 2201–2212.

Coppolillo, P.; Gomez, H.; Maisels, F.; Wallace, R. 2008. Selection criteria for suites of landscape species as a basis for site-based conservation. Biological Conservation. 115: 419–430.

Corn, P.S.; Bury, R.B. 1989. Logging in western Oregon: responses of headwater habitats and stream amphibians. Forest Ecology and Management. 29: 39–57.

Corn, P.S.; Jennings, M.L.; Muths, E. 1997. Survey and assessment of amphibian populations in Rocky Mountain National Park. Northwestern Naturalist. 78: 34–55.

Cornish, T.E.; Linders, M.J.; Little, S.E.; Vander Haegen, W.M. 2001. Notoedric mange in western gray squirrels from Washington. Journal of Wildlife Diseases. 37: 630–633.

Coulter, M. 2008. A multi-spatial-scale characterization of lark sparrow habitat and the management implications. Bowling Green, OH: Bowling Green State University. 63 p. M.S. thesis.

Courtney, S.P.; Blakesley, J.A.; Bigley, R.E.; Cody, M.L.; Dumbacher, J.P.; Fleischer, R.C.; Franklin, A.B.; Franklin, J.F.; Gutierrez, R.J.; Marzluff, J.M.; Sztukowski, L. 2004. Scientific evaluation of the status of the Northern Spotted Owl. Portland, OR: Sustainable Ecosystems Institute. 357 p.

Cowardin, L.M.; Carter, V.; Golet, F.C.; LaRoe, E.T. 1979. Classification of wetlands and deepwater habitats of the United States. Washington, DC: U.S. Fish and Wildlife Service, Biological Services Program.

Craig, E.H.; Craig, T.H.; Powers, L.R. 1986. Habitat use by wintering golden eagles and rough-legged hawks in southeastern Idaho. Raptor Research. 20: 69–71.

Crampton, L.H.; Barclay, R.M.R. 1996. Habitat selection by bats in fragmented and unfragmented aspen mixedwood stands of different ages. In: Barclay, R.M.R.; Brigham, M.R., eds. Bats and forest symposium. Victoria, BC: Research Branch, British Columbia Ministry of Forests: 238–259.

Crisafulli, C.M. 1999. Survey protocol for larch mountain salamander (*Plethodon larselli*). In: Olson, D.H., ed. Survey protocols for amphibians under the Survey and Manage provision of the Northwest Forest Plan. Version 3.0. Portland, OR: U.S. Department of Agriculture, Forest Service: 253–310.

Crivelli, A.J. 1983. The destruction of aquatic vegetation by carp: a comparison between southern France and the United States. Hydrobiologia. 106: 37–41.

Cuellar, O. 1994. Ecological observation on *Rana pretiosa* in western Utah. Alytes. 12: 109–121.

Cullen, S.A.; Jehl, J.R., Jr.; Nuechterlein, G.L. 1999. Eared grebe (*Podiceps nigricollis*). In: Poole, A.; Gill, F., eds. The Birds of North America, No. 433. Philadelphia, PA: The Birds of North America, Inc.

Cunningham, J.B.; Balda, R.P.; Gaud, W.S. 1980. Selection and use of snags by secondary cavity-nesting birds of the ponderosa pine forest. Res. Pap. RM-222. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 15 p.

Cushman, S.A. 2006. Effects of habitat loss and fragmentation on amphibians: a review and prospectus. Biological Conservation. 128: 231–240.

Cushman, S.A.; McKelvey, K.S.; Flather, C.H.; McGarigal, K. 2007. Do forest community types provide a sufficient basis to evaluate biological diversity? Frontiers in Ecology and Environment. 6(1): 13–17.

Czech, B.; Krausman, P.R. 1997. Distribution and causation of species endangerment in the United States. Science. 277: 1116–1117.

Dahl, T.E. 1990. Wetlands losses in the United States 1780's to 1980's. Washington, DC: U.S. Fish and Wildlife Service. http://www.npwrc.usgs.gov/resource/wetlands/wetloss/index.htm. (May 15, 2010).

Daugherty, C.H.; Sheldon, A.L. 1982. Age-determination, growth, and life history of a Montana population of the tailed frog. Herpetologica. 38(4): 461–468.

Davis, R.; Lint, J. 2005. Habitat status and trend. In: Lint, J. ed. Status and trends of northern spotted owl populations and habitat. Gen. Tech. Rep. PNW-GTR-648. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 21–82.

Davis, R.J.; Dugger, K.M.; Mohoric, S.; Evers, L.; Aney, W.C. 2011. Northwest Forest Plan first 15 years (1984–2008):–status and trends of northern spotted owl populations and habitats. Gen. Tech. Rep. PNW-GTR-850. Portland, OR: U.S Department of Agriculture, Forest Service, Pacific Northwest Research Station. 147 p.

Daw, S.K.; DeStefano, S. 2001. Forest characteristics of northern goshawk nest stands and post-fledgling areas in Oregon. Journal of Wildlife Management. 65: 59–65.

de Jong, J. 1994. Distribution patterns and habitat use by bats in relation to landscape heterogeneity, and consequences for conservation. Uppsala, Sweden: University of Agricultural Sciences. 132 p. Ph.D. thesis.

DeMaynadier, P.G.; Hunter, M.L., Jr. 2000. Road effects on amphibian movements in a forested landscape. Natural Areas Journal. 20: 56–65.

Denoel, M.; Ficetola, G.F. 2007. Landscape-level thresholds and newt conservation. Ecological Applications. 17(1): 302–309.

Desimone, S.M.; DeStefano, S. 2005. Temporal patterns of northern goshawk nest area occupancy and habitat: a retrospective analysis. Journal of Raptor Research. 39(3): 310–323.

Desimone, S.M.; Hays, D.W. 2003. Northern goshawk. In: Management recommendations for Washington's priority habitats and species. Volume IV: Birds. Olympia, WA: Washington Department of Fish and Wildlife: 6–1 to 6–16.

Dixon, R.D. 1995a. Density, nest-site and roost-site characteristics, home-range, habitat-use, and behavior of white-headed woodpeckers: Deschutes and Winema National Forests, Oregon. Nongame Rep. 93-3-01. Portland, OR: Oregon Department of Fish and Wildlife.

Dixon, R.D. 1995b. Ecology of white-headed woodpeckers in the central Oregon Cascades. Moscow, ID: University of Idaho, Moscow. 148 p. M.S. thesis.

Dixon, R.D.; Saab, V.A. 2000. Black-backed woodpecker (*Picoides arcticus)*. In: Poole, A.; Gill, F., eds. Birds of North America, No. 509. Philadelphia, PA: The Birds of North America, Inc. 20 p.

Dobler, F.C.; Eby, J.; Perry, C.; Richardson, S.; Vander Haegen, M. 1996. Status of Washington's shrub-steppe ecosystem: extent, ownership, and wildlife/vegetation relationships. Olympia, WA: Washington Department of Fish and Wildlife.

Doubledee, R.A.; Muller, E.B.; Nisbet, R.M. 2003. Bullfrogs, disturbance regimes, and the persistence of California red-legged frogs. Journal of Wildlife Management. 67: 424–438.

Drennen, J.E.; Beier, P. 2003. Forest structure and prey abundance in winter habitat of northern goshawks. Journal of Wildlife Management. 67: 177–185.

Drever, M.C.; Aitken, K.E.H.; Norris, A.R.; Martin, K. 2008. Woodpeckers as reliable indicators of bird richness, forest health and harvest. Biological Conservation. 141: 624–634.

Duebbert, H.F.; Lokemoen, J.T. 1977. Upland nesting of American bitterns, marsh hawks, and short-eared owls. Prairie Naturalist. 9: 33–40.

Dumyahn, J.B.; Zollner, P.A.; Gilbert, J.H. 2007. Winter home-range characteristics of American marten in northern Wisconsin. American Midland Naturalist. 158: 382–394.

Dunham, J.B.; Pilliod, D.S.; Young, M.K. 2004. Assessing the consequences of nonnative trout in headwater ecosystems in western North America. Fisheries. 29(6): 18–26.

Dupuis, L.A. 1997. Effects of logging on terrestrial amphibians of coastal British Columbia. Herpetological Conservation. 1: 185–190.

Dupuis, L.; Bunnell, F.L.; Friele, P.A. 2000. Determinants of the tailed frog's range in British Columbia. Northwest Science. 74: 109–115.

Dupuis, L.; Steventon, D. 1999. Riparian management and the tailed frog in northern coastal forests. Forest Ecology and Management. 124: 35–43.

Dvornich, K.M.; McAllister, K.R.; Aubry, K.B. 1997. Amphibians and reptiles of Washington state: location data and predicted distributions. Volume 2. In: Washington State Gap Analysis–final report. Seattle, WA: Washington Cooperative Fish and Wildlife Research Unit, University of Washington. 146 p.

Dwyer, J.K.; Block, W.M. 2000. Effects of wildfire on densities of secondary cavity-nesting birds in ponderosa pine forests of northern Arizona. In: Moser, W.K.; Moser, C.E., eds. Fire and forest ecology: innovative silviculture and vegetation management. Tall Timbers Fire ecology proceedings, No. 21. Tallahassee, FL: Tall Timbers Research Station: 151–156.

Eaton, R.L. 1976. Golden eagle (*Aquila chrysaetos*). In: Brittell, J.D.; Brown, J.M.; Eaton, R.L., eds. Marine shoreline fauna of Washington. Volume 2. Washington State Department of Game Coastal Zone Environmental Studies Report. 3: 82–118.

Edelmann, F.; Copeland, J. 1999. Wolverine distribution in the northwestern United States and a survey in the Seven Devils Mountains of Idaho. Northwest Science. 73: 295–300.

Ellison, L.E.; Everette, A.L.; Bogan, M.A. 2005. Examining patterns of bat activity in Bandelier National Monument, New Mexico, by using walking point transects. Southwest Naturalist. 50: 197–208.

Erdman, T.C.; Brinker, D.F.; Jacobs, J.P. [and others]. 1998. Productivity, population trend, and status of northern goshawks, *Accipiter gentilis*, in northeastern Wisconsin. The Canadian Field Naturalist. 112(1): 17–27.

Evans, D.L. 1982. Status reports on twelve raptors. Special Scientific Report-Wildlife No. 238. Washington, DC: U.S. Fish and Wildlife Service. 70 p.

Everett, R.; Hessburg, P.; Jensen, M.; Borman, B. 1994. Eastside forest health assessment. Volume 1: Executive summary. Gen. Tech. Rep. PNW-GTR-317. [Portland, OR]: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 57 p.

Everett, R.; Lehmkuhl, J.; Schellhaas, R.; Ohlson, P.; Keenum, D.; Riesterer, H.; Spurbeck, D. 1999. Snag dynamics in a chronosequence of 26 wildfires on the east slope of the Cascade Range in Washington State, USA. International Journal of Wildland Fire. 9(4): 223–234.

Everett, R.L.; Schellhaas, R.; Keenum, D.; Spurbeck, D.; Ohlson, P. 2000. Fire history in the ponderosa/Douglas-fir forests on the east slope of the Washington Cascades. Forest Ecology and Management. 129: 207–225.

Faaborg, J. 1976. Habitat selection and territorial behavior of the small grebes of North Dakota. Wilson Bulletin. 88: 390–399.

Faanes, C.A. 1983. Breeding birds of wooded draws in western North Dakota. Prairie Naturalist. 15: 173–187.

Fahrig, L.; Pedlar, J.H.; Pope, S.E.; Taylor, P.D.; Wegner, J.F. 1995. Effect of road traffic on amphibian density. Biological Conservation. 73: 177–182.

Fecske, D.M.; Jenks, J.A.; Smith, V.J. 2002. Field evaluation of a habitat-relation model for the American marten. Wildlife Society Bulletin. 30: 775–782.

Feller, G.M.; Pierson, E.D. 2002. Habitat use and foraging behavior of Townsend's big-eared bat *(Corynorhinus townsendii)* in coastal California. Journal of Mammalogy. 83: 167–177.

Feminella, J.W.; Hawkins, C.P. 1994. Tailed frog tadpoles differentially alter their feeding behavior in response to non-visual cues from four predators. Journal of the North American Benthological Society. 13: 310–320.

Fernandez, C. 1993. The choice of nesting cliffs by golden eagles, *Aquila chrysaetos*: the influence of accessibility and disturbance by humans. Alauda. 61: 105–110.

Ficetola, G.F.; Sacchi, R.; Scali, S.; Gentilli, A.; DeBarnardi, F.; Galeotti, P. 2007. Vertebrates respond differently to human disturbance: implications for the use of a focal species approach. Acta Oecologica. 31: 109–118.

Findlay, C.S.; Bourdages, J. 2000. Response time of wetland biodiversity to road construction on adjacent lands. Conservation Biology. 14: 86–94.

Findlay, C.S.; Houlahan, J. 1997. Anthropogenic correlates of species richness in southeastern Ontario wetlands. Conservation Biology. 11: 1000–1009.

Finn, S.P.; Marzluff, J.; Varland, D.E. 2002a. Does northern goshawk breeding occupancy vary with nest-stand characteristics on the Olympic Peninsula, Washington? Journal of Raptor Research. 36: 265–279.

Finn, S.P.; Marzluff, J.; Varland, D.E. 2002b. Effects of landscape and local habitat attributes on northern goshawk site occupancy in western Washington. Forest Science. 48: 1–10.

Fischer, D.L.; Ellis, K.L.; Meese, R.J. 1984. Winter habitat selection of diurnal raptors in central Utah. Raptor Research. 18: 98–102.

Fisher, R.N.; Shaffer, H.B. 1996. The decline of amphibians in California's Great Central Valley. Conservation Biology. 10(5): 1387–1397.

Fitzgerald, J.P. 1978. Vertebrate associations in plant communities along the South Platte River in northeastern Colorado. In: Graul, W.D.; Bissell, S.J. eds. Lowland river and stream habitat in Colorado: a symposium. Greeley, CO: Colorado Chapter Wildlife Society and Colorado Audubon Council: 73–88.

Flanders, A.A.; Kuvlesky, W.P.; Ruthven, D.C.; Zaiglin, R.E.; Bingham, R.L.; Fulbright, T.E.; Hernandez, F.; Brennan, L.A. 2006. Effects of invasive exotic grasses on south Texas rangeland breeding birds. Auk. 123: 171–182.

Fletcher, A.R.; Morison, A.K.; Hume, D.J. 1985. Effects of carp, *Cyprinus carpio* L., on communities of aquatic vegetation and turbidity of waterbodies in the lower Goulburn River basin. Australian Journal of Marine and Freshwater Research. 36: 311–327.

Fletcher, R.J.; McKinney, S.T.; Bock, C.E. 1999. Effects of recreational trails on wintering diurnal raptors along riparian corridors in a Colorado grassland. Journal of Raptor Research. 33(3): 233–239.

Fontaine, J.B.; Donato, D.C.; Robinson, W.D.; Law, B.E.; Kauffman, J.B. 2009. Bird communities following high-severity fire: response to single and repeat fires in a mixed-evergreen forest, Oregon, USA. Forest Ecology and Management. 257: 1496–1504.

Foreyt, W.J. 1989. Fatal *Pasturella haemolytica pneumonia* in bighorn sheep after direct contact with clinically normal domestic sheep. American Journal of Veterinary Research. 50: 341–344.

Foreyt, W.J.; Jessup, D. 1982. Fatal pneumonia of bighorn sheep following association with domestic sheep. Journal of Wildlife Diseases. 18: 163–167.

Foreyt, W.J.; Snipes, K.P.; Kasten, R.W. 1994. Fatal pneumonia following inoculation of healthy bighorn sheep with *Pasturella haemolytica* from healthy domestic sheep. Journal of Wildlife Diseases. 30: 137–145.

Forristal, C.D. 2009. Influence of postfire salvage logging on black-backed woodpecker nest-site selection and nest survival. Bozeman, MT: Montana State University. 93 p. M.S. thesis.

Forsman, E.D.; Anthony, R.G.; Dugger, K.M. [and others]. 2011. Population demography of northern spotted owls. Studies in Avian Biology. 40. 106 p.

Franklin, J.F.; Hemstrom, M.A.; VanPelt, R.; Buchanan, J.B. 2008. The case for active management of dry forests types in eastern Washington: perpetuating and creating old forest structures and functions. Olympia, WA: Washington State Department of Natural Resources.

Franklin, J.F.; Johnson, K.N. 2012. A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry. 110(8): 429–439.

Franzreb, K.E. 1977. Bird population changes after timber harvesting of a mixed conifer forest in Arizona. Res. Pap. RM-184. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 26 p.

Fraser, J.D.; Frenzel, L.D.; Mathisen, J.E. 1985. The impact of human activities on breeding bald eagles in north-central Minnesota. Journal of Wildlife Management. 49: 585–592.

Freudenberger, D.; Brooker, L. 2004. Development of the focal species approach for biodiversity conservation in the temperate agricultural zones of Australia. Biodiversity and Conservation. 13: 253–274.

Friedmann, H. 1963. Host relations of the parasitic cowbirds. U.S. National Museum Bulletin. 233.

Fule', P.Z.; Covington, W.W.; Moore, M.M. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications. 7(3): 895–908.

Funk, W.C.; Greene, A.E.; Corn, P.S.; Allendorf, F.W. 2005. High dispersal in a frog species suggests that it is vulnerable to habitat fragmentation. Biology Letters (Royal Society of London). 1: 13–16.

Fyfe, R.W.; Olendorff, R.R. 1976. Minimizing the dangers of nesting studies to raptors and other sensitive species. Canadian Wildlife Service. Occas. Pap. 23. 17 p.

Gabrielson, I.N.; Jewett, S.G. 1940. Birds of Oregon. Corvallis, OR: Oregon State College. 650 p.

Gaines, W.L. 2000. Disturbance ecology, land allocations, and wildlife management. In: Proceedings of the management of fire maintained ecosystems workshop. Whistler, BC: British Columbia Forest Service: 2–9–34.

Gaines, W.L.; Haggard, M.; Begley, J.; Lehmkuhl, J.; Lyons, A.L. 2010a. Short-term effects of thinning and burning restoration treatments on avian community composition, density, and nest survival in the eastern Cascades dry forests, Washington. Forest Science. 56(1): 88–99.

Gaines, W.L.; Haggard, M.; Lehmkuhl, J.F.; Lyons, A.L.; Harrod, R.J. 2007. Short-term response of land birds to ponderosa pine restoration. Restoration Ecology. 15(4): 666–674.

Gaines, W.L.; Harrod, R.J.; Lehmkuhl, J.F. 2003. Monitoring biodiversity for ecoregional initiatives. In: Busch, D.E.; Trexler, J.C., eds. Monitoring ecosystems: interdisciplinary approaches for evaluating ecoregional initiatives. Washington, DC: Island Press: 377–404.

Gaines, W.L.; Peterson, D.W.; Thomas, C.A.; Harrod, R.J. 2012. Adaptations to climate change: Colville and Okanogan-Wenatchee National Forests. Gen. Tech. Rep. PNW-GTR-862. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 34 p.

Gaines, W.L.; Singleton, P.H.; Ross, R.C. 2003. Assessing the cumulative effects of linear recreation routes on wildlife habitats on the Okanogan and Wenatchee National Forests. Gen. Tech. Rep. PNW-GTR-586. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 79 p.

Galen, C. 1989. A preliminary assessment of the status of Lewis' woodpecker in Waso County, Oregon. Tech. Rep. 88-3-01. Portland, OR: Oregon Department of Fish and Wildlife.

Gallo, K.; Lanigan, S.H.; Eldred, P.; Gordon, S.N.; Moyer, C. 2005. Northwest Forest Plan—the first ten years (1994–2003): preliminary assessment of the condition of watersheds. Gen. Tech. Rep. PNW-GTR-647. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 133 p.

Garrett, K.L.; Raphael, M.G.; Dixon, R.D. 1996. White-headed woodpecker (*Picoides albolarvatus*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 252. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union.

Garrett, M.G.; Watson, J.W.; Anthony, R.G. 1993. Bald eagle home range and habitat use in the Columbia River Estuary. Journal of Wildlife Management. 57: 19–27.

Gartner, S.; Reynolds, K.M.; Hessburg, P.F.; Hummel, S.; Twery, M. 2008. Decision support for evaluating forest departure and prioritizing forest management activities in a changing environment. Forest Ecology and Management. 256: 1666–1676.

Gashwiler, J.S. 1977. Bird populations in four vegetational types in central Oregon. Spec. Sci. Rep.–Wildl. 205. Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service. 20 p.

Gentry, D.J.; Vierling, K.T. 2007. Old burns as source habitats for Lewis's woodpeckers breeding in the black hills of South Dakota. The Condor. 109(1): 122–131.

Germaine, H.L.; Germaine, S.S. 2006. Forest restoration treatment effects on the nesting success of western bluebirds. (*Sialia mexicana*). Restoration Ecology. 10: 326–367.

Gerrard, J.M.; Bortolotti, G.R. 1988. The bald eagle, haunts and habits of a wilderness monarch. Washington, DC: Smithsonian Institution Press. 177 p.

Gibbs, J.P. 1998. Amphibian movements in response to forest edges, roads, and streambeds in southern New England. Journal of Wildlife Management. 62: 584–589.

Gibbs, J.P.; Longcore, J.R.; McAuley, D.G.; Ringelman, J.K. 1991. Use of wetland habitats by selected nongame waterbirds in Maine. Fed. Gov. Ser. Fish and Wildlife Research 9. U.S. Department of the Interior, Fish and Wildlife Service. http://pubs.er.usgs.gov/. (June 5, 2014).

Gibbs, J.P.; Melvin, S.M. 1990. An assessment of wading birds and other wetlands avifauna and their habitats in Maine. Report to Endangered and Nongame Wildlife Grants Program. Bangor, ME: Maine Department of Inland Fisheries and Wildlife.

Giblisco, C.J. 1994. Distributional dynamics of modern Martes in North America. In: Buskirk, S.W.; Harestad, A.S.; Raphael, M.G.; Powell, R.A. eds. Martens, sables, and fishers: biology and conservation. Ithaca, NY: Cornell University Press: 59–70.

Gilbert, J.H.; Wright, J.L.; Lauten, D.J.; Probst, J.R. 1997. Den and rest-site characteristics of American marten and fisher in northern Wisconsin. In: Proulx, G.; Bryant, H.N.; Woodard, P.M., eds. Martes: taxonomy, ecology, techniques, and management. Edmonton, AB: Provincial Museum of Alberta: 135–145.

Gilligan, J.; Smith, M.; Rogers, D.; Contreras, A. 1994. Birds of Oregon: status and distribution. McMinnville, OR: Cinclus Publishing.

Gillis, E. 1989. Western bluebirds, tree swallows & violet-green swallows west of the Cascade Mountains in Oregon, Washington and Vancouver Island, BC. Sialia. 11(4): 127–130.

Godbout, G.; Ouellet, J. 2008. Habitat selection of American marten in a logged landscape at the southern fringe of the boreal forest. Ecoscience. 15: 332–342.

Goggans, R. 1989. Black-backed woodpecker (*Picoides arcticus*). In: Clark, T.W.; Harvey, A.H.; Dorn, R.D.; Genter, D.L.; Groves, C., eds. Rare, sensitive, and threatened species of the greater Yellowstone ecosystem. Jackson, WY: Northern Rockies Conservation Cooperative: 88–89.

Goggans, R.; Dixon, R.D.; Seminara, L.C. 1988. Habitat use by three-toed and black-backed woodpeckers, Deschutes National Forest, Oregon. Nongame Wildlife Program Report 87-3-02. Salem, OR: Oregon Department of Fish and Wildlife. 43 p.

Goguen, C.B.; Mathews, N.E. 1998. Songbird community composition and nesting success in grazed and ungrazed pinyon-juniper woodlands. Journal of Wildlife Management. 62: 474–484.

Goguen, C.B.; Mathews, N.E. 1999. Review of the causes and implications of the association between cowbirds and livestock. In: Morrison, M.L.; Hall, L.S.; Robinson, S.K.; Rothstein, S.I.; Hahn, D.C.; Rich, T.D., eds. Research and management of the brown-headed cowbird in western landscapes. Studies in Avian Biology. 18: 10–17.

Goguen, C.B.; Mathews, N.E. 2000. Local gradients of cowbird abundance and parasitism relative to livestock grazing in a western landscape. Conservation Biology. 14: 1862–1869.

Goldberg, C.S.; Waits, L.P. 2009. Using habitat models to determine conservation priorities for pond-breeding amphibians in a privately-owned landscape of northern Idaho, USA. Biological Conservation. 142: 1096–1104.

Gosse, J.W.; Cox, R.; Avery, S.W. 2005. Home-range characteristics and habitat use by American martens in eastern Newfoundland. Journal of Mammalogy. 86: 1156–1163.

Grant, T.A.; Madden, E.; Berkey, G.B. 2004. Tree and shrub invasion in northern mixed-grass prairie: implications for breeding grassland birds. Wildlife Society Bulletin. 32: 807–818.

Grindal, S.D.; Morissette, J.L.; Bringham, R.M. 1999. Concentration of bat activity in riparian habitats over an elevational gradient. Canadian Journal of Zoology. 77: 972–977.

Gross, J.E.; Singer, F.J.; Moses, M.E. 2000. Effects of disease, dispersal, and area on bighorn sheep restoration. Restoration Ecology. 8(4S): 25–37.

Groves, C. 2003. Drafting a conservation blueprint: a practitioner's guide to planning for biodiversity. Washington, DC: Island Press. 480 p.

Groves, C.; Yenson, E. 1989. Rediscovery of the northern bog lemming (*Synaptomys borealis*) in Idaho. Northwest Naturalist. 70: 14–15.

Grubb, T.G.; King, R.M. 1991. Assessing human disturbance of breeding bald eagles with classification tree models. Journal of Wildlife Management. 55: 500–511.

Grubb, T.G.; Pater, L.L.; Delaney, D.K. 1998. Logging truck noise near nesting northern goshawks. Res. Note RMRS-RN-3. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 30 p.

Guinan, J.A.; Gowaty, P.A.; Eltzroth, E.K. 2000. Western bluebird (*Sialia mexicana*). In: Poole, A.; Gill, F., eds. The birds of North America, No. 510. Philadelphia, PA: The Birds of North America, Inc.

Guynn, D.C., Jr.; Guynn, S.T.; Wigley, T.B.; Miller, D.A. 2004. Herbicides and forest biodiversity–What do we know and where do we go from here? Wildlife Society Bulletin. 32: 1085–1092.

Hagar, D.C. 1960. The interrelationships of logging, birds, and timber regeneration in the Douglas-fir region of northwestern California. Ecology. 41: 116–125.

Hager, H.A.; McCoy, K.D. 1998. The implications of accepting untested hypotheses: a review of the effects of purple loosestrife (*Lythrum salicaria*) in North America. Biodiversity and Conservation. 7: 1069–1079.

Haggard, M.E.; Gaines, W.L. 2001. Effects of stand-replacement fire and salvage logging on a cavity-nesting bird community in eastern Cascades, Washington. Northwest Science. 75(4): 387–396.

Haggard, M.E.; Gaines, W.L. 2008. Management considerations for forest raptors on the Okanogan-Wenatchee, and Colville National Forests. Wenatchee, WA: U.S. Department of Agriculture, Forest Service, Wenatchee National Forest.

Hahn, T.P. 1996. Cassin's finch (*Carpodacus cassinii*). In: Poole, A.; Gill, F., eds. The Birds of North America, No. 240. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American, Ornithologist's Union.

Hamann, B.; Johnston, H.; McClelland, P. [et al.]. 1999. Birds. In: Joslin, G.; Youmans, H., coords. Effects of recreation on Rocky Mountain wildlife: a review for Montana. Helena, MT: Committee on Effects of Recreation on Wildlife, Montana Chapter of The Wildlife Society: 3.1–3.34.

Hamer, T.L.; Flather, C.H.; Noon, B.R. 2006. Factors associated with grassland bird species richness: the relative roles of grassland area, landscape structure, and prey. Landscape Ecology. 21: 569–583.

Hamerstrom, F. 1969. A harrier population study. In: Hickey, J.J., ed. Peregrine falcon populations, their biology and decline. Madison, WI: University of Wisconsin Press: 367–383.

Hamerstrom, F. 1986. Harrier, hawk of the marshes: the hawk that is ruled by a mouse. Washington, DC: Smithsonian Institution Press. 171 p.

Hamerstrom, F.; Kopeny, M. 1981. Harrier nest-site vegetation. Journal of Raptor Research. 15: 86–88.

Hann, W.J.; Jones, J.L.; Karl, M.G. [et al.]. 1997. Landscape dynamics of the basin. Gen. Tech. Rep. PNW-GTR-405. In: Quigley, T.M.; Arbelbide, S.J., tech. eds. An assessment of ecosystem components in the interior Columbia Basin and portions of the Klamath and Great Basins: volume 2. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 307–1055. 4 vol. (Quigley, T.M., tech. ed.; The Interior Columbia Basin Ecosystem Management Project: Scientific Assessment).

Hansen, A.; McComb, W.; Vega, R.; Raphael, M.; Hunter, M. 1995. Bird habitat relationships in natural and managed forests in west Cascades of Oregon. Ecological Applications. 5: 555–569.

Hanus, S.; Wollis, H.; Wilkinson, L. 2002. Western (*Aechmophors occidentalis*) and eared (*Podiceps nigricollis*) grebes of central Alberta: inventory, survey techniques and management concerns. Edmanton, AB; Alberta Sustainable Resource Development Fish and Wildlife Division Species at Risk Report 41.

Hargis, C.D.; Bissonette, J.A. 1997. Effects of forest fragmentation on populations of American marten in the intermountain West. In: Proulx, G.; Bryant, H.N.; Woodard, P.M., eds. Martes: taxonomy, ecology, techniques, and management. Edmonton, AB; Provincial Museum of Alberta: 437–451.

Hargis, C.D.; Bissonette, J.A.; Turner, D.L. 1999. The influence of forest fragmentation and landscape pattern on American martens. Journal of Applied Ecology. 36: 157–172.

Hargis, C.D.; McCarthy, C.; Perloff, R.D. 1994. Home ranges and habitats of northern goshawks in eastern California. Studies in Avian Biology. 16: 66–74.

Harmata, A.R.; Oakleaf, B. 1992. Bald eagles in the greater Yellowstone ecosystem: an ecological study with emphasis on the Snake River, Wyoming. Cheyenne, WY: Wyoming Game and Fish Department.

Harris, M.A. 1982. Habitat use among woodpeckers in forest burns. Missoula, MT: University of Montana. 62 p. M.S. thesis.

Harrison, H.H. 1979. A field guide to western birds' nests. Boston, MA: Houghton Mifflin Company. 279 p.

Harrod, R.J.; Gaines, W.L.; Hartl, W.E.; Camp, A. 1998. Estimating historical snag densities in dry forests east of the Cascade Range. Gen. Tech. Rep. PNW-GTR-428. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 16 p.

Harrod, R.J.; McRae, B.H.; Hartl, W.E. 1989. Historical stand reconstruction in ponderosa pine forests to guide silvicultural prescriptions. Forest Ecology and Management. 114: 433–446.

Harrod, R.J.; Povak, N.K.; Peterson, D.W. 2007. Comparing the effectiveness of thinning and prescribed fire for modifying structure in dry coniferous forests. [CD-ROM]. In: Bulter, B.W.; Cook, W., comps. The fire environment-innovations, management, and policy. RMRS-P-46CD. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 329–346.

Harvey, D.; Knight, J.; Cantebury, G. 2000. Proposed rule: California tiger salamander *Ambystoma californiense*. Sacramento, CA: U.S. Department of the Interior, Fish and Wildlife Service.

Havera, S.P.; Boens, L.R.; Georgi, M.M.; Shealy, R.T. 1992. Human disturbance of waterfowl on Keokuk Pool, Mississippi River. Wildlife Society Bulletin. 20: 290–298.

Hayes, G.E.; Buchanan, J.B. 2002. Washington State status report for the peregrine falcon. Olympia, WA: Washington Department of Fish and Wildlife. 77 p.

Hayes, J.P. 1997. Temporal variation in activity of bats and the design of echolocation-monitoring studies. Journal of Mammalogy. 62: 233–243.

Hayes, J.P. 2003. Habitat ecology and conservation of bats in western coniferous forests. In: Zabel, C.J.; Anthony, R.G., eds. Mammal community dynamics in coniferous forests of western North America: management and conservation. Cambridge, MA: Cambridge University Press: 81–119 p.

Hayes, J.P.; Loeb, S.C. 2007. The influences of forest management on bats in North America. In: Lacki, M.J.; Hayes, J.P.; Kurta, A., eds. Bats in forests conservation and management. Baltimore, MD: John Hopkins University Press: 207–235.

Hayes, M. P. 1997. Status of the Oregon spotted frog (*Rana pretiosa* sensu *stricto*) in the Deschutes Basin and selected other systems in Oregon and northeastern California with a rangewide synopsis of species' status. Final report prepared for The Nature Conservancy under contract to the U.S. Fish and Wildlife Service. Portland, OR. 57 p. plus appendices.

Hayes, M.P.; Jennings, M.R. 1986. Decline of ranid frog species in Western North America: are bullfrogs (*Rana catesbeiana*) responsible? Journal of Herpetology. 20: 490–509.

Hays, D.W.; Milner, R.L. 1999. Peregrine falcon, *Falco peregrinus*. In: Management recommendations for Washington's priority species. Volume IV: Birds. Olympia, WA: Washington Department of Fish and Wildlife: 11–1 to 11–4.

Hayward, G.D.; Hayward, P.H.; Garton, E.O. 1993. Ecology of boreal owls in the northern Rocky Mountains. Wildlife Monograph No. 124: 1–59.

Heady, P.H.; Frick, W.F. 2001. Bat inventory of Muir Woods National Monument. Final report. Aptos, CA; Central Coast Bat Research Group.

Hecht, W.R. 1951. Nesting of the marsh hawk at Delta, Manitoba. Wilson Bulletin. 63: 167–176.

Hecnar, S.J.; M'Closkey, R.T. 1997. The effects of predatory fish on amphibian species richness and distribution. Biological Conservation. 79: 123–131.

Hedges, S. 1994. Utah's bluebirds. Utah Birds. 10(1): 7–8.

Herkert, J.R.; Simpson, S.A.; Westemeier, R.L.; Esker, T.L.; Walk, J.W. 1999. Response of northern harriers and short-eared owls to grassland management in Illinois. Journal of Wildlife Management. 63: 517–523.

Herlugson, C.J. 1980. Biology of sympatric populations of western and mountain bluebirds. Pullman, WA: Washington State University. 133 p. Ph.D. dissertation.

Herrington, R.E.; Larsen, J.H. 1985. Current status, habitat requirements, and management of the larch mountain salamander *Plethodon larselli*. Biological Conservation. 34: 169–179.

Hessburg, P.F.; Agee, J.K. 2003. An environmental narrative of inland Northwest US forests, 1800-2000. Forest Ecology and Management. 178: 23–59.

Hessburg, P.F.; Agee, J.K.; Franklin, J.F. 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management. 211: 117–139.

Hessburg, P.F.; James, K.M.; Salter, R.B. 2007. Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology. 22(1): 5–24.

Hessburg, P.F.; Smith, B.G.; Kreiter, S.G.; Miller, C.A.; Salter, R.B.; McNicholl, C.H.; Hann, W.J. 1999a. Historical and current forest and range landscapes in the interior Columbia River basin and portions of the Klamath and Great Basins. Part 1. Linking vegetation patterns and landscape vulnerability to potential insect and pathogen disturbances. Gen. Tech. Rep. PNW-GTR-458. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 357 p.

Hessburg, P.F.; Smith, B.G.; Salter, R.B. 1999b. Using estimates of natural variation to detect ecologically important change in forest spatial patterns: a case study, Cascade Range, eastern Washington. Res. Rep. PNW-RP-514. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 65 p.

Hessburg, P.F.; Smith, B.G.; Salter, R.B.; Ottmar, R.D.; Alvarado, E. 2000. Recent changes (1930s–1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management. 136: 53–83.

Hessburg, P.F.; Reynolds, K.M.; Salter, R.B.; Dickinson, J.D.; Gaines, W.L.; Harrod, R.J. 2013. Landscape evaluation for restoration planning on the Okanogan-Wenatchee National Forest, USA. Sustainability. 5: 805–840.

Hicks, L.L.; Elder, J.M. 1979. Human disturbance of Sierra Nevada bighorn sheep. Journal of Wildlife Management. 43(3): 909–915.

Hill, R.A. 1976. Host-parasite relationships of the brown-headed cowbird in a prairie habitat of west-central Kansas. Wilson Bulletin. 88: 555–565.

Hirner, J.L.M.; Cox, S.P. 2007. Effects of rainbow trout (*Oncorhynchus mykiss*) on amphibians in productive recreational fishing lakes of British Columbia. Canadian Journal of Fisheries and Aquatic Sciences. 64: 1770–1780.

Hodges, K.E. 1999. Ecology of snowshoe hares in southern boreal and montane forests. In: Ruggiero, L.F.; Aubry, K.B.; Buskirk, S.W.; Koehler, G.M.; Krebs, C.J.; McKelvey, K.S.; Squires, J.R. Ecology and conservation of lynx in the United States. Gen. Tech. Rep. RMRS-GTR-30WWW. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 163–206. Chapter 7.

Hodgman, T.P.; Harrison, D.J.; Katnik, D.D.; Elowe, K.D. 1994. Survival in an intensively trapped marten population in Maine. Journal of Wildlife Management. 58: 593–600.

Hollenbeck, R.R. 1974. Growth rates and movements within a population of *Rana pretiosa pretiosa* Baird and Girard in south central Montana. Bozeman, MT: Montana State University. 65 p. Ph.D. dissertation.

Holmes, A.L. 2007. Short-term effects of a prescribed burn on songbirds and vegetation in mountain big sagebrush. Western North American Naturalist. 67: 292–298.

Holmes, T.L.; Knight, R.L.; Stegall, L.; Craig, G.R. 1993. Responses of wintering grassland raptors to human disturbance. Wildlife Society Bulletin. 21: 461–468.

Holthausen, R. 2002. White paper on managing for population viability. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Washington, DC.

Holthausen, R.S.; Raphael, M.G.; Samson, F.B.; Ebert, D.; Heibert, R.; Manesco, K. 1999. Population viability in ecosystem management. In: Sexton, W.T.; Malk, A.J.; Szaro, R.C.; Johnson, N.C., eds. Ecological stewardship: a common reference for ecosystem management, Vol. II. Kidlington, The Netherlands: Elsevier Science: 135–156.

Holthuijzen, A.M.A.; Eastland, W.G.; Ansell, A.R.; Kochert, M.N.; Williams, R.D.; Young, L.S. 1990. Effects of blasting on behavior and productivity of nesting prairie falcons. Wildlife Society Bulletin. 18: 270–281.

Houlahan, J.E.; Findlay, C.S. 2003. The effects of adjacent land use on wetland amphibian species richness and community composition. Canadian Journal of Fish and Aquatic Science. 60: 1078–1094.

Hoyt, S.F. 1957. The ecology of the pileated woodpecker. Ecology. 38: 246–256.

Huff, M.; Brown, M. 1998. Four yeas of bird point count monitoring in late-successional conifer forests and riparian areas from the Pacific Northwest national forests, interim results. Sandy, OR: U.S. Department of Agriculture, Forest Service. 142 p.

Huggett, A.J. 2005. The concept and utility of 'ecological thresholds' in biodiversity conservation. Biological Conservation. 124: 301–310.

Humes, M.L.; Hayes, J.P.; Collopy, M.W. 1999. Bat activity in thinned, unthinned, and old-growth forests in western Oregon. Journal of Wildlife Management. 63: 553–561.

Humphrey, S.R.; Kunz, T.H. 1976. Ecology of a Pleistocene relict, the western big-eared bat (*Plecotus townsendii*), in the southern Great Plains. Journal of Mammalogy. 57: 470–494.

Hunt, W.G.; Jackman, R.E.; Brown, T.L.; Gilardi, J.G.; Driscoll, D.E.; Culp, L. 1995. A pilot golden eagle population study in the Altamont Pass Wind Resource Area, California. Report to National Renewable Energy Laboratory. Subcontract No. XCG-4-14200. Santa Cruz, CA: Predatory Bird Research Group, University of California.

Hunt, W.G.; Jackman, R.E.; Hunt, T.L.; Driscoll, D.E.; Culp, L. 1999. A population study of golden eagles in the Altamont Pass Wind Resource Area; population trend analysis 1994–1997. Santa Cruz, CA: Predatory Bird Research Group, University of California.

Hunter, M.L.; Jacobson, G.L., Jr.; Webb, T., III. 1988. Paleoecology and the coarse-filter approach to maintaining biological diversity. Conservation Biology. 4: 375–384.

Hutto, R.L. 1995a. U.S.F.S. Northern Region Songbird Monitoring Program: distribution and habitat relationships. Internal report. Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Region.

Hutto, R.L. 1995b. Composition of bird communities following stand-replacement fires in northern Rocky Mountain (U.S.A.) conifer forests. Conservation Biology. 9: 1041–1058.

Hutto, R.L. 2006. Toward meaningful snag-management guidelines for postfire salvage logging in North American conifer forests. Conservation Biology. 20(4): 984–993.

Huxel, G.R.; Hastings, A. 1999. Habitat loss, fragmentation and restoration. Restoration Ecology. 7: 309–315.

Idaho State Conservation Effort. 1995. Habitat conservation assessment and conservation strategy for the Townsend’s big-eared bat. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, U.S. Department of the Interior, Bureau of Land Management; Interior Columbia Basin Ecosystem Management Project, 304 N. 8th Street, Boise, ID 83702.

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

Interagency Lynx Biology Team. 2013. Canada lynx conservation assessment and strategy. 3rd ed. Forest Service Publication R1-13-19. Missoula, MT: USDA Forest Service, USDI Fish and Wildlife Service, USDI Bureau of Land Management, and USDI National Park Service. 128 p.

International Union for the Conservation of Nature [IUCN]. 2000. International Union for the Conservation of Nature Red List Categories. Gland, Switzerland.

Jacobson, W.B. 1972. Relative abundance of the avian population along the South Platte River flood plain at the proposed Narrows Reservoir site. Greeley, CO: University of Northern Colorado. 82 p. M.S. thesis.

Jewett, S.G.; Taylor, W.P.; Shaw, W.T.; Aldrich, J.W. 1953. Birds of Washington state. Seattle, WA: University of Washington Press. 768 p.

Johnsgard, P.A. 1987. Diving birds of North America. Lincoln, NE: University of Nebraska Press. http://digitalcommons.unl.edu/cgi/viewcontent.cgi?article=1012&context=bioscidivingbirds. (June 6, 2014).

Johnsgard, P.A. 1990. Hawks, eagles, and falcons of North America: biology and natural history. Washington, DC: Smithsonian Institution.

Johnson, D.H.; Igle, L.D. 2001. Area requirements of grassland birds: a regional perspective. The Auk. 118(1): 24–34.

Johnson, D.H.; O'Neil, T.A., eds. 2001. Wildlife-habitat relationships in Oregon and Washington. Corvallis, OR: Oregon State University Press. 736 p.

Johnson, R. 1999. Washington. In: Toweill, D.E.; Geist, V. eds. Return of royality: wild sheep of North America. Missoula, MT: Boone and Crockett Club and Foundation for North American Wild Sheep: 112, 152.

Johnson, R.E. 1977. An historical ananysis of wolverine abundance and distribution in Washington. The Murrelet. 58: 13–16.

Johnson, R.E.; Cassidy, K.M. 1997. Terrestrial mammals of Washington state: Location data and predicted distributions. Volume 3 in Washington State Gap Analysis–final report. In: Cassidy, K.M.; Grue, C.E.; Smith, M.R.; Dvornich, K.M., eds. Seattle, WA: Washington Cooperative Fish and Wildlife Research Unit, University of Washington. 304 p.

Johnson, R.G.; Temple, S.A. 1990. Nest predation and brood parasitism of tallgrass prairie birds. Journal of Wildlife Management. 54: 106–111.

Johnson, R.R.; Haight, L.T. 1984. Riparian problems and initiatives in the American Southwest: a regional perspective. In: Warner, R.E.; Hendrix, K.M., eds. California riparian systems: ecology, conservation, and productive management. Berkeley, CA: University of California Press: 404–412.

Johnson, T.H.; Wauer, R.H. 1996. Avifaunal response to the 1977 La Mesa fire. In: Allen, C.D., tech. ed. fire effects in southwestern forests. Proceedings of the second La Mesa Fire symposium. Gen. Tech. Rep. RM-GTR-286. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 70–94.

Johnson, T.L.; Swift, D.M. 2000. A test of a habitat evaluation procedure for Rocky Mountain bighorn sheep. Restoration Ecology. 8(4S): 47–56.

Jones, Z.F.; Bock, C.E 2002. Conservation of grassland birds in an urban landscape: a historical perspective. Condor. 104: 643–651.

Kadera, J. 1987. Trees' future obscured by forest of rules. Oregonian. January 25; Sect. C: 1–4.

Kaisanlahti-Jokimäki, M.-L.; Jokimäki, J.; Huhta, E.; Ukkola, M.; Helle, P.; Ollila, T. 2008. Territory occupancy and breeding success of the golden eagle (*Aquila chrysaetos*) around tourist destinations in northern Finland. Ornis Fennica. 85: 2–12.

Kalcounis-Ruppell, M.C.; Psyllakis, J.M.; Brigham, R.M. 2005. Tree roost selection by bats: an empirical synthesis using meta-analysis. Wildlife Society Bulletin. 33: 1123–1132.

Kantrud, H.A.; Higgins, K.F. 1992. Nest and nest site characteristics of some ground-nesting, non-passerine birds of northern grasslands. Prairie Naturalist. 24: 67–84.

Kantrud, H.A.; Kologiski, R.L. 1982. Effects of soils and grazing on breeding birds of uncultivated upland grasslands of the northern Great Plains. Wildlife Research Report 15. Washington, DC: U.S. Fish and Wildlife Service. 33 p.

Kantrud, H.A.; Stewart, R.E. 1984. Ecological distribution and crude density of breeding birds on prairie wetlands. Journal of Wildlife Management. 48: 426–437.

Keinath, D.A. 2004. Fringed myotis (*Myotis thysanodes*): a technical conservation assessment. [Online]. USDA Forest Service, Rocky Mountain Region. http://www.fs.fed.us/r2/projects/scp/assessments/fringedmyotis.pdf. (June 15, 2010).

Kelsall, J.P. 1981. COSEWIC status report on the wolverine, *Gulo gulo*, in Canada. Ottawa, ON: Committee on the Status of Endangered Wildlife in Canada. 50 p.

Kennedy, P.L.; Ward, J.M.; Rinker, G.A.; Gessaman, J.A. 1994. Post-fledgling areas in northern goshawk home ranges. Studies in Avian Biology. 16: 75–82.

Kerley, L.L.; Anderson, S.H. 1995. Songbird responses to sagebrush removal in a high elevation sagebrush steppe ecosystem. Prairie Naturalist. 27: 129–146.

Kibbe, D.P. 1975. The nesting season June 1–July 31, 1975. Niagara-Champlain Region. American Birds. 29: 967–970.

Kimsey, B.; Conley, M.R. 1988. Habitat use by raptors in southcentral New Mexico. In: Glinski, R.L.; Pendleton, B.G.; Moss, M.B.; LeFranc, M.N., Jr.; Millsap, B.A.; Hoffman, S.W., eds. Proceedings of the southwest raptor management symposium and workshop. National Wildlife Federation Scientific and Technical Series. 11: 197–203.

King, M.M.; Workman, G.W. 1986. Response of desert bighorn sheep to human harassment: management implications. North American Wildlife and Natural Resources Conference. 51: 74–85.

Kirk, D.A.; Hobson, K.A. 2001. Bird-habitat relationships in jack pine boreal forests. Forest Ecology and Management. 147: 217–243.

Kirk, T.A.; Zielinski, W.J. 2009. Developing and testing a landscape habitat suitability model for the American marten (*Martes americana*) in the Cascades mountains of California. Landscape Ecology. 24: 759–773.

Klopatek, J.M.; Olson, R.J.; Emerson, J.L. 1979. Land use conflicts with natural vegetation in the United States. Environmental Conservation. 6: 191–200.

Knapp, R.A.; Hawkins, C.P.; Ladau, J.; McClory, J.G. 2005. Fauna of Yosemite National Park lakes has low resistance but high resilience to fish introductions. Ecological Applications. 15: 835–847.

Knapp, R.A.; Matthews, K.R.; Preisler, H.K.; Jellison, R. 2003. Developing probabilistic models to predict amphibian site occupancy in a patchy landscape. Ecological Applications. 13: 1069–1082.

Knapp, R.A.; Matthews, K.R.; Sarnelle, O. 2001. Resistance and resilience of alpine lake faunal assemblages to fish introductions. Ecological Monographs. 71: 401–421.

Knick, S.T.; Rotenberry, J.T. 1995. Habitat relationships and breeding birds on the Snake River Birds of Prey Area. Boise, ID: Idaho Bureau of Land Management Technical Bulletin 95-5. 9 p.

Knick, S.T.; Rotenberry, J.T. 1997. Landscape characteristics of disturbed shrubsteppe habitats in southwestern Idaho. Landscape Ecology. 12: 287–297.

Knick, S.T.; Rotenberry, J.T. 2002. Effects of habitat fragmentation on passerine birds breeding in intermountain shrub steppe. In: George, T.L.; Dobkin, D.S., eds. Effects of habitat fragmentation on birds in western landscapes: contrasts with paradigms from the eastern United States. Studies in Avian Biology. 25. 130–140.

Knopf, F.L.; Sedgwick, J.A.; Cannon, R.W. 1988. Guild structure of a riparian avifauna relative to seasonal cattle grazing. Journal of Wildlife Management. 52: 280–290.

Kochert, M.N. 1989. Responses of raptors to livestock grazing in the Western United States. In: Pendleton, B.G., ed. Proceedings of the western raptor management symposium. National Wildlife Federation Scientific and Technical Series 12: 194–203.

Kochert, M.N.; Steenhof, K.; Carpenter, L.B.; Marzluff, J.M. 1999. Effects of fire on golden eagle territory occupancy and reproductive success. Journal of Wildlife Management. 63: 773–780.

Kochert, M.N.; Steenhof, K.; McIntyre, C.L.; Craig, E.H. 2002. Golden eagle (*Aquila chrysaetos*). Number 684. In: Poole, A.; Gill, F., eds. The birds of North America. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union.

Koehler, G.M. 1990. Population and habitat characteristics of lynx and snowshoe hares in north central Washington. Canadian Journal of Zoology. 68: 845–851.

Koehler, G.M.; Aubry, K.B. 1994. Lynx. In: Ruggerio, L.F.; Aubry, K.B.; Buskirk, S.W.; Lyon, L.J.; Zielinski, W.J., tech. coords. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the United States. Gen. Tech. Rep. GTR-RM-25. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 74–98.

Koehler, G.M.; Maletzke, B.J.; Von Kienast, J.A.; Aubry, K.B.; Wielgus, R.B.; Naney, R.H. 2008. Habitat fragmentation and the persistence of lynx populations in Washington state. Journal of Wildlife Management. 72(7): 1518–1524.

Koehler, G.M.; Moore, W.R.; Taylor, R.A. 1975. Preserving the pine marten: management guidelines for western forests. Western Wildlife. 2: 31–36.

Kolbe, J.A.; Squires, J.R.; Pletscher, D.H.; Ruggiero, L.F. 2007. The effect of snowmobile trails on coyote movements within lynx home ranges. Journal of Wildlife Management. 71: 1409–1418.

Kotliar, N.B.; Hejl, S.J.; Hutto, R.L.; Saab, V.A.; Melcher, C.P.; McFadzen, M.E. 2002. Effects of fire and post-fire salvage logging on avian communities in conifer-dominated forests of the western United States. Studies in Avian Biology. 25: 49–64.

Krausman, P.R., ed. 1996. Rangeland wildlife. Denver, CO: Society for Range Management. 440 p.

Krebs, J.A.; Lewis, D. 1999. Wolverine ecology and habitat use in the North Columbia Mountains: progress report. British Columbia, Canada: Columbia Basin Fish and Wildlife Compensation Program.

Krebs, J.A.; Lofroth, E.C.; Parfitt, I. 2007. Multiscale habitat use by wolverines in British Columbia, Canada. Journal of Wildlife Management. 71(7): 2180–2192.

Kreisel, K.J.; Stein, S.J. 1999. Bird use of burned and unburned coniferous forests during winter. Wilson Bulletin. 111: 243–250.

Kroodsma, D.E.; Verner, J. 1997. Marsh wren (*Cistothorus palustris*). In: Poole, A.; Gill, F. eds. The Birds of North America, No. 308. Philadelphia, PA: The Birds of North America, Inc.

Kuchel, C.R. 1977. Some aspects of the behavior and ecology of harlequin ducks in Glacier National Park, Montana. Missoula, MO: University of Montana. 314 p. M.S. thesis.

Kunz, T. H.; Martin, R.A. 1982. *Plecotus townsendii*. Mammalian Species. 175: 1–6.

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

Lambeck, R.J. 1997. Focal species: a multi-species umbrella for nature conservation. Conservation Biology. 11: 849–856.

Landres, P.B.; Morgan, P.; Swanson, F.J. 1999. Overview of the use of natural range of variability concepts in managing ecological systems. Ecological Applications. 9(4): 1179–1188.

Landres, P.B.; Verner, J.; Thomas, J.W. 1988. Ecological use of vertebrate indicator species: a critique. Conservation Biology. 2: 316–328.

Lanier, J.W.; Joseph, R.A. 1989. Managing human recreational impacts on hacked or free-ranging peregrines. In: Pendleton, B.A.G., ed. Proceedings of the western raptor management symposium and workshop. National Wildlife Federation Scientific and Technical Series No. 12: 149–153.

Lawler, J.J.; Mathias, M. 2007. Climate change and the future of biodiversity in Washington. Report prepared for the Washington Biodiversity Council. Seattle, WA: University of Washington. 42 p.

Layser, E.F.; Burke, T.E. 1973. The northern bog lemming and its unique habitat in northeastern Washington. The Murrelet. 54(1): 7–8.

Leege, T.A.; Herman, D.J.; Zamora, B. 1981. Effects of cattle grazing on mountain meadows in Idaho. Journal of Range Management. 34: 324–328.

Lehmkuhl, J.F.; Burger, E.D.; Drew, E.K.; Lindsey, J.P.; Haggard, M.; Woodruff, K.Z. 2007b. Breeding birds in riparian and upland dry forests of the Cascade range. Journal of Wildlife Management. 71: 2632–2643.

Lehmkuhl, J.F.; Everett, R.L.; Schellhaas, R. [et al.]. 2003. Cavities in snags along a wildfire chronosequence in eastern Washington. Journal of Wildlife Management. 67(1): 219–228.

Lehmkuhl, J.F.; Lyons, A.L.; Bracken, E.; Leingang, J.; Gaines, W.L.; Dodson, E.K.; Singleton, P.H. 2013. Forage composition, productivity, and utilization in eastern Washington Cascade Range. Northwest Science. 87(3): 207–231.

Lehmkuhl, J.F.; Kennedy, M.; Ford, E.D.; Singleton, P.H.; Gaines, W.L.; Lind, R.L. 2007a. Seeing the forest for the fuels: integrating ecological values and fuels management. Forest Ecology and Management. 246: 73–80.

Lehmkuhl, J.F.; Kie, J.G.; Bender, L.C.; Servheen, G.; Nyberg, H. 2001. Evaluating the effects of ecosystem management alternatives on elk, mule deer, and white-tailed deer in the interior Columbia River basin, U.S.A. Forest Ecology and Management. 153: 89–104.

Lehmkuhl, J.F.; Raphael, M.G.; Holthausen, R.S.; Hickenbottom, J.R.; Naney, R.H.; Shelly, J.S. 1997. Historical and current status of terrestrial species and the effects of the proposed alternatives. In: Quigley, T.M.; Lee, K.M.; Arbelbide, S.J., eds. Evaluation of EIS alternatives by the Science Integration Team. Gen. Tech. Rep. PNW-GTR-406. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 537–730.

Leonard, M.L.; Picman, J. 1986. Why are nesting marsh wrens and yellow-headed blackbirds spatially segregated? Auk. 103: 135–140.

Leonard, W.P.; Brown, H.A.; Jones, L.L.C.; McAllister, K.R.; Storm, R.M. 1993. Amphibians of Washington and Oregon. Seattle, WA: Seattle Audubon Society. 169 p.

Leslie, D.M.; Douglas, C.L. 1980. Human disturbance at water sources of the desert bighorn sheep. Wildlife Society Bulletin. 8(4): 284–290.

Lewis, J.C.; Kraege, D. 2000. Cavity nesting ducks. In: Management recommendations for Washington's priority species. Volume IV: Birds. Olympia, WA: Washington Department of Fish and Wildlife: 4–1 to 4–6.

Lindenmayer, D.B.; Franklin, J.F.; Fischer, J. 2006. General management principles and a checklist of strategies to guide forest biodiversity conservation. Biological Conservation. 131: 433–445.

Lindenmayer, D.B.; Luck, G. 2005. Synthesis: thresholds in conservation and management. Biological Conservation. 124: 351–354.

Lindenmayer, D.B.; Manning, A.D.; Smith, P.L.; Possingham, H.P.; Fischeer, J.; Oliver, I.; McCarthy, M.A. 2002. The focal-species approach and landscape restoration: a critique. Conservation Biology. 16(2): 338–345.

Linder, E.T.; Klaus, N.A.; Buehler, D.A. 2004. Population viability as a measure of forest sustainability. In: Rauscher, H.M.; Johnsen, K., eds. Southern forest science: past, present, and future. Gen. Tech. Rep. GTR-SRS-75. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeast Research Station: 307–317.

Linder G.; Wyant, J.; Meganck, R.; Williams, B. 1991. Evaluating amphibian responses in wetlands impacted by mining activities in the Western United States. In: Comer, R.D.; Davis, P.R.; Foster, S.Q.; Grant, C.V.; Rush, R.; Thorne, O., II; Todd, J., eds. Issues and technology in the management of impacted wildlife. Boulder, CO: Thorne Ecological Institute: 17–25.

Linder, K.A.; Anderson, S.H. 1998. Nesting habitat of Lewis's woodpeckers in southeastern Wyoming. Journal of Field Ornithology. 69: 109–116.

Linders, M.J.; Stinson, D.W. 2007. Washington State Recovery Plan for the Western Gray Squirrel. Olympia, WA: Washington Department of Fish and Wildlife. 128 p.

Linders, M.J.; West, S.D.; Vander Haegen, M. 2004. Seasonal variability in the use of space by western gray squirrels in southcentral Washington. Journal of Mammalogy. 85: 511–516.

Linner, S.C. 1980. Resource partitioning in breeding populations of marsh hawks and short-eared owls. Logan, UT: Utah State University. 66 p. M.S. thesis.

Linz, G.M.; Blixt, D.C.; Bergman, D.L.; Bleier, W.J. 1996. Responses of red-winged blackbirds, yellow-headed blackbirds and marsh wrens to glyphosate-induced alterations in cattail density. Journal of Field Ornithology. 67: 167–176.

Loeb, S.C.; Waldrop, T.A. 2007. Bat activity in relation to fire and fire surrogate treatments in southern pine stands. Forest Ecology and Management. 255: 3185–3192.

López-López, P.; Garcia-Ripollés, C.; Soutullo, Á.; Cadahia, L.; Urios, V. 2007. Identifying potentially suitable nesting habitat for golden eagles applied to 'important bird areas' design. Animal Conservation. 10: 208–218.

Ludwig, J.A.; Reynolds, J.F. 1988. Statistical ecology. New York: John Wiley and Sons. 337 p.

Lunde, R.E.; Harestad, A.S. 1986. Activity of little brown bats in coastal forests. Northwest Science. 60: 206–209.

Lunney, D.; Curtin, A.; Ayers, D.; Cogger, H.J.; Dickman, C.R. 1996. An ecological approach to identifying the endangered fauna of New South Wales. Pacific Conservation Biology. 2: 212–231.

Lusk, J.J.; Wells, K.S.; Guthery, F.S.; Fuhlendorf, S.D. 2003. Lark sparrow (*Chondestes grammacus*) nest-site selection and success in a mixed-grass prairie. Auk. 120: 120–129.

Lyons, A.L.; Gaines, W.L.; Lehmkuhl, J.F.; Harrod, R.J. 2008. Short-term effects of fire and fire surrogate treatments on foraging tree selection by cavity-nesting birds in dry forests of central Washington. Forest Ecology and Management. 255: 3203–3211.

MacArthur, R.A.; Geist, V.; Johnston, R.H. 1982. Cardiac and behavioral responses of mountain sheep to human disturbance. Journal of Wildlife Management. 46(2): 351–358.

MacArthur, R.A.; Johnston, R.H.; Geist, V. 1979. Factors influencing heart rate in free-ranging bighorn sheep: a physiological approach to the study of wildlife harassment. Canadian Journal of Zoology. 57: 2010–2021.

Machtans, C.S.; Latour, P.B. 2003. Boreal forest songbird communities of the Liard Valley, Northwest Territories, Canada. Condor. 105: 27–44.

Maclaren, P.A. 1986. Resource partitioning in an assemblage of breeding raptors from southeast Wyoming. Laramie, WY: University of Wyoming. 64 p. M.S. thesis.

MacWhirter, R.B.; Bildstein, K.L. 1996. Northern harrier (*Circus cyaneus*). In: Poole, A.; Gill, F., eds. The birds of North America, No. 210. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union.

Maletzke, B.J.; Koehler, G.M.; Wielgus, R.B.; Aubry, K.B.; Evans, M.A. 2008. Habitat conditions associated with lynx hunting behavior during winter in northern Washington. Journal of Wildlife Management. 72(7): 1473–1478.

Maletzke, B.T. 2004. Winter habitat selection of lynx in northern Washington. Pullman, WA: Washington State University. 48 p. M.S. thesis.

Manci, K.M.; Rusch, D.H. 1988. Indices to distribution and abundance of some inconspicuous waterbirds on Horicon Marsh. Journal of Field Ornithology. 59: 67–75.

Mannan, R.W. 1984. Summer area requirements of pileated woodpeckers in western Oregon. Wildlife Society Bulletin. 12: 265–268.

Marcot, B.G. 1984. Winter use of some northwestern California caves by western big-eared bats and long-eared myotis. Murrelet. 65: 46.

Marcot, B.G. 2006. Habitat modeling for biodiversity conservation. Northwest Naturalist. 87: 56–65.

Marcot, B.G.; Hohenlohe, P.A.; Morey, S.; Holmes, R.; Molina, R.; Turley, M.C.; Huff, M.H.; Laurence, J.A. 2006a. Characterizing species at risk II: using Bayesian belief networks as decision support tools to determine species conservation categories under the Northwest Forest Plan. Ecology and Society. URL:http://www.ecologyandsociety.org/vol11/iss2/art12. (June 15, 2010).

Marcot, B.G.; Holthausen, R.S.; Raphael, M.G.; Rowland, M.M.; Wisdom, M.J. 2001. Using Bayesian belief networks to evaluate fish and wildlife population viability under land management alternatives from an environmental impact statement. Forest Ecology and Management. 153: 29–42.

Marcot, B.G.; Murphy, D.D. 1996. On population viability analysis and management. In: Szaro, R.C.; Johnson, D.W., eds. Biodiversity in managed landscapes: theory and practice. New York: Oxford University Press: 58–76.

Marcot, B.G.; Steventon, J.D.; Sutherland, G.D.; McCann, R.K. 2006b. Guidelines for developing and updating Baysian belief networks for ecological modeling. Canadian Journal of Forest Research. 36: 3063–3074.

Marsh, D.M.; Milam, G.S.; Gorham, N.P.; Beckman, N.G. 2005. Forest roads as partial barriers to terrestrial salamander movement. Conservation Biology. 19(6): 2004–2008.

Martin, J.W. 1987. Behavior and habitat use of breeding northern harriers in southwestern Idaho. Journal of Raptor Research. 21: 57–66.

Martin, J.; McIntyre, C.L.; Hines, J.E.; Nichols, J.D.; Schmutz, J.A.; MacCluskie, M.C. 2009. Dynamic multistate site occupancy models to evaluate hypotheses relevant to conservation of golden eagles in Denali National Park, Alaska. Biological Conservation. 142: 2726–2731.

Martin, J.W.; Parrish, J.R. 2000. Lark sparrow (*Chondestes grammacus*). In: Poole, A.; Gill, F., eds. The birds of North America, No. 488. Philadelphia, PA: The Birds of North America, Inc.

Martin, S.K. 1987. The ecology of the pine marten (*Martes americana*) at Sagehen Creek, California. Berkeley, CA: University of California. 223 p. Ph.D. dissertation.

Martin, S.K.; Barrett, R.H. 1991. Resting site selection by marten at Sagehen Creek, California. Northwest Naturalist. 72: 37–42.

Marzluff, J.M.; Knick, S.T.; Vekasy, M.S.; Schueck, L.S.; Zarriello, T.J. 1997. Spatial use and habitat selection of golden eagles in southwestern Idaho. Auk. 114: 673–687.

Master, L.L. 1991. Assessing threats and setting priorities for conservation. Conservation Biology. 5: 559–563.

Master, L.L.; Stein, B.A.; Kutner, L.S.; Hammerson, G.A. 2000. Vanishing assets: conservation status of US species. In: Stein, B.A.; Kutner, L.S.; Adams, J.S., eds. Precious heritage: the status of biodiversity in the United States. New York, NY: Oxford University Press: 93–118.

Mazurek, M.J. 2004. A maternity roost of Townsend's big-eared bats *(Corynorhinus townsendii)* in coast redwood basal hollows in northwestern California. Northwestern Naturalist. 85: 60–62.

McAdoo, J.K.; Longland, W.S.; Evans, R.A. 1989. Nongame bird community responses to sagebrush invasion of crested wheatgrass seedlings. Journal of Wildlife Management. 53: 489–502.

McAllister, K.R.; Leonard, W.P. 1997. Washington state status report for the Oregon spotted frog. Olympia, WA: Washington Department of Fish and Wildlife. 47 p.

McClelland, B.R. 1977. Relationships between hole-nesting birds, forest snags, and decay in western larch-Douglas-fir forests of the northern Rocky Mountains. Missoula, MO: University of Montana. 498 p. Ph.D. dissertation.

McClelland, B.R. 1979. The pileated woodpecker in forests of the northern Rocky Mountians. In: Dickson, J.G.; Conner, R.N.; Fleet, R.R.; Jackson, J.A.; Kroll, J.C., eds. The role of insectivorous birds in forest ecosystems. New York, NY: Academic Press: 283–299.

McCord, C.M.; Cardoza, J.E. 1982. Bobcat and lynx. In: Chapman, J.A.; Feldhammer, G.A., eds. Wild mammals of North America. Baltimore, MD: Johns Hopkins University Press: 728–766.

McGahan, J. 1968. Ecology of the golden eagle. Auk. 85: 1–12.

McGarigal, K.; Anthony, R.G.; Isaacs, F.B. 1991. Interactions of humans and bald eagles on the Columbia River Estuary. Wildlife Monographs. 115: 1–47.

McGrady, M.J.; Grant, J.R.; Bainbridge, I.P.; McLeod, D.R.A. 2002. A model of golden eagle ranging behavior. Journal of Raptor Research. 36(supplement): 62–69.

McGrath, M.T.; DeStefano, S.; Riggs, R.; Irwin, L.L.; Roloff, G.J. 2003. Spatially explicit influences on northern goshawk nesting habitat in the interior Pacific Northwest. Wildlife Monograph No. 154: 1–63.

McIntyre, C.L.; Collopy, M.W.; Kidd, J.G.; Stickney, A.A.; Paynter, J. 2006. Characteristics of the landscape surrounding golden eagle nest sites in Denali National Park and Preserve, Alaska. Journal of Raptor Research. 40: 46–51.

McIntyre, K.K. 2002. Species composition and beta diversity of avian communities in burned, mixed, and unburned sagebrush steppe habitat at Sheldon National Wildlife Refuge, Nevada. Alpine, TX: Sul Ross University. M.S. thesis.

McKelvey, K.S.; Aubry, K.B.; Ortega, Y.K. 2000. History and distribution of lynx in the contiguous United States. In: Ruggerio, L.F.; Aubry, K.B.; Buskirk, S.W.; Koehler, G.M.; Krebs, C.J.; McKelvey, K.S.; Squires, J.R., eds. Ecology and conservation of lynx in the United States. Denver, CO: University Press of Colorado: 207–264.

McKibben, L.A.; Hofmann, P. 1985. Breeding range and populations studies of common snipe in California. California Fish and Game. 71: 68–75.

McLeod, D.R.; Whitfield, A.D.P.; McGrady, M.J. 2002. Improving prediction of golden eagle (*Aquila chrysaetos*) ranging in western Scotland, using GIS and terrain modelling. Journal of Raptor Research. 36(supplement): 72–79.

Medin, D.E.; Clary, W.B. 1991. Breeding bird populations in a grazed and ungrazed riparian habitat in Nevada. Res. Pap. INT-441. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 4 p.

Mellen-McLean, K.; Marcot, B.G.; Ohmann, J.L; Waddell, K.; Livingston, S.A.; Willhite, B.A.; Hostetler, B.B.; Ogden, C.; Dreisbach, T. 2009. DecAID, the decayed wood advisor for managing snags, partially dead trees, and down wood for biodiversity in forests of Washington and Oregon. Version 2.1. U.S. Department of Agriculture, Forest Service, Pacific Northwest Region and Pacific Northwest Research Station; U.S. Department of the Interior, Fish and Wildlife Service, Oregon State Office; Portland, OR. http://wwwnotes.fs.fed.us:81/pnw/DecAID/DecAID.nsf. (June 15, 2010).

Mellen-McLean, K.; Wales, B.; Bresson, B. 2013. A conservation assessment for whiteheaded woodpecker *(Picoides albolarvatus)*. Portland, OR. U.S. Department of Agriculture, Forest Service, Pacific Northwest Region and U.S. Department of the Interior, Bureau of Land Management, Oregon State Office. 48 p.

Mellen, T.K. 1987. Home range and habitat use by pileated woodpeckers. Corvallis, OR: Oregon State University. 96 p. M.S. thesis.

Menkens, G.E.; Anderson, S.H. 1983. How selective are golden eagles when choosing a nest site? Colorado-Wyoming Academy of Sciences. 15: 55–56.

Meslow, E.C.; Wight, H.M. 1975. Avifauna and succession in Douglas-fir forests of the Pacific Northwest. In: Smith, D.R., coord. Proceedings of the symposium on management of forest and range habitats for nongame birds. Gen. Tech. Rep. WO-1. Washington, DC: U.S. Department of Agriculture, Forest Service: 266–271.

Messmer, T.A. 1990. Influence of grazing treatments on nongame birds and vegetation structure in south central North Dakota. Fargo, ND: North Dakota State University. 164 p. Ph.D. dissertation.

Meyer, S.W. 2003. Comparative use of *Phragmites australis* and other habitats by birds, amphibians, and small mammals at Long Point, Ontario. Ontario, Canada: University of Western Ontario. 127 p. M.S. thesis.

Millar, C.I.; Woolfenden, W.B. 1999. The role of climate change in interpreting historical variability. Ecological Applications. 9(4): 1207–1216.

Millsap, B.A.; Gore, J.A.; Runder, D.E.; Cerulean, S.I. 1990. Setting priorities for the conservation of fish and wildlife species in Florida. Wildlife Monographs. 111: 1–57.

Milne, K.A.; Hejl, S.J. 1989. Nest-site characteristics of white-headed woodpeckers. Journal of Wildlife Management. 53: 50–55.

Moen, R.; Burdett, C.L.; Niemi, G.J. 2008. Movement and habitat use of Canada lynx during denning in Minnesota. Journal of Wildlife Management. 72(7): 1507–1513.

Molloy, J.; Davis, A. 1992. Setting priorities for conservation of New Zealand's threatened plants and animals. Wellington, New Zealand: Department of Conservation. 26 p.

Monello, R.J.; Dennehy, J.J.; Murray, D.L.; Wirsing, A.J. 2006. Growth and behavioral responses of tadpoles of two native frogs to an exotic competitor, *Rana catesbeiana*. Journal of Herpetology. 40: 403–407.

Monello, R.J.; Wright, R.G. 1999. Amphibian habitat preferences among artificial ponds in the Palouse region of northern Idaho. Journal of Herpetology. 33: 298–303.

Morneau, F.; Brodeur, S.; Decarie, R.; Carriere, S.; Bird, D.M. 1994. Abundance and distribution of nesting golden eagles in Hudson Bay, Quebec. Journal of Raptor Research. 28: 220–225.

Morris, R.L.; Tanner, W.W. 1969. The ecology of the western spotted frog, *Rana pretiosa pretiosa* Baird and Girard, a life history study. Great Basin Naturalist. 29: 45–81.

Morrison, M.L. 1981. The structure of western warbler assemblages: analysis of foraging behavior and habitat selection in Oregon. Auk. 98: 578–588.

Morrison, M.L.; Meslow, E.C. 1983. Bird community structure on early growth clear cuts in western Oregon. American Midland Naturalist. 110: 129–137.

Morgan, P.L.; Aplet, G.H.; Haufler, J.B.; Humphries, H.C.; Moore, M.M.; Wilson, W.D. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. In: Sampson, R.N.; Adams, D.L., eds. Assessing forest ecosystem health in the inland West. New York: Hawthorn Press: 87–111.

Mosconi, D.L.; Hutto, R.L. 1982. The effects of grazing on land birds of a western Montana riparian habitat. In: Wildlife-livestock relationships symposium. Peek, J.M.; Dalke, P.D., eds. University of Idaho, Forest and Range Experimental Station. Moscow, ID: 221–223

Mosher, J.A.; White, C.M. 1976. Directional exposure of golden eagle nests. Canadian Field-Naturalist. 90: 356–359.

Mowat, G.; Poole, K.G.; O' Donoghue, M. 1999. Ecology of lynx in northern Canada and Alaska. In: Ruggerio, L.F.; Aubry, K.B.; Buskirk, S.W.; Koehler, G.M.; Krebs, C.J.; McKelvey, K.S.; Squires, J.R. Ecology and conservation of lynx in the United States. Gen. Tech. Rep. RMRS-GTR-30WWW. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 265–306.

Mueller, H. 1999. Wilson's snipe (*Gallinago delicata*). In: Poole, A.; Gill, F., eds. The birds of North America, No. 417. Philadelphia, PA: The Birds of North America, Inc.

Munger, J.C.; Garber, M.; Madrid, K.; Carroll, M.; Peterson, W.; Heberger, L. 1998. U.S. national wetland inventory classifications as predictors of the occurrence of Columbia spotted frogs (*Rana luteiventris*) and Pacific treefrogs (*Hyla regilla*). Conservation Biology. 12: 320–330.

Muradian, R. 2001. Ecological thresholds: a survey. Ecological Economics. 38: 7–24.

Murphy, E.C.; Lehnhausen, W.A. 1998. Density and foraging ecology of woodpeckes following a stand-replacement fire. Journal of Wildlife Management. 62:4: 1359–1372.

Myers, L. 1991. Managing livestock to minimize impacts on riparian areas. In: Roth, R.; Bridges, C.; Zimmerman, C., eds. Riparian: What does it mean to me? Pueblo, CO: Third annual Colorado Riparian Association Conference: 24–30.

Nagorsen, D.W.; Brigham, R.M. 1993. Bats of British Columbia. Vancouver, WA: University of British Columbia Press. 164 p.

Nams, V.O.; Bourgeois, M. 2004. Fractal dimension measures habitat use at different spatial scales: an example with marten. Canadian Journal of Zoology. 82: 1738–1747.

Nappi, A.; Drapeau, P.; Giroux, J.; Savard, J.L. 2003. Snag use by foraging blackbacked woodpeckers (*Picoides arcticus*) in a recently burned eastern boreal forest. Auk. 120: 505–511.

National Forest Management Act of 1976 [NFMA]; Act of October 22, 1976; 16 U.S.C. 1600.

NatureServe. 2009. NatureServe Explore: an online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, VA: U.S.A. http://www.natureserve.org/explorer. (19 January 2010).

Naugle, D.E.; Higgins, K.F.; Nusser, S.M. 1999. Effects of woody vegetation on prairie wetland birds. Canadian Field-Naturalist. 113: 487–492.

Naugle, D.E.; Johnson, R.; Estey, M.; Higgins, K. 2001. A landscape approach to conserving wetland bird habitat in the Prairie Pothole Region of eastern South Dakota. Wetlands. 21: 1–17.

Nauman, R.S.; Olson, D.H. 1999. Survey and manage salamander known sites. In: Olson, D.H., ed. Survey protocols for amphibians under the survey and manage provision of the Northwest Forest Plan. Version 3.0. Interagency Special Status and Sensitive Species Program. Portland, OR: U.S. Department of Agriculture, Forest Service and U.S. Department of the Interior, Bureau of Land Management: 43–78.

Nelson, S.K. 1988. Habitat use and densities of cavity nesting birds in the Oregon Coast Ranges. Corvallis, OR: Oregon State University. 157 p. M.S. thesis.

Newman, G.A. 1970. Cowbird parasitism and nesting success of lark sparrows in southern Oklahoma. Wilson Bulletin. 82: 304–309.

Noon, B.R. 2003. Principles of ecosystem monitoring design. In: Busch, D.E.; Trexler, J.C., eds. Monitoring ecosystems: interdisciplinary approaches for evaluating ecoregional initiatives. Washington, DC: Island Press: 27–72. Chapter

Nordstrom, N.; Milner, R. 1997. Larch mountain salamander, *Plethodon larselli*. In: Larson, E.M., tech. ed. Management recommendations for Washington's priority species. Volume III: Amphibians and reptiles. Olympia, WA: Washington Department of Fish and Wildlife: 3-1–3-8.

Noss, R.F. 1990. Indicators for monitoring biodiversity: a hierarchial approach. Conservation Biology. 4(4): 355–364.

Noss, R.F.; O'Connell, M.A.; Murphy, D.D. 1997. The science of conservation planning: habitat conservation under the Endangered Species Act. Washington, DC: Island Press. 246 p.

Nussbaum, R.A.; Brodie, E.D.; Storm, R.M. 1983. Amphibians and reptiles of the Pacific Northwest. Moscow, ID: University of Idaho Press. 332 p.

Nyberg, J.B.; Marcot, B.G.; Sulyma, R. 2006. Using Bayesian belief networks in adaptive management. Canadian Journal of Forest Research. 36: 1–13.

Ogden, J.C.; Davis, S.M.; Brandt, L.A. 2003. Science strategy for a regional ecosystem monitoring and assessment program: The Florida Everglades examples. In: Busch, D.E.; Trexler, J.C., eds. Monitoring ecosystems: interdisciplinary approaches for evaluating ecoregional initiatives. Washington, DC: Island Press: 135–166.

Ohmann, J.L.; Gregory, M.J. 2002. Predictive mapping of forest composition and structure with direct gradient analysis and nearest neighbor imputation in the coastal province of Oregon, U.S.A. Canadian Journal of Forest Research. 32: 725–741.

Olechnowski, B.F.M.; Debinski, D.M . 2008. Response of songbirds to riparian willow habitat structure across two regions of the Greater Yellowstone Ecosystem. Wilson Journal of Ornithology. 120: 830–839.

Olendorff, R.R. 1976. The food habits of North American golden eagles. American Midland Naturalist. 95: 231–236.

Olson, G.S.; Glenn, E.M.; Anthony, R.G.; Forsman, E.D.; Reid, J.A.; Loschl, P.J.; William, J.R. 2004. Modeling demographic performance of northern spotted owls relative to forest habitat in Oregon. Journal of Wildlife Management. 68(4): 1039–1053.

Oregon Department of Fish and Wildlife. 2005. Oregon Conservation Strategy. Salem, OR.

Organ, J.F.; Vashon, J.H.; McDonald, J.E., Jr.; Vashon, A.D.; Crowley, S.M.; Jakubas, W.J.; Matula, G.J., Jr.; Meehan, A.L. 2008. Within-stand selection of Canada lynx natal dens in northwest Maine, USA. Journal of Wildlife Management. 72(7): 1514–1517.

Ozesmi, S.L.; Ozesmi, U. 1999. An artificial neural network approach to spatial habitat modelling with interspecific interaction. Ecological Modelling. 116: 15–31.

Page, J.L.; Dodd, N.; Osborne, T.O.; Carson, J.A. 1978. The influence of livestock grazing on non-game wildlife. California Nevada Wildlife: 159–173.

Papouchis, C.M.; Singer, F.J.; Sloan, W.B. 2001. Responses of desert bighorn sheep to increased human recreation. Journal of Wildlife Management. 65(3): 573–582.

Patla, S.M. 1997. Nesting ecology and habitat of the northern goshawk in undisturbed and timber harvest areas on the Targhee National Forest, Greater Yellowstone Ecosystem. Pocatello, ID: Idaho State University. 164 p. M.S. thesis.

Patriquin, K.J.; Barclay, R.M.R. 2003. Foraging by bats in cleared, thinned and unharvested boreal forest. Journal of Applied Ecology. 40: 646–657.

Payer, R.D.; Harrison, D.J. 1999. Effects of forest structure on spatial distribution of American marten. Tech. Bull. 787. Research Triangle Park, NC: National Council of the Paper Industry for Air and Stream Improvement. 37 p.

Pearl, C.A.; Adams, M.J.; Wente, W.H. 2007. Characteristics of Columbia spotted frog (*Rana luteiventris*) oviposition sites in northeastern Oregon, USA. Western North American Naturalist. 67: 86–91.

Pearl, C.A.; Bull, E.L.; Green, D.E.; Bowerman, J.; Adams, M.J.; Hyatt, A.; Wente, W.H. 2007. Occurrence of the amphibian pathogen *Batrachochytrium dendrobatidis* in the Pacific Northwest. Journal of Herpetology. 41: 145–149.

Pedrini, P.; Sergio, F. 2002. Regional conservation priorities for a large predator: golden eagles (*Aquila chrysaetos*) in the alpine range. Biological Conservation. 103: 163–172.

Perry, D.A.; Hessburg, P.E.; Skinner, C.N.; Spies, T.A.; Stephens, S.L.; Henry Taylor, A.; Franklin, J.F.; McComb, B.; Riegel, G. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and northern California. Forest Ecology and Management. 262: 703–717.

Peters, D.D. 1990. Wetlands and deep water habitats in the state of Washington. Olympia, WA: U.S. National Wetlands Inventory, Fish and Wildlife Service. 6 p.

Peterson, B.; Gauthier, G. 1985. Nest site use by cavity-nesting birds of Cariboo Parkland, British Columbia. Wilson Bulletin. 97: 319–331.

Petersen, K.L.; Best, L.B. 1987. Effects of prescribed burning on nongame birds in a sagebrush community. Wildlife Society Bulletin. 15: 317–329.

Peterson, K.L.; Best, L.B. 1999. Design and duration of perturbation experiments: implications for data interpretation. Studies in Avian Biology. 19: 230–236.

Phillips, R.L.; Beske, A.E. 1990. Distribution and abundance of golden eagles and other raptors in Campbell and Converse Counties, Wyoming. Tech. Rep. 27. Washington, DC: U.S. Fish and Wildlife Service.

Phillips, R.L.; McEneaney, T.P.; Beske, A.E. 1984. Population densities of breeding golden eagles in Wyoming. Wildlife Society Bulletin. 12: 269–273.

Phillips, R.L.; Wheeler, A.H.; Lockhart, J.M.; McEneaney, T.P.; Forrester, N.C. 1990. Nesting ecology of golden eagles and other raptors in southeastern Montana and northern Wyoming. Tech. Rep. 26. Washington, DC: U.S. Fish and Wildlife Service.

Picman, J.; Milks, M.L.; Leptich, M. 1993. Patterns of predation on passerine nests in marshes: effects of water depth and distance from edge. Auk. 110: 89–94.

Pierson, E.D.; Rainey, W.E. 1998. Distribution, status, and management of Townsend's big-eared bat (*Corynorhinus townsendii*) in California. Technical Report 96-7. Davis, CA: Birds and Mammals Conservation Program, California Department of Fish and Game.

Pilliod, D.S.; Peterson, C.R. 2001. Local and landscape effects of introduced trout on amphibians in historically fishless watersheds. Ecosystems. 4: 322–333.

Pilliod, D.S.; Peterson, C.R.; Ritson, P.I. 2002. Seasonal migration of Columbia spotted frogs (*Rana luteiventris*) among complementary resources in a high mountain basin. Canadian Journal of Zoology. 80: 1849–1862.

Piper, S.D. 1996. Terrestrial amphibian and passerine bird abundance in riparian habitats of eastern Washington Cascade Forests. Pullman, WA: Washington State University. 131 p. M.S. thesis.

Pitocchelli, J. 1995. MacGillivray's warbler (*Oporornis tolmiei*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 159. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union.

Potvin, F.; Belanger, L.; Lowell, K. 2000. Marten habitat selection in a clearcut boreal landscape. Conservation Biology. 14: 844–857.

Prescott, D.R.C. 1997. Avian communities and NAWMP habitat priorities in the northern prairie biome of Alberta. NAWMP-029. St. Albert, AB: Land Stewardship Centre of Canada. 41 p.

Prescott, D.R.C.; Murphy, A.J.; Ewaschuk, E. 1995. An avian community approach to determining biodiversity values of NAWMP habitats in the aspen parkland of Alberta. NAWMP-012. Edmonton, AB: Alberta NAWMP Centre. 58 p.

Rabe, M.J.; Morrell, T.E.; Green, H.; DeVos, J.C., Jr.; Miller, C.R. 1998. Characteristics of ponderosa pine snag roosts used by reproductive bats in northern Arizona. Journal of Wildlife Management. 62: 612–621.

Radford, J.Q.; Bennett, A.F.; Cheers, J.G. 2005. Landscape-level thresholds of habitat cover for woodland-dependent birds. Biological Conservation. 124: 317–337.

Raesly, R.L.; Gates, J.E. 1987. Winter habitat selection by north temperate cave bats. American Midland Naturalist. 118: 15–31.

Rancourt, S.J.; Rule, M.I.; O'Connell, M.A. 2008. Maternity roost site selection of big brown bats in ponderosa pine forests of the Channeled Scablands of northeastern Washington state, USA. Forest Ecology and Management. 248: 183–192.

Randall-Parker, T.; Miller, R. 2002. Effects of prescribed fire in ponderosa pine on key wildlife habitat components: preliminary results and a method for monitoring. Gen. Tech. Rep. PSW-GTR-181. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 823–834.

Raphael, M.G. 1981. Interspecific differences in nesting habitat of sympatric woodpeckers and nuthatches. In: Capen, D.E., ed. The use of multivariate statistics in studies of wildlife habitat. Gen. Tech. Rep. RM-87. Albany, CA: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experimental Station: 142–151.

Raphael, M.G.; Marcot, B.G. 1994. Key questions and issues-species and ecosystem viability. Journal of Forestry. 92: 45–47.

Raphael, M.G.; Morrison, M.L.; Yoder-Williams, M.P. 1987. Breeding bird populations during twenty-five years of postfire succession in the Sierra Nevada. Condor. 89: 614–626.

Raphael, M.G.; White, M. 1984. Use of snags by cavity-nesting birds in the Sierra Nevada. Wildlife Monograph No. 86: 1–66.

Raphael, M.G.; Wisdom, M.J.; Rowland, M.M.; Holthausen, R.S.; Wales, B.C.; Marcot, B.G.; Rich, T.D. 2001. Status and trends of habitats of terrestrial vertebrates in relation to land management in the interior Columbia river basin. Forest Ecology and Management. 153: 63–88.

Ratcliffe, D.A. 1993. The peregrine falcon. 2nd ed. London, England: T and A D Poyser. 454 p.

Rawinski, T.J.; Malecki, R.A. 1984. Ecological relationships among purple loosestrife, cattail, and wildlife at the Montezuma National Wildlife Refuge. New York Fish and Game Journal. 31: 81–87.

Reaser, J.K. 2000. Demographic analysis of the Columbia spotted frog (*Rana luteiventris*): case study in spatio-temporal variation. Canadian Journal of Zoology. 78: 1158–1167.

Reaser, J.K.; Pilliod, D.S. 2005. *Rana luteiventris* Thompson 1913, Columbia spotted frog. In: Lannoo, M., ed. Amphibian declines: the conservation status of United States species. Berkeley, CA: University of California Press: 559–563.

Reichel, J.D.; Beckstrom, S.G. 1994. Northern bog lemming survey. Helena, MO: Montana Natural Heritage Program. 87 p.

Reinkensmeyer, D.P.; Miller, R.F.; Anthony, R.G.; Marr, V.E. 2007. Avian community structure along a mountain big sagebrush successional gradient. Journal of Wildlife Management. 71: 1057–1066.

Renwald, J.D. 1977. Effect of fire on lark sparrow nesting densities. Journal of Range Management. 30: 283–285.

Reynolds, R.T.; Graham, R.T.; Reiser, M.H.; Bassett, R.L.; Kennedy, P.L.; Boyce, D.A.; Goodwin, G.; Smith, R.; Fisher, E.L. 1992. Management recommendations for the northern goshawk in the southwestern United States. Gen. Tech. Rep. GTR-RM-217. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 90 p.

Reynolds, T.D.; Rich, T.D.; Stephens, D.A. 1999. Sage thrasher (*Oreoscoptes montanus*). Number 463. In: Poole, A.; Gill, F., eds. The birds of North America. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union.

Reynolds, T.D.; Trost, C.H. 1980. The response of native vertebrate populations to crested wheatgrass planting and grazing by sheep. Journal of Range Management. 33: 122–125.

Reynolds, T.D.; Trost, C.H. 1981. Grazing, crested wheatgrass and bird populations in southeastern Idaho. Northwest Science. 55: 225–234.

Richardson, C.T.; Miller, C.K. 1997. Recommendations for protecting raptors from human disturbance: a review. Wildlife Society Bulletin. 25: 634–638.

Richter, W. 1984. Nestling survival and growth in the yellow-headed blackbird, *Xanthocephalus xanthocephalus*. Ecology. 65: 597–608.

Rieman, B.; Peterson, J.T.; Clayton, J; Howell, P.; Thurow, R.; Thompson, W.; Lee, D. 2001. Evaluation of potential effects of federal land management alternatives on trends of salmonids and their habitats in the interior Columbia River basin. Forest Ecology and Management. 153: 43–62.

Robb, J.R.; Bookhout, T.A. 1995. Factors influencing wood duck use of natural cavities. Journal of Wildlife Management. 59: 372–383.

Roberge, J.; Angelstam, P. 2004. Usefulness of the umbrella species concept as a conservation tool. Conservation Biology. 18: 76–85.

Roberts, J.; Chick, A.; Oswald, L.; Thompson, P. 1995. Effect of carp, *Cyprinus carpio* L., an exotic benthivorous fish, on aquatic plants and water quality in experimental ponds. Marine and Freshwater Research. 46: 1171–1180.

Rodrick, E.A. 1986. Survey of historic habitats of the western gray squirrel (*Sciurus griseus*) in the southern Puget Trough and Klickitat County, Washington. Seattle, WA: University of Washington. 41 p. M.S. thesis.

Rogers J.A., Jr.; Smith, H.T. 1995. Set-back distances to protect nesting bird colonies from human disturbance in Florida. Conservation Biology. 9: 89–99.

Rompre, G.; Boucher, Y.; Belanger, L.; Cote, S.; Robinson, W.D. 2010. Conserving biodiverisity in managed forest landscapes: the use of critical thresholds for habitat. Forestry Chronicle. 86(5): 589–596.

Rose, C.; Marcot, B.; Mellen, T.K.; Ohmann, J.L.; Waddell, K.; Lindley, D.; Schreiber, B. 2001. Decaying wood in Pacific Northwest forests: concepts and tools for habitat management. In: Johnson, D.; O'Neal, T., eds. Wildlife-habitat relationships in Oregon and Washington. Corvallis, OR: Oregon State Unversity Press: 580–623.

Rosenstock, S.S. 1996. Habitat relationships of breeding birds in northern Arizona ponderosa pine and pine-oak forests. Tech. Rep. 23. Arizona Game and Fish Department.

Ross, A. 1967. Ecological aspects of the food habits of insectivorous bats. Western Foundation of Vertebrate Zoology. 1: 205–264.

Ross, D.; Reaser, J.K.; Kleeman, P.; Drake, D.L. 1999. *Rana luteiventris* (Columbia spotted frog) mortality and site fidelity. Herpetological Review. 30: 163.

Rowland, M.M.; Wisdom, M.J.; Johnson, D.H.; Wales, B.C.; Copeland, J.P.; Edelmann, F.B. 2003. Evaluation of landscape models for wolverines in the interior Northwest, United States of America. Journal of Mammalogy. 84(1): 92–105.

Ruediger, B.; Claar, J.; Gniadek, S. [et al.]. 2000. Canada lynx conservation assessment and strategy. Publ. R1-00-53. Missoula, MT: U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Fish and Wildlife Service, Bureau of Land Management, National Park Service. 142 p.

Ruggiero, L.F.; Aubry, K.B.; Buskirk, S.W.; Lyon, L.J.; Zielinski, W.J. 1994. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the Western United States. Gen. Tech. Rep. GTR-RM-254. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 184 p.

Ruggerio, L.F.; McKelvey, K.S.; Aubry, K.B.; Copeland, J.P.; Pletscher, D.H.; Hornocker, M.G. 2007. Wolverine conservation and management. Journal of Wildlife Management. 71(7): 2145–2146.

Ruggiero, L.F.; Pearson, D.E.; Henry, S.E. 1998. Characteristics of American marten den sites in Wyoming. Journal of Wildlife Management. 62: 663–673.

Russell, R.E.; Saab, V.A.; Dudley, J.G. 2007. Habitat-suitability models for cavity-nesting birds in a postfire landscape. Journal of Wildlife Management. 71(8): 2600–2611.

Ryan, L.A.; Carey, A.B. 1995. Biology and management of the western gray squirrel and Oregon white oak woodlands: with emphasis on the Puget Trough. Gen. Tech. Rep. PNW-GTR-348. Portland, OR: U.S.Department of Agricullture, Forest Service, Pacific Northwest Research Station. 36 p.

Saab, V.A.; Bock, C.E.; Rich, T.D.G.; Dobkin, D.S. 1995. Livestock grazing effects in western North America. In: Martin, T.E.; Finch, D.M., eds. Ecology and management of Neotropical migratory birds: a synthesis and review of critical issues. New York, NY: Oxford University Press: 311–353.

Saab, V.A.; Dudley, J. 1998. Responses of cavity-nesting birds to stand-replacement fire and salvage logging in ponderosa pine/Douglas-fir forests of southwestern Idaho. Res. Pap. RMRS-RP-11. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 17 p.

Saab, V.A.; Powell, H.D.W.; Kotliar, N.B.; Newlon, K.R. 2005. Variation in fire regimes of the Rocky Mountains: implications for avian communities and fire management. Studies in Avian Biology. 30: 76–96.

Saab, V.A.; Russell, R.E.; Dudley, J.G. 2009. Nest-site selection by cavity-nesting birds in relation to postfire salvage logging. Forest Ecology and Management. 257: 151–159.

Saab, V.A.; Vierling, K.T. 2001. Reproductive success of Lewis's woodpecker in burned pine and cottonwood riparian forests. Condor. 103: 491–501.

Salafsky, S.R.; Reynolds, R.T. 2005. Patterns of temporal variation in goshawk reproduction and prey resources. Journal of Raptor Research. 39(3): 237–246.

Sallabanks, R. 1995. Avian biodiversity and bird-habitat relationships in conifer forest of the inland Northwest: an ecosystem management approach. Annual rep. to Boise Cascade Corp. Meridian, ID: National Fish and Wildlife Foundation, National Council for Air and Stream Improvement and U.S. Department of Agriculture, Forest Service, Sustainable Ecosystems Institute.

Sallabanks, R.; Marcot, B.G.; Riggs, R.A.; Mehl, C.A.; Arnett, E.B. 2001. Wildlife of eastside (interior) forests and woodlands. In: Johnson, D.H.; O'Neil, T.A., eds. Wildlife-habitat relationships in Oregon and Washington. Corvallis, OR: Oregon State University Press: 213–238.

Sample, B.E. 1991. Effects of Dimilin on food of the endangered Virginia big-eared bat. Morgantown, WV: West Virginia University. 201 p. Ph.D. dissertation.

Samson, F.B. 2002. Population viability analysis, management, and conservation planning at large scales. In: Beissinger, S.R.; McCullough, D.R., eds. Population viability analysis. Chicago, Il: University of Chicago Press: 425–441.

Samson, F.B.; Knopf, F.L.; McCarthy, C.W. [et al.]. 2003. Planning for population viability on Northern Great Plains national grasslands. Wildlife Society Bulletin. 31(4): 986–999.

Samuels, I.A.; Gardali, T.; Humple, D.L.; Geupel, G.R. 2005. Winter site fidelity and body condition of three riparian songbird species following a fire. Western North American Naturalist. 65: 45–52.

Sarell, M.J. 2004. Tiger salamander *Ambystoma tigrinum*. I: accounts and measures for managing identified wildlife—accounts V. Victoria, BC: British Columbia Ministry of Water, Land and Air Protection. 9 p.

Saunders, A.A. 1913. A study of the nesting of the marsh hawk. Condor. 15: 99–104.

Savard, J.-P.L.; Boyd, W.S.; Smith, G.E.J. 1994. Waterfowl-wetland relationships in the aspen parkland of British Columbia: comparison of analytical methods. Hydrobiologia. 279/280: 309–325.

Schirato, G. 1994. Population and distribution of harlequin ducks in Washington state. In: Galiano, B.C., ed. Proceeding of the second harlequin duck symposium. Harlequin Duck Working Group: 18.

Schmalzried, J.T. 1976. Nesting and food habits of the golden eagle on the Laramie Plains. Laramie, WY: University of Wyoming. 86 p. M.S. thesis.

Schmid, W.D. 1972. Snowmobile activity, subnivean microclimate and winter mortality of small mammals. Proceedings of the American Institute of Biological Scientists. Bulletin of the Ecological Society of American. 53(2): 37.

Schommer, T.J.; Woolever, M.M. 2008. A review of disease related conflicts between domestic sheep and goats and bighorn sheep. Gen. Tech. Rep. RMRS-GTR-209. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 16.

Schulz, T.T.; Leininger, W.C. 1991. Non-game wildlife communities in grazed and ungrazed Montana riparian sites. Great Basin Naturalist. 51: 286–292.

Schwab, F.E.; Simon, N.P.; Sinclair, A.R. 2006. Bird-vegetation relationships in Southeastern British Columbia. Journal of Wildlife Management. 70(1): 189–197.

Scott, T.A. 1985. Human impacts on the golden eagle population of San Diego County. San Diego, CA: San Diego State University. 202 p. M.S. thesis.

Scott, V.E.; Evans, K.E.; Patton, D.R.; Stone, C.P. 1977. Cavity-nesting birds of North American forests. Agric. Handb. 511. Washington, DC: U.S. Department of Agriculture. 112 p.

Sealy, S.G. 1967. Notes on the breeding biology of the marsh hawk in Alberta and Saskatchewan. Blue Jay. 25: 63–69.

Sedivec, K.K. 1994. Grazing treatment effects on and habitat use of upland nesting birds on native rangeland. Fargo, ND: North Dakota State University. 124 p. Ph.D. dissertation.

Semlitsch, R.D. 1998. Biological delineation of terrestrial buffer zones for pond-breeding salamanders. Conservation Biology. 12: 1113−1119.

Sergio, F.; Pedrini, P.; Rizzolli, E.; Marchesi, L. 2006. Adaptive range selection by golden eagles in a changing landscape: a multiple modelling approach. Biological Conservation. 133: 32–41.

Serrentino, P. 1992. Northern harrier, *Circus cyaneus*. In: Schneider, K.J.; Pence, D.M., eds. Migratory nongame birds of management concern in the Northeast. Newton Corner, MA: U.S. Fish and Wildlife Service: 89–117.

Shaffer, J.A.; Goldade, C.M.; Dinkins, M.F.; Johnson, D.H.; Igl, L.D.; Euliss, B.R. 2003. Brown-headed cowbirds in grasslands: their habitats, hosts, and response to management. Prairie Naturalist. 35: 146–186.

Shaffer, M.L.; Stein, B.A. 2000. Safeguarding our precious heritage. In: Stein, B.A.; Kutner, L.S.; Adams, J.S., eds. Precious heritage: the status of biodiversity in the United States. New York: Oxford University Press: 301–321.

Simmons, R.; Smith, P.C. 1985. Do northern harriers (*Circus cyaneus*) choose nest sites adaptively? Canadian Journal of Zoology. 63: 494–498.

Simon, N.P.P.; Schwab, F.E.; Otto, R.D. 2002. Songbird abundance in clear-cut and burned stands: a comparison of natural disturbance and forest management. Canadian Journal of Forest Research. 32: 1343–1350.

Singer, F.J.; Moses, M.E.; Bellew, S.; Sloan, W. 2000. Correlates to colonizations of new patches by translocated populations of bighorn sheep. Restoration Ecology. 8(4S): 66–74.

Singleton, P.H.; Gaines, W.L.; Lehmkuhl, J.F. 2002. Landscape permeability for large carnivores in Washington: a geographic information system weighted-distance and least-cost corridor assessment. Gen. Tech. Rep. PNW-GTR-549. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 13 p.

Singleton, P.H.; Lehmkuhl J.F. 1998. Wildlife and roadway interactions: a bibliography and review of roadway and wildlife interactions. Wenatchee, WA: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 162 p.

Singleton, P.H.; Lehmkuhl, J.F. 2000. I-90 Snoqualmie Pass wildlife habitat linkage assessment. Publ. No. WA-RD 489.1. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 106 p.

Singleton, P.H.; Lehmkuhl, J.F.; Gaines, W.L.; Graham, S.A. 2010. Barred owl space use and habitat selection in the eastern Cascades, Washington. Journal of Wildlife Management. 74(2): 285–294.

Skagen, S.K.; Knight, R.L.; Orians, G.H. 1991. Human disturbance of an avian scavenging guild. Ecological Applications. 1(2): 215–225.

Skagen S.K.; Melcher, C.P.; Muths, E. 2001. The interplay of habitat change, human disturbance and species interactions in a waterbird colony. American Midland Naturalist. 145: 18–28.

Slauson, K.M.; Zielinski, W.J. 2009. Characteristics of summer and fall diurnal resting habitat used by American martens in coastal northwestern California. Northwest Science. 83: 35–45.

Slauson, K.M.; Zielinski, W.J.; Hayes, J.P. 2007. Habitat selection by American martens in coastal California. Journal of Wildlife Management. 71: 458–468.

Slough, B.B. 1999. Characteristics of Canada lynx, *lynx canadensis*, maternal dens and denning habitat. Canadian Field-Naturalist. 113(4): 605–608.

Smith, D.G.; Murphy, J.R. 1979. Breeding responses of raptors to jackrabbit density in the eastern Great Basin Desert of Utah. Journal of Raptor Research. 13: 1–14.

Smith, D.G.; Murphy, J.R. 1982. Spatial relationships of nesting golden eagles in central Utah. Journal of Raptor Research. 16: 127–132.

Smith, G.W.; Nydegger, N.C. 1985. A spotlight, line-transect method for surveying jack rabbits. Journal of Wildlife Management. 49: 699–702.

Smith, M.R.; Mattocks, P.W., Jr.; Cassidy, K.M. 1997. Breeding birds of Washington state. Volume 4. In: Cassidy, K.M.; Grue, C.E.; Smith, M.R.; Dvornich, K.M., eds. Washington state gap analysis; final report. Audubon Society Publications in Zoology 1. Seattle, WA: Seattle Audubon Society. 146 p.

Smith, T.G. 2002. Small mammal relationships with downed wood and antelope bitterbrush in ponderosa pine forest of central Oregon. Corvallis, OR: Oregon State University. 146 p. M.S. thesis.

Smith, T.S.; Flinders, J.T.; Winn, D.S. 1991. A habitat evaluation procedure for Rocky Mountain bighorn sheep in the intermountain West. Great Basin Naturalist. 51: 205–225.

Smucker, K.M.; Hutto, R.L.; Steele, B.M. 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecological Applications. 15(5): 1535–1549.

Snedecor, G.W.; Cochran, W.G. 1989. Statistical methods. 6th ed. Ames, IA: Iowa State University Press. 593 p.

Snyder, J.E.; Bissonette, J.A. 1987. Marten use of clear-cuttings and residual forest stands in western Newfoundland. Canadian Journal of Zoology. 65: 169–174.

Society for Ecological Restoration. 1993. Mission statement, Society for Ecological Restoration. Restoration Ecology. 1: 206–207.

Soulliere, G. 1988. Density of suitable wood duck nest cavities in a northern hardwood forest. Journal of Wildlife Management. 52: 86–89.

Sousa, P.J.; Farmer, A.H. 1983. Habitat suitability index models: wood duck. FWS/OBS-82/10.43. Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service. 38 p.

Soutiere, E.C. 1979. Effects of timber harvesting on marten in Maine. Journal of Wildlife Management. 43: 850–860.

Spencer, W.D.; Barrett, R.H.; Zielinski, W.J. 1983. Marten habitat preferences in the northern Sierra Nevada. Journal of Wildlife Management. 47: 1181–1186.

Spring, L.W. 1965. Climbing and pecking adaptations in some North American woodpeckers. Condor. 67: 457–488.

Squires, J.R.; Decesare, N.J.; Kolbe, J.A.; Ruggiero, L.F. 2008. Hierarchical den selection of Canada lynx in western Montana. Journal of Wildlife Management. 72(7): 1497–1506.

Squires, J.R.; Laurion, T. 2000. Lynx home range and movements in Montana and Wyoming: preliminary results. In: Ruggerio, L.F.; Aubry, K.B.; Buskirk, S.W.; Koehler, G.M.; McKelvey, K.S.; Squires, J.R. Ecology and conservation of lynx in the United States. Gen. Tech. Rep. RMRS-GTR-30WWW. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 337–349.

Stalmaster, M.V. 1987. The bald eagle. New York, NY: Universe Books.

Stalmaster, M.V.; Newman, J.R. 1978. Behavioral responses of wintering bald eagles to human activity. Journal of Wildlife Management. 42: 506–513.

Steen, D.A.; Gibbs, J.P.; Timmermans, S.T.A. 2006. Assessing the sensitivity of wetland bird communities to hydrologic change in the eastern Great Lakes region. Wetlands. 26: 605–611.

Steenhof, K.; Kochert, M.N.; Doremus, J.H. 1983. Nesting of subadult golden eagles in southwestern Idaho. Auk. 100: 743–747.

Steenhof, K.; Kochert, M.N.; McDonald, T.L. 1997. Interactive effects of prey and weather on golden eagle reproduction. Journal of Animal Ecology. 66: 350–362.

Steenhof, K.; Kochert, M.N.; Roppe, J.A. 1993. Nesting by raptors and ravens on electrical transmission line towers. Journal of Wildlife Management. 57: 271–281

Steidl, R.J.; Kozie, K.D.; Dodge, G.J.; Pehovski, T.; Hogan, E.R. 1993. Effects of human activity on breeding behavior of golden eagles in Wrangell-St. Elias National Park and Preserve: a preliminary assessment. WRST Reserve Resource Management Report 93-3. Copper Center, AK: National Park Service, Wrangell-St. Elias National Park.

Stewart, R.E. 1975. Breeding birds of North Dakota. Fargo, ND: Tri-College Center for Environmental Studies. 295 p.

Stewart, R.E.; Kantrud, H.A. 1965. Ecological studies of waterfowl populations in the prairie potholes of North Dakota. 1965 Progress Report. [Place of publication unknown]: U.S. Fish and Wildlife Service, Bureau of Sport Fisheries and Wildlife. 14 p.

Stinson, D.W.; Watson, J.W.; McAllister, K.R. 2007. Washington state status report for the bald eagle. Olympia, WA: Washington Department of Fish and Wildlife. 86 p.

Stoddard, M.A.; Hayes, J.P. 2005. The influence of forest management on headwater stream amphibians at multiple spatial scales. Ecological Applications. 15(3): 811–823.

Studier, E.H.; O'Farrell, M.J. 1980. Physiological ecology of Myotis. In: Wilson, D.E.; Gardner, A.L. eds. Proceedings of the fifth international bat research conference. Lubbock, TX: Texas Press: 415–423.

Suding, K.N.; Gross, K.L.; Houseman, G. 2004. Alternative states and positive feedbacks in restoration ecology. Trends in Ecology and Evolution. 19: 46–53.

Sullivan, S.L.; Pyle, W.H.; Herman, S.G. 1986. Cassin's finch nesting in big sagebrush. Condor. 88: 378–379.

Suring, L.H.; Gaines, W.L.; Wales, B.C.; Mellen-McLean, K.; Begley, J.S.; Mohoric, S. 2011. Maintaining populations of terrestrial wildlife through land management planning: a case study. Journal of Wildlife Management. 75(4): 945–958.

Svancara, L.K.; Brannon, R.; Scott, J.M.; Groves, C.R.; Noss, R.F.; Pressey, R.L. 2005. Policy-driven versus evidence-based conservation: a review of political targets and biological needs. BioScience. 55(11): 989–995.

Swanson, F.J.; Jones, J.A.; Wallin, D.O.; Cissel, J.H. 1994. Natural variability—implications for ecosystem management. In: Mensen, M.E.; Bourgeron, P.S., eds. Volume II: Ecosystem mangement: principles and applications. Gen. Tech. Rep. PNW-GTR-318. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 80–94.

Szaro, R.C. 1976. Population densities, habitat selection, and foliage use by the birds of selected ponderosa pine areas in the Beaver Creek Watershed, Arizona. Flagstaff, AZ: Northern Arizona University. 264 p. Ph.D. dissertation.

Taylor, D.M. 1986. Effects of cattle grazing on passerine bird nesting in riparian habitat. Journal of Range Management. 39: 254–258.

Tazik, D.J. 1991. Effects of army training activities on bird communities at the Pinon Canyon maneuver site, Colorado. USACERL Tech. Rep. N-91/31. Champaign, IL: U.S. Army Corps of Engineers, Construction Engineering Research Laboratory.

Tear, T.H.; Kareiva, P.; Angermeier, P.L. [et al.]. 2005. How much is enough? The recurrent problem of setting measurable objectives in conservation. BioScience. 55(10): 835–849.

Terres, J.K. 1980. The Audubon Society encyclopedia of North American birds. New York, NY: Alfred A. Knopf.

Tewksbury, J.J.; Martin, T.E.; Hejl, S.J.; Redman, T.S.; Wheeler, F.J. 1999. Cowbirds in a western valley: effects of landscape structure, vegetation, and host density. Studies in Avian Biology. 18: 23–33.

Thelander, C.G. 1974. Nesting territory utilization by golden eagles (*Aquila chrysaetos*) in California during 1974. Special Wildlife Investigations. Sacramento, CA: California Department of Fish and Game.

Thompson, I.D. 1994. Marten populations in uncut and logged boreal forests in Ontario. Journal of Wildlife Management. 58: 272–280.

Thompson, I.D.; Davidson, I.J.; O'Donnell, S.; Brazeau, F. 1989. Use of track transects to measure the relative occurrence of some boreal mammals in uncut forest and regeneration stands. Canadian Journal of Zoology. 67: 1816–1823.

Tibbels, A.E.; Kurta, A. 2003. Bat activity is low in thinned and unthinned stands of red pine. Canadian Journal of Forest Research. 33: 2436–2442.

Timmermans, S.T.A.; Badzinski, S.S.; Ingram, J.W. 2008. Associations between breeding marsh bird abundances and great lakes hydrology. Journal of Great Lakes Research. 34: 351–364.

Titus, J.R.; Vandruff, L.W. 1981. Response of the common loon (*Gavia immer*) to recreational pressure in the Boundary Waters Canoe Area northeastern Minnesota. Wildlife Monograph No. 79.

Tobalske, B.W. 1997. Lewis woodpecker (*Melanerpes lewis*). No. 284. In: Poole, A.; Gill, F., eds. The birds of North America. Philadelphia, PA: The Birds of North America, Inc.

Toland, B.R. 1985. Nest site selection, productivity, and food habits of northern harriers in southwest Missouri. Natural Areas Journal. 5: 22–27

Touchan, R.; Allen, C.D.; Swetnam, T.W. 1996. Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, northwestern New Mexico. In: Allen, C.D., tech. ed. Fire effects in Southwestern forests: proceedings of the second La Mesa fire symposium. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 33–46.

Townsley, J.; Gaines, B.; Hadfield, J.; Harrod, R.; Mehmel, C.; Leyda, E. 2004. Forest health assessment: Okanogan and Wenatchee National Forests. Wenatchee, WA: U.S. Department of Agriculture, Forest Service, Okanogan and Wenatchee National Forests. 105 p.

Tozer, D.C. 2002. Point count efficiency and nesting success in marsh-nesting birds. Peterborough, ON: Trent University. 162 p. M.S. thesis.

Tremblay, J.A.; Ibarzabal, J.; Dussault, C.; Savard, J.-P.L. 2009. Habitat requirements of breeding black-backed woodpeckers (*Picoides arcticus*) in managed, unburned boreal forest. Avian Conservation and Ecology. 4(1): 2. URL: http://www.ace-eco.org/vol4/iss1/art2/.

Trombulak, S.C.; Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology. 14: 18–30.

Tuck, L.M. 1972. The snipes: a study of the genus *Capella*. Canadian Wildlife Service Monograph 5.

Turner, F.B. 1960. Population structure and dynamics of the western spotted frog, *Rana pretiosa* Baird and Girard, in Yellowstone Park, Wyoming. Ecological Monographs. 30: 251–278.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2000. Land type associations of north central Washington. Wenatchee, WA: Okanogan and Wenatchee National Forests. 98 p.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2006. Conducting species viability assessments for forest plan revision in Region 6. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region.

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2004. The PLANTS database, Version 3.5. Baton Rouge, LA: National Resources Conservation Service. http://plants.usda.gov. (June 15, 2010).

U.S. Department of the Interior, Bureau of Land Management [USDI BLM]. 1979. Snake River birds of prey special research report to the Secretary of the Interior. Boise, ID.

U.S. Department of the Interior, Fish and Wildlife Service [USDI FWS]. 2002. Birds of conservation concern. Arlington, VI: Division of Migratory Bird Management. 99 p. http://migratorybirds.fws.gov/reports/bcc2002.pdf. (June 15, 2010).

U.S. Department of the Interior, Fish and Wildlife Service [USDI FWS]. 2005. Recovery outline for the Canada lynx. Missoula, MO.

U.S. Department of the Interior, Fish and Wildlife Service [USDI FWS]. 2011. Final recovery plan for the northern spotted owl. Pacific Northwest Region, Portland, OR.

U.S. Department of the Interior, Geological Services [USGS]. 2004. Wood duck (*Aix sponsa*): requirements and limiting factors. www.npwrc.usgs.gov/resource/1999/woodduck/wdreqlim. (June 15, 2010).

Vander Haegen, W.M. 2007. Fragmentation by agriculture influences reproductive success of birds in a shrubsteppe landscape. Ecological Applications. 17: 934–947.

Vander Haegen, W.M.; Dobler, F.C.; Pierce, D.J. 2000. Shrubsteppe bird response to habitat and landscape variables in eastern Washington, U.S.A. Conservation Biology. 14: 1145–1160.

Vander Haegen, W.M.; Schroeder, M.A.; DeGraaf, R.M. 2002. Predation on real and artificial nests in shrubsteppe landscapes fragmented by agriculture. Condor. 101: 496–506.

Verner, J.; Engelsen, G.H. 1970. Territories, multiple nest building, and polygyny in the long-billed marsh wren. Auk. 87: 557–567.

Vesely, D.G.; McComb, W.C. 2002. Salamander abundance and amphibian species richness in riparian buffer strips in the Oregon Coast Range. Forest Science. 48(2): 291–297.

Villard, P.; Benniger, C.W. 1993. Foraging behavior of male black-backed and hairy woodpeckers in a forest burn. Journal of Field Ornithology. 64(1): 71–76.

Von Kienast, J.A. 2003. Winter habitat selection and food habits of lynx on the Okanogan Plateau, Washington. Seattle, WA: University of Washington. 64 p. M.S. thesis.

Vos, C.C.; Chardon, J.P. 1998. Effects of habitat fragmentation and road density on the distribution pattern of the moor frog *Rana arvalis*. Journal of Applied Ecology. 35: 44–56.

Wackenhut, M.C. 1990. Bat species over-wintering in lava-tube caves in Lincoln, Gooding, Blaine, Bingham, and Butte Counties, Idaho. Pocatello, ID: Idaho State University. M.S. thesis.

Wade, P.R. 2002. Bayesian population viability analysis. In: Beissinger, S.R.; McCullough, D.R., eds. Population viability analysis. Chicago, Il: University of Chicago Press: 213–238.

Walcheck, K.C. 1970. Nesting bird ecology of four plant communities in the Missouri River Breaks, Montana. Wilson Bulletin. 82: 370–382.

Waldien, D.L.; Hayes, J.P. 2001. Activity areas of female long-eared myotis in coniferous forests in western Oregon. Northwest Science. 75: 307–314.

Wales, B.C. 2001. The management of insects, disease, fire, and grazing and implications for terrestrial vertebrates using riparian habitats in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 119–127.

Wallen, R.L.; Groves, C.R. 1989. Distribution, breeding biology, and nesting habitat of harlequin ducks in northern Idaho. Boise ID: Idaho Department of Fish and Game. 39 p.

Walston, L.J.; Mullin, S.J. 2007. Responses of a pond-breeding amphibian community to the experimental removal of predatory fish. American Midland Naturalist. 157: 63–73.

Washington Department of Fish and Wildlife [WDFW]. 1993. Status of the western gray squirrel (*Sciurus griseus*) in Washington. Olympia, WA. 33 p.

Washington Department of Fish and Wildlife [WDFW]. 2006. Fish and wildlife geographic information system digital data documentation. Olympia, WA.

Washington Department of Fish and Wildlife [WDFW]. 2005. Warmwater fish of Washington. Report FM93-9. Olympia, WA.

Washington, Department of Fish and Wildlife [WDFW]. 2008. Bighorn sheep Management Plan. Olympia, WA.

Washington Wildlife Habitat Connectivity Working Group [WWHCWG]. 2010. Washington Connected Landscape Project: statewide analysis. Washington Departments of Fish and Wildlife, and Transportation, Olympia, WA. http://www.waconnected.org. (January 15, 2011).

Waters, T.F. 1995. Sediment in streams: sources, biological effects and control. American Fisheries Society Monograph 7.

Watson, D. 1977. The hen harrier. Hertfordshire, England: T.E.A.D. Poyser, Limited, Berkhamsted. 307 p.

Watson, J. 1997. The golden eagle. First ed. London, United Kingdom: T. and A. D. Poyser.

Watson, J.; Freidenberger, D.; Paull, D. 2001. An assessment of the focal species approach for conserving birds in variegated landscapes in southeastern Australia. Conservation Biology. 15(5): 1364–1373.

Watson, J.W. 1993. Responses of nesting bald eagles to helicopter surveys. Wildlife Society Bulletin. 21: 171–178.

Watson, J.W.; Pierce, D.J. 1998. Ecology of bald eagles in western Washington with an emphasis on the effects of human activity. Final report, Washington Department of Fish and Wildlife, Olympia, WA.

Watson, J.W.; Rodrick, E.A. 2000. Bald eagle. In: Management recommendations for Washington's priority habitats and species. Olympia, WA: Washington Department of Fish and Wildlife: 9–1 to 9–15.

Webb, S.M.; Boyce, M.S. 2009. Marten fur harvests and landscape change in west-central Alberta. Journal of Wildlife Management. 73: 894–903.

Weber, W.C. 1980. A proposed list of rare and endangered bird species for British Columbia. In: Stace-Smith, R.; Johns, L.; Joslin, P., eds. Threatened and endangered species and habitats in British Columbia and the Yukon. Victoria, BC: British Columbia Ministry of the Environment Fish and Wildlife Branch: 160–182.

Webster, J.D. 1975. The fox sparrow in southwestern Yukon and adjacent areas. Condor. 77: 215–216.

Weckstein, J.D.; Kroodsma, D.E.; Faucett, R.C. 2002. Fox sparrow (*Passerella iliaca*) In: Poole, A.; Gill, F. eds. The Birds of North America, No. 715. The Birds of North America, Inc., Philadelphia, Pennsylvania, U.S.A.

Wegner, S.J. 2008. Use of surrogates to predict the stressor response of imperiled species. Conservation Biology. 22(6): 1564–1571.

Welch, B.L. 2002. Bird counts of burned versus unburned big sagebrush sites. Res. Note. RMRS-RN-16. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Welch, N.E.; MacMahon, J.A. 2005. Identifying habitat variables important to the rare Columbia spotted frog in Utah (U.S.A.): an information-theoretic approach. Conservation Biology. 19: 473–481.

Weller, T.J.; Zabel, C.J. 2001. Characteristics of fringed myotis day roosts in northern California. Journal of Wildlife Management. 65: 489–497.

Welsh, H.H.; Ollivier, L.M. 1998. Stream amphibians as indicators of ecosystem stress: a case study from California's redwoods. Ecological Applications. 8(4): 1118–1132.

Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; Swetnam, T.W. 2006. Warming and earlier spring increase Western U.S. forest wildfire activity. Science. 313: 940–943.

Wethington T.A.; Leslie, D.M., Jr.; Gregory, M.S.; Wethington, M.K. 1997. Vegetative structure and land use relative to cave selection by endangered Ozark big-eared bats (*Corynorhinus townsendii ingens*). Southwestern Naturalist. 42: 177–181.

Whitaker J.O., Jr.; Cross, S.P.; Skovlin, J.M.; Maser, C. 1983. Food habits of the spotted frog (*Rana pretiosa*) from managed sites in Grant County, Oregon. Northwest Science. 57: 147–154.

Whitaker, J.O., Jr.; Maser, C.; Keller, L.E. 1977. Food habits of bats of western Oregon. Northwest Science. 51: 46–55.

Whitfield, D.P.; Fielding, A.H.; McLeod, A.; Haworth, P.F. 2004. The effects of persecution on age of breeding and territory occupation in golden eagles in Scotland. Biological Conservation. 118: 249–259.

Whitt, M.B.; Prince, H.H.; Cox, R.R., Jr. 1999. Avian use of purple loosestrife dominated habitat relative to other vegetation types in a Lake Huron wetland complex. Wilson Bulletin. 111: 105–114.

Wiens, J.A.; Hayward, G.D.; Holthausen, R.S.; Wisdom, M.J. 2008. Using focal species and groups for conservation planning and management. BioScience. 58(3): 241–252.

Wiens, J.A.; Hayward, G.D.; Safford, H.D.; Giffen, C.M. 2012. Historical environmental variation in conservation and natural resource management. Hoboken, NJ: Wiley-Blackwell.

Wiens, J.A.; Rotenberry, J.T. 1981. Habitat associations and community structure of birds in shrubsteppe environments. Ecological Monographs. 5: 21–41.

Wilbert, C.J.; Buskirk, S.W.; Gerow, K.G. 2000. Effects of weather and snow on habitat selection by American martens. Canadian Journal of Zoology. 78: 1691–1696.

Wimberly, M.C.; Spies, T.A.; Long, C.J.; Whitlock, C. 2000. Simulating historical variability in the amount of old forests in the Oregon Coast Range. Conservation Biology. 14: 167–180.

Windsor, J. 1975. The response of peregrine falcons to aircraft and human disturbance. Canadian Wildlife Service Note 87.

Winter, M.; Johnson, D.H.; Faaborg, J. 2000. Evidence for edge effects on multiple levels in tallgrass prairie. Condor. 102: 256–266.

Wisdom, M.J.; Bate, L.J. 2008. Snag density varies with intensity of timber harvest and human access. Forest Ecology and Management. 255: 2085–2093.

Wisdom, M.J.; Holthausen, R.S.; Wales, B.C.; Hargis, C.D.; Saab, V.A.; Lee, D.C.; Hann, W.J.; Rich, T.D.; Rowland, M.M.; Murphy, W.J.; Eames, M.R. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia Basin: broad-scale trends and management implications. Gen. Tech. Rep. PNW-GTR-485. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 3 vol.

Wisdom, M.J.; Wales, B.C.; Holthausen, R.S.; Hann, W.J.; Hemstrom, M.A.; Rowland, M.M. 2002. A habitat network for terrestrial wildlife in the interior Columbia Basin. Northwest Science. 76(1): 1–14.

Woodbridge, B.; Hargis, C.D. 2006. Northern goshawk inventory and monitoring technical guide. Gen. Tech. Rep. GTR-WO-71. Washington, DC: U.S. Department of Agriculture, Forest Service. 80 p.

Woodruff, K.; Ferguson, H. 2005. Townsend's big-eared bat *(Corynorhinus townsendii)*. In: Management recommendations for Washington's priority species. Volume V: Mammals. Olympia, WA: Washington Department of Fish and Wildlife: 1–13.

Worthing, P. 1993. Endangered and threatened wildlife and plants: finding on petition to list the spotted frog. Federal Register. 58: 38553.

Wright, C.S.; Agee, J.K. 2004. Fire and vegetation history in the eastern Cascade Mountains, Washington. Ecological Applications. 14(2): 443–459.

Wrightman, C.S.; Germaine, S.S. 2006. Forest stand characteristics altered by restoration affect western bluebird habitat quality. Restoration Ecology. 14 (4): 653–661.

Yanes, M.; Velasco, J.M.; Suarez, F. 1995. Permeability of roads and railways to vertebrates: the importance of culverts. Biological Conservation. 71: 217–222.

Yokom, C.F.; Harris, S.W.; Hansen, H.A. 1958. Status of grebes in eastern Washington. Auk. 75: 36–47

Young, J.S.; Hutto, R.L. 1999. Habitat and landscape factors affecting cowbird distribution in the Northern Rockies. Studies in Avian Biology. 18: 41–51.

Zeigenfuss, L.C.; Singer, F.J.; Gudorf, M.A. 2000. Test of a modified habitat suitability model for bighorn sheep. Restoration Ecology. 8(4S): 38–46.

Zielinski, W.J.; Slauson, K.M.; Carroll, C.; Kent, C.; Kudrna, D. 2001. Status of American martens in coastal forests of the Pacific states. Journal of Mammalogy. 82: 478–490.

Zuckerberg, B.; Porte, W.F. 2010. Thresholds in the long-term responses of breeding birds to forest cover and fragmentation. Biological Conservation. 143: 952–962.

Zwartjes, P.W.; Cartron, J.-L. E.; Stoleson, P.L.L.; Haussamen, W.C.; Crane, T.E. 2005. Assessment of native species and ungulate grazing in the Southwest: terrestrial wildlife. Gen. Tech. Rep. RMRS-GTR-142. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 73 p.

Appendix 1: Common and Scientific Names

Common name	Scientific name
American marten	*Martes americana*
Bald eagle	*Haliaeetus leucocephalus*
Bighorn sheep	*Ovis canadensis*
Black-backed woodpecker	*Picoides arcticus*
Canada lynx	*Lynx canadensis*
Cassin’s finch	*Carpodacus cassini*
Columbia spotted frog	*Rana luteiventrus*
Eared grebe	*Podiceps nigricollis*
Fox sparrow	*Passerella lliaca*
Golden eagle	*Aquila chrysaetos*
Harlequin duck	*Histrionicus histrionicus*
Larch mountain salamander	*Plethodon larselli*
Lark sparrow	*Chondestes grammacus*
Lewis’s woodpecker	*Melanerpes lewis*
MacGillivray’s warbler	*Oporornis tolmiei*
Marsh wren	*Cistothorus palustris*
Northern bog lemming	*Synaptomys borealis*
Northern goshawk	*Accipiter gentilis*
Northern harrier	*Circus cyanus*
Peregrine falcon	*Falco peregrinus*
Pileated woodpecker	*Dyrocopus pileatus*
Sage thrasher	*Oreoscoptes montanus*
Tailed frog	*Ascaphus truei*
Tiger salamander	*Abystoma tigrinum*
Townsend’s big-eared bat	*Corynorhinus townsendii*
Western bluebird	*Sialia mexicana*
Western gray squirrel	*Sciurus griseus*
White-headed woodpecker	*Picoides albolarvatus*
Wilson’s snipe	*Gallinago delicata*
Wolverine	*Gulo gulo*
Wood duck	*Aix sponsa*

Appendix 2: Species of Conservation Concern

Family	Group	Common name	Surrogate
Alpine/boreal	Alpine	Gray-crowned rosy-finch	F
Alpine/boreal	Boreal forest	Spruce grouse	F*
Alpine/boreal	Boreal forest	Boreal owl	F*
Alpine/boreal	Boreal forest	Boreal chickadee	
Alpine/boreal	Boreal forest	Pine grosbeak	
Alpine/boreal	Boreal forest	Pygmy shrew	
Alpine/boreal	Boreal forest	Water vole	F
Alpine/boreal	Boreal forest	Northern bog lemming	F
Alpine/boreal	Boreal forest	Canada lynx	F*
Alpine/boreal	Boreal forest	Moose	
Forest mosaic	All forest communities	Northern goshawk	F
Forest mosaic	All forest communities	Blue grouse	
Forest mosaic	All forest communities	Band-tailed pigeon	
Forest mosaic	All forest communities	Great gray owl	
Forest mosaic	All forest communities	Long-eared owl	
Medium/large trees	All forest communities	Sharp-shinned hawk	
Medium/large trees	All forest communities	Rufous hummingbird	
Medium/large trees	All forest communities	Williamson’s sapsucker	
Medium/large trees	All forest communities	Hammond’s flycatcher	
Medium/large trees	All forest communities	Cordilleran flycatcher	
Medium/large trees	All forest communities	Mountain chickadee	
Medium/large trees	All forest communities	Cassin’s finch	F
Medium/large trees	All forest communities	Long-legged myotis	
Medium/large trees	All forest communities	Silver-haired bat	
Medium/large trees	All forest communities	Hoary bat	
Medium/large trees	All forest communities	Red-tailed chipmunk	
Medium/large trees	All forest communities	Northern flying squirrel	
Medium/large trees	Cool/moist forest	Larch Mountain salamander	F
Medium/large trees	Cool/moist forest	Spotted owl	
Medium/large trees	Cool/moist forest	Vaux’s swift	
Medium/large trees	Cool/moist forest	Pileated woodpecker	F

Family	Group	Common name	Surrogate
Medium/large trees	Cool/moist forest	Chestnut-backed chickadee	
Medium/large trees	Cool/moist forest	Brown creeper	
Medium/large trees	Cool/moist forest	Winter wren	
Medium/large trees	Cool/moist forest	Golden-crowned kinglet	
Medium/large trees	Cool/moist forest	Ruby-crowned kinglet	
Medium/large trees	Cool/moist forest	Varied thrush	
Medium/large trees	Cool/moist forest	Hermit warbler	
Medium/large trees	Cool/moist forest	American marten	F
Medium/large trees	Cool/moist forest	Fisher	
Medium/large trees	Cool/moist forest	Caribou	
Medium/large trees	Dry forest	Flammulated owl	
Medium/large trees	Dry forest	White-headed woodpecker	F
Medium/large trees	Dry forest	Purple martin	
Medium/large trees	Dry forest	White-breasted nuthatch	
Medium/large trees	Dry forest	Pygmy nuthatch	
Medium/large trees	Dry forest	Ringtail	
Open forest	All forest communities	Rubber boa	
Open forest	All forest communities	Sharptail snake	
Open forest	All forest communities	Cassin’s vireo	
Open forest	All forest communities	Western bluebird	F
Open forest	All forest communities	Nashville warbler	
Open forest	All forest communities	Purple finch	
Open forest	All forest communities	Pine siskin	
Open forest	All forest communities	Evening grosbeak	
Open forest	All forest communities	California myotis	
Open forest	All forest communities	Fringed myotis	F
Open forest	All forest communities	Long-eared myotis	
Open forest	Early successional	Townsend’s solitaire	F
Open forest	Early successional	Fox sparrow	F
Open forest	Early successional	Lazuli bunting	
Open forest	Pine/oak - (medium to large tree)	California mountain kingsnake	F
Open forest	Pine/oak - (medium to large tree)	Acorn woodpecker	F

Family	Group	Common name	Surrogate
Open forest	Pine/oak - (medium to large tree)	Western gray squirrel	F
Open forest	Postfire habitat	American kestrel	
Open forest	Postfire habitat	Lewis’s woodpecker	F
Open forest	Postfire habitat	Three-toed woodpecker	
Open forest	Postfire habitat	Black-backed woodpecker	F
Open forest	Postfire habitat	Olive-sided flycatcher	
Open forest	Postfire habitat	Western wood-pewee	
Upland grassland	Upland grassland	Upland sandpiper	F
Human disturbance	Habitat generalist	Peregrine falcon	F
Human disturbance	Habitat generalist	Gray wolf	
Human disturbance	Habitat generalist	Grizzly bear	
Human disturbance	Habitat generalist	Wolverine	F
Woodland/grass/shrub	Woodland/grass/shrub	Pygmy horned lizard	
Woodland/grass/shrub	Woodland/grass/shrub	Side-blotched lizard	
Woodland/grass/shrub	Woodland/grass/shrub	Ringneck snake	
Woodland/grass/shrub	Woodland/grass/shrub	Striped whipsnake	
Woodland/gass/shrub	Woodland/grass/shrub	Ferruginous hawk	
Woodland/grass/shrub	Woodland/grass/shrub	Golden eagle	F
Woodland/grass/shrub	Woodland/grass/shrub	Prairie falcon	
Woodland/grass/shrub	Woodland/grass/shrub	Common poorwill	
Woodland/grass/shrub	Woodland/grass/shrub	White-throated swift	
Woodland/grass/shrub	Woodland/grass/shrub	Black-billed magpie	
Woodland/gass/shrub	Woodland/grass/shrub	Canyon wren	
Woodland/grass/shrub	Woodland/grass/shrub	Lark sparrow	F
Woodland/grass/shrub	Woodland/grass/shrub	Brewer’s blackbird	
Woodland/grass/shrub	Woodland/grass/shrub	Western small-footed myotis	
Woodland/grass/shrub	Woodland/grass/shrub	Yuma myotis	
Woodland/grass/shrub	Woodland/grass/shrub	Spotted bat	
Woodland/grass/shrub	Woodland/grass/shrub	Pallid bat	F
Woodland/grass/shrub	Juniper woodland	Ash-throated flycatcher	F
Woodland/grass/shrub	Juniper woodland	Pinyon jay	
Woodland/grass/shrub	Juniper woodland	Lesser goldfinch	

Family	Group	Common name	Surrogate
Woodland/grass/shrub	Woodland/shrub	Sagebrush lizard	
Woodland/grass/shrub	Woodland/shrub	Night snake	
Woodland/grass/shrub	Woodland/shrub	Gray flycatcher	
Woodland/grass/shrub	Woodland/shrub	Loggerhead shrike	F
Woodland/grass/shrub	Woodland/shrub	Green-tailed towhee	
Woodland/grass/shrub	Woodland/shrub	Merriam’s shrew	
Woodland/grass/shrub	Shrub	Desert horned lizard	
Woodland/grass/shrub	Shrub	Greater sage grouse	F*
Woodland/grass/shrub	Shrub	Sage thrasher	F*
Woodland/grass/shrub	Shrub	Brewer’s sparrow	
Woodland/grass/shrub	Shrub	Black-throated sparrow	
Woodland/grass/shrub	Shrub	Sage sparrow	
Woodland/grass/shrub	Shrub	Pygmy rabbit	
Woodland/grass/shrub	Shrub	Black-tailed jackrabbit	
Woodland/grass/shrub	Grass/shrub	Tiger salamander	F
Woodland/grass/shrub	Grass/shrub	Sharp-tailed grouse	
Woodland/grass/shrub	Grass/shrub	Long-billed curlew	
Woodland/grass/shrub	Grass/shrub	Burrowing owl	
Woodland/grass/shrub	Grass/shrub	Horned lark	
Woodland/grass/shrub	Grass/shrub	Oregon vesper sparrow	
Woodland/grass/shrub	Grass/shrub	Western meadowlark	
Woodland/grass/shrub	Grass/shrub	Preble’s shrew	
Woodland/grass/shrub	Grass/shrub	White-tailed jackrabbit	
Woodland/grass/shrub	Grass/shrub	Ord’s kangaroo rat	
Woodland/grass/shrub	Grass/shrub	Sagebrush vole	
Woodland/grass/shrub	Grass/shrub	American badger	
Woodland/grass/shrub	Grass/shrub	Pronghorn	
Woodland/grass/shrub	Grass/shrub	Mountain goat	
Woodland/grass/shrub	Grass/shrub	Rocky mountain bighorn sheep	f
Woodland/grass/shrub	Grass/shrub	California bighorn	f
Woodland/grass/shrub	Grassland	Northern harrier	F*
Woodland/grass/shrub	Grassland	Swainson’s hawk	F*

Family	Group	Common name	Surrogate
Woodland/grass/shrub	Grassland	Short-eared owl	
Woodland/grass/shrub	Grassland	Grasshopper sparrow	F*
Chambers/caves	Chambers/caves	Townsend's big-eared bat	F
Chambers/caves	Chambers/caves	Brazilian free-tailed bat	
Riparian	Conifer riparian	Inland tailed frog	F
Riparian	Conifer riparian	Black swift	F
Riparian	riparian/lg tree or snag/open water	Wood duck	F
Riparian	riparian/lg tree or snag/open water	Harlequin duck	F
Riparian	riparian/lg tree or snag/open water	Bufflehead	
Riparian	riparian/lg tree or snag/open water	Common goldeneye	
Riparian	riparian/lg tree or snag/open water	Barrow's goldeneye	
Riparian	riparian/lg tree or snag/open water	Hooded merganser	
Riparian	riparian/lg tree or snag/open water	Common merganser	
Riparian	riparian/lg tree or snag/open water	Bald eagle	F
Riparian	Shrubby/deciduous riparian	Mountain quail	
Riparian	Shrubby/deciduous riparian	Yellow-billed cuckoo	
Riparian	Shrubby/deciduous riparian	Western screech-owl	
Riparian	Shrubby/deciduous riparian	Red-naped sapsucker	F
Riparian	Shrubby/deciduous riparian	Willow flycatcher	
Riparian	Shrubby/deciduous riparian	Red-eyed vireo	
Riparian	Shrubby/deciduous riparian	Veery	
Riparian	Shrubby/deciduous riparian	Yellow warbler	
Riparian	Shrubby/deciduous riparian	American redstart	
Riparian	Shrubby/deciduous riparian	Northern waterthrush	
Riparian	Shrubby/deciduous riparian	MacGillivray's warbler	F
Riparian	Shrubby/deciduous riparian	Wilson's warbler	
Riparian	Shrubby/deciduous riparian	Yellow-breasted chat	
Riparian	Shrubby/deciduous riparian	Water shrew	
Riparian	Marsh with adjacent large trees	Great blue heron	
Riparian	Marsh with adjacent large trees	Great egret	
Riparian	Marsh with adjacent large trees	Green heron	
Riparian	Marsh with adjacent large trees	Black-crowned night-heron	F

Family	Group	Common name	Surrogate
Riparian	Pond/small lake/backwater	Painted turtle	F
Riparian	Pond/small lake/backwater	Western pond turtle	F
Riparian	Pond/small lake/backwater	Western toad	
Riparian	Pond/small lake/backwater	Woodhouse’s toad	F
Riparian	Pond/small lake/backwater	Cascades frog	F
Riparian	Pond/small lake/backwater	Oregon spotted frog	F*
Riparian	Pond/small lake/backwater	Columbia spotted frog	F*
Riparian	Pond/small lake/backwater	Spotted sandpiper	
Riparian	Banks	Northern rough-winged swallow	F
Wetland	Marsh	American bittern	
Wetland	Marsh	Least bittern	
Wetland	Marsh	Snowy egret	
Wetland	Marsh	Yellow rail	
Wetland	Marsh	Virginia rail	
Wetland	Marsh	Marsh wren	F
Wetland	Marsh	Tricolored blackbird	
Wetland	Marsh	Yellow-headed blackbird	
Wetland	Marsh/wet meadow	White-faced ibis	
Wetland	Marsh/wet meadow	Sandhill crane	
Wetland	Marsh/wet meadow	Killdeer	
Wetland	Marsh/wet meadow	Black-necked stilt	
Wetland	Marsh/wet meadow	American avocet	F
Wetland	Marsh/wet meadow	Greater yellowlegs	
Wetland	Marsh/wet meadow	Willet	
Wetland	Marsh/wet meadow	Wilson’s snipe	F
Wetland	Marsh/wet meadow	Wilson’s phalarope	
Wetland	Marsh/wet meadow	Franklin’s gull	
Wetland	Marsh/wet meadow	Bobolink	
Wetland	Marsh/open water	Common loon	
Wetland	Marsh/open water	Pied-billed grebe	
Wetland	Marsh/open water	Horned grebe	
Wetland	Marsh/open water	Red-necked grebe	

Family	Group	Common name	Surrogate
Wetland	Marsh/open water	Eared grebe	F
Wetland	Marsh/open water	Western grebe	
Wetland	Marsh/open water	Clark's grebe	
Wetland	Marsh/open water	American white pelican	
Wetland	Marsh/open water	Trumpeter swan	
Wetland	Marsh/open water	Blue-winged teal	
Wetland	Marsh/open water	Northern shoveler	
Wetland	Marsh/open water	Northern pintail	
Wetland	Marsh/open water	Green-winged teal	
Wetland	Marsh/open water	Canvasback	
Wetland	Marsh/open water	Redhead	
Wetland	Marsh/open water	Ring-necked duck	
Wetland	Marsh/open water	Greater scaup	
Wetland	Marsh/open water	Lesser scaup	
Wetland	Marsh/open water	Ruddy duck	
Wetland	Marsh/open water	Caspian tern	
Wetland	Marsh/open water	Forster's tern	
Wetland	Marsh/open water	Black tern	

F-indicates a surrogate species for the group that could be addressed in the development of management actions.

F*-indicates a choice of which surrogate species to use. Managers from different areas may choose different species primarily based on the distribution of the species.

f-indicates a species that had localized populations that were confined to very specific habitats. Proposed management alternatives for these species would be applied only to local areas.

Appendix 3: Members of Working Groups

USFS Pacific Northwest Region Species Viability Assessment Workgroup

Shawne Mohoric–Team Co-Leader
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Olympia Lab

Bruce Marcot
Research ecologist
USDA Forest Service
Pacific Northwest Research Station
Portland Laboratory

Richard Holthausen
National wildlife ecologist (retired)
USDA Forest Service
Washington Office

Experts Consulted During Species Model Development

Name	Affiliation	Area of expertise
Robert Altman	American Bird Conservancy	Avian ecology
Peter Singleton	USDA Pacific Northwest Research Station	Carnivore ecology
		Landscape permeability
Robert Naney	USDA Forest Service, Pacific Northwest Region Carnivore Species Leader	Carnivore ecology and management
Evelyn Bull	USDA Pacific Northwest Research Station	Woodpecker and amphibian ecology
Patricia Garvey-Darda	USDA Forest Service, Wenatchee National Forest	Amphibian ecology
Joseph Buchanan	Washington Department of Fish and Wildlife	Raptor ecologist
Victoria Saab	USDA Forest Service, Rocky Mountain Research Station	Avian ecology

Field Biologist Review Teams

Forest reviewed	Name and position	District/agency
Okanogan	John Rohrer, district wildlife biologist	Methow Ranger District
Okanogan	Scott Fitkin, district biologist	Washington Department of Fish and Wildlife
Okanogan	Jeremy Anderson, district wildlife biologist	Tonasket Ranger District
Okanogan	Robert Naney, forest wildlife biologist	Okanogan National Forest
Colville	Chris Loggers, district wildlife biologist	Colville National Forest
Colville	James McGowan, forest wildlife biologist	Colville National Forest
Wenatchee	Mallory Lenz, district wildlife biologist	Chelan Ranger District
Wenatchee	Ann Sprague, district wildlife biologist	Entiat Ranger District
Wenatchee	Don Youkey, district wildlife biologist	Wenatchee River Ranger District
Wenatchee	JoEllen Richards, district wildlife biologist	Cle Elum Ranger District
Wenatchee	Peter Forbes, district wildlife biologist	Naches Ranger District
Wenatchee	Beau Patterson, district biologist	Washington Department of Fish and Wildlife

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