November 25, 2021

Grand Mesa, Uncompahgre, and Gunnison National Forests
Attn: Plan Revision Team
2250 Highway 50
Delta, CO 81416
Submitted via the Online Feedback Tool

Re: Comments on the Draft Revised Land Management Plan and Draft Environmental Impact Statement for the Grand Mesa, Uncompahgre, and Gunnison National Forest

Dear GMUG Planning Team,

Please accept the attached documents as supplemental information we referenced in our comment letter on the Draft Plan and Draft Environmental Impact Statement. Please feel free to contact Matt Reed at 970.349.7104 or matt@hccacb.org if you have any questions.

Sincerely,

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High Country Conservation Advocates
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United States
Department of
Agriculture

Forest Service

Pacific Northwest
Research Station

General Technical Report
PNW-GTR-780

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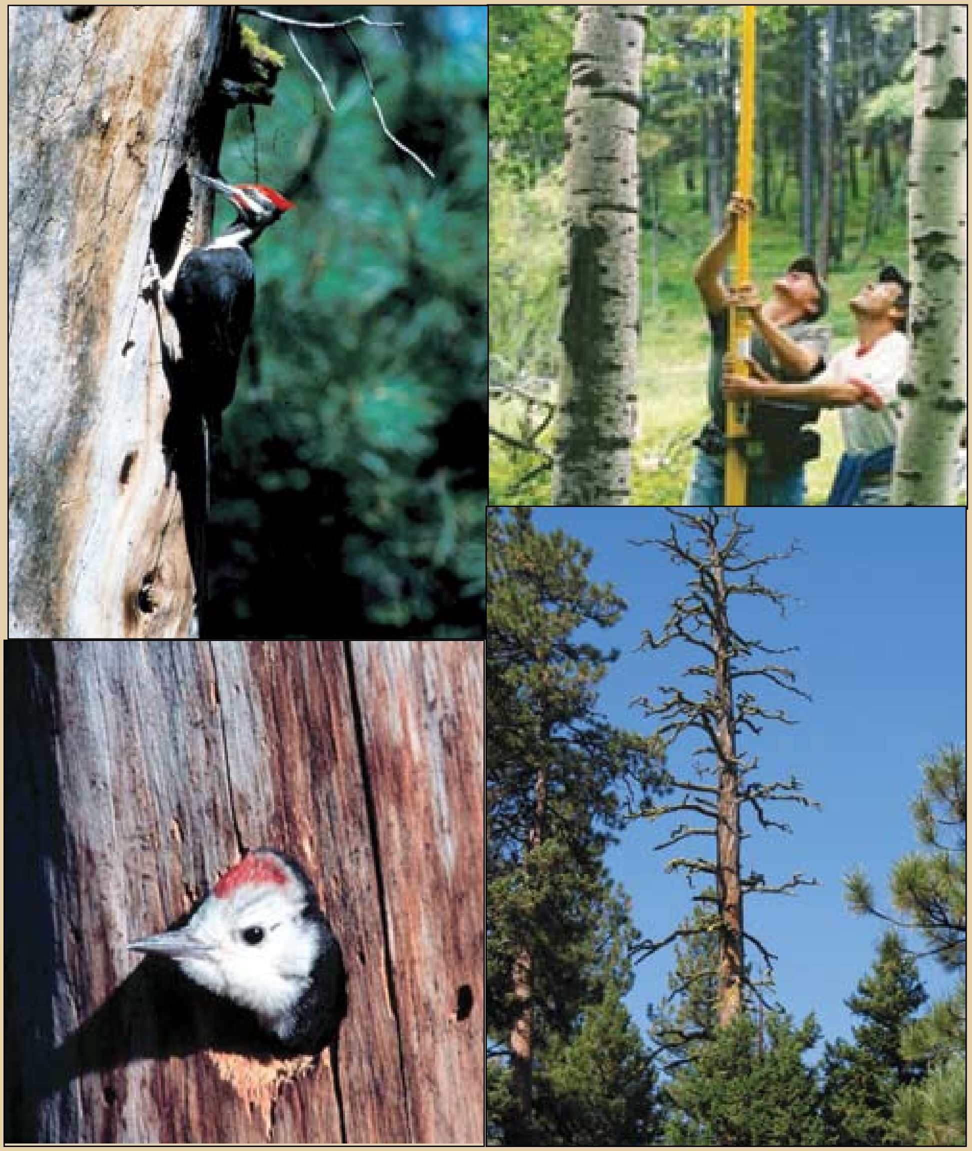
SnagPRO: Snag and Tree Sampling and Analysis Methods for Wildlife

Lisa J. Bate, Michael J. Wisdom, Edward O. Garton, and Shawn C. Clabough



Authors

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Cover

Clockwise from upper left, pileated woodpecker (Evelyn Bull), researchers monitoring nest in aspen tree (Lisa Bate), large old-growth snag with nesting and roosting cavities (Evelyn Bull), and white-headed woodpecker (Lisa Bate).

Abstract

Bate, Lisa J.; Wisdom, Michael J.; Garton, Edward O.; Clabough, Shawn C. 2008. SnagPRO: snag and tree sampling and analysis methods for wildlife. Gen. Tech. Rep. PNW-GTR-780. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 80 p.

We describe sampling methods and provide software to accurately and efficiently estimate snag and tree densities at desired scales to meet a variety of research and management objectives. The methods optimize sampling effort by choosing a plot size appropriate for the specified forest conditions and sampling goals. Plot selection and data analyses are supported by SnagPRO, a software program designed specifically to serve our sampling methods.

We present two sampling methods to estimate density and associated characteristics of snags and trees. The first method requires sampling until a desired precision is obtained for a density estimate. The second method compares estimated densities with target densities, such as target snag densities specified under a land management plan.

Our methods of snag and tree sampling are compatible with recently developed methods of log sampling, thereby improving efficiencies by enabling the simultaneous collection of all three habitat components—snags, large trees, and logs—to meet research or management objectives for a variety of resource disciplines, including wildlife, silviculture, fuels, and soils. Recently developed methods of log sampling also use SnagPRO for data collection and analysis.

Our methods and software are particularly relevant to forest management, given that nearly all federal land use plans require monitoring of snag and tree densities in relation to management direction for wildlife. Staffing and budgets available to estimate snag and tree densities, however, are extremely limited, and thus require efficient methods to achieve acceptable accuracy. Our methods are an efficient approach for estimating snag and tree densities, particularly when combined with use of the supporting SnagPRO software.

Keywords: Cavity nester, density, foraging, large tree, nesting, monitoring, sampling technique, snags, SnagPRO, wildlife management, wildlife use, woodpecker.

Contents

Introduction

The ecological roles and importance of dead and dying wood in forest ecosystems have been the subject of increasing interest and awareness over the past decades. For many vertebrate species, standing dead trees (snags) provide essential habitat in the form of cover and food. Snags with internal pockets of decay provide insulated and protected nest, roost, or den sites (Bull and others 1997, Laudenslayer 2002, Mellen and others 2006, Rose and others 2001). Other types of snags, colonized by invertebrates, provide a rich foraging resource (Bate 1995, Bull and Holthausen 1993).

Living trees with decay also provide nest, roost, and den sites (Bull and others 1997). In Oregon, for example, Rose and others (2001) documented a myriad of wildlife species associated with tree cavities (51 species), with decayed portions of trees (45 species), with hollow trees (28 species), with bark crevices of trees (21 species), and with mistletoe clumps found in large trees (18 species). In addition, large, mature trees provide an essential foraging resource for wildlife in forest ecosystems. White-headed woodpeckers (*Picoides albolarvatus*) and other wildlife species depend on the seeds produced by mature ponderosa pines (*Pinus ponderosa* Dougl. ex Laws.) for spring and autumn foraging (Dixon 1995, Ligon 1973). As certain tree species age, they develop deep furrows that harbor increased arthropod densities for foraging birds (Bull and others 1986, Mariani and Manuwal 1990). Live trees with internal pockets of decay may be colonized by ants (for example, *Formica* spp.), which serve as a key food for several vertebrate species (Bull and others 1997). Finally, large trees are the pool for recruitment of future snags.

For many vertebrate species, standing dead trees (snags) provide essential habitat in the form of cover and food. Living trees with decay also provide nest, roost, and den sites.

As primary cavity-nesters, the role of woodpeckers is integral to healthy forest ecosystems because these species excavate cavities in decayed portions of snags or live trees for nest and roost sites. These cavities are subsequently used by secondary cavity-nesting or nonexcavating vertebrates. Because of their role in providing cavities needed by many other vertebrates, woodpeckers often are considered **indicator species** (Brown 1985, Thomas and others 1979). That is, if the needs of woodpeckers are met, then the needs of the larger set of species that depend on the snags and live trees that woodpeckers modify also are met (Rose and others 2001).

Although most federal land use agencies have adopted retention and recruitment standards to maintain adequate densities of snags and large trees for wildlife, these structures have declined in abundance for various reasons (Hann and others 1997). Snags are systematically removed because of their commercial and firewood values (Bate and others 2007, Wisdom and Bate 2008) and to reduce estimated risks associated with safety, fire, and disease (Dickson and others 1983, Ffolliot 1983, Hann and others 1997, Styskel 1983). Large trees also are targeted for removal during timber harvest as well as for firewood. Both snags and dying trees are routinely removed during salvage-logging operations (Saab and Dudley 1998). In addition, snag retention programs on national forests are hampered by problems with safety, funds, and inconsistent standards and guidelines (Hope and McComb 1994). Consequently, the density, size, and condition of snags on national forests often do not meet management standards (Bate 1995, Morrison and others 1986).

Reduced snag densities affect more than the species that depend on snags for survival. In addition, the commodity value of timber may be diminished. Most cavity-nesters are insectivores, and are instrumental in preventing or retarding insect outbreaks (Beebe 1974, Otvos 1979). Some species of woodpeckers are known to aggregate in areas of insect outbreaks, helping to accelerate the decline of the insect populations (Otvos 1979). Foraging woodpeckers chip and probe at the bark of beetle-infested trees, altering the microenvironment of any eggs and larvae and increasing beetle susceptibility to mortality from parasites and extreme temperature fluctuations. Thomas and others (1979) provided compelling arguments and evidence in support of maintaining viable populations of woodpeckers and other insectivores to benefit forest-based economies.

Managing densities of snags and large trees is essential for ensuring that the needs of cavity-nesting and decay-dependent species are met. Recognizing the integral role of woodpeckers in forest ecosystems, Thomas and others (1979) and Brown (1985) provided some of the first guidelines for managing snag densities for woodpeckers and other snag-dependent wildlife. These guidelines, however, focused only on the nesting needs of woodpeckers. Since then, new studies indicate that more snags are required than recommended in either of these publications to provide for all needs of snag-dependent species (Bull and others 1997, Mellen and others 2006, Rose and others 2001). In Oregon, for example, at least 93 vertebrate species use snags for nesting, roosting, denning, feeding, or related life functions (Rose and others 2001). In addition, foraging structures differ from nesting and roosting structures for woodpeckers (Bate 1995, Bull and Holthausen 1993, Caton 1996, Dixon 1995), and some secondary cavity nesters, such as bats (Betts 1998, Campbell and others 1996, Ormsbee and McComb 1998) and Vaux's swifts (Bull and others 1997), use hollow trees or snags for nesting and roosting.

Monitoring snags and large trees can be inherently difficult because their densities and distributions differ extensively, as do forest conditions that hamper sampling, such as topography, seral stage, and sampling visibility.

Monitoring snags and large trees can be inherently difficult because their densities and distributions differ extensively, as do forest conditions that hamper sampling, such as topography, seral stage, and sampling visibility (Bate and others 2007, Wisdom and Bate 2008). Therefore, to improve the efficiency of snag and large-tree monitoring programs, resource specialists must first determine the shape and size of plot that works best in a given area. Bull and others (1990) tested the efficiency and accuracy of both fixed- and variable-radius circular plots to determine snag densities. They found that 1-acre (0.4-ha) fixed and variable plots, with a factor-5 prism, worked best for areas with snag densities ranging from 0.7 to 2 snags per acre (1.7 to 4.9 snags/ha). Although large circular plots may be adequate for sampling open forests with relatively low snag densities, it is not possible to accurately count snags with plots of this size and shape in areas obscured by vegetation or in steep terrain (Bate and others 1999). In addition, where snag densities are high, such as in beetle-killed or burned areas, the use of large, circular plots will increase sample variance, making it difficult to obtain a precise estimate (Bate and others 1999).

Although large circular plots may be adequate for sampling open forests with relatively low snag densities, it is not possible to accurately count snags with plots of this size and shape in areas obscured by vegetation or in steep terrain.

Prisms or gauges can also be used to sample snags or trees along a transect line, referred to as horizontal line sampling (Husch and others 1972). Ducey and others (2002) presented a modification of horizontal line sampling (MHLS) that uses shorter segments and then adds one-half of a conventional horizontal point sample at the end of each line. Ducey and others (2002) found the modified line sampling to be more efficient and precise than traditional line sampling. As with variable-radius circular plots, however, a small prism usually is required to obtain adequate samples when snags are rare. This poses a substantial bias, however, of being unable to detect snags often hidden at the longer distances required by the sampling method, resulting in underestimation of snag densities (Harmon and Sexton 1996).

Kenning and others (2005) investigated the efficiency and bias of various snag inventory methods including fixed circular plots (1/20th acre [0.02 ha]), MHLS (Ducey and others 2002), N-tree distance sampling, and distance-limited N-tree sampling. The N-tree sampling method measures snag characteristics on a specified number of snags (N) from a center point. Under N-tree distance sampling, the maximum sampling distance was unlimited. In distance-limited N-tree sampling, the maximum sampling distance was 8 m. Kenning and others (2005) tested N-tree sampling with N = 1, N = 2, and N = 3. They found that small, fixed-area plots were most efficient for estimating density and that MHLS was best for estimating basal area.

Bütler and Schlaepfer (2004) tested a new method of quantifying large snags by coupling color infrared aerial photographs and a geographical information system (GIS) in spruce forests of Switzerland. They were encouraged by their results for large snags in these forest conditions, but did observe different degrees of accuracy based on tree diameter, treetop condition (intact or broken), and canopy closure. Other factors such as aspect, surface slope, weather, and hour of flight also affected the snag detection rate. Consequently, Bütler and Schlaepfer (2004) suggested further testing to obtain the appropriate coefficient to correct for underestimation of snag densities when using this method.

As Krebs (1989) documented, most ecologists have found that rectangular or other long, thin plots are more accurate and efficient than circular or square plots. Forest habitat components are never uniformly distributed, and clumps or patches of habitat, such as snags and trees, are common. Rectangular plots are better for sampling because they cross more clumps of snags or trees, rather than either encircling or missing them completely. Consequently, use of rectangular or other long, thin plots results in lower sample variance, which translates into smaller sample sizes required to obtain desired precision.

SnagPRO was designed to identify optimal plot size, help design field surveys, facilitate data collection with electronic field forms, estimate sample sizes, and analyze results.

Whereas rectangular plots are recognized as the optimal plot **shape** for sampling in patchy or clumped habitats (Krebs 1989), the optimal plot size differs among forest types depending on the abundance and distribution of the snag or tree size of interest. The determination of optimal plot size is affected by a variety of conditions and objectives, all of which can be efficiently and accurately considered with the use of SnagPRO. The SnagPRO program was designed not only to identify optimal plot size, but to help users design field surveys, to guide and facilitate data collection with use of standard, electronic field forms, to estimate required sample sizes needed to achieve desired precision, and to analyze all results in ways that are statistically valid and that meet sampling objectives. SnagPRO provides practical tutorials with sample data sets to demonstrate use of the software in survey design, field sampling, and data analysis.

In the following sections, we describe our sampling methods and provide examples with SnagPRO to design surveys, conduct sampling, and analyze data for estimating snag and tree densities at desired scales. We provide example tutorials and address all aspects of the estimation process.

General Information

Downloading and Installing SnagPRO

Download SnagPRO (version 1.0) from the USDA Forest Service Pacific Northwest (PNW) Web site at http://www.fs.fed.us/pnw/publications/tools-databases.shtml. SnagPRO installation requires at least 5 MB of space. SnagPRO requires another 10 to 50 MB of space to operate. Users may choose where to install SnagPRO; the default location is C:\Program Files. Once installed, users may create a shortcut to SnagPRO for their desktops or Quick Launch bar.

There will be a Microsoft Excel file—Snag_Tutorial_Data.xls—accompanying the zipped SnagPRO file that needs to be downloaded. This file contains four worksheets. Two contain sample snag data sets for use with the tutorials found at the end of this report. One worksheet contains a sample data form for printing and hard-copy use in the field. The fourth worksheet is for users who want to enter their data directly into a spreadsheet file while in the field. Other electronic formats can be used in the field but then need to be formatted as shown in the examples below before importing to SnagPRO for analysis.

User's existing resource data—from spreadsheet or database—must be correctly formatted as a comma-separated value (CSV) file before importing to SnagPRO. For simplicity, this report addresses only spreadsheet examples, and data files for the tutorial are in spreadsheet format.

Our sampling methods can be used to gain knowledge about snag or tree habitats for any wildlife species of interest and are appropriate for other resource disciplines needing statistically valid esitmates of snag and tree densities.

Sampling Applications

Our sampling methods can be used to gain knowledge about snag or tree habitats for any wildlife species of interest. For example, knowledge of the difference in large (≥16 in [40 cm] diameter at breast height [d.b.h.]) wildlife tree densities between two foraging areas for brown creeper (*Certhia americana*) or red tree voles (*Arborimus longicaudus*) may be of interest. Similarly, a land use plan may call for monitoring snag and wildlife tree densities for white-headed woodpecker in a landscape dominated by intensive timber production versus another landscape dominated by wilderness designation. Or, mitigation of timber harvest practices may call for retention of snags >20 in (5.1 cm) d.b.h. that are likely to serve as nest structures for pileated woodpeckers (*Dryocopus pileatus*).

Our methods also are appropriate for other resource disciplines needing statistically valid estimates of snag and tree densities. Plot sizes can be adjusted easily to accommodate small-diameter (for example, saplings) or large-diameter snags or large trees as necessary for different resource objectives. The methods may also complement the data collected in other projects (for example, project planning, effects analyses, stand exam or Forest Inventory and Analysis [FIA] data) by converting data to similar units of measurement (for example, number/acre [number/ha]) to provide additional baseline comparisons for resource planning and management.

Methods

General Snag and Large-Tree Sampling Guidelines

A condensed outline of the guidelines for sampling snags and trees can be found in appendix 1. A more detailed discussion of topics in the outline follows.

Sampling Objectives (Step 1)

Most ecological studies are designed to answer some form of the question: How many are there? For example, do harvested areas comply with snag density standards of the land management plan? How many large trees are available for future snag recruitment? Or, how many large snags suitable for nesting Lewis' woodpeckers (*Melanerpes lewis*) are available in a burned area? Therefore, the first step in any sampling program is to specify the sampling objective(s). The objectives ultimately determine the amount of time and resources needed to obtain estimates at a desired precision. Answering the following questions will help determine the objectives:

Time spent on extraneous data collection limits sample size and the subsequent results.

1. What snag (tree) size(s) will be surveyed (diameter and height)?
2. How will data be used? For land use allocation? For compliance monitoring? Or to respond to land use appeals or other legal actions? The purpose often dictates answers to the following questions.
3. How precise does the estimate need to be?
4. Is snag/tree species important? If so, why?
5. Will signs of wildlife use be recorded (for example, woodpecker foraging, nest, or roost cavities)?
6. Are estimates for separate areas needed?

As Krebs (1989) stated, "Not everything that can be measured should be." It is common to collect information on everything possible while in the field. Yet, the time spent on extraneous data collection limits sample size and the subsequent results. For example, examining each snag for cavities may seem like a simple addition to the field protocol. Yet, the time spent examining a tall snag on all sides for cavities can substantially increase the amount of time spent surveying a given transect length, especially for inexperienced field crews. Therefore, it is important to establish clear objectives and explicitly describe how data will be used before starting fieldwork.

Regarding precision levels for most sampling activities, we recommend a design to obtain estimates within 20 percent of the true mean 90 percent of the time. We have set these values as defaults in SnagPRO. Sampling to achieve a higher precision (for example, within 10 percent of the true mean 95 percent of the time) would be cost and area prohibitive for habitat components that are relatively rare and have clumped distributions. Only when habitat components are relatively abundant and randomly distributed would higher precision be manageable.

Landscape Definition and Selection (Step 2)

The second step is to define the landscape, or area of interest, by delineating the boundaries. This area is the sampling frame, within which a random sample is drawn for the purpose of making inferences to the entire area. Our sampling methods are designed to be compatible with the snag and large-tree sampling methods previously developed by Bate and others (1999). These methods were based on a sampling unit defined as a landscape (sampling frame) ranging from about 3,000 to 6,900 acres or 1200 to 2800 hectares (Bull and others 1991). These sampling methods can also be used on a subwatershed scale with a few modifications. See "Establishing Transects" and "Compare to Target" sections for details. Subwatersheds within the Columbia River Basin region can be as large as 20,000 acres, or 8100 hectares (Quigley and others 1996).

The sample area need not be a delineated subwatershed, but may be a smaller area like a research natural area including less than 1,000 acres (400 ha). Whether these sampling methods can be used within even smaller areas (<100 acres [40 ha]) depends on the density and distribution of snags and trees in the sizes of interest. Burned habitats will likely have enough snags to make these sampling procedures practical with small plot sizes. To obtain a density estimate of large (>20-in [51-cm] d.b.h.) snags in an unburned forest of the same area, a complete count may be more appropriate.

Stratification should reduce the final requirement of resources and provide a more precise estimate.

Landscape Stratification (Step 3)

Perhaps the most critical step in snag or tree sampling is the stratification process. Although the initial investment of time spent in the stratification process may seem large, if done correctly, it should reduce the final requirement of resources and provide a more precise estimate (Krebs 1989). Existing stratifications, such as those used by silviculturists to conduct stand exams, can be readily adapted for stratifications used to sample snags and trees. If snag and tree sampling is to occur simultaneously with log sampling, stratification designs based on snag abundance are appropriate because obtaining precise estimates of snags is often more difficult than for trees or logs, owing to the low abundance and patchy distribution of snags.

Whether to stratify a landscape before sampling depends on several factors. Cochran (1977) identified the three most common reasons.

- Stratification may produce a gain in precision of the estimate. If the landscape is heterogeneous (highly variable) in abundance of snags or trees, establishing individual strata that are homogeneous (same) within each stratum can substantially improve precision.
- Sampling problems can differ for parts of the landscape with different forest community types, timber harvest methods, and seral stages; stratifying by these conditions will allow appropriate sample size allocation among these different conditions, again increasing precision.
- Separate estimates are desired for certain subdivisions of the landscape. For example, part of a subwatershed may be managed for timber production,

another managed as a research natural area, and a third managed as a wilderness area.

If one or more of the above reasons are relevant, it is beneficial to stratify. SnagPRO can accommodate up to four strata. We set the limit at four strata because it is rare that more than four sampling categories will be used. In particular, with increasing number of strata comes the law of diminishing returns. That is, for each additional stratum, there needs to be an additional 10 transects (400 ft or 100 m) of sampling line. If, however, resource specialists find that they need to divide a landscape into five or more strata, they can use the **Simple-Random Sampling Equation** page within SnagPRO to obtain their stratum means (equation 2) and variances (equation 9) and then calculate a stratified mean estimate and its bound using equations 12 through 14. If the landscape is homogeneous throughout in regard to snag and tree densities, there is probably little to be gained from stratification.

Use the following steps to stratify your survey area:

1. Visit the area to identify areas with general differences in snag and tree densities, vegetation types, and structural conditions. These differences should be noted and marked on the map.
2. Following the initial field visit, obtain more accurate reference maps for field use, such as GIS maps, U.S. Geological Survey (USGS) orthoquad maps, or both. Make sure that appropriate metadata (data definitions) are included for all GIS layers or maps to be used. Maps should include the following information:
   a. Road system, road types, and maintenance level of roads.
   b. Polygon or vegetation units and their respective unique numeric identifiers.
   c. Current seral stage of vegetation at a scale of 1:31,680, or better resolution. Keep in mind that scale is a ratio or fraction, so polygons mapped at 1:24,000-scale will appear larger than they do in the 1:31,680-scale map. This information may be on one or more maps.
3. Query the polygon database for detailed information about each polygon such as forest type (low versus high elevation, dry versus moist), management history, seral stage, disturbance history (wind, fire, insects, and disease), and any other factors that may affect snag/tree abundance. The output of your query will be a simple report of polygon data attributes. Ensure that the report includes types of management activities, such as harvest method used, slash and burn prescriptions, thinning, and snag and tree retention standards that potentially apply to each area or land use allocation.

4. Ground check the map and polygon data using aerial photographs. Generally, the amount of time that must be spent to stratify the polygons in the field is inversely proportional to the quality of the GIS layers available. Carefully review the metadata and discuss any concerns with the GIS specialist to ensure that characteristics of the spatial data, particularly its accuracy and how it was collected or derived, are well understood.
5. Revisit the survey area with the field maps. Plan to spend at least one day to validate the information on the map(s) and in the report from the database query.
6. Assign each polygon to a stratum. Estimate the number of acres (ha) within each polygon or stratum.

Most landscapes surveyed for snags/trees have undergone some amount of timber harvest. Consequently, depending on the method of timber harvest, the placement of each polygon within a stratum may or may not be straightforward. For example, if snags are of interest, most unharvested mature/old-growth stands in mixed-conifer forests support a high abundance of snags. By contrast, older harvest units that have been clearcut may have few snags. Finally, more recent clearcut units may have snags distributed throughout the polygon, reflecting more recent policy changes.

For these conditions, combine all unharvested mature/old-growth stands into a single stratum. Then determine the time period when snag retention began in timber harvest units, and ground check some example units. Combine these stands into a stratum. Finally, combine all older harvest units into another stratum. Generate a new map of all stands categorized as one of three strata: (1) stands that were clearcut before adoption of snag retention standards, (2) stands that were clearcut since adoption of snag retention standards, and (3) unharvested mature/old-growth stands.

Further designating the individual strata is more time-consuming for areas where selection harvest has occurred, especially if GIS stand data are unavailable. In this situation, use ocular stratification by tree composition and varying snag densities. For example, in a subwatershed composed of a mix of ponderosa pine and lodgepole pine (*Pinus contorta* Dougl. ex Loud.) stands, three strata might be possible: (1) stands dominated by ponderosa pine with few snags observed; (2) stands represented by co-dominance of ponderosa and lodgepole pine trees, usually with 1 or 2 snags per acre observed; and (3) stands dominated by lodgepole pine with 5 or more snags per acre observed.

The primary criterion in stratifying a subwatershed is the **sampling objective**. If sampling is intended to estimate the density of large trees only, stratification is dictated solely by this variable. If both large trees and snags will be sampled, base the stratification on which structure varies most in abundance. The secondary criterion is either **seral stage** or **timber harvest technique**, which affect not only precision but also the level of sampling difficulty during fieldwork. The tertiary criterion is **forest community type**, especially for stands affected by insect- or disease-based mortality events. Certain tree species are more susceptible to insects or diseases, and these stands will have higher densities of snags, such as mixed stands of lodgepole and ponderosa pine. Finally, consider **land management use**. Do you need separate estimates for areas that are managed for different purposes (for example, riparian versus timber production areas)?

Establishing Transects (Step 4)

Conducting a pilot survey is one of the most important steps of any snag or tree survey. In a pilot survey, there are two primary objectives:

- Collect preliminary data by which to identify the optimal plot size.
- Obtain an estimate of the total number of samples required to meet a user's objectives.

Pilot data are not extraneous data to be discarded. Rather, they are the first samples collected, and are included in the variable estimates for the entire sampling area. In areas where snags or trees in the targeted size classes are abundant, the pilot survey may provide an adequate number of samples to meet a user's objectives. By contrast, in areas where snag or tree abundance is low, analyzing the pilot data to determine the optimal plot size can minimize the number of samples needed to achieve the desired precision. Use the optimal plot size to collect the remainder of the data.

We designed the transects for snag and tree sampling to be compatible with transects used for log sampling, thus improving the efficiency of the fieldwork.

We designed the transects for snag and tree sampling to be compatible with transects used for log sampling (Bate and others 2008), thus improving the efficiency of the fieldwork by allowing all three structural components to be sampled simultaneously. The original snag and large-tree sampling protocol recommended 800 ft (200 m) within each stand on stratified landscapes (Bate and others 1999); however, instead of using the single 800-ft (200-m) transect, split it into two 400-ft (100-m)-long sections called transects. These two smaller transects capture more of the variability occurring in a single stand and increase compatibility with log sampling (Bate and others 2008). Subdivide each transect into smaller increments, called subsegments, and sample for the three habitat components of snags, large trees, and logs. This standardizes the sampling protocol and allows SnagPRO to determine the optimal transect length for each variable in relation to the specific forest conditions.

Two options exist for establishing transects: the single-stratum landscape method and the stratified method. For the single-stratum landscape method, follow these steps to establish transects within a single stand or a nonstratified landscape:

1. Randomly place a grid over the area.
2. Randomly select 10 grid points for sampling.
3. Randomly select compass bearings for each of the 10 transect starting points.

For the stratified method on heterogeneous landscapes composed of numerous stands or units, it may be more efficient to randomly select stands for sampling. To do this:

1. Select stands for sampling by randomly picking stand unit numbers from the complete list of stands within that stratum.
2. Place a grid over the stand.
3. Randomly pick two grid points within each stand.
4. Randomly pick compass bearings for each point.

Use a random number generator or random numbers table for either method, or generate random numbers for compass bearings using the second hand of a watch. If a watch is used to generate random starting direction, multiply the number of seconds (60) by six to obtain numbers from 6 to 360 that can be used as compass bearings for the starting point.

The pilot survey should include:

- A minimum of two transects per stand (fig. 1) to adequately represent the variability in each stand and stratum, providing a better estimate of the sample size required to meet objectives.
- A total of at least eight 400-ft-long transects (English users), or ten 100-m-long transects (metric users), within each stratum.

When establishing transects, it is important to realize that the equations used in SnagPRO assume a normal distribution (Krebs 1989). However, snags are rarely normally distributed, instead occurring in clumps. Therefore, a minimum of 60 samples is usually needed to achieve a normal distribution. Users should consult with a statistician if they are unsure as to whether their data are normally distributed in relation to the number of samples. Avoid overlapping the transects because the equations assume that no snags or trees are sampled more than once.

For larger subwatersheds, the stands in the pilot survey should not be close together, especially for subwatersheds encompassing several plant communities.

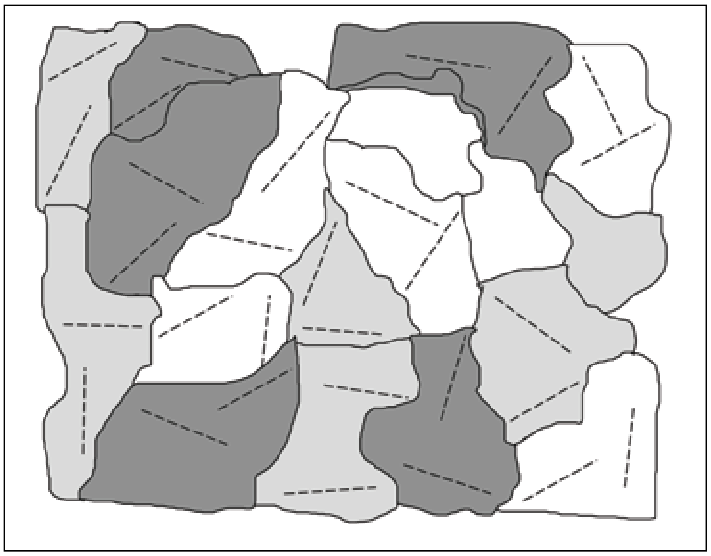


Figure 1—Illustration of transect establishment for snag or large-tree pilot survey on a landscape with three strata. Five stands within each stratum should be selected. Within each stand, two 100-m or 400-ft transects are established. Each transect within the entire landscape is given a unique numeric identifier and is divided into eight 12.5-m or 50-ft-long subsegments. Subsegments are numbered from 1 to 8 on each transect.

In this situation, divide the subwatershed into three sections and equally divide the samples throughout the sections.

Field Techniques (Step 5)

Fieldwork requires some or all of the field equipment listed in table 1. Where shrub cover is thick, the 100- or 200-ft (English users) or 50-m (metric users) fiberglass surveyor's tape (with a logger's nail taped to one end) is very efficient for marking the center transect line. One person walks the centerline, locating targeted size classes of snags or trees, and taking all measurements. The second person ensures quality control and records data on field forms.

Quality control is best accomplished by having the data recorder walk some distance away from the centerline.

Quality control is best accomplished by having the data recorder walk some distance away from the centerline. This ensures that all snags on the centerline are counted (surprisingly, snags on the centerline are the ones most likely to be missed because observers look mainly to the side). The data recorder also helps ensure that the tape is held perpendicular to the centerline when measuring the distance of a snag or tree from the line. Relascopes may also be used to gauge the distance of a snag or tree from the transect line. However, a relascope estimates the distance from the closest edge of a snag or tree and not to its central axis. Therefore, measuring the actual distance with a logger's tape is needed for all borderline cases.

Table 1—Field equipment for snag and tree sampling

Item	Use
Accurate map of polygon units or vegetation cover types	Record correct stratum number
Road map	Determine location and access
Aerial photographs	Determine stratum and locations
Orthophoto quads	Determine stratum and locations
Field data forms (hard or electronic)	Record survey information
Engineer's surveyor tape (50 m or 100 or 200 ft long)	Measure transect distances; mark centerline
Logger's tape	Measure distance of snag or tree from transect or required distance away for heights
Calipers	Measure diameter of snags or trees
Relascope	Measure distance of snag or tree from transect or required distance away for heights
Compass	Determine bearings
Pocket knife	Determine species and decay class of snags
Flagging	Mark ends of subsegments, if necessary

SnagPRO's standardized field forms include the snag or tree information needed for all analyses. Field forms can be customized for each location and survey. For simultaneous collection of data on snags, trees, and logs, data are recorded in separate files for each component.

We found that hand-held computers are useful for fieldwork, and SnagPRO is designed accordingly. Users can avoid entering data twice by using the Data_entry worksheet to enter data directly on a hand-held computer while sampling in the field. The Date_entry worksheet is found in the Snag_Tutorial_Data.xls file. If hand-held computers are not used for fieldwork, create hardcopy field forms from the worksheet labeled "Field form" found within the Snag_Tutorial_Data.xls file. Open the Snag_Tutorial_Data.xls file, highlight the entire page that has gridlines, and choose **Selection**, instead of Sheet, under the Print options for a hardcopy form with gridlines.

Appendix 2 provides a sampling protocol to collect data for snag and large-tree habitat. Copy this appendix to a new file and customize it for your fieldwork. Customizing options include:

- Defining a qualifying snag or tree by diameter and height.
- Using either numeric or four- to six-letter alpha codes for snag and tree species.
- Altering data requirements for each variable to meet sampling objectives, such as recording heights to the nearest foot or meter.
- Defining snag decay classes or tree structural classes.
- Recording wildlife signs, if desired.

Default plot sizes—

There are four default widths available for both English (fig. 2) and metric systems (fig. 3): 33, 66, 99, and 132 ft; and 10, 20, 30, and 40 m. These widths are whole widths of the plot, measuring from one side, across the centerline, to the other side. In SnagPRO, these plot sizes in English units are labeled Width33, Width66, Width99, and Width132. For metric units, they are labeled Width10, Width20, Width30, and Width40.

The half-width is half the distance from the centerline in which you count all snags or trees, based on the chosen plot width. The half-width distance of these plots is 16.5, 33, 49.5, and 66 ft (or 5, 10, 15, and 20 m). When measuring distance, be sure to measure all snags and trees to their midpoints, or central axis.

Four default plot lengths are available: 50, 100, 200, and 400 ft (12.5, 25, 50, and 100 m). For studies that use only one transect length such as segments (100-ft or 25-m lengths), it is still necessary to assign a transect and subsegment (50-ft or 12.5-m length) number to each length and keep track of the smallest increments (subsegments). Later, users may indicate in SnagPRO that only segment lengths are desired.

Custom plot sizes—

Different sampling objectives may require different plot sizes. Remember that for optimal transect length analyses, transects should be twice as long as sections; sections twice as long as segments; and segments twice as long as subsegments. SnagPRO can also accept customized plot widths. Both customized widths and lengths are adjusted under Custom Dimensions found under the Plot Dimensions menu.

Survey—

Conduct the pilot survey to determine the optimal plot size with these steps:

1. Use an engineer's surveying or measuring tape to establish transects, starting each transect from the randomly selected points (described above).
2. Assign a unique numeric identifier to each transect, delineating the subsegment lengths (50 ft [or 12.5 m]) as you walk along the transect (400 ft or 100 m).
3. Number each transect's subsegments 1 through 8.
4. Conduct a complete count of all snags or trees of interest within 66 ft (20 m) of each side of the centerline, using the tape as centerline. A snag or tree is "in" if its midpoint is ≤66 ft (20 m), as measured perpendicularly, from the centerline.

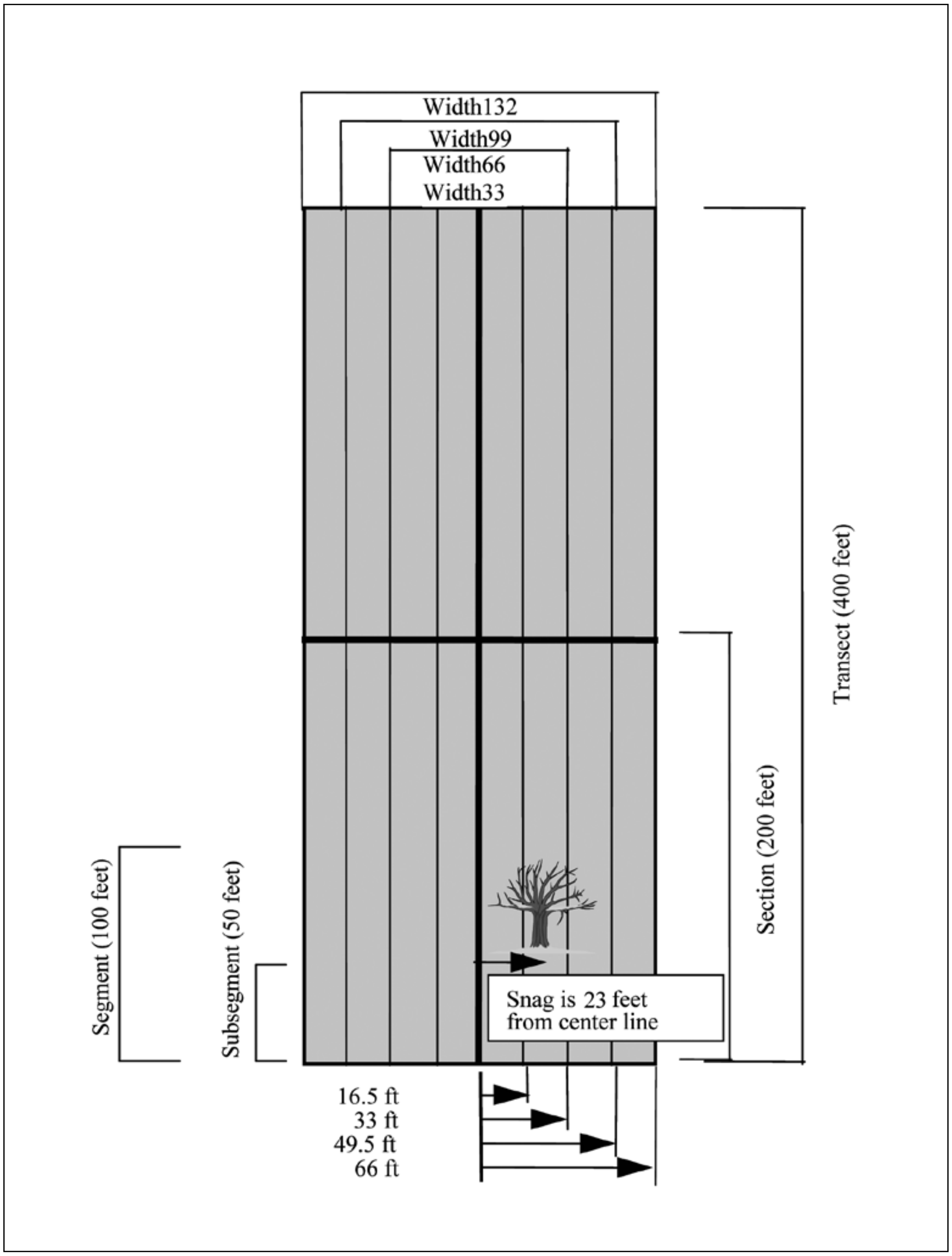


Figure 2—There are 16 default plot sizes available using the English measurement system. During the pilot survey, the distance of all qualifying snags or trees is measured in the maximum half width (66 ft). Then after the optimal plot size has been determined, this optimal plot width is used for the remainder of the survey. The Width33 to Width132 refer to the entire width of the plot (in feet). Therefore, only snags or trees whose measured distance (from centerline) is ≤ half of the entire width qualify for a specific plot size when determining the optimal plot size. For example, a snag that is 23 ft from the centerline would be counted in the Width66, Width99, and Width132 plots. It does not qualify, however, for the Width33 plot because its distance is more than 16.5 ft (the half width) from the centerline.

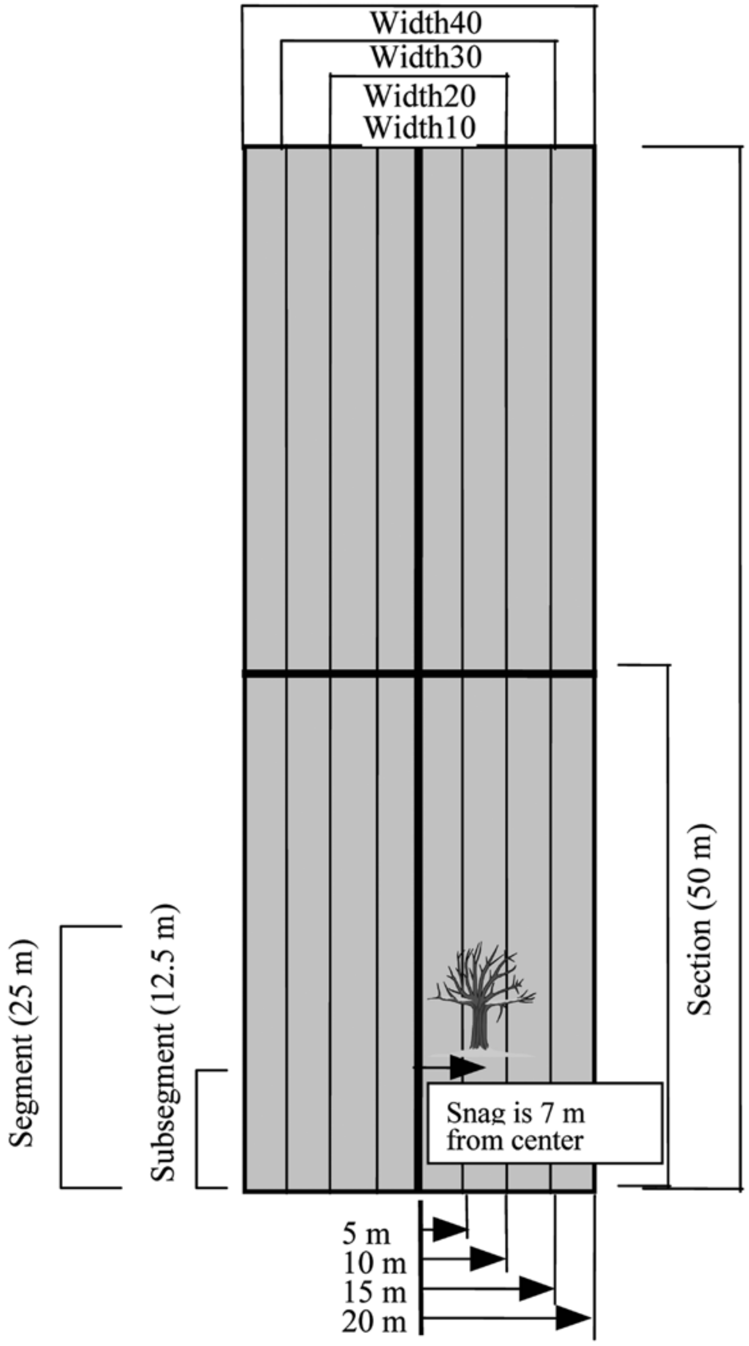


Figure 3—There are 16 default plot sizes available using the metric measurement system. During the pilot survey, the distance of all qualifying snags or trees is measured in the maximum half width (20 m). Then after the optimal plot size has been determined, this optimal plot width is used for the remainder of the survey. The Width10 to Width40 plots refer to the entire width of the plot (in meters). Therefore, only snags or trees whose measured distance (from centerline) is ≤ half of the entire width qualify for a specific plot size when determining the optimal plot size. For example, a snag that is 7 m from the centerline would be counted in the Width20, Width30, and Width40 plots. It does not qualify, however, for the Width10 plot because its distance is more than 5 m (the half width) from the centerline.

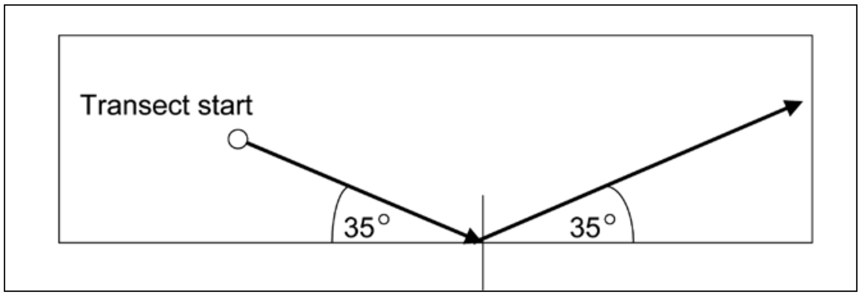


Figure 4—Illustration of randomly oriented transect hitting the edge and "bouncing" back within the sampling area. This ensures edges are included in the sampling population, while maintaining the option to analyze data for the optimal transect length.

Exceptions—

Modify the plot width when sampling visibility is ≤33 ft [10 m]) or high densities of snags or trees result in >15 snags or trees recorded in each plot. In these conditions, only count snags and trees out 33 ft (10 m) from either side of the centerline for the optimal plot size analysis. See "Presampling plot size selection" below for other exceptions.

Occasionally, the random compass bearing for a transect will cause it to continue outside the boundary of the sampling area. Use the "bounce back" method to keep the transect within the stand. The bounce-back method is similar to hitting a billiard ball or racquetball against a sidewall, and having it travel back away from the wall at the same angle. In your sample area, determine the angle at which the transect hits the edge, then use this same angle to continue (fig. 4) back into the sample area. This technique allows resource specialists to determine the optimal length and include the edges of the stand in the sampling pool.

Presampling plot size selection—

Rather than conducting the optimal plot size analysis (See Optimal Plot Size in the "SnagPRO Analysis" section), it may be more practical to preselect a plot size for sampling, based on information gathered during the stratification process. For example, in a clearcut stratum where travel is easy and snag density is low (<1 snag per acre [0.4 snag/ha]), a statistical review of the data may reveal little about the optimal plot size. In such cases, use a wider plot (132 ft or 40 m wide) to collect data efficiently on as many snags as possible. For small clumps of snags retained within harvest units, recording the distance from the centerline to each snag may be advantageous if the variability proves too great with use of the wider plots. This will enable you to use only snags in a narrower plot if deemed necessary.

In mature/old-growth forests, 20- by 50-m plots are commonly used to sample snags, especially where terrain is steep and rugged, or vegetation is dense. In these conditions, the wider plots are inefficient and prone to inaccuracies. By contrast, the narrower plots make it easier to detect snags and large trees within the plot boundaries and to measure their distances.

In areas where snag numbers tend to be low, but clumps of snags are present, the longer, narrower plots work best to minimize the variance. Burned habitats with high densities of evenly distributed snags present a different challenge. Here, the shorter plots would likely work best. This is also true for sampling live trees that are high in abundance and evenly distributed.

The advantage of preselecting a sample plot width is that the distance of each snag or tree from the centerline does not have to be recorded, saving time in the field during the pilot survey. The disadvantage is that users lose the option of determining the optimal plot width. Note that once you determine the optimal plot width, there is no need to continue measuring distances; enter a “1” in the distance column as a placeholder after a plot width is established.

SnagPRO guides users in their selection of the optimal plot size.

Postsampling selection—

For many conditions, the optimal plot size for sampling is unknown until a pilot sample provides estimates of the density and distribution of the snags or trees in the area. This is a key strength of SnagPRO in that it guides users in their selection of the optimal plot size that minimizes the sampling effort while attaining the sampling objectives. Once the plot optimal width is identified, the Optimal Plot Size option can continue to be used for analysis of optimal length, but without the need to measure the distance of each snag or tree from the centerline. See the “Optimal Plot Size” section for details.

Distribution and abundance of snags or trees influence the optimal plot size. Generally, the more abundant the habitat component of interest, the smaller the plot needed. In areas of clumped snags or trees of high abundance, narrower plots are the better choice. How narrow can be difficult to ascertain without first collecting pilot data. In the pilot survey, collect data for the length and at both the 10- and 20-m (33- and 66-ft) widths.

Data Collection

The following are mandatory fields requiring information for SnagPRO to operate correctly (fig. 5). Refer to appendix 2 for details about each field variable.

For each qualifying snag or tree (that meets the stipulated criteria) along a transect, record the following:

1. Stratum number
2. Transect number
3. Subsegment number

Stratum	Location	Transect	Subsegment	Distance	Species	Class	Dbh	Height	Cavity	Forage
1	2345	1	1	9999						
1	2345	1	2	2	2	1	64	32.6	1	3
1	2345	1	3	5	2	5	41	2.7	2	4
1	2345	1	4	13	2	2	74	26.5	1	1
1	2345	1	5	7	2	3	76	17.7	5	4
1	2345	1	6	11	2	2	69	25.9	4	3
1	2345	1	7	8	2	2	61	23.8	3	4
1	2345	1	8	9999						
1	2345	2	1	6	2	2	33	19.2	4	4
1	2345	2	2	10	2	3	46	13.7	4	2
1	2345	2	3	3	2	5	28	2.4	4	3
1	2345	2	4	11	2	5	31	9.8	4	4
1	2345	2	5	15	2	4	89	15.2	4	4
1	2345	2	6	10	2	4	41	15.2	4	4
1	2345	2	7	8	2	4	38	13.7	4	2
1	2345	2	8	0	2	3	61	14.3	2	4
1	4567	3	1	12	2	4	48	19.2	5	2
1	4567	3	2	1	2	1	94	38.1	5	4
1	4567	3	3	9	2	3	56	35.4	5	3
1	4567	3	4	2	2	5	56	2.4	5	4
1	4567	3	5	3	2	3	102	30.8	2	4
1	4567	3	6	1	2	1	94	38.1	5	4
1	4567	3	7	9999						
1	4567	3	8	10	2	1	32	26.2	4	3
1	4567	4	1	9999						
1	4567	4	2	9999						
1	4567	4	3	9999						
1	4567	4	4	9	2	4	76	29	3	3
1	4567	4	5	4	2	5	28	2.1	4	2
1	4567	4	6	16	2	2	33	12.5	4	3
1	4567	4	7	3	2	3	102	30.8	2	4
1	4567	4	8	9999						
1	6789	5	1	5	2	5	41	2.7	2	4
1	6789	5	2	9999						
1	6789	5	3	9999						
1	6789	5	4	4	2	2	33	12.5	4	3
1	6789	5	5	9999						
1	6789	5	6	12	2	3	49	30	4	4
1	6789	5	7	5	2	3	64	37.2	4	4
1	6789	5	8	9999						
1	6789	6	1	0	2	2	61	23.5	4	2
1	6789	6	2	9	2	2	64	16.2	5	2
1	6789	6	3	15	2	3	43	15.5	4	2
1	6789	6	4	9999						
1	6789	6	5	8	2	2	71	32	3	4
1	6789	6	6	6	2	1	58	32.6	4	2
1	6789	6	7	10	2	3	46	13.7	4	2
1	6789	6	8	11	2	2	48	21.3	4	1

Figure 5—Example of properly formatted data. This format is required before saving as a comma-separated, .csv file and importing to SnagPRO. Class refers to decay (snags) or structural (trees) stage.

4. Perpendicular distance of the midpoint of the snag or tree from the centerline (when using preselected widths just enter “1” as a place holder)
5. Species
6. Decay (snags) or structural (large tree) class
7. D.b.h.
8. Height (for surveys where all snags ≥6 ft (1.8 m) are recorded, record a minimum height).

Optional fields are Location, Cavity, and Forage. Location can correspond to (1) the stand number from which the transect originates, (2) the transect starting position determined by a global positioning system (GPS), or (3) the universal transverse meridian (UTM) coordinates of the transect starting point.

User-defined fields may also be recorded during surveys, but only include this data in columns to the right of those in the CSV file (fig. 5) that are needed for importing to SnagPRO. Additional habitat variables can be added, such as seral stage of the stand, distance to the nearest edge, and immediate habitat surrounding a snag or tree.

Distance is the most important variable, so take care to record it accurately. For cases where snags or trees are not encountered, record “9999” in the distance column. This is a critical step; it allows SnagPRO to distinguish plots without snags or trees from plots that have snags or trees with a distance of “0” because the structures are located directly on the centerline.

Distance measurements should be checked periodically by the person overseeing the fieldwork. Consistently over- or underestimating this variable will affect results. Borderline cases, in which the distance of the surveyed snag or tree falls on the edge of a width interval, need to be carefully checked. For example, if you estimate a tree to be 33 ft (10 m) away from the line, it is important to measure this distance exactly. Recording 33 ft (10 m) when the actual distance is 36 ft (11 m) biases the accuracy of your data. Estimating the distance by pacing often is accurate in open, flat areas, but for borderline cases, measuring the distance with a tape is required. In addition, when vegetation or steep terrain make pacing difficult, measure rather than estimate the distances to maintain accuracy.

Header row variables may also be recorded for each snag or tree encountered: (1) Forest, (2) District, (3) Subwatershed, (4) Observer, (5) Date, and (6) Pages. Because the data recorded for each of these variables may be redundant, the columns are set to the far right of the data entry spreadsheet. This enables easy viewing of the data while providing a permanent record of each of these variables for future referencing.

As with the log sampling protocol (Bate and others 2008), we recommend sampling 10 transects (4,000 ft or 1000 m) within each stratum for a pilot sample. For smaller trees and abundant snags, these samples are all that may be needed if the snag or tree size of interest is also normally distributed.

Although results from the pilot survey will identify an optimal plot length, we recommend continuing to sample afterwards with 400-ft or 100-m transects rather than switching to a shorter length, **unless** a serial correlation problem is detected (See "SnagPRO Analysis" section for more details). There are two reasons for this approach. First, snags and trees in different size classes usually differ in abundance and distribution; hence the optimal transect length for each differs. Second, most of the time required to sample snags and trees is to locate random points and establish transects. Transect lengths of 400 ft or 100 m are long enough to be efficient, yet short enough to ensure that sampling effort is not concentrated within a small area. Make sure to continue sampling with only the recommended width to save time in the field.

SnagPRO Analysis (Step 6)

In this section, we provide the general background, statistics, and discussion of each function and page within SnagPRO. Refer to the "Tutorial" section for detailed operating instructions and examples. See appendixes 3 and 4 for brief outlines of steps needed to conduct analyses on single-stratum and stratified landscapes, respectively.

No two data sets will be the same size. Data sets will differ depending on snag or tree characteristics, the abundance of qualifying snags or trees, number of strata, and the total number of samples taken. SnagPRO has been designed to accommodate these variations.

SnagPRO has been designed to accommodate variations in data sets including tree characteristics and abundance, number of strata, and number of samples.

Data entry—

To prepare for data entry and analysis, follow these steps:

1. Open the **Snag_Tutorial_Data.xls** file.
2. Activate the **Data Entry** sheet.
3. Click on **Move or Copy Sheet** under the **Edit** menu.
4. Check the box **Create a copy**.
5. Under **To book** click on (**new book**).
6. Rename the new file, and then use this sheet to make hardcopies for fieldwork.

To use hand-held computers during fieldwork, activate the data sheet and complete the process from step 3. Depending on the sampling objectives, not all fields on the data form may be necessary during field surveys or data entry, and you may choose to hide some columns. All mandatory columns, however, must be present (Unhide) in the CSV import file (fig. 5) or the SnagPRO import will fail.

To save the entered data as a CSV file:

1. Activate the **Data Entry** sheet.
2. Select **Save As** from the **File** menu.
3. Scroll to find CSV (comma delimited) (***.csv**).
4. Click **Save.**

Only the active sheet is saved. This keeps the original file intact, by saving the file with a different extension. Figure 5 illustrates the correct formatting needed to successfully import to SnagPRO.

Consecutive plots—

Scroll through the entire data set before importing it to SnagPRO to ensure that each transect has a unique numeric identifier and eight subsegment lengths, with the first subsegment numbered as "1." Otherwise, the analysis for optimal transect length will join subsegments from different transects.

Importing files—

To import data to SnagPRO, the application prompts users for some initial information. For example, the first message box to appear in SnagPRO asks users to indicate what habitat component will be analyzed:

- Logs
- Snags or Trees

Select **Snags or Trees** so that SnagPRO will expect the specific field names and column arrangement from the import file. SnagPRO opens the Snag and Tree Analysis portion. Selecting Logs will cause the SnagPRO import to fail. See Bate and others (2008) for correct formatting of log data.

This opens to a window that says "SnagPRO-Snag and Tree Analysis:"

1. From the **Measurement** menu, select **Metric** or **English**.
2. From the **File** menu, select **Open**.
3. Navigate to the location of the saved CSV data file, and select the file by clicking on **Open**.

Correctly formatted files will open promptly to the Single/Combined page in SnagPRO with the message, "Status: Data file read" in the bottom left-hand corner. This page is where the entire data set is stored while working in SnagPRO.

If SnagPRO fails to import the file, the message, "An invalid column header was found" will appear. If users know they selected the correct file to import the first time, there may be a problem with formatting. Copy the entire data set into a new file, including only the rows and columns with data. Then repeat the process above.

SnagPRO automatically inserts two "length" columns into the data set after a successful import, labeled Section and Segment. SnagPRO combines the subsegments of varying lengths into newly created sections and segments, resulting in four transect lengths: 50, 100, 200, and 400 ft, or 12.5, 25, 50, and 100 m. SnagPRO also inserts four "width" columns, labeled Width33, Width66, Width99, and Width132 for English, and Width10, Width20, Width30, and Width40 for metric measurement. These fields are later populated by your choice of formula. See "Formulas" below.

Default plot dimensions—

A user's sampling objectives may require different plot dimensions. To override SnagPRO's defaults, navigate to **Plot Dimensions** and select **Custom Dimensions**, then place the cursor within each box to enter the correct length(s) and/or width(s). Remember that for optimal transect length analyses, transects should be twice as long as sections, sections twice as long as segments, and segments twice as long as subsegments.

For optimal transect length analyses, transects should be twice as long as sections, sections twice as long as segments, and segments twice as long as subsegments.

Preselected transect lengths and widths—

For analyses using a single transect length and width, navigate to **Settings** and select **Optimal Selection**. Check the plot dimension to be included in the analysis. Check **Automatic** to again include all plot sizes in the analysis. If you did not collect data using long transects, but wanted only segment lengths, data entry must follow the same protocol for SnagPRO analysis. That is, still identify each transect with a unique numeric identifier, and then divide into smaller subsegments. During the CSV import, SnagPRO creates and populates the Segment column, so users only need to check it to include it for the analysis.

Species—

Users may select from three options for analyzing snag or tree data by species:

1. All species
2. One species, excluding all others
3. Exclude a single species

SnagPRO's default values include all species in the analysis, providing a choice to exclude a single species (**Multiple** button). For analysis of a single species, select **Single** at the bottom of the screen.

Formulas

SnagPRO evaluates each snag or tree by using five criteria for data entries before a value based on a formula is placed in each of the width columns:

1. D.b.h.
2. Height
3. Class: decay (snags) or structural (trees)
4. Species
5. Distance

The first four criteria are relative fields. The values accepted are those entered by the user.

The distance criterion is an absolute field. SnagPRO truncates the distances for each snag or tree in 16.5-ft (5-m) intervals from the centerline, creating four plot widths. These four plot widths correspond to the four width columns that are blank.

SnagPRO's formulas are "If, then" statements. **If** the snag or tree meets all criteria specified by the user, **plus** meets the distance requirement, **then** a "1" is placed in that specific width column. **If** the snag or tree fails to meet all specified criteria, **then** it places a "0" in the column.

Only those snags or trees meeting **all** requirements of the user-created formula receive a "1" and are included in the statistical analysis for each plot size. For example, if the perpendicular distance (half width) of a snag from the centerline was measured at 23 ft (7 m) and it meets all other criteria, the snag qualifies for the Width66(20), Width99(30), and Width132(40) columns. The snag is not included in the Width33(10) column because it falls beyond that width interval around the centerline. The Width33(10) column represents a plot with a 33-ft (10-m) total width, or a 16.5-ft (5-m) half width. Only snags or trees ≤16.5 ft (5 m) away from the centerline, in either direction, will be accepted within the Width33(10) column. See figures 2 and 3 as examples.

SnagPRO evaluates d.b.h. and height characteristics based on the minimum value the user provides, and decay or structural classes based on a maximum value specified by the user. Decay values are based on Cline and others (1980), who considered decay classes I through III as hard snags, and decay classes IV and V as soft snags, based on their five-decay-class system. If only hard snags are to be included, enter a "3" as the maximum value.

Parks and others (1997) reduced the total number of decay classes to three by combining Cline and other's (1980) decay classes 2 and 3, and classes 4 and 5. The division between hard and soft snags therefore remains the same, so the two systems are compatible. Bull and others (1997) also provided descriptions of large-tree structural variations important to wildlife. For example, hollow trees, trees with partial decay, and trees with brooms provide valuable wildlife habitats. Use of structural classes can include such important information.

We recommend referring to Bull and others (1997) before starting a large-tree survey to ensure that large-tree structural classes are designed to meet objectives. See appendix 2 for an example. Values used for large-tree structural classes should be arranged so that with increasing values, the tree is increasingly sound. This is the opposite of the decay class values for snags.

Cavity and Foraging Signs

The objectives of snag or tree sampling may include collection of data on wildlife use. For example, to determine the snag species in which most of the cavities exist, use SnagPRO's **Cavity** function. Or, to determine which tree species exhibits the most foraging, choose **Forage** for your analysis.

To determine a value for Percent Use, each function evaluates every snag or tree for the following five factors:

1. D.b.h.
2. Height.
3. Class: decay (snag) or structural (tree).
4. Species.
5. Cavity or foraging use.

Percent Use is calculated by dividing the number of snags or trees with cavities or foraging signs by the total number of snags or trees encountered.

$$P_u = \frac{S_s}{S_t} \qquad (1)$$

where

P_u = percentage of use,

S_s = number of snags or trees with nesting or foraging signs, and

S_t = total number of snags or trees encountered.

In the field, it is not always possible to determine whether a snag or tree has a cavity because the bole is partially hidden. To exclude such snags in the calculation of percentage use, leave the cell blank.

Few subwatersheds will be homogeneous enough to forgo stratification.

Sorting Data Sets

Few subwatersheds will be homogeneous enough to forgo stratification. In addition, the optimal plot size will likely differ among heterogeneous strata, because of differences in the means and variances. Consequently, data need to be separated so that each stratum can be analyzed individually. SnagPRO automatically sorts data sets into separate strata once the Single or Multiple button is clicked and values are placed in the Width columns. The entire data set is retained on the Single/Combined page.

SnagPRO automatically sorts data into separate strata so that each stratum can be analyzed individually. The entire data set remains on the Single/Combined page and can be analyzed as a single stratum. This is helpful for situations in which it is not certain whether stratification was helpful in increasing precision.

Analysis for Nonstratified Stand or Landscape

Before means, standard deviations, and sample sizes are computed, the values in the Width columns need to be summed and subtotaled for each of the four transect lengths. The means, standard deviations, current number of samples, and sample size required are then calculated from these values.

Once the Width columns are populated, sum the qualifying snags or trees for a **nonstratified stand or landscape** with these steps:

1. Click on the **Optimal** tab.
2. Select **Single** in the "Stratum to Process" section (highlight the circle).
3. Click **Compute**.

SnagPRO calculates subtotals, displaying the average, standard deviation (std. dev.), and current number of samples (N) for each plot size on the Summary Statistics page. These calculated averages, standard deviations, and current number of samples also are copied to the Optimal page.

Analysis for individual strata on a stratified landscape—

Sum the qualifying snags or trees for each stratum on a stratified landscape with these steps:

1. Click on the **Optimal** tab.
2. Select **Stratum 1** in the "Stratum to Process" section.
3. Click **Compute**.

SnagPRO prompts users for:

- Number of strata
- Numeric code for the General Cost per Sample Guideline (see the "Optimal Plot Size" section for details) for stratum 1
- Size of each stratum (acres or hectares).

Repeat the process above for each additional stratum. Again, the results are displayed on the Summary Statistics and Optimal pages.

Averages are calculated by using the equation:

$$\bar{x} = \frac{\sum x_i}{n} \qquad (2)$$

where

$\bar{x}$ = sample mean,

x_i = value of x observed in sample *i,* and

n = total number of samples.

Standard deviations are obtained by the equation:

$$s = \sqrt{\frac{n\sum x_i^2 - \left(\sum x_i\right)^2}{n(n-1)}} \qquad (3)$$

where

s = sample standard deviation.

Optimal Plot Size

After running the summary statistics, SnagPRO provides the information on the Optimal page to aid users in selecting the optimal plot size. The statistics for each subsegment, segment, section, and transect within a unique stratum are displayed across three consecutive pages. For each stratum, the optimal plot size analysis needs to be run separately. The Stratum box immediately above the Optimal tabs allows for a text description of the stratum, which is useful when working with different size classes and for multiple strata. Users may print these results before proceeding to other strata.

There are two options to determine the optimal plot size for a stratum or area. The sample size option examines the number of plots required for sampling in comparison to the number of acres or hectares that would be sampled using that plot size. The second option is Wiegert's (1962) method, which incorporates a cost factor into the analysis. Default settings for both options estimate the number of samples needed to obtain a density estimate that is within 20 percent of the true mean, 90 percent of the time.

Weigert's method incorporates a cost factor into sample size analysis.

The sample size option considers three factors:

- The sample size required in plots.
- The sample size required in acres (ha).
- Whether the estimated sample size (n) meets the minimum requirements.

The required sample size (Cochran 1977) is determined by:

$$n = \left(\frac{t_\alpha s}{d}\right)^2 \qquad (4)$$

where

n = sample size required to estimate the mean density,

s = standard deviation of the mean within each plot size,

t_{α} = student's t-value for a 90-percent confidence interval ($\alpha = 0.10$), and

d = desired absolute error (calculated as 20 percent of the pilot mean).

After selecting **Optimal 2** tab, locate the required number of sample plots in the Sample Size (plots) column. Then this number is converted to the area requirements and reported in the Sample Size (acres [ha]) column.

The optimal plot size typically is one that requires sampling the minimal number of plots and acres (ha), once the requirements are met, but several sizes may be appropriate. Results from the pilot sample should enable users to determine if it is more efficient and accurate to use a wider or a narrower plot width, given the forest conditions of this stratum.

Example: Results on the Optimal 2 sheet estimate that a Section length with a 132-ft (40-m) plot width requires 20 acres (49 ha) more than one that is only 66 ft (20 m) wide to obtain the same level of precision. This difference may or may not take a considerable amount of time to survey, depending on sampling conditions. In areas where snags or trees are present in low densities, and visibility is open to 66 ft (20 m) from the centerline, the Section lengths that are 132 ft (40 m) wide would probably be the best choice to ensure that every possible snag is surveyed. By contrast, if snag or tree densities are moderate or high, narrower plots are likely the better choice. This will reduce sampling effort by decreasing the amount of time required to complete each plot.

Users also need to consider how difficult it is to see and reach snags or trees out to the specified distance. In steep terrain where shorter snags may be obscured by vegetation, it is important to select a narrower plot size to maintain accuracy.

The option that incorporates Wiegert's (1962) method demonstrates that the optimal plot size is that which minimizes the product of the Relative Cost and the Relative Variance. If both relative costs and variances are available, Wiegert's method is considered preferable (Krebs 1989).

Although it is more accurate to estimate costs by conducting time trials, there are some logistical difficulties. Time trials are conducted by surveying one plot size at a time, and cannot be conducted simultaneously with the pilot survey, which usually requires surveying snags or large trees to 66 ft (20 m) from the centerline. Consequently, the time and costs to conduct time trials may quickly offset any benefits because of the additional field effort required beyond the pilot survey. Therefore, we recommend against conducting time trials. Instead, we suggest using the cost factors provided in SnagPRO, which were developed on a relative scale. We have outlined our methods below so that users may understand how the costs were derived.

There are six cost scenarios in SnagPRO. Click on **Compute** and choose the category that best describes the forest situation within each stratum to see how the relative cost affects the outcome of your decision about the optimal plot size. The cost data are transferred to the Optimal page. Try several Costs per Sample categories if forest conditions are between categories.

We modified Wiegert's method by calculating the total cost expected for each plot size selected. This is valuable information, because all costs are relative to each other within the same area. Total cost also incorporates the minimum sample size required. Although the actual cost for sampling will vary for a particular area, selecting the plot size that demonstrates the lowest total cost allows users to select the optimal plot size for the forest conditions sampled.

The cost per plot is mainly a function of three factors: visibility, terrain, and density.

Estimating Costs

The cost per plot is mainly a function of three factors: visibility, terrain, and density. Visibility is the unobstructed viewing distance from centerline to snags or trees for a given stratum. It is most strongly affected by seral stage or young tree or shrub cover within a stand. Terrain includes slope, young tree or shrub cover, and density of logs, all of which affect difficulty in traversing an area. Density is the number of snags or trees per unit area for a given size class of interest. Stands of higher density will require substantially more time, and thus higher costs, to sample.

Each survey has an associated fixed cost and a cost estimate for each plot, calculated at $10 per observer/hour. Fixed Cost is the time spent selecting and locating each beginning transect point, including the time spent returning to a vehicle upon completion. Costs may be minimal in clearcut areas or quite high in areas with difficult terrain. We therefore calculated an average fixed cost for each of the four possible plot lengths (transect, section, segment, and subsegment) based on a moderate situation with a low snag density.

SnagPRO's cost estimates were based on time trials conducted in forests in northern Idaho, adjusted with cost estimates from a snag study in the central Oregon Cascade Range. During the time trials, all snags >10 in (25 cm) d.b.h. were surveyed. For each snag, we recorded the following snag characteristics: species, d.b.h. (measured with a Biltmore diameter stick), decay class, height (ocular estimate), distance of the midpoint of the snag from the centerline, nesting evidence (ocular), and foraging signs (ocular).

In flat, open areas, distances to each snag were either paced or measured with a tape. Paced distances were calibrated with a measuring tape to ensure accurate and precise estimation of distances. Paced distances, following calibration, were then used to count snags that were clearly within the plot width, but snags potentially on the boundary, referred to as “marginal,” were always measured with a tape. For example, a snag was considered marginal if its distance of 16.5 or 18 ft (5 or 5.5 m) away from the line was uncertain. For these cases, all distances were measured. In addition, periodic calibration of pacing was conducted. In steep areas, all distances >12 ft (3.5 m) were measured.

We developed six hypothetical situations based on three categories of terrain and snag visibility coupled with two snag densities. “Easy” refers to an area that is relatively flat (<30 percent slope) and where snags or trees are easily observed to 66 ft (20 m) in both directions. “Moderate” refers to situations where the slope is 30 to 50 percent and visibility of snags or trees averages 50 ft (15 m). “Difficult” describes situations in which a combination of factors makes travel difficult and slow and visibility is low. In dense regeneration stands, it is not possible to accurately detect snags beyond a particular distance. Travel could be difficult owing to slope, type of seral stage, amount of shrub cover, or density of logs.

Cost estimates were then developed for each forest situation given two densities:

- Low—two snags per acre (0.8 snags/ha).
- High—eight snags per acre (3.2 snags/ha) as shown in table 2.

Total time was computed by summing:

- Average time required for an observer to walk a 164-ft (50-m) line while looking for snags in the various forest conditions, without encountering any snags.
- Average amount of time per snag in the different forest conditions needed to record the seven snag characteristics listed above.

Total time was then multiplied by $10 per hour to obtain the cost estimate (table 2). Costs for all other transect lengths were derived either by doubling the cost, or dividing by 2, for the shorter lengths. These costs where then placed into the General Cost per Sample Guidelines tables found under the View menu in SnagPRO for both English (table 3) and metric (table 4) analyses. Note that costs jump substantially within the moderate and difficult categories for the Width99(30) and Width132(40) plot sizes, owing to the observer having to periodically leave the centerline to survey beyond the point of visibility.

One of the basic assumptions of all analyses presented here is that sampling units are independent

Analysis for independence—

One of the basic assumptions of all analyses presented here is that sampling units are independent (Hurlbert 1984, Krebs 1989, Swihart and Slade 1985). This means

Table 2—Cost estimates based on time trials to conduct snag (≥10-in [25-cm] diameter at breast height) surveys in different forest conditions

Visibility,[a] terrain,[b] density[c]	Plot dimensions	Time to walk plot length (no snags)	Average number of snags per plot	Time per snag	Total time	Total cost[d]
		Minutes		*- - - Minutes*	*- - -*	*Dollars*
High, easy, low	Section33(10)	2	0 .25	2	2.50	0.43
	Section66(20)	2	.50	2	3.00	.51
	Section99(30)	2	.75	2	3.50	.60
	Section132(40)	2	1.00	2	4.00	.68
High, easy, high	Section33(10)	2	1.00	2	4.00	.68
	Section66(20)	2	2.00	2	6.00	1.02
	Section99(30)	2	3.00	2	8.00	1.36
	Section132(40)	2	4.00	2	10.00	1.70
Medium, moderate, low	Section33(10)	3	.25	3	3.75	.64
	Section66(20)	3	.50	3	4.50	.77
	Section99(30)	4.5	.75	3	6.75	1.15
	Section132(40)	6.75	1.00	3	9.75	1.66
Medium, moderate, high	Section33(10)	3	1.00	3	5.00	.85
	Section66(20)	3	2.00	3	8.00	1.36
	Section99(30)	4.5	3.00	3	13.50	2.30
	Section132(40)	6.75	4.00	3	18.75	3.19
Low, difficult, low	Section33(10)	6	.25	4	7.0	1.19
	Section66(20)	6	.50	4	8.0	1.36
	Section99(30)	24	.75	4	27.0	4.59
	Section132(40)	30	1.00	4	34.0	5.78
Low, difficult, high	Section33(10)	6	1.00	4	10.0	1.70
	Section66(20)	6	2.00	4	14.0	2.38
	Section99(30)	24	3.00	4	36.0	6.12
	Section132(40)	30	4.00	4	46.0	7.82

Note: Section33(10) = 33 ft (10 m) wide, section66(20) = 66 ft (20 m) wide, section 99(30) = 99 ft (30 m) wide, section132(40) = 132 ft (40 m) wide.

[a] High is ≥66 ft (20 m); medium is 49.5 ft (15 m); low is <33 ft (10 m).

[b] Easy is ≤30 percent slope, moderate is >30 but <50 percent slope; difficult is ≥50 percent slope.

[c] Low = 2 snags/ac (0.8/ha), high = 8 snags/ac (3.2/ha).

[d] Cost calculated at $10 per hour per person.

Table 3—Cost per sample using English-unit plots

Plot dimensions[a]	1 Easy,[b] low density[c]	2 Easy, high density	3 Moderate, low density	4 Moderate, high density	5 Difficult, low density	6 Difficult high density
	Dollars per sample					
Subsegment33	0.13	0.21	0.20	0.26	0.36	0.52
Subsegment66	0.16	0.31	0.23	0.41	0.41	0.73
Subsegment99	0.18	0.41	0.35	0.70	1.40	1.87
Subsegment132	0.21	0.52	0.51	0.97	1.76	2.38
Segment33	0.26	0.41	0.39	0.52	0.73	1.04
Segment66	0.31	0.62	0.47	0.83	0.83	1.45
Segment99	0.37	0.83	0.70	1.40	2.80	3.73
Segment132	0.41	1.04	1.01	1.95	3.52	4.77
Section33	0.52	0.83	0.78	1.04	1.45	2.07
Section66	0.62	1.24	0.94	1.66	1.66	2.90
Section99	0.73	1.66	1.40	2.80	5.60	7.46
Section132	0.83	2.07	2.02	3.89	7.05	9.54
Transect33	1.05	1.66	1.56	2.07	2.90	4.15
Transect66	1.24	2.49	1.88	3.32	3.32	5.80
Transect99	1.46	3.32	2.80	5.61	11.20	14.93
Transect132	1.66	4.15	4.05	7.78	14.10	19.07

[a] Dimensions are 33, 66, 99, or 132 ft wide.

[b] Easy is ≤30 percent slope; moderate is >30 but <50 percent slope; difficult is ≥50 percent slope.

[c] Low density = 2 snags/ac, high density = 8 snags/ac.

Table 4—Costs per sample using metric-unit plots

Plot dimensions[a]	1 Easy,[b] low density[c]	2 Easy, high density	3 Moderate, low density	4 Moderate, high density	5 Difficult, low density	6 Difficult high density
	Dollars per sample					
Subsegment10	0.11	0.17	0.16	0.21	0.30	0.43
Subsegment20	0.13	0.26	0.19	0.34	0.34	0.60
Subsegment30	0.15	0.34	0.29	0.58	1.15	1.53
Subsegment40	0.17	0.43	0.42	0.80	1.45	1.96
Segment10	0.22	0.34	0.32	0.43	0.60	0.85
Segment20	0.26	0.51	0.39	0.68	0.68	1.19
Segment30	0.30	0.68	0.58	1.15	2.30	3.06
Segment40	0.34	0.85	0.83	1.60	2.89	3.91
Section10	0.43	0.68	0.64	0.85	1.19	1.70
Section20	0.51	1.02	0.77	1.36	1.36	2.38
Section30	0.60	1.36	1.15	2.30	4.59	6.12
Section40	0.68	1.70	1.66	3.19	5.78	7.82
Transect10	0.86	1.36	1.28	1.70	2.38	3.40
Transect20	1.02	2.04	1.54	2.72	2.72	4.76
Transect30	1.20	2.72	2.30	4.60	9.18	12.24
Transect40	1.36	3.40	3.32	6.38	11.56	15.64

[a] Dimensions are 10, 20, 30, or 40 m wide.

[b] Easy is ≤30 percent slope; moderate is >30 and <50 percent slope; difficult is ≥50 percent slope.

[c] Low density is <0.8 snags/ha, high density is >3.2 snags/ha.

that whatever length of transect is chosen as optimal (subsegments, segments, or sections), the user can assume that the snag density in one sampling unit is not predicted by the snag density on the previous sampling unit of the same transect. Sampling units that are serially correlated would violate the assumption of sampling independence (Krebs 1989).

SnagPRO tests for serial correlations between increments of similar length along transects. Users will find this function on the Summary Statistics page. To conduct the test:

1. Fill in the Width columns on the **Single/Combined** page by using the appropriate formula (**Single** or **Multiple**).
2. **Compute** statistics on the **Optimal** page for the stratum of interest.
3. Click **Correlation** on the Summary Statistics page.
4. Enter the name of the transect length increment to test for serial correlation.
5. Enter the width of the plot size you are interested in testing.

Results provide a Pearson's correlation coefficient and the coefficient of determination. The correlation coefficient (r) estimates the strength of linear association between two variables (Sokal and Rohlf 1981). The coefficient of determination (r^2) is the correlation coefficient squared. It estimates the linear dependence of one variable upon another. In this instance, the r^2 value indicates how much the density in one transect increment is predicted by another transect increment.

The range for correlation coefficients is -1 to +1 (Sokal and Rohlf 1981). A high correlation coefficient suggests that adjacent increments along the same transect (for example, subsegments, segments, or sections) are correlated with each other and cannot be considered independent sampling units.

As a general guide, a correlation coefficient <0.45 ($r^2 < 0.2$) suggests that adjacent increments are independent and the increment selected can be used as the sampling unit. Values higher than this suggest adjacent increments are correlated. In the latter case, alternative transect lengths (combined segments or subsegments) must be tested for independence, and this process continued until an optimal transect length is identified that is not serially correlated.

Generally, a correlation coefficient <0.45 suggests that adjacent increments are independent and the increment selected can be used as the sampling unit.

Sample Size Determination

The estimated sample size for unstratified subwatersheds is found under Sample Size (plots) of Optimal 2 tab. For stratified subwatersheds, go to the Sample Size page. SnagPRO provides both the proportional allocation and optimal methods for estimating the total sample size required, identified as the number of plots and acres (ha) required for sampling within each stratum.

The proportional allocation method allocates the samples among the strata based on the proportion of the total area in each stratum (weight W_i). By contrast, optimal allocation incorporates both the stratum proportional area (W_i) and variance (s_i^2) to determine the number of samples required within each stratum (Krebs 1989). Both methods calculate the number of samples required to obtain a density estimate within 20 percent of the true mean 90 percent of the time.

The sample size (Krebs 1989) required by the proportional allocation method is determined by the equation:

$$n = \frac{t_\alpha^2 \sum W_i s_i^2}{B^2} \tag{5}$$

where

B = desired bound for 1 - α (x_{st} · 20 percent),[1]

t_α = student's *t* value for 90-percent confidence limits (1 - α),

n = total sample size required in stratified sampling,

W_i = stratum weight (A_i/A), and

s_i^2 = variance in stratum *I*.

Then the number of samples within each stratum (n_i) is determined by multiplying the total number of samples needed (n) by the weight (W_i) of each stratum.

$$n_i = nW_i \tag{6}$$

Sample size for the optimal allocation method (Krebs 1989) is found by using the following equation:

$$n = \frac{\left(\sum W_i s_i\right)^2}{\left(\frac{B}{t_\alpha}\right)^2 + \left(\frac{1}{A}\right)\left(\sum W_i s_i^2\right)} \tag{7}$$

where

A = total number of acres (ha) in subwatershed, and

s_i = standard deviation in stratum *i*.

Then the number of samples needed within each stratum is estimated by:

$$n_i = n\left(\frac{A_i s_i}{\sum A_i s_i}\right) \tag{8}$$

---

[1] We have substituted the symbol *B* (to denote bound) for the *d* listed in Krebs (1989) equations.

where

A_i = number of acres (ha) in stratum i, and

n_i = total sample size required in stratum i.

Both allocation methods have advantages and disadvantages. Proportional allocation offers the advantage of dropping the strata and combining all samples after sampling is done, which is appropriate when there is little or no difference in densities among strata. This yields a larger sample size (n) and a smaller variance (s^2). This option is not available if the optimal allocation method is used. Optimal allocation, however, provides the best estimate for the least cost in situations where large differences in density exist among strata. With this method, sampling is concentrated in the stratum that has highest variance. By contrast, proportional allocation concentrates sampling effort in the largest stratum, regardless of the variance within each stratum.

Again, remember that the sample sizes given are only estimates of the number required to obtain a desired level of precision. Consequently, data should be analyzed in SnagPRO periodically to gauge the precision of estimates.

Estimating Densities

A minimum of 60 samples for the landscape, or 20 samples from each stratum (whichever is higher), are required before the mean density of snags or trees can be estimated. At this point, users can decide whether enough samples have been collected to achieve their objectives. See the earlier section, “Establishing Transects,” which describes an exception to the above requirements for sample size.

The two density options provided are **Estimate Average Density** and **Compare to Target Density**. The first allows users to obtain an average snag or tree density that is within 20 percent of the true mean at a desired confidence level. The second allows users to determine whether the estimated density is significantly different from the targeted density. Users may choose both options. Go to the Densities page to use the Estimate Average Density option. For the Compare to Target Density option, you must first obtain a density estimate from the Densities page, and transfer this information to the Statistical Test page.

Estimate Average Density allows users to obtain an average snag or tree density that is within 20 percent of the true mean at a desired confidence level. Compare to Target Density allows users to determine whether the estimated density is significantly different from the targeted density.

Estimate Average Density

The Estimate Average Density option requires one of two equations based on which sampling method you use: simple or stratified random sampling. To see these equations, go to the Densities page.

For the simple random sampling method, the average is calculated in the standard way (equation 2). Then the variance is calculated by:

$$s^2 = \frac{\sum\left(x_i - \overline{x}\right)^2}{n-1} \tag{9}$$

and the standard error of the mean is determined by:

$$s_{\overline{x}} = \sqrt{\frac{s^2}{n}} \tag{10}$$

where

$\overline{x}$ = population mean,

x_i = observed x value in sample i,

n = sample size,

s^2 = variance of the measurements, and

$s_{\overline{x}}$ = standard error of the mean $\overline{x}$.

The confidence interval is then calculated using a normal approximation (Krebs 1989):

$$\overline{x} \pm t_{\alpha}\, s_{\overline{x}} \tag{11}$$

where

t_{α} = student's t value for 90-percent confidence limits $(1 - \alpha)$.

The t-value is preset at 1.67 for a sample size equal to 60 ($n = 60$) to obtain a 90-percent confidence interval. If a different level of confidence is desired, the t-value may be changed. On the Simple-Random Sampling Equation page, an estimated mean is given based on simple random sampling methods.

In the second method, a density estimate with a bound is calculated based on stratified random sampling methods. The stratified mean density is computed by the following equation:

$$\overline{x}_{st} = \frac{\sum_{i=1}^{L} A_i \overline{x}_i}{A} \tag{12}$$

where

$\overline{x}_{st}$ = stratified population mean (number per acres [ha]),

$\overline{x}_i$ = observed mean in stratum i,

A_i = number of acres (ha) in stratum i,

A = total number of acres (ha) in subwatershed,

i = stratum number, and

L = total number of strata.

To calculate a confidence interval, the stratified variance must first be determined:

$$\text{Variance of } (\bar{x}_{st}) = \sum_{i=1}^{L} \left(\frac{W_i^2 \; s_i^2}{n_i} \right) \tag{13}$$

where

n_i = number of samples in stratum i,

s_i^2 = variance in stratum i, and

W_i = stratum weight or proportion of area in stratum i (A_i/A).

Then the confidence interval is calculated by the normal approximation:

$$\bar{x}_{st} \pm t_{\alpha} \left(\sqrt{\text{var } (\bar{x}_{st})} \right) \tag{14}$$

Because SnagPRO is designed to accommodate landscapes with different numbers of strata, the user must enter the correct number when prompted by the "Number of Strata" message box. This tells SnagPRO which equation to use.

Compare to Target Density

The second density option is an informal statistical test that allows users to determine whether the estimated snag or tree density is significantly different from the targeted density, such as a targeted density identified in standards and guidelines for land use plans.

A minimum of 60 samples for the landscape, or 20 samples from each stratum (whichever is higher), are required. For subwatersheds >7,000 acres (2834 ha), it may be necessary to increase sampling effort to compensate for the natural variability of snags and trees in relation to elevation gradient.

An example is a 20,000-acre (8097-ha) subwatershed that encompasses three distinct forest community types; this situation may require about 100 samples to adequately conduct the compare-to-target-density test. This represents an increase of about three sample plots for every 1,000 acres (405 ha) surveyed above 7,000 acres (2834 ha). This option is especially useful in situations where densities are low and the sampling effort is extremely high to obtain an estimate within 20 percent of the true mean (90 percent of the time). It is intended for surveys where the main objective is to determine whether the subwatershed meets the targeted guidelines for retention of snags or large trees.

If the target mean falls within the confidence interval of the estimated mean, then the two values are not different, indicating management compliance with the target density.

The t-test is the most common way to test for a significant difference between two means. The t-test compares the mean within each plot to the target mean. This works well in single-stratum landscapes, but there are some problems using this approach on stratified landscapes, where differences among multiple means must be tested. Consequently, SnagPRO calculates confidence intervals about each estimated mean snag density.

If the target mean falls within the confidence interval of the estimated mean, then the two values are not different, indicating management compliance with the target density. If the target mean is significantly lower than the estimated mean density, this situation would also indicate management compliance. That is, the observed mean density is higher than the target density for management.

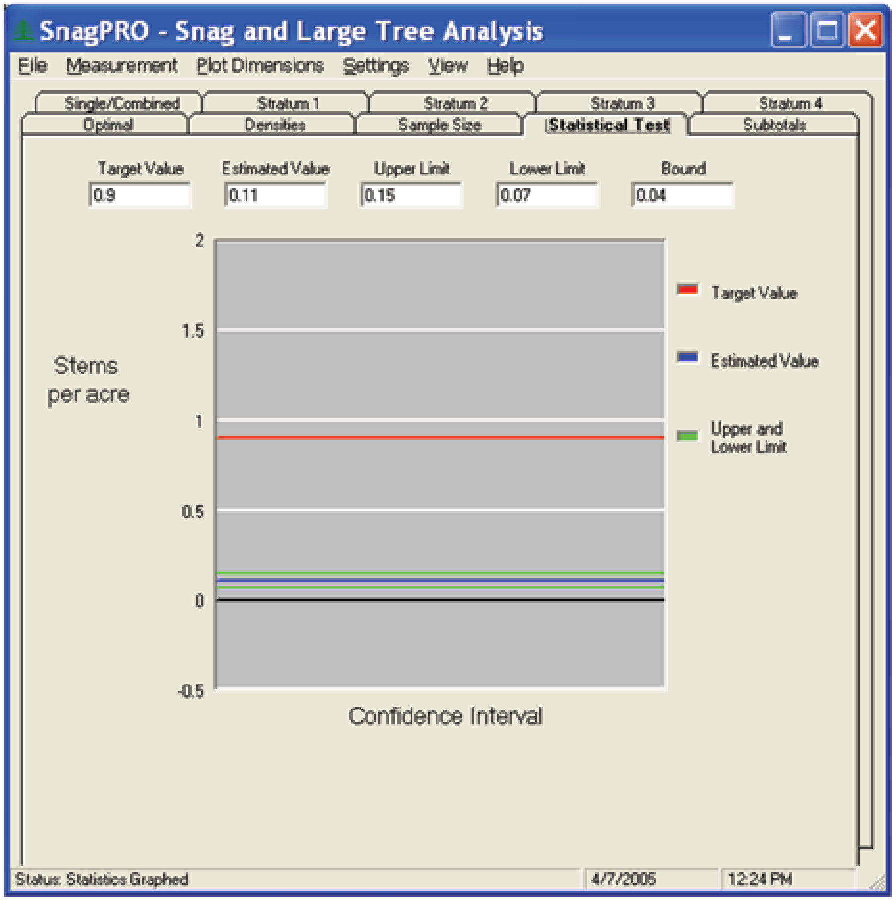
The Statistical Test page enables users to visually assess whether the estimated and targeted densities of a survey are significantly different from each other. Users simply enter the targeted density and the estimated density and its bounds from the snag or tree survey; results are automatically plotted on a graph. An example is a homogeneous 6,044-acre (2447-ha) area of ponderosa pine forest in the Oregon Cascade Range. The objective was to determine whether the area supported the targeted hard snag densities identified in the forest plan. The forest plan stipulated that the area support at least 0.9 hard snags per acre (2.2 hard snags/ha). On this site, the estimated (n = 175) hard snag density was 0.11 ± 0.04 snags per acre (0.3 ± 0.1 snags/ha [Bate 1995]). It was obvious that the area did not meet the targeted snag densities identified in the forest plan, as demonstrated statistically.

The null hypothesis for this test was:

H_0: There is no difference between the targeted and estimated hard snag densities.

To evaluate the results, we checked whether the target value (0.9 hard snags per acre [2.2 snags/ha]) fell between the values 0.07 and 0.15 on the Statistical Test page in SnagPRO (fig. 6). These values are the upper and lower limits on the mean estimate of 0.11 hard snags per acre (0.27 hard snags/ha). In this case, the targeted density did not fall within the confidence interval. We therefore rejected the null hypothesis. Because the target value was above the upper limit of the confidence (higher than the estimated mean density), we also concluded that snag density in the area did not comply with management direction.

These are the results when a 90-percent confidence interval is used. That is, there is a 90-percent probability that the estimated mean lies within the stated interval. If users want to increase the probability that a given interval will contain the true value of the estimated mean density, a 95- or a 99-percent confidence interval can be used (resulting in a 95-percent or 99-percent probability that the true value of the estimated mean lies within the interval). We recommend consulting with a statistician for more details regarding the choice of interval. In general, a 95-percent confidence interval is typically used for most analyses and considered appropriate for most situations where the cost is not prohibitive.



Target Value: 0.9
Estimated Value: 0.11
Upper Limit: 0.15
Lower Limit: 0.07
Bound: 0.04

Status: Statistics Graphed
4/7/2005
12:24 PM

Figure 6—Statistical Test page. Results of the statistical test found a difference between the targeted density of 0.9 snags per acre (2.2 snags/ha) and the estimated density of 0.11 snags per acre (0.3 snags/ha). Data are from an area of intensive timber harvest in ponderosa pine forest on the Deschutes National Forest.

Tutorials

Example 1: Snag Density Analysis for Single-Stratum Landscape Using Metric Units

Background information—

A 364-ha stand of old-growth ponderosa pine, located on the east side of the central Oregon Cascade Range, is to be sampled to determine whether the area meets the targeted densities of hard snags dictated by the forest plan. The forest plan defines hard snags as those in decay classes I through III [Cline and others 1980]) that also are ≥25.4 cm d.b.h. and ≥1.8 m tall.

The forest plan dictates that at least 10 hard snags/ha be maintained to provide adequate nesting habitat. Also of interest is the estimated density of all snags in this same size class, to be used as baseline data for mimicking old-growth characteristics in adjacent areas. Therefore, sampling all decay classes is planned, but with hard snags as the primary objective. An additional objective is to estimate the percentage of snags (all decay classes) exhibiting new cavities.

Stratification—

Use aerial photographs and vegetation cover maps to determine whether any apparent strata can be delineated. The area appears to contain a variety of seral stages, owing to a mix of past management activities. Some areas have undergone controlled burns and others have not, which could result in different snag densities across the landscape. A more thorough ground check, however, reveals a relatively homogeneous forest with respect to snag densities based on 0.4-ha ocular estimates. Consequently, the area was not stratified. Visibility averages about 15 m, and the area is flat and easy to traverse.

Pilot survey—

Ten 100-m-long transects are initially established within the area by placing a grid over a map and randomly selecting 10 grid intersection points, which are the starting points of each transect. The direction of each transect is then established by randomly choosing the compass direction. Each transect is then labeled with a unique numeric identifier and delineated into eight 12.5-m subsegments, numbered 1 through 8. For transects heading outside the landscape boundaries, use the bounce-back method to keep the transect within the sampling area while continuing to sample with standardized transect lengths (fig. 4).

The Dataform sheet found in the file named Snag_Tutorial_Data.xls (fig. 5) is used as a hardcopy field form to record field data. See “Field Forms” under “General Surveying Procedures” for complete details. Appendix 2 can be customized to further explain to field crews what information and methods are required under each field heading.

In the field, all snags 20 m either side of the centerline of specified size are tallied within each subsegment. Distance is measured from the centerline to each snag’s midpoint (that is, the center of a snag as opposed to the snag’s outer bark area that is closest to centerline). For each snag, record:

- D.b.h. (cm)
- Height (m)
- Distance of the snag’s midpoint from centerline (in meters)
- Numeric code for species
- Decay class (Cline and others 1980)
- Numeric codes are assigned to snags that have evidence of nesting. For

snags where it is not possible to determine whether a cavity exists because of vegetation, the cell is left blank (see app. 2 for details).

Data entry—

For this tutorial, the data are found in the Snag_Tutorial_Data.xls file on the Tutorial_data_I_metric page. This file can be found at the PNW Web site at http://www.fs.fed.us/pnw/publications/tools-databases.shtml. Open this file in the spreadsheet program Excel[2] for Windows. Ten transects of eight subsegments of data are available, following the same format provided on the field form (fig. 5).

Consecutive subsegments—

Before starting any analyses, sort transects and subsegments in ascending order to ensure that there are eight subsegments for each transect. In Excel, click **Data | Sort**, then select **Sort By Transect** and **Then By Subsegment**. Scroll through the entire data set to ensure that eight subsegment lengths have been entered for each transect, and the beginning subsegment of each transect is numbered "1."

Saving as a CSV file—

SnagPRO imports only CSV files. To create a CSV file, follow these steps:

1. Activate the **Tutorial_data_I_metric** sheet by clicking anywhere on the sheet: Select **File | Save As**.
2. Click **Save as Type** at the bottom of the Save As message box.
3. Select CSV (comma delimited) (***.csv**).
4. Assign a new file name in the file name box.
5. Click **Save.** When saved as a CSV file, only the active sheet is retained. Saving the file with a different name keeps the original file intact.

Importing to SnagPRO—

Import the CSV file of snag data by using these steps:

1. Launch SnagPRO by double-clicking on the desktop icon or the executable file—**SnagPRO .exe**.
2. Click **Snags or Trees**.
3. Go to **Measurement**, and click **Metric**.
4. Go to **File | Open**. In the message box "Look in," browse to the folder containing the CSV data and select the file name.

This should successfully import the CSV file. Note that additional columns have been added to your file:

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[2] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

- The Segment and Section columns were inserted between Transect and Subsegment.
- Width10, Width20, Width30, and Width40 columns have been added.

SnagPRO combined consecutive subsegments (12.5-m lengths) into segments (25-m lengths), and segments into sections (50-m lengths). The Width columns are populated after you select a formula (see below).

Formula entry—

Create the appropriate formulas for the **Width** columns. These formulas determine which snags are included in the current analysis.

First obtain estimates of hard snags only. To do this, locate and click on the **Multiple** button on the Single/Combined page to have SnagPRO include multiple species in the analysis. Several input boxes will then appear.

To create the correct formula, based on your survey objectives, enter:

- "25.4" (cm) for D.b.h.
- "1.8" (m) for Height.
- "3" for Decay Class (hard snags based on Cline and others 1980).
- "9999" for Species (all species are included).

SnagPRO evaluates each snag for the criteria listed above plus its distance from the centerline. For snags meeting all criteria, a value of "1" is placed in the cell; otherwise, the cell receives a "0."

Analyzing by plot size—

SnagPRO now calculates means and standard deviations for each plot size, transferring the results to the Optimal pages. First, review General Cost per Sample Guidelines under the View menu to select one of the six cost categories that best applies to the forest conditions (see the "Estimating Costs" section for details). For this example, choose **Code 4** because the forest conditions have moderate visibility and terrain, and snag densities appear higher than 3.2 snags per hectare (table 4).

To sum and subtotal the values for each plot size, click the **Optimal** tab; go to **Stratum to Process**; select **Single,** and click **Compute.**

A series of message boxes will appear. Enter:

- "1" for the Number of Strata.
- "4" for the General Cost per Sample code.
- "364" (ha) for the Stratum Size.

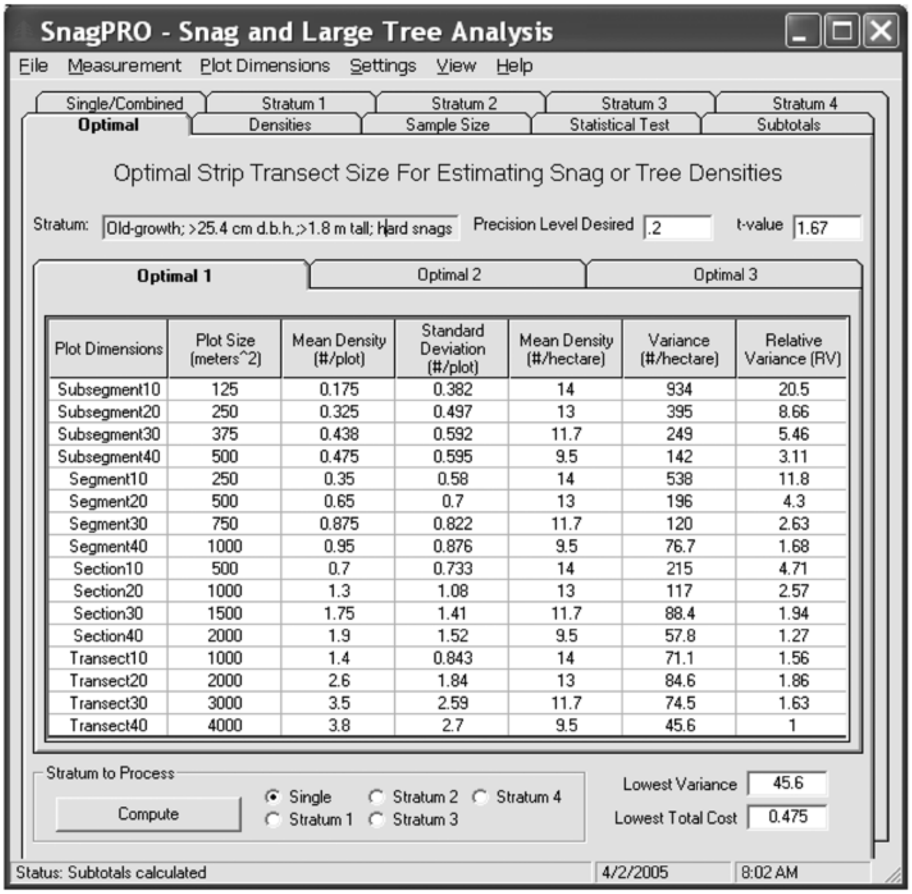
SnagPRO calculates subtotals on the Summary Statistics page and simultaneously transfers the results to the Optimal page. On the bottom of the Summary Statistics page, you will find the mean, standard deviation, and current sample size for each plot size and length.

Optimal plot size—

To determine the plot size that optimizes sampling in the current forest conditions, switch to the Optimal page. There are three Optimal sheets. Activate the **Optimal 1** worksheet by clicking on this tab. Write a brief description of the study area in the shaded box labeled Stratum. For example, for this analysis you might write: "Old-growth; ≥25.4 cm d.b.h.; ≥1.8 m tall; hard snags."

On Optimal 1 tab, the Plot Dimensions column lists the 16 available plot sizes (fig. 7). The next column, Plot Size (meters ^2), displays the plot in square meters. Mean Density (#/plot) displays the estimated average for the 16 plot sizes, with the appropriate standard deviation values in Standard Deviation (#/plot).

Under Mean Density (#/hectare), the number of snags per plot is converted to the number of snags per hectare (fig. 7). Similarly, the standard deviation of each



Optimal Strip Transect Size For Estimating Snag or Tree Densities

Stratum: Old-growth; >25.4 cm d.b.h.;>1.8 m tall; hard snags

Precision Level Desired .2

t-value 1.67

Plot Dimensions	Plot Size (meters^2)	Mean Density (#/plot)	Standard Deviation (#/plot)	Mean Density (#/hectare)	Variance (#/hectare)	Relative Variance (RV)
Subsegment10	125	0.175	0.382	14	934	20.5
Subsegment20	250	0.325	0.497	13	395	8.66
Subsegment30	375	0.438	0.592	11.7	249	5.46
Subsegment40	500	0.475	0.595	9.5	142	3.11
Segment10	250	0.35	0.58	14	538	11.8
Segment20	500	0.65	0.7	13	196	4.3
Segment30	750	0.875	0.822	11.7	120	2.63
Segment40	1000	0.95	0.876	9.5	76.7	1.68
Section10	500	0.7	0.733	14	215	4.71
Section20	1000	1.3	1.08	13	117	2.57
Section30	1500	1.75	1.41	11.7	88.4	1.94
Section40	2000	1.9	1.52	9.5	57.8	1.27
Transect10	1000	1.4	0.843	14	71.1	1.56
Transect20	2000	2.6	1.84	13	84.6	1.86
Transect30	3000	3.5	2.59	11.7	74.5	1.63
Transect40	4000	3.8	2.7	9.5	45.6	1

Stratum to Process

● Single ○ Stratum 2 ○ Stratum 4
○ Stratum 1 ○ Stratum 3

Lowest Variance 45.6

Lowest Total Cost 0.475

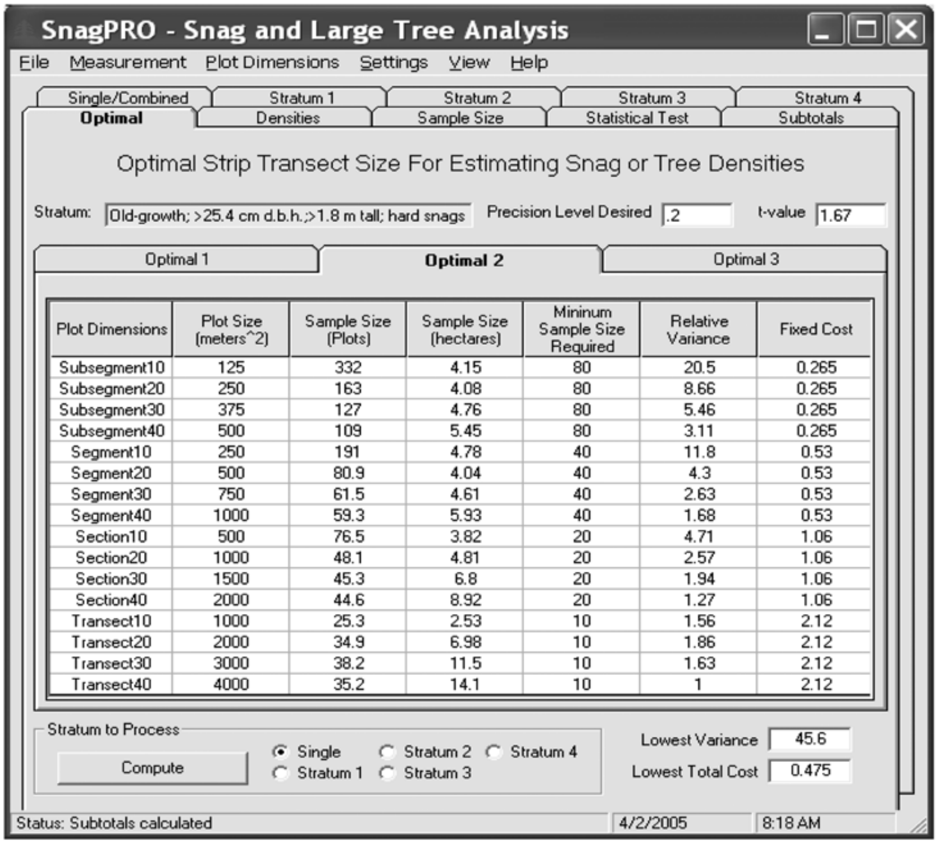
Status: Subtotals calculated 4/2/2005 8:02 AM

Figure 7—Optimal 1 page: first of three optimal pages showing size (m^2), mean, standard deviation, variance, and relative variance for each plot size for hard snags in a single-stratum ponderosa pine landscape.

plot is squared to get the variance, and then converted to number per hectare in the Variance column. The mean snag density ranges from a low of 9.5 in the widest plots to a high of 14 in the narrowest plots. If you find that the density varies from low to high among plot widths, it is best to select one of the midrange plot widths so that you are not over- or underestimating snag densities. The Relative Variance column uses the lowest variance calculated among the 16 plot sizes as the divisor for all variances to determine the optimal plot size.

The **Optimal 2** tab page (fig. 8) repeats the first two columns from Optimal 1—Plot Dimensions and Plot Size (meters^2). Sample Size (plots) calculates the total number of plots and Sample Size (hectares) calculates the number of hectares needed to obtain a density estimate within 20 percent of the true mean 90 percent of the time, based on the sample mean and standard deviation.

Earlier, we stated that SnagPRO's equations require a minimum of 60 samples. The Minimum Number of Samples Required column represents the minimum



Optimal Strip Transect Size For Estimating Snag or Tree Densities

Stratum: Old-growth; >25.4 cm d.b.h.;>1.8 m tall; hard snags

Precision Level Desired .2

t-value 1.67

Plot Dimensions	Plot Size (meters^2)	Sample Size (Plots)	Sample Size (hectares)	Mininum Sample Size Required	Relative Variance	Fixed Cost
Subsegment10	125	332	4.15	80	20.5	0.265
Subsegment20	250	163	4.08	80	8.66	0.265
Subsegment30	375	127	4.76	80	5.46	0.265
Subsegment40	500	109	5.45	80	3.11	0.265
Segment10	250	191	4.78	40	11.8	0.53
Segment20	500	80.9	4.04	40	4.3	0.53
Segment30	750	61.5	4.61	40	2.63	0.53
Segment40	1000	59.3	5.93	40	1.68	0.53
Section10	500	76.5	3.82	20	4.71	1.06
Section20	1000	48.1	4.81	20	2.57	1.06
Section30	1500	45.3	6.8	20	1.94	1.06
Section40	2000	44.6	8.92	20	1.27	1.06
Transect10	1000	25.3	2.53	10	1.56	2.12
Transect20	2000	34.9	6.98	10	1.86	2.12
Transect30	3000	38.2	11.5	10	1.63	2.12
Transect40	4000	35.2	14.1	10	1	2.12

Stratum to Process

● Single ○ Stratum 2 ○ Stratum 4
○ Stratum 1 ○ Stratum 3

Lowest Variance 45.6

Lowest Total Cost 0.475

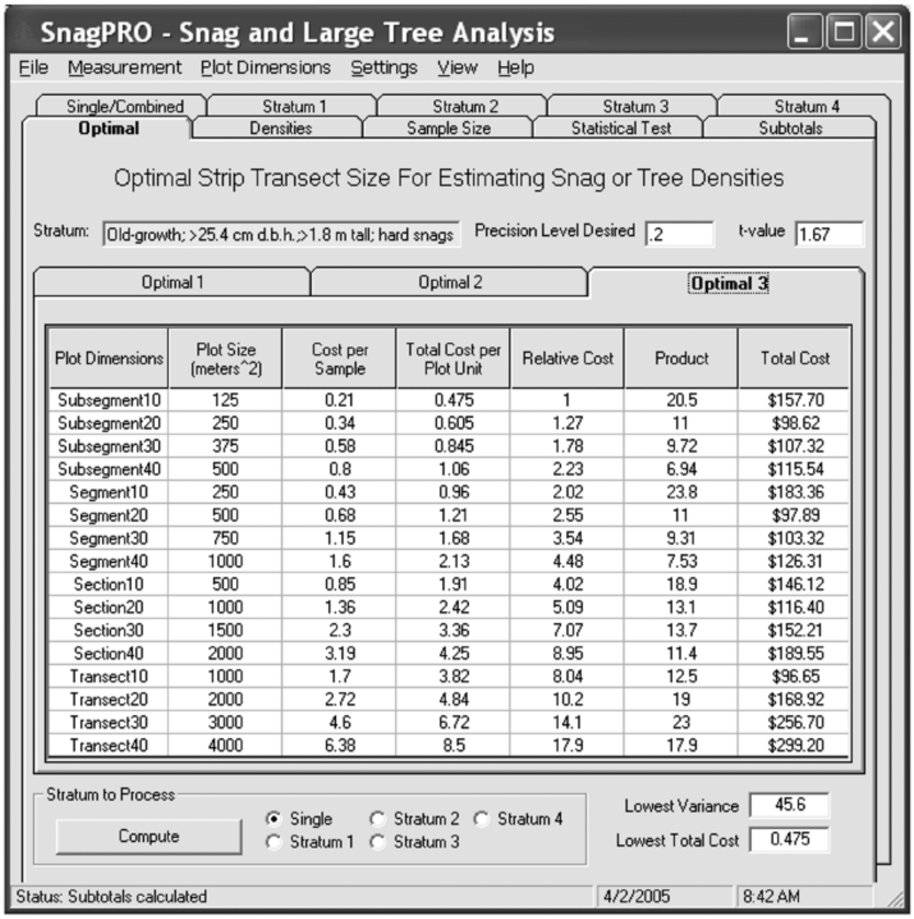
Status: Subtotals calculated 4/2/2005 8:18 AM

Figure 8—Optimal 2 page: second of three optimal pages showing sample size (plots), sample size (ha), minimum sample required, relative variance, and fixed cost for each plot size for hard snags in a single-stratum ponderosa pine landscape.

number of plots required for each transect length. See the “Establishing Transects” section for more detail on this subject. The final column in Optimal 2 gives the fixed cost associated with each plot size. See the “Estimating Costs” section for more detail.

The first two columns in **Optimal 3** (fig. 9) repeat each plot name and area. The Cost per Sample column reports the cost of each sample in addition to the Fixed Cost from Optimal 2, providing the Total Cost per Plot Unit.

SnagPRO calculates Relative Cost in the same way as Relative Variance from Optimal 1 and 2, dividing each total cost by the lowest cost in this column. Following Wiegert’s method (1962), SnagPRO multiplies the relative variance by the relative cost to get Product. The optimal plot size is the one that minimizes the product of these two factors. Finally, the Total Cost column provides users with an idea of the costs they could expect for one plot size versus another.



Optimal Strip Transect Size For Estimating Snag or Tree Densities

Stratum: Old-growth; >25.4 cm d.b.h.;>1.8 m tall; hard snags

Precision Level Desired .2

t-value 1.67

Plot Dimensions	Plot Size (meters^2)	Cost per Sample	Total Cost per Plot Unit	Relative Cost	Product	Total Cost
Subsegment10	125	0.21	0.475	1	20.5	$157.70
Subsegment20	250	0.34	0.605	1.27	11	$98.62
Subsegment30	375	0.58	0.845	1.78	9.72	$107.32
Subsegment40	500	0.8	1.06	2.23	6.94	$115.54
Segment10	250	0.43	0.96	2.02	23.8	$183.36
Segment20	500	0.68	1.21	2.55	11	$97.89
Segment30	750	1.15	1.68	3.54	9.31	$103.32
Segment40	1000	1.6	2.13	4.48	7.53	$126.31
Section10	500	0.85	1.91	4.02	18.9	$146.12
Section20	1000	1.36	2.42	5.09	13.1	$116.40
Section30	1500	2.3	3.36	7.07	13.7	$152.21
Section40	2000	3.19	4.25	8.95	11.4	$189.55
Transect10	1000	1.7	3.82	8.04	12.5	$96.65
Transect20	2000	2.72	4.84	10.2	19	$168.92
Transect30	3000	4.6	6.72	14.1	23	$256.70
Transect40	4000	6.38	8.5	17.9	17.9	$299.20

Stratum to Process

● Single ○ Stratum 2 ○ Stratum 4
○ Stratum 1 ○ Stratum 3

Lowest Variance 45.6

Lowest Total Cost 0.475

Status: Subtotals calculated | 4/2/2005 | 8:42 AM

Figure 9—Optimal 3 page: third of three optimal pages showing cost per sample, total cost per plot unit, relative cost, product (cost x relative cost), and total cost for each plot size for hard snags in a single-stratum ponderosa pine landscape.

These variables are used in the decisionmaking process regarding sample design. On Optimal 2, review the number of plots required in **Sample Size (Plots)** for each **Plot Size (meters ^2)** (fig. 8). Which plot sizes require the fewest samples?

Consider the plot size in terms of area, as shown in Sample Size (hectares). Note that Transect10 requires the minimal number of plots (25.3) and hectares (2.53). Now check the Minimum Sample Size Required. With a sample size of only 10 it is not possible to assume a normal distribution because SnagPRO's equations work from a minimum of 60 samples.

Switch to **Optimal 3 | Total Cost** (fig. 9) to check the costs based on the minimum number of required samples. Transect10 plots could cost approximately $97; using Segment20 or Subsegment20 plots, the estimated total cost is similar, and these plots provide a larger sample size and thus would be a better choice, assuming these shorter transects are independent. To test for independence, switch to the **Summary Statistics** page and run the serial correlation test.

First test the independence of the Segment20 plots. To do this:

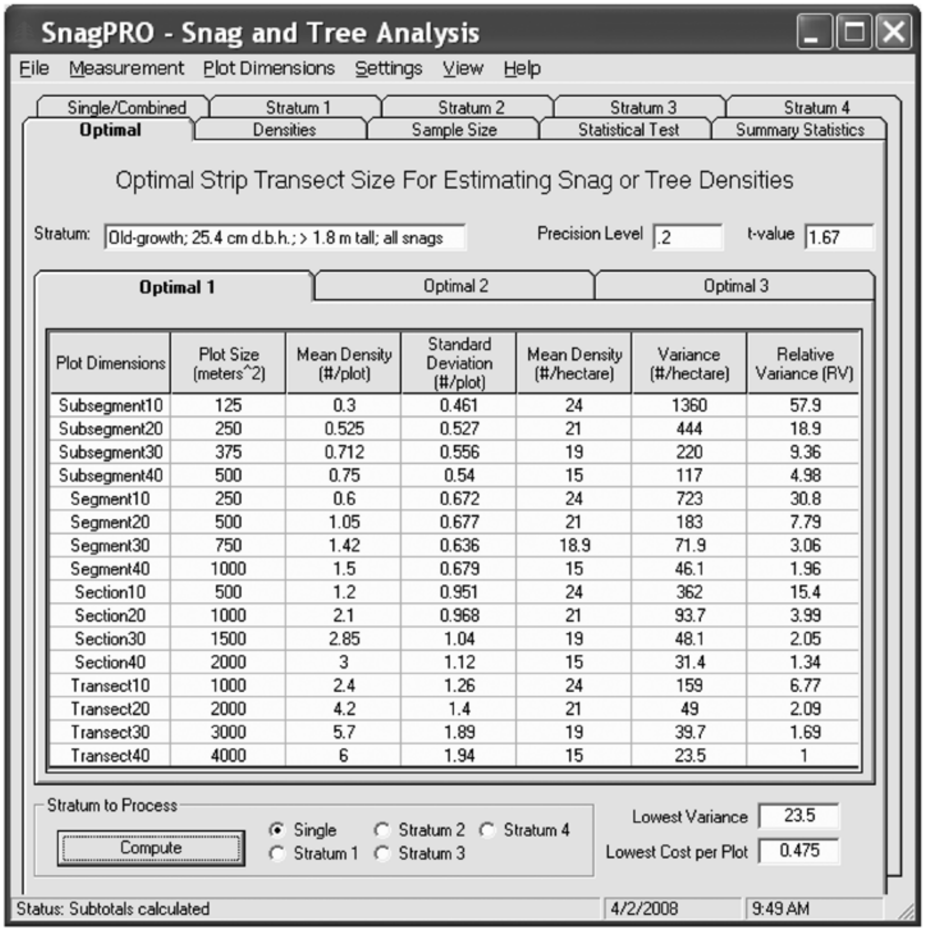
1. Click on the **Correlation** button in the bottom-right corner of the screen.
2. Enter "Segment" when the first message box appears labeled "Correlation Length."
3. Enter "20" into the box labeled "Correlation Width."

The message box displays the correlation coefficient (r = 0.23) and coefficient of determination ($r^2 = 0.05$). The low r^2 value (0.05) indicates that the adjacent 20-m segments are independent sampling units.

When this process is repeated for Subsegment20 plots, adjacent plots also appear to be independent, with a correlation coefficient (r) of 0.06 and a coefficient of determination (r^2) of 0.0. Thus, it is acceptable to use either plot size for analysis, but Subsegment20 plots appear to be slightly better. Because snags of all decay classes also are of interest, running the optimal plot analysis on both hard and soft snags may aid the decisionmaking process.

To analyze snags in all decay classes, return to **Single/Combined** page and click on the **Multiple** button. Refer to the "Formula Entry" section above and enter the same responses, with the exception of the decay class. Enter the value "5" instead of "3" to include both hard and soft snags in the analysis.

Go to **Optimal 1** and click on **Compute | Single**. Results show that snag densities for all decay classes are relatively high, ranging from an estimated 15 to 24 snags per hectare (fig. 10). In addition, the Sample Size (plots) required is low for a number of the plots as shown on Optimal 2 (fig. 11). This suggests a random distribution of snags (low variance) rather than a highly clumped one. For Subsegment20



Optimal Strip Transect Size For Estimating Snag or Tree Densities

Stratum: Old-growth; 25.4 cm d.b.h.; > 1.8 m tall; all snags

Precision Level .2

t-value 1.67

Plot Dimensions	Plot Size (meters^2)	Mean Density (#/plot)	Standard Deviation (#/plot)	Mean Density (#/hectare)	Variance (#/hectare)	Relative Variance (RV)
Subsegment10	125	0.3	0.461	24	1360	57.9
Subsegment20	250	0.525	0.527	21	444	18.9
Subsegment30	375	0.712	0.556	19	220	9.36
Subsegment40	500	0.75	0.54	15	117	4.98
Segment10	250	0.6	0.672	24	723	30.8
Segment20	500	1.05	0.677	21	183	7.79
Segment30	750	1.42	0.636	18.9	71.9	3.06
Segment40	1000	1.5	0.679	15	46.1	1.96
Section10	500	1.2	0.951	24	362	15.4
Section20	1000	2.1	0.968	21	93.7	3.99
Section30	1500	2.85	1.04	19	48.1	2.05
Section40	2000	3	1.12	15	31.4	1.34
Transect10	1000	2.4	1.26	24	159	6.77
Transect20	2000	4.2	1.4	21	49	2.09
Transect30	3000	5.7	1.89	19	39.7	1.69
Transect40	4000	6	1.94	15	23.5	1

Stratum to Process

● Single
○ Stratum 1
○ Stratum 2
○ Stratum 3
○ Stratum 4

Lowest Variance 23.5

Lowest Cost per Plot 0.475

Status: Subtotals calculated

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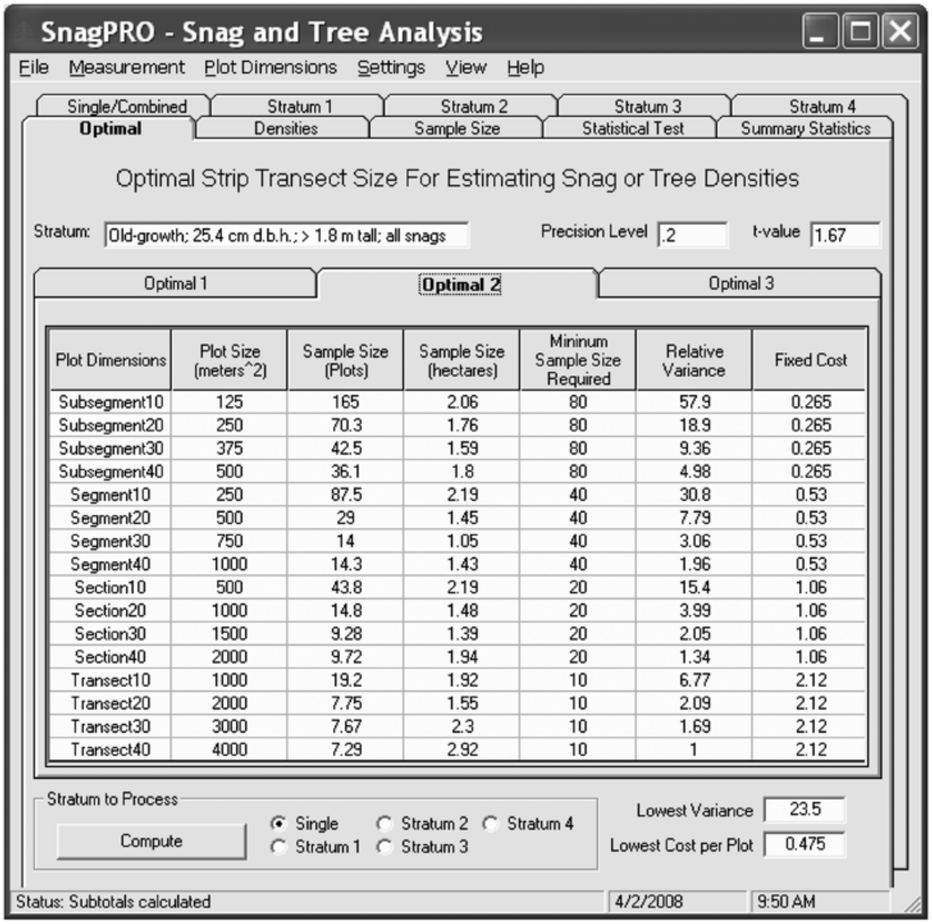
Figure 10—Optimal 1 page: first of three optimal pages showing size (m^2), mean, standard deviation, variance, and relative variance for each plot size for both hard and soft snags in a single-stratum ponderosa pine landscape.



Optimal Strip Transect Size For Estimating Snag or Tree Densities

Stratum: Old-growth; 25.4 cm d.b.h.; > 1.8 m tall; all snags

Precision Level .2

t-value 1.67

Plot Dimensions	Plot Size (meters^2)	Sample Size (Plots)	Sample Size (hectares)	Mininum Sample Size Required	Relative Variance	Fixed Cost
Subsegment10	125	165	2.06	80	57.9	0.265
Subsegment20	250	70.3	1.76	80	18.9	0.265
Subsegment30	375	42.5	1.59	80	9.36	0.265
Subsegment40	500	36.1	1.8	80	4.98	0.265
Segment10	250	87.5	2.19	40	30.8	0.53
Segment20	500	29	1.45	40	7.79	0.53
Segment30	750	14	1.05	40	3.06	0.53
Segment40	1000	14.3	1.43	40	1.96	0.53
Section10	500	43.8	2.19	20	15.4	1.06
Section20	1000	14.8	1.48	20	3.99	1.06
Section30	1500	9.28	1.39	20	2.05	1.06
Section40	2000	9.72	1.94	20	1.34	1.06
Transect10	1000	19.2	1.92	10	6.77	2.12
Transect20	2000	7.75	1.55	10	2.09	2.12
Transect30	3000	7.67	2.3	10	1.69	2.12
Transect40	4000	7.29	2.92	10	1	2.12

Stratum to Process

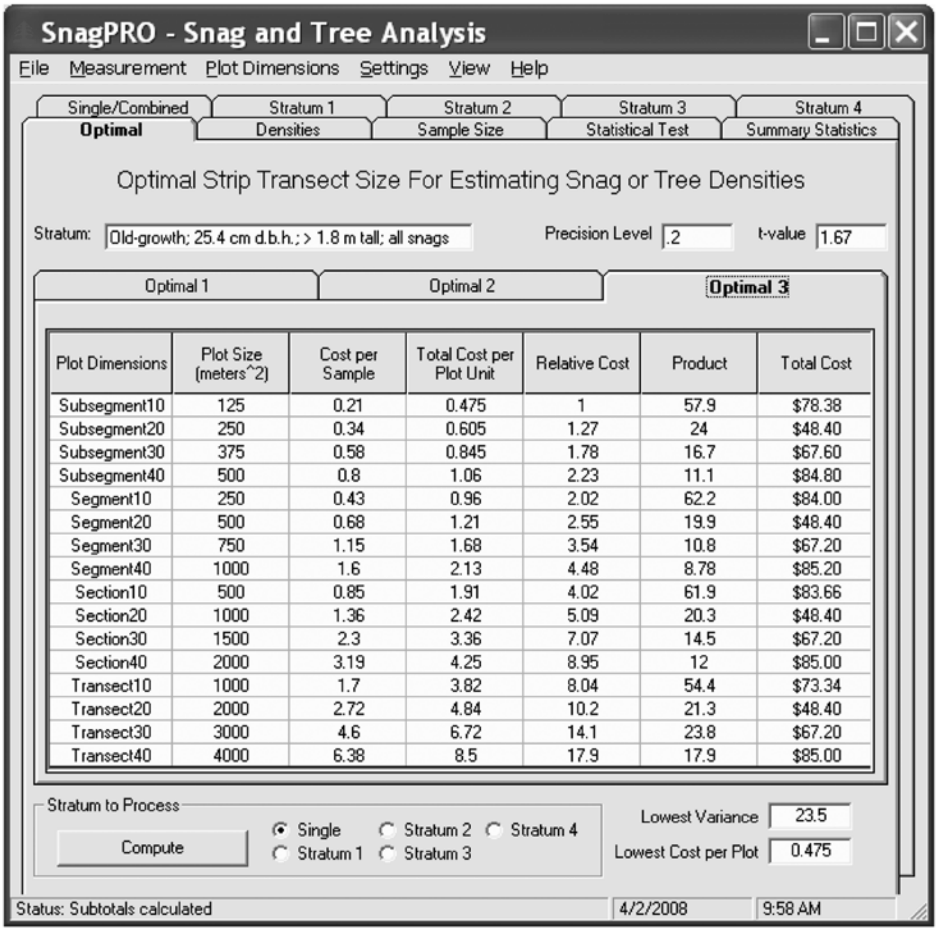
● Single ○ Stratum 2 ○ Stratum 4
○ Stratum 1 ○ Stratum 3

Lowest Variance 23.5

Lowest Cost per Plot 0.475

Status: Subtotals calculated | 4/2/2008 | 9:50 AM

Figure 11─Optimal 2 page: second of three optimal pages showing sample size (plots), sample size (ha), minimum sample required, relative variance, and fixed costs for each plot size for both hard and soft snags in a single-stratum ponderosa pine landscape.



Optimal Strip Transect Size For Estimating Snag or Tree Densities

Stratum: Old-growth; 25.4 cm d.b.h.; > 1.8 m tall; all snags

Precision Level .2

t-value 1.67

Plot Dimensions	Plot Size (meters^2)	Cost per Sample	Total Cost per Plot Unit	Relative Cost	Product	Total Cost
Subsegment10	125	0.21	0.475	1	57.9	$78.38
Subsegment20	250	0.34	0.605	1.27	24	$48.40
Subsegment30	375	0.58	0.845	1.78	16.7	$67.60
Subsegment40	500	0.8	1.06	2.23	11.1	$84.80
Segment10	250	0.43	0.96	2.02	62.2	$84.00
Segment20	500	0.68	1.21	2.55	19.9	$48.40
Segment30	750	1.15	1.68	3.54	10.8	$67.20
Segment40	1000	1.6	2.13	4.48	8.78	$85.20
Section10	500	0.85	1.91	4.02	61.9	$83.66
Section20	1000	1.36	2.42	5.09	20.3	$48.40
Section30	1500	2.3	3.36	7.07	14.5	$67.20
Section40	2000	3.19	4.25	8.95	12	$85.00
Transect10	1000	1.7	3.82	8.04	54.4	$73.34
Transect20	2000	2.72	4.84	10.2	21.3	$48.40
Transect30	3000	4.6	6.72	14.1	23.8	$67.20
Transect40	4000	6.38	8.5	17.9	17.9	$85.00

● Single

Lowest Variance 23.5

Lowest Cost per Plot 0.475

Status: Subtotals calculated

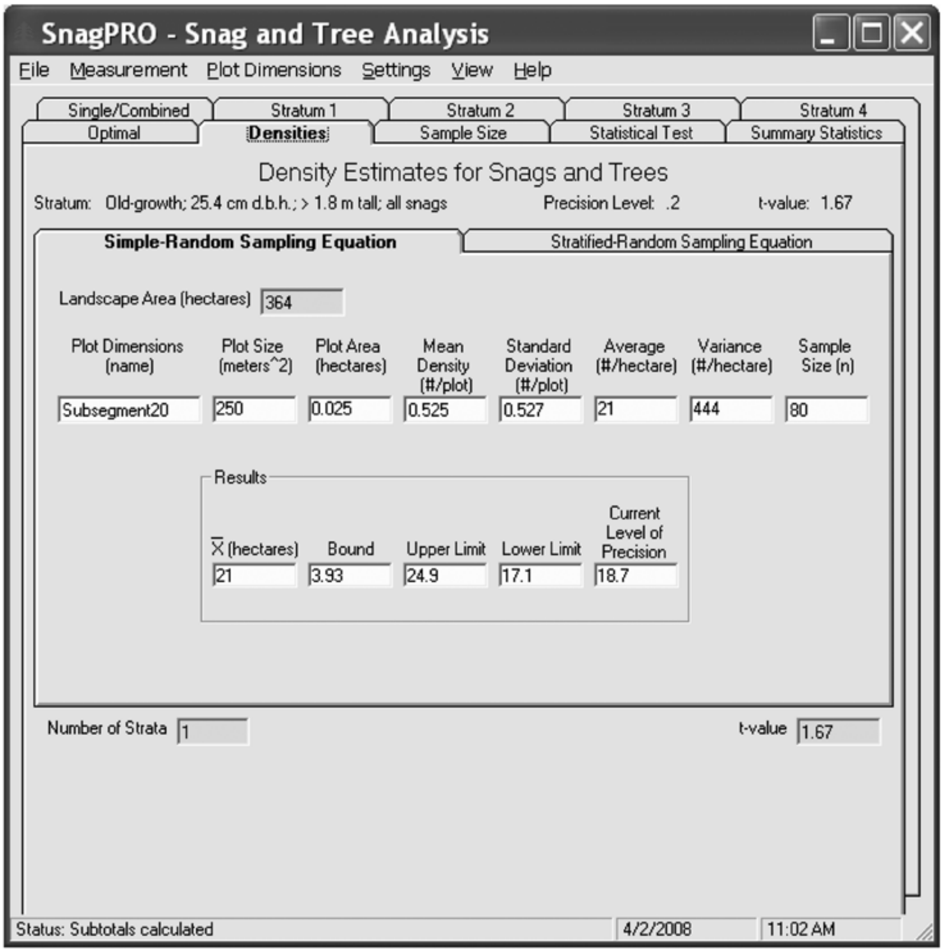
4/2/2008

9:58 AM

Figure 12─Optimal 3 page: third of three optimal pages showing cost per sample, total cost per plot unit, relative cost, product (cost x relative cost), and total cost for each plot size for both hard and soft snags in a single-stratum ponderosa pine landscape.

plots, 80 samples have been collected (even though Sample Size (Plots) shows 70). This number exceeds the minimum number of 60 samples required to meet the assumptions of normality.

On the **Optimal 3** page (fig. 12), examine the **Total Cost** for Subsegment20 plots and repeat steps described above for Segment20 plots. Lowest cost is the same as for the other width 20 plots ($48.40). To check for the independence of these plots, repeat steps described above for Segment20 plots. Results for independence show that Subsegment20 plots may be considered independent ($r = 0.11$ and $r^2 = 0.01$). Consequently, Subsegment20 plots are used for the remainder of the analyses and field sampling because this plot size appears best in meeting both objectives.



Density Estimates for Snags and Trees

Stratum: Old-growth; 25.4 cm d.b.h.; > 1.8 m tall; all snags

Precision Level: .2

t-value: 1.67

Landscape Area (hectares): 364

Plot Dimensions (name)	Plot Size (meters^2)	Plot Area (hectares)	Mean Density (#/plot)	Standard Deviation (#/plot)	Average (#/hectare)	Variance (#/hectare)	Sample Size (n)
Subsegment20	250	0.025	0.525	0.527	21	444	80

Results

$\overline{X}$ (hectares)	Bound	Upper Limit	Lower Limit	Current Level of Precision
21	3.93	24.9	17.1	18.7

Number of Strata: 1

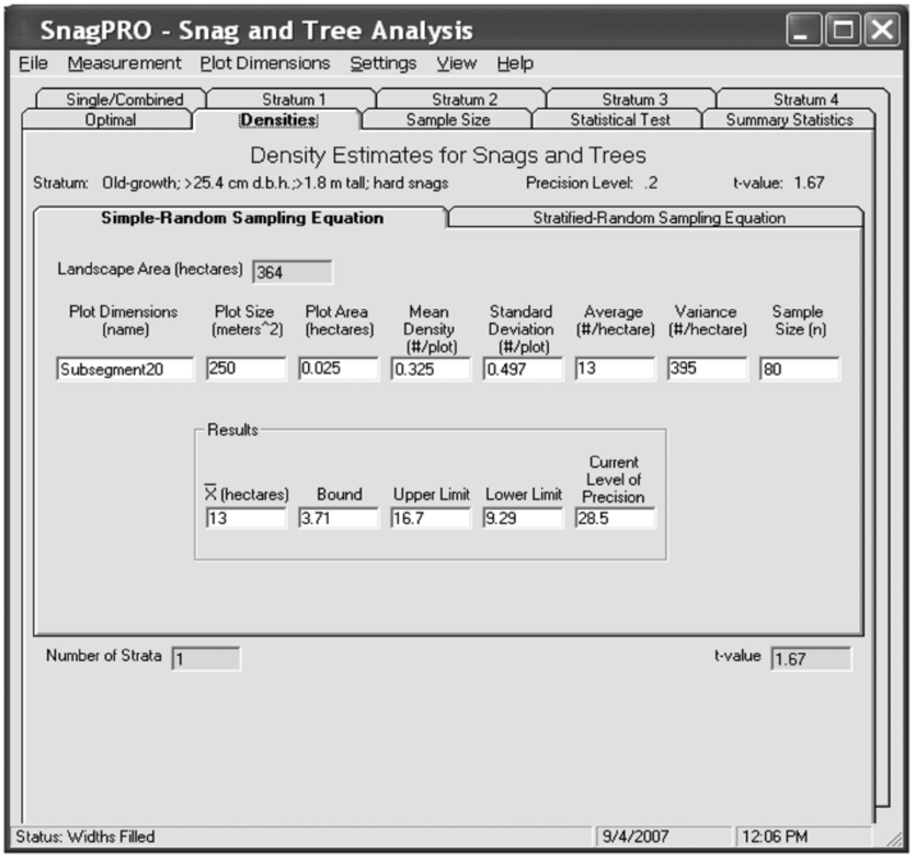
t-value: 1.67

Status: Subtotals calculated | 4/2/2008 | 11:02 AM

Figure 13—Densities page. Estimated density results using Subsegment20 plots for single-stratum ponderosa pine landscape. Results are for both hard and soft snags from the Tutorial_data_I_metric data set.

Density analysis—
First obtain a density estimate for snags in all decay classes because these are the data that currently fill the Optimal page. SnagPRO transfers the density statistics from the Optimal page to the Densities page based on which plot has the lowest total cost.

Go to **Densities | Simple-Random Sampling Equation** to obtain a density estimate for single-stratum landscapes. Check to ensure that **Subsegment20** is listed in the box labeled “Plot Dimensions (name)” to verify that the correct data have been transferred (fig. 13). Results show that this landscape supports 21 ± 3.93 snags per hectare. To determine the precision of your estimate, the bound is divided by the mean. For this example, the bound 3.93 is divided by the mean of 21, and then multiplied by 100. The result is 18.7. This indicates a 90-percent probability that the estimated mean density is within 18.7 percent of the true mean.



Stratum: Old-growth; >25.4 cm d.b.h.;>1.8 m tall; hard snags
Precision Level: .2
t-value: 1.67

Landscape Area (hectares) 364

Plot Dimensions (name)	Plot Size (meters^2)	Plot Area (hectares)	Mean Density (#/plot)	Standard Deviation (#/plot)	Average (#/hectare)	Variance (#/hectare)	Sample Size (n)
Subsegment20	250	0.025	0.325	0.497	13	395	80

X̄ (hectares)	Bound	Upper Limit	Lower Limit	Current Level of Precision
13	3.71	16.7	9.29	28.5

Number of Strata 1
t-value 1.67

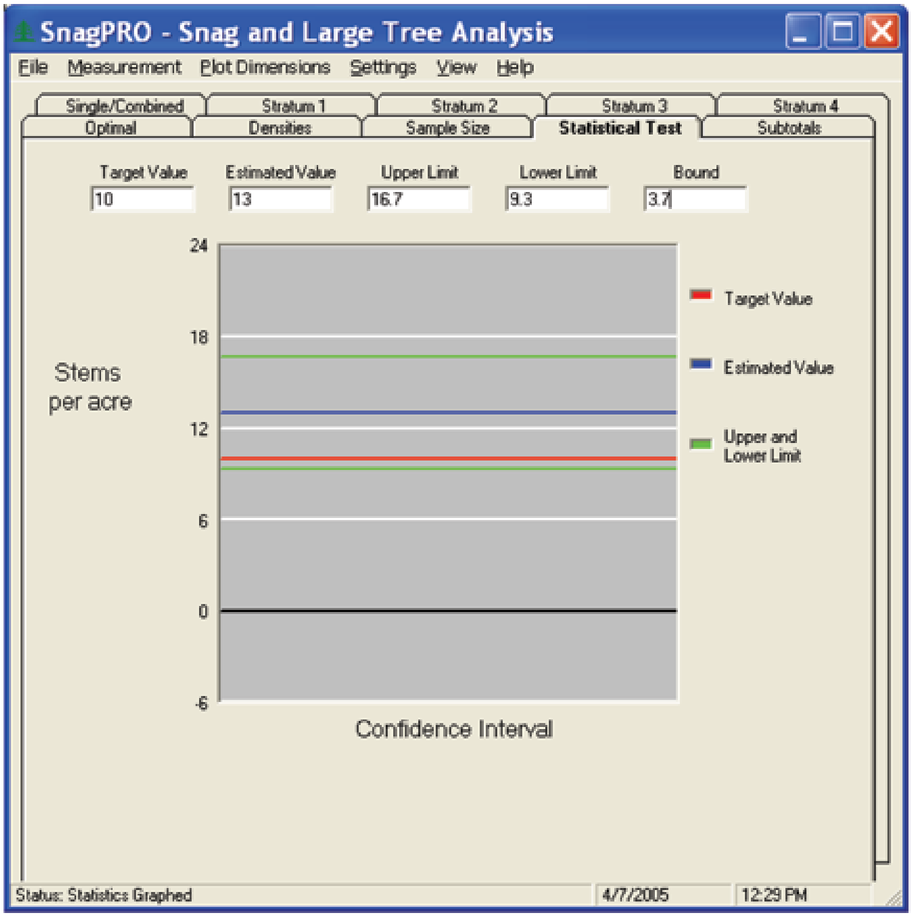
Status: Widths Filled
9/4/2007
12:06 PM

Figure 14—Densities page. Estimated density results using Subsegment20 plots for single-stratum ponderosa pine landscape. Optimal plot size was overridden by selecting Segment20 plots from the Optimal Selection on the Settings menu. Results are for hard snags from the Tutorial_data_I_metric data set.

Now obtain the density results for hard snags. Based on the previous analysis of hard snags, the lowest total cost comes with the Transect20 plots. However, it was determined that Subsegment20 plots would likely be best for both categories of snags. Consequently, the default Optimal plot selection needs to be overridden. To do this, follow these steps:

1. Go to the **Single/Combined** page.
2. Rerun the **Multiple** formula just for hard snags (Decay Class = **3**).
3. Go to **Settings | Optimal Selection**, and click on **Single Stratum**.
4. Select **Subsegment20** instead of Automatic.
5. Switch to **Optimal**, and click **Compute** in the lower left-hand corner.

Next, switch back to the **Densities** page (fig. 14). Results show that data for Subsegment20 plots have been transferred and that there are an estimated 13 ± 3.71 hard snags per hectare on this landscape. The current level of precision is 28.5 percent, which is less precise than the targeted goal.



Target Value: 10
Estimated Value: 13
Upper Limit: 16.7
Lower Limit: 9.3
Bound: 3.7

Status: Statistics Graphed | 4/7/2005 | 12:29 PM

Figure 15—Statistical Test page. Graph depicting test for significant difference between estimated and targeted densities of qualifying hard snags in single-stratum ponderosa pine landscape. Results are from Tutorial_data_I_metric data set.

The graph shows that the line representing the target density for snags falls just within the boundaries of the upper and lower limits of the estimated density.

Compare to target density—

Assume that you have finished sampling and want to test whether the estimated density of qualifying snags meets the targeted density of 10 hard snags per hectare reported in the forest plan. Go to **Statistical Test** page to conduct the test, using the following steps:

1. Affirm the null hypothesis for this test: Ho: There is no difference between the estimated and the targeted hard snag densities.
2. Enter the targeted density of "10" into the shaded box labeled Target Value.
3. Enter the estimated snag density of "13" snags per hectare into the shaded box labeled Estimated Value.
4. Enter the estimated Bound of "3.7" for a 90-percent confidence interval.

Results are automatically plotted as a graph (fig. 15). The graph shows that the line representing the target density for snags falls just within the boundaries of the upper and lower limits of the estimated density. Consequently, the null hypothesis of "no difference between the estimated and targeted snag densities" is not rejected. The result indicates a 90-percent probability that the snag density on this landscape meets forest plan guidelines.

Cavity analysis—

To conduct the cavity analysis, return to the **Single/Combined** page. The cavity use codes are based on definitions described in appendix 2. To calculate the percentage of snags containing new cavities, follow these steps:

1. Click **Cavity** in the bottom-right corner of the **Single/Combined** page.
2. Enter "25.4" for the D.b.h. message box.
3. Enter "1.8" for Height.
4. Enter "5" for Decay Class.
5. Enter "9999" when prompted for any species to exclude (allows all species).
6. Enter "1" for Cavity Code (new cavity) (see app. 2).

Results show that of 48 available snags, 4 contained a new cavity. That is, 8 percent of the snags surveyed showed new signs of nesting. Note that several snags were excluded from the analysis because it was not possible to view the entire snag for signs of cavities, owing to its height or vegetation obstructing the view. In these cases, no value was placed in the cell in the Cavity column.

Conclusions for single stratum—

From this analysis, we may conclude:

- The estimated density of hard snags (13 ± 3.71 snags per hectare) on this landscape meets or exceeds the targeted densities listed in the forest plan (10 snags per hectare).
- The goal of obtaining an estimate of hard snags within 20 percent of the true mean, however, has not been achieved. To obtain this desired level of precision would require 83 additional plots. This translates to about 10 additional 100-m transects of 12.5-m subsegments.
- To save time in the field, distances will no longer be measured with a tape, but instead estimated by pacing because the plot width has been selected. Borderline cases, however, will continue to be measured.
- Snags in all decay classes averaged about 21 snags (± 3.93) per hectare.
- About 8 percent of all snags exhibit signs of recent use for nesting.

Example 2: Density Analysis for Subwatershed With Multiple Strata Using English Units

Background information—

To conduct forestwide compliance monitoring, a snag survey must be done on a representative subwatershed to determine if management activities have maintained designated snag densities for woodpeckers and other cavity-nesting species. Stands to be monitored are dominated by Douglas-fir/western hemlock communities. Owing to limited resources, monitoring will focus on whether the subwatershed meets forest plan standards for hard snags. However, data on snags for all decay classes will be collected if this does not substantially increase sampling effort.

Subwatershed stratification—

Aerial photographs show that the subwatershed is highly fragmented, and geospatial data verify that clearcutting has been the primary method of timber harvest. Ten years ago, a retention program was initiated for snags ≥18 in d.b.h., at least 20 ft tall, and in the decay classes I through III (Cline and others 1980). Owing to the timber harvest techniques used and the recent initiation of the snag retention program, snag densities are anticipated to be low in second-growth and most clearcut stands. In the old-growth stands, high densities of snags are anticipated.

Based on these expected differences, two strata may be sufficient: old-growth stands and a combined stratum of second-growth and clearcut stands. A thorough ground check, however, suggests that combining the second-growth and clearcut stands may not be feasible because the older clearcut stands have a dense understory that obstructs viewing. Stands that were clearcut >10 years ago are placed in their own stratum labeled “second growth.” All clearcut stands harvested within the past 10 years are placed in a stratum labeled “clearcut.”

Three strata have been delineated: clearcut, second growth, and old growth. Make a field map of the stand polygons assigned to each stratum. Attribute each stand with its number of acres and the total acres for that stratum within the entire subwatershed. Strata 1, 2, and 3 have 1,759, 1,381, and 1,243 acres, respectively, for a total of 4,383 acres. Spend a day in the field, validating that each stand has been placed in the proper stratum.

Pilot survey—

Although topography of the subwatershed ranges from flat benches to steep (>100 percent) slopes, clearcut harvest has occurred mostly in flat areas. Visibility within the clearcut stratum is unobstructed, travel is easy, and snag densities appear low. Therefore, preselect the widest plot width—132 ft—to maximize sampling efficiency.

A preliminary review of the second-growth stands, however, reveals difficult travel conditions and low visibility. Few, if any, snags are expected in these stands because they were harvested before a snag retention policy was adopted. Consequently, choose a narrower plot—66 ft—to ensure that snags are not missed because of viewing obstructions.

Snag densities within the old growth are highly variable. Thus, the best choice for plot width is not apparent. Visibility averages about 49.5 feet, and travel is moderate to difficult, owing to a large volume of logs and dense patches of understory trees and shrubs. Thus, plots 99-ft wide are chosen, allowing a formal optimal plot size analysis to be conducted for the stratum.

Across the watershed, five stands of each stratum are randomly selected in which two 400-ft-long transects are established. Starting points for each transect are selected by placing a grid over each stand and then randomly choosing a grid intersection for the starting point (fig. 1). The compass direction also is randomly determined.

Each transect is assigned a unique numeric identifier, and divided into eight 50-ft subsegments, numbered 1 through 8. The bounce-back method is used for any transects running into the stand boundaries (fig. 4). In each subsegment, all snags of interest are tallied from the center line: 66 ft for stratum 1; 33 ft for stratum 2; and 49.5 ft for stratum 3.

Data entry—
Data for all strata within this subwatershed are found in Snag_Tutorial_Data, on the Tutorial_data_II_English page, found at the PNW Web site at http://www.fs.fed.us/pnw/publications/tools-databases.shtml. Data in this MS Excel spreadsheet are for 30 transects, each divided into eight subsegments. Data are in the same format as the field form (fig. 5), with one exception. An additional column, Plot width, was added on the field form to help track sampling within each stratum. This extraneous column is not included in the CSV import file, however, because its header information is not recognized by SnagPRO.

Consecutive subsegments—
Before starting the analysis, sort transects and subsegments in ascending order to verify that there are eight subsegments for each transect. To do this, go to the menu, **Data | Sort**, and select **ascending** for both **Sort By** and **Then By**. Make sure that eight subsegment lengths are entered for each transect, and that the beginning subsegment of each transect is numbered “1.”

Saving as a CSV file—

SnagPRO imports only CSV files. To create a CSV file, follow these steps:

1. Activate the **Tutorial_data_II_English** sheet by clicking anywhere on the sheet.
2. Select **File | Save As**.
3. Click **Save as Type** at the bottom of the Save As message box.
4. Select CSV (comma delimited) (***.csv**).
5. Assign a new file name in the file name box.
6. Click **Save**. When saved as a CSV file, only the active sheet is retained. Saving the file with a different name keeps the original file intact.

Importing to SnagPRO—

Import the CSV file of snag data using these steps:

1. Launch SnagPRO by double-clicking on the desktop icon or the executable file— **SnagPRO.exe**.
2. Click **Snags or Trees**.
3. Go to **Measurement**, and click **English**.
4. Go to **File | Open**. In the message box “Look in,” browse to the folder containing the CSV data and select the file name.

This should successfully import the CSV file. Additional columns have been added to your file:

- The Segment and Section columns were inserted between Transect and Subsegment.
- Width33, Width66, Width99, and Width132 columns have been added.

SnagPRO combined consecutive subsegments (50-ft lengths) into segments (100-ft lengths), and segments into sections (200-ft lengths). The Width columns are populated after you select a formula (see below).

Formula entry—

Create the formulas so SnagPRO places the correct values into the Width columns. These formulas determine which snags are included in the current analysis.

First obtain estimates of hard snags only. To do this, locate and click on the **Single/Combined** page. Then click the **Multiple** button to have SnagPRO include multiple species in the analysis. Several input boxes will then appear. To create the correct formula, based on survey objectives, enter:

- “18” for D.b.h.
- “20” for Height.
- “3” for Decay Class (hard snags based on Cline and others [1980]).
- “9999” for Species (all snag species are included).

SnagPRO evaluates each snag in each Width column for the criteria listed above plus its distance from the centerline. For snags meeting all criteria, a value of "1" is placed in the cell; otherwise, the cell receives a "0." After the formulas have been created, SnagPRO will sort the data by stratum and place the appropriate data on each stratum page.

Analyzing by plot size—

SnagPRO now calculates averages and standard deviations for each plot size, transferring the results to the Optimal pages. First, review General Cost per Sample Guidelines under the **View** menu to select one of the six cost categories that best applies to your forest conditions (see the "Estimating Costs" section and table 3 for details).

Choose **Code 1** for stratum 1 because the clearcut harvesting resulted in open conditions and low snag densities. Choose **Code 5** in stratum 2 because travel conditions are difficult, visibility is limited, and snag densities are low. Choose **Code 6** for the stratum 3 because of difficult travel conditions, limited visibility, and higher snag densities.

To sum and subtotal the values for each plot size in stratum 1:

- Click the **Optimal** tab.
- Go to **Stratum to Process.**
- Select **Stratum 1**.
- Click **Compute**.

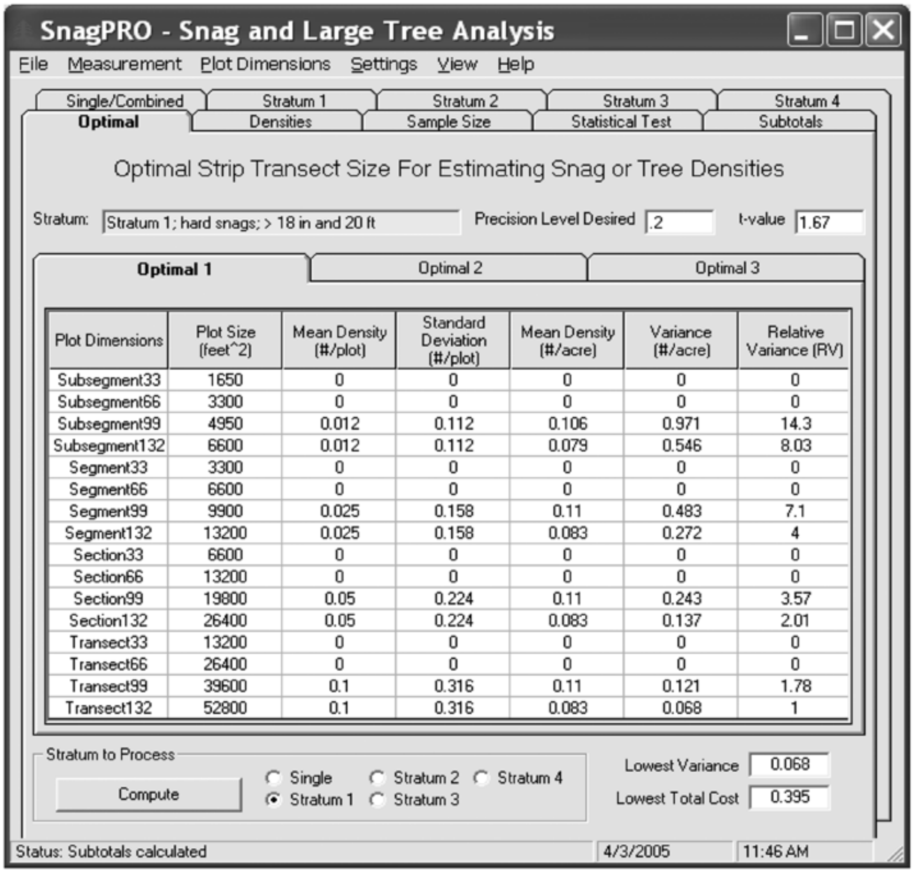
A series of message boxes will appear. Enter:

- "3" into the Number of Strata message box.
- "1" into the General Cost per Sample box (this is the code we chose for stratum 1).
- "1759" for Stratum 1 Size.
- "1381" for Stratum 2 Size.
- "1243" for Stratum 3 Size.

Stratum 1 analysis—

The results on the Optimal page are for hard snags in stratum 1. Write this description in the Stratum box at the top of the page. On the Optimal 1 page (fig. 16), the narrower plots (Width33 and Width66) have zeros for Mean Density (#/plot) because only one qualifying snag was found in stratum 1, at a distance of 46 ft from the centerline (shown on the Summary Statistics page). Mean Density (#/acre) for the wider plots is estimated to be about 0.1 snag per acre.

On the Optimal 3 page, the Transect99 plots have the lowest Total Cost value ($2,491.68), making this the optimal plot size. This excludes plots without snags



Optimal Strip Transect Size For Estimating Snag or Tree Densities

Stratum: Stratum 1; hard snags; > 18 in and 20 ft
Precision Level Desired .2
t-value 1.67

Plot Dimensions	Plot Size (feet^2)	Mean Density (#/plot)	Standard Deviation (#/plot)	Mean Density (#/acre)	Variance (#/acre)	Relative Variance (RV)
Subsegment33	1650	0	0	0	0	0
Subsegment66	3300	0	0	0	0	0
Subsegment99	4950	0.012	0.112	0.106	0.971	14.3
Subsegment132	6600	0.012	0.112	0.079	0.546	8.03
Segment33	3300	0	0	0	0	0
Segment66	6600	0	0	0	0	0
Segment99	9900	0.025	0.158	0.11	0.483	7.1
Segment132	13200	0.025	0.158	0.083	0.272	4
Section33	6600	0	0	0	0	0
Section66	13200	0	0	0	0	0
Section99	19800	0.05	0.224	0.11	0.243	3.57
Section132	26400	0.05	0.224	0.083	0.137	2.01
Transect33	13200	0	0	0	0	0
Transect66	26400	0	0	0	0	0
Transect99	39600	0.1	0.316	0.11	0.121	1.78
Transect132	52800	0.1	0.316	0.083	0.068	1

Stratum to Process
○ Single ● Stratum 1 ○ Stratum 2 ○ Stratum 3 ○ Stratum 4

Lowest Variance 0.068
Lowest Total Cost 0.395

Status: Subtotals calculated 4/3/2005 11:46 AM

Figure 16—Optimal 1 page: first of three optimal pages showing size (m^2), mean, standard deviation, variance, and relative variance for each plot size for hard snags in a stratified Douglas-fir/western hemlock forest landscape. No qualifying snags were found in the two narrowest plots. Therefore, these cells only contain "0."

Total Cost is high for the nonstratified landscape with 696 plots required; if we analyze strata, the sample size required would be substantially lower.

in them. The Total Cost is extremely high within this stratum, because SnagPRO is currently treating it as a nonstratified landscape and the estimated sample size required within this stratum is 696 plots (shown on the Optimal 2 page). If we analyze as strata within a subwatershed, however, the sample size required would be substantially lower.

Stratum 2—

To sum and subtotal the values for each plot size in stratum 2:

- Click the **Optimal** tab.
- Go to **Stratum to Process**.
- Select **Stratum 2**.
- Click **Compute**.

Enter “5” when asked for the General Cost per Sample code. Note on the Stratum 2 page that, as in stratum 1, only one snag was encountered. The decay class for this snag, however, was “4,” which makes it a soft snag, therefore not qualifying in this analysis and making the hard snag density equal to zero.

Stratum 3—

To sum and subtotal the values for each plot size in stratum 3:

- Click the **Optimal** tab.
- Go to **Stratum to Process**.
- Select **Stratum 3**.
- Click **Compute**.

Enter “6” when asked for the General Cost per Sample code. In contrast to the previous analyses, many hard snags were encountered in stratum 3.

Note the estimated mean snag densities in each of the plot sizes on the Optimal 1 page. The three narrowest plots are in close agreement with each other (3.52 to 3.8 snags per acre), whereas for plots that are 132 ft wide, the density drops considerably, to 2.6 snags per acre. In this situation, it is best to select one of the narrower plots, because the lower density may have been caused by observers “missing” some of the snags on the outer boundaries of plots.

Looking at the Optimal 2 page for stratum 3, note that Transect66 plots require the lowest number of samples (23.7 plots), but when these samples are converted to acres, the Section66 plots have the minimum number of acres (13.9 acres), suggesting these as the better plot size.

Skim the values listed in the Product column on the Optimal 3 page. When Relative Cost is multiplied by the Relative Variance, we find the Section66 plots have the lowest product (14.5) of all plot sizes. Consequently, based on Wiegert’s (1962) method, this is the optimal plot size for sampling snags in this forest stratum.

Look at Total Cost to see if total costs support the values found in the Product column. If so, then this plot size would minimize our costs ($182) and achieve the desired precision. Additionally, we would not jeopardize accuracy by sampling beyond the point of clear visibility from the centerline. Section66 plots, therefore, seem to be the optimal plot size for use within stratum 3, but are these plots independent?

To test for independence, switch to the **Summary Statistics** page and run the serial correlation test for Section66 plots in stratum 3. To do this:

1. Click on the **Correlation** button in the bottom-right corner of the screen.
2. Enter “Section” when the first message box appears labeled “Correlation Length.”
3. Enter “66” into the Correlation Width box.

The message box displays the correlation coefficient (r = 0.0) and coefficient of determination ($r^2 = 0.0$). The extremely low r^2 value (0.0) verifies that adjacent Section66 plots appear to be independent sampling units. Therefore, this plot size is used for the remainder of the analyses.

Stratified density analysis—

To obtain an estimate of the required sample size for this subwatershed, we first need an estimate of the stratified mean density to enter in the sample size equation. To do this:

- Click on the **Densities** page.
- Activate the **Stratified-Random Sampling Equation** page.
- Click the **Calculate Stratified Values** button, toward the bottom of the page.

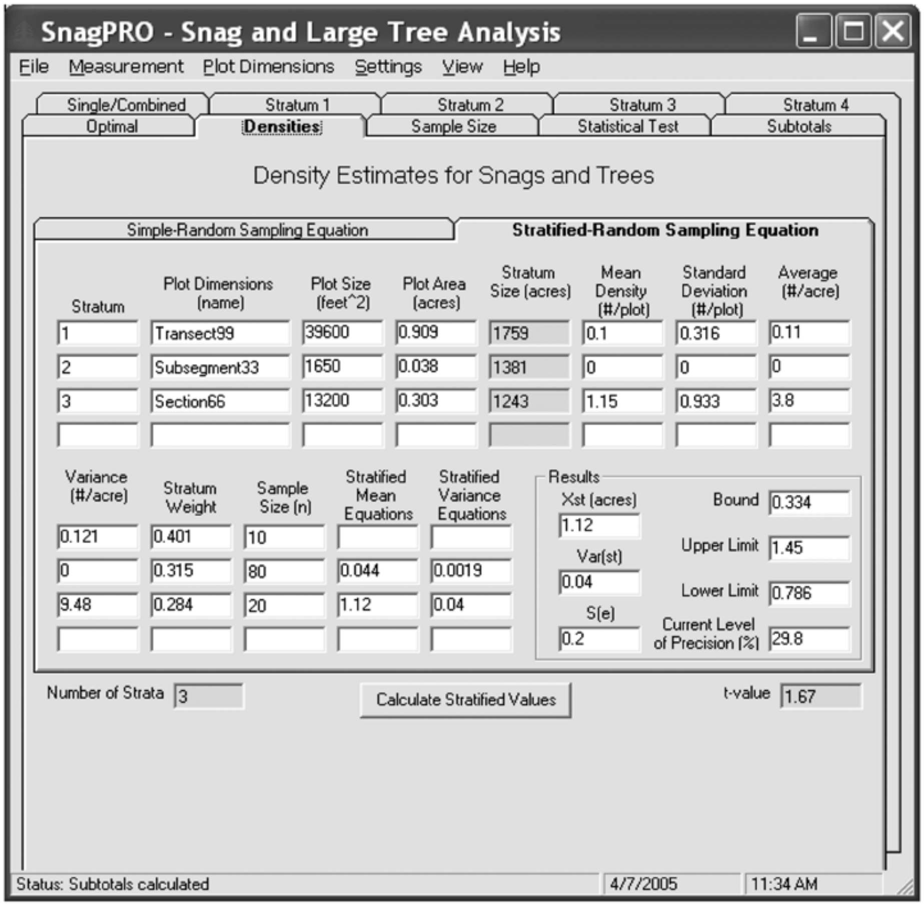
SnagPRO transfers all statistics to the Densities page and fills in the Stratum Sizes (acres) with previous entries. If necessary, values in the shaded boxes can be changed. It is estimated that there are 1.12 hard snags (± 0.334) per acre in this subwatershed (fig. 17) at a 29.8 percent level of precision. Results indicate that the estimated hard snag density for this subwatershed is within 29.8 percent of the true mean under a 90-percent confidence interval, which is not as precise as desired.

Sample size determination—

The next step is to determine the sample size needed to achieve the desired precision. Sample sizes for stratified subwatersheds are calculated on the Sample Size page, so activate this page. SnagPRO transferred the statistics to the Sample Size page once the stratified density estimate was calculated.

In the lower portion of the Sample Size page (fig. 18), see the output from two sample size equations—Optimal Allocation and Proportional Allocation. The Optimal allocation method incorporates the strata variances into its calculations, estimating that 55.2 samples are required to obtain a stratified mean within 20 percent of the true mean 90 percent of the time. These 55 samples are then divided among the three strata: 7.61 plots in stratum 1 (clearcuts), no plots in stratum 2 (second growth), and 47.6 plots in stratum 3 (old growth). With rounding, this results in 56 total samples required: 8 in stratum 1 and 48 in stratum 3.

To the right of the page is the heading Proportional Allocation. This method uses the overall variance of the subwatershed to allocate the samples, based on the



Stratum	Plot Dimensions (name)	Plot Size (feet^2)	Plot Area (acres)	Stratum Size (acres)	Mean Density (#/plot)	Standard Deviation (#/plot)	Average (#/acre)
1	Transect99	39600	0.909	1759	0.1	0.316	0.11
2	Subsegment33	1650	0.038	1381	0	0	0
3	Section66	13200	0.303	1243	1.15	0.933	3.8

Variance (#/acre)	Stratum Weight	Sample Size (n)	Stratified Mean Equations	Stratified Variance Equations
0.121	0.401	10		
0	0.315	80	0.044	0.0019
9.48	0.284	20	1.12	0.04

Results

Xst (acres) 1.12

Var(st) 0.04

S(e) 0.2

Bound 0.334

Upper Limit 1.45

Lower Limit 0.786

Current Level of Precision (%) 29.8

Number of Strata 3

t-value 1.67

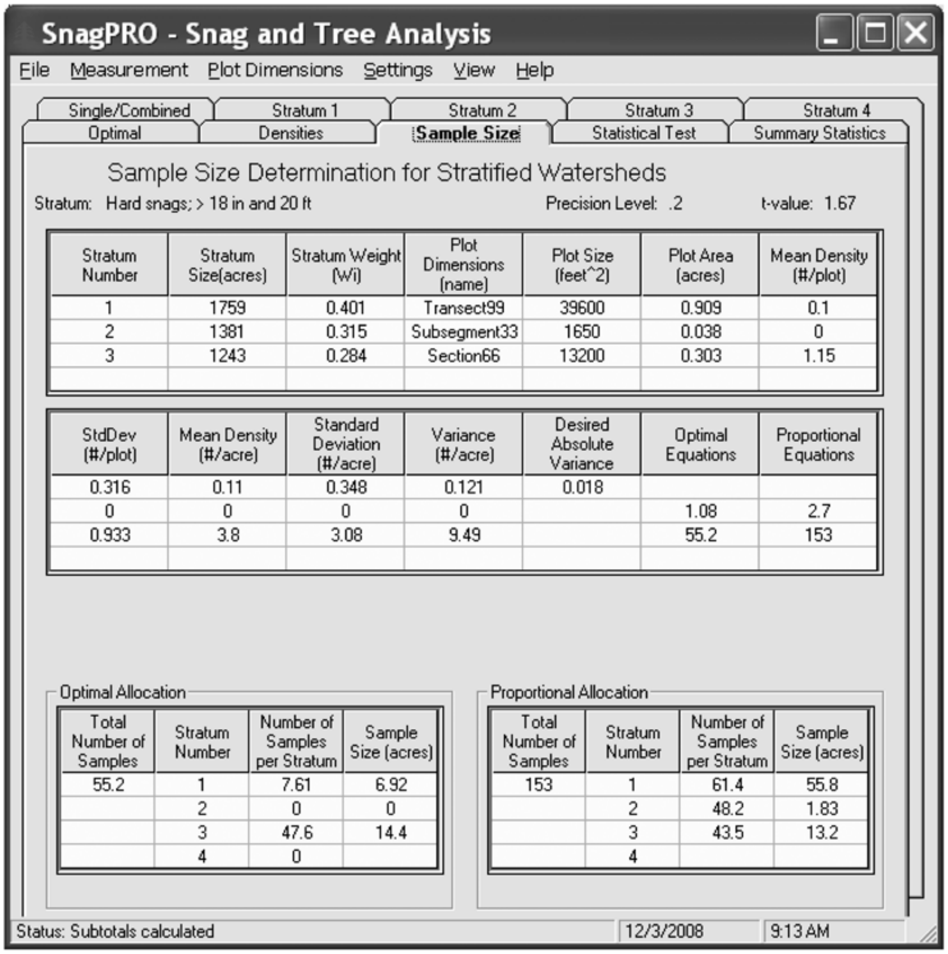
Status: Subtotals calculated | 4/7/2005 | 11:34 AM

Figure 17—Stratified mean density estimate with a 90-percent confidence interval for qualifying hard snags for the Douglas-fir/western hemlock landscape. Data are from page labeled “Tutorial_data_II_English” in the Snag_Tutorial_Data file.

relative size of each stratum. The results state that 153 samples are needed to obtain the same precision (20 percent of the true mean 90 percent of the time), which, after rounding to whole numbers for each stratum, yields 61 plots in stratum 1, 48 plots in stratum 2, and 44 plots in stratum 3.

Both sample size equations are in agreement regarding the number of samples needed for stratum 3. There are large discrepancies however, for stratum 1 and stratum 2. The optimal method suggests focusing most of the effort in stratum 3 (n = 47.6), which has the largest variance relative to the other strata. The optimal method recommends no samples in stratum 2, whereas the proportional method suggests 48.2 sample plots.

In this situation, it is best to follow the numbers suggested by the optimal allocation equation. That is, no additional sampling is needed within strata 1 and 2, unless it is suspected that encountering such low snag numbers within these



Sample Size Determination for Stratified Watersheds

Stratum: Hard snags; > 18 in and 20 ft

Precision Level: .2

t-value: 1.67

Stratum Number	Stratum Size(acres)	Stratum Weight (Wi)	Plot Dimensions (name)	Plot Size (feet^2)	Plot Area (acres)	Mean Density (#/plot)
1	1759	0.401	Transect99	39600	0.909	0.1
2	1381	0.315	Subsegment33	1650	0.038	0
3	1243	0.284	Section66	13200	0.303	1.15

StdDev (#/plot)	Mean Density (#/acre)	Standard Deviation (#/acre)	Variance (#/acre)	Desired Absolute Variance	Optimal Equations	Proportional Equations
0.316	0.11	0.348	0.121	0.018		
0	0	0	0		1.08	2.7
0.933	3.8	3.08	9.49		55.2	153

Optimal Allocation

Total Number of Samples	Stratum Number	Number of Samples per Stratum	Sample Size (acres)
55.2	1	7.61	6.92
	2	0	0
	3	47.6	14.4
	4	0	

Proportional Allocation

Total Number of Samples	Stratum Number	Number of Samples per Stratum	Sample Size (acres)
153	1	61.4	55.8
	2	48.2	1.83
	3	43.5	13.2
	4		

Status: Subtotals calculated

12/3/2008

9:13 AM

Figure 18—Sample size page. Optimal and proportional sample size calculations for sampling hard snags in each of three strata on the Douglas-fir/western hemlock landscape. Data are from page labeled ‘Tutorial_data_II_English’ in the file named Snag_Tutorial_Data file.

As additional data are collected, entered, and analyzed, the variances and thus the required sample size may change within a stratum.

stands for the pilot sample was an inaccurate representation of the conditions. Instead, focus sampling effort in stratum 3 by surveying 14 additional transects (400-ft lengths) to obtain 28 additional Section66 plots. Sampling will proceed more quickly now that the optimal plot width has been identified. This eliminates the need to measure all the distances of snags from the centerline. After completion of these 28 plots, the estimated sample size requirement of 48 will have been met. Data can then be analyzed to determine whether to continue sampling to increase precision, or to stop because precision meets the sampling objectives.

Again, remember that the sample size equations simply provide an estimate of the number of samples required, given current sample data. As additional data are collected, entered, and analyzed, the variances and thus the required sample size may change within a stratum. This possibility increases if the pilot sample data are a poor representation of the variation within a stratum. Consequently, the best way to avoid oversampling (where large sample sizes are required) is to continually enter data in SnagPRO and periodically calculate a mean density and its bound to determine the current precision. See the discussion of sample size determination in the “SnagPRO Analysis” section for a description of the advantages and disadvantages of optimal and proportional allocation methods for determining sample size.

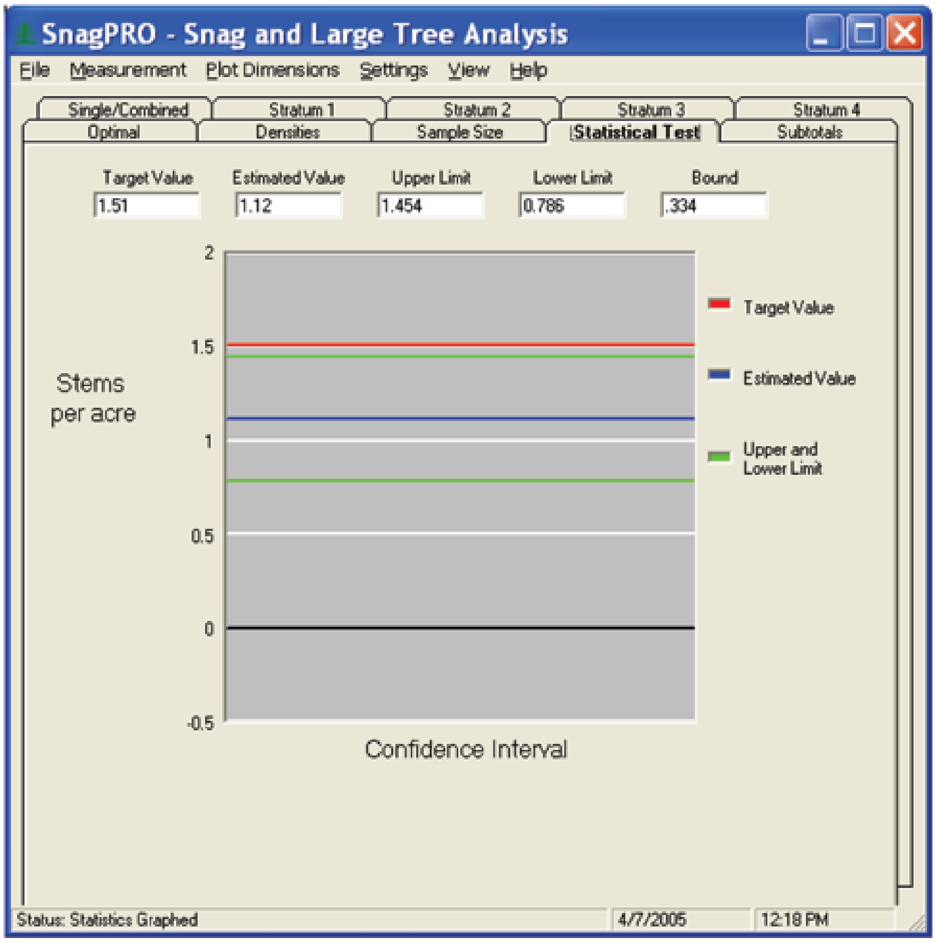
Compare to target density—
Assume that 60 samples have been collected, and thus it is appropriate to test whether the estimated density of snags meets the targeted density identified in the forest plan. The bound on the density estimate for the 90-percent confidence interval has already been calculated, which is 1.12 ± 0.334 target snags per acre (see Data Sheet page). Now activate the page labeled **Statistical Test** by clicking on this tab. To conduct the test, follow these steps:

1. The analysis is based on the null hypothesis: H_o: There is no difference between the estimated and the targeted hard snag densities.
2. Assume that the target density for hard snags in this subwatershed is “1.51” snags per acre. Enter this value in the shaded Target Value box.
3. Enter the estimated snag density of “1.12” snags per acre into the shaded Estimated Value box.
4. Enter the estimated Bound for a 90-percent confidence interval: “0.334.”

Once the necessary information has been entered onto the Statistical Test page, a graph depicting the results is automatically created (fig. 19). The graph shows that the line representing the target density for snags does not overlap the upper and lower limits of the estimated density. Consequently, the null hypothesis of no difference between the estimated and targeted snag densities is rejected. These test results, however, are inconclusive until we have attained our desired level of precision. The 14 additional transects (which contain 28 section samples) in stratum 3 should meet these requirements. Data from the additional samples can then be considered and new results entered on the Statistical Test page.

Conclusions for multiple strata—

- Results suggest that the estimated density of hard snags (1.12 ± 0.334 snags per acre) on this landscape fails to meet the targeted densities listed in the forest plan (1.51 snags per acre).
- Results also indicate that snags needed to sustain woodpecker populations may be inadequate in this subwatershed, because large portions of the subwatershed do not contain hard snags. This is a problem because woodpeckers are territorial.



Target Value: 1.51
Estimated Value: 1.12
Upper Limit: 1.454
Lower Limit: 0.786
Bound: .334

Status: Statistics Graphed | 4/7/2005 | 12:18 PM

Figure 19—Statistical test page. Graph depicting test for significant difference between estimated and targeted densities of qualifying snags on Douglas-fir/western hemlock landscape. Input data are from figure 17.

- These tests are inconclusive, however, until we attain the specified level of precision.
- Snags in all decay classes averaged about 2.48 snags (± 0.64) per acre.

Management options include (1) continue sampling to increase the precision of estimates and to determine whether results of the analysis will change; (2) sample recent clearcut areas for the number and quality of retained snags; (3) increase snag retention efforts as part of timber harvest planning, layout, and implementation; and (4) initiate snag creation programs to increase the density and improve the distribution of snags in the subwatershed.

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Metric Equivalents

When you know:	Multiply by:	To find:
Inches (in)	2.54	Centimeters (cm)
Feet (ft)	0.305	Meters (m)
Acres	0.405	Hectares (ha)

Literature Cited

Bate, L.J. 1995. Monitoring woodpecker abundance and habitat in the central Oregon Cascades. Moscow, ID: University of Idaho. 116 p. M.S. thesis.

Bate, L.J.; Garton, E.O.; Wisdom, M.J. 1999. Estimating snag and large tree densities and distributions on a landscape for wildlife management. Gen. Tech. Rep. PNW-425. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 76 p.

Bate, L.J.; Torgersen, T.R.; Wisdom, M.J.; Garton, E.O.; Clabough, S.C. 2008. Log sampling methods and software for stand and landscape analyses. Gen. Tech. Rep. PNW-746. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 93 p.

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.

Beebe, S.B. 1974. Relationships between insectivorous hole-nesting birds and forest management. New Haven, CT: Yale University, School of Forestry and Environmental Studies. 49 p.

Betts, B.J. 1998. Roosts used by maternity colonies of silver-haired bats in northeastern Oregon. Journal of Mammology. 79(2): 643-650.

Brown, R.E., tech. ed. 1985. Management of wildlife and fish habitats in forests of western Oregon and Washington. R6-F&WL-192-1985. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 332 p.

Bull, E.L.; Carter, B.; Henjum, M.; Holthausen, R.; Johnson, J.; Mellen, K. 1991. Monitoring protocol—October 1991. Protocol for monitoring woodpeckers and snags in the Pacific Northwest Region of U.S. Forest Service. 8 p. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Forestry and Range Sciences Laboratory, 1401 Gekeler Lane, La Grande, OR 97850.

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.; Marx, D.B. 1990. How to determine snag density. Western Journal of Applied Forestry. 5(2): 56-58.

Bull, E.L.; Parks, C.G.; Torgersen, T.R. 1997. Trees and logs important to wildlife in the Interior Columbia River basin. Gen. Tech. Rep. PNW-391. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 55 p.

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 Forest and Range Experiment Station. 19 p.

Bütler, R.; Schlaepfer, R. 2004. Spruce snag quantification by coupling colour infrared aerial photos and a GIS. Forest Ecology and Management. 195: 325-339.

Campbell, L.A.; Hallett, J.G.; O'Connell, M.A. 1996. Conservation of bats in managed forests: use of roosts by *Lasionycteris noctivagans*. Journal of Mammalogy. 77: 976-984.

Caton, E. 1996. Effects of fire and salvage-logging on a cavity-nesting bird community in northwestern Montana. Missoula, MT: University of Montana. 115 p. Ph.D. dissertation.

Cline, S.P.; Berg, A.B.; Wight, H.M. 1980. Snag characteristics and dynamics in Douglas-fir forests, western Oregon. Journal of Wildlife Management. 44(4): 773-786.

Cochran, W.G. 1977. Sampling techniques. 3rd ed. New York: John Wiley and Sons. 428 p.

Dickson, J.G.; Conner, R.N.; Williamson, J.H. 1983. Snag retention increases bird use of a clear cut. Journal of Wildlife Management. 47(3): 799-804.

Dixon, R. 1995. Ecology of white-headed woodpeckers in the central Oregon Cascades. Moscow, ID: University of Idaho. 148 p. M.S. thesis.

Ducey, M.J.; Jordan, G.J.; Gove, J.H.; Valentine, H.T. 2002. A practical modification of horizontal line sampling for snag and cavity tree inventory. Canadian Journal of Forest Research. 32: 1217-1224.

Ffolliott, P.F. 1983. Implications of snag policies on management of southwestern ponderosa pine forests. In: Davis, J.W.; Goodwin, G.A.; Ockenfels, R.A., tech. ed. Snag habitat management symposium. Gen. Tech. Rep. RM-99. Flagstaff, AZ: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 28-32.

Hann, W.J.; Jones, J.L.; Karl, M.G.; Hessburg, P.F.; Keane, R.E.; Long, D.G.; Menakis, J.P.; McNicoll, C.H.; Leonard, S.G.; Gravenmier, R.A.; Smith, B.G. 1997. Landscape dynamics of the basin. 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. Gen. Tech. Rep. PNW-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 334-1056.

Harmon, M.E.; Sexton, J. 1996. Guidelines for measurements of woody detritus in forest ecosystems. Publ. No. 20. Seattle, WA: U.S. Long-Term Ecological Research (LTER) Network Office, University of Washington. 73 p.

Hope, S.; McComb, W.C. 1994. Perceptions of implementing and monitoring wildlife tree prescriptions on national forests in western Washington and Oregon. Wildlife Society Bulletin. 22: 383-393.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs. 54(2): 187-211.

Husch, T.; Beers, W.; Kershaw, J. 1972. Forest mensuration. 2nd ed. New York: The Ronald Press Company. 410 p.

Kenning, R.; Ducey, M.; Brisette, J.; Gove, J. 2005. Field efficiency and bias of snag inventory methods. Canadian Journal of Forest Research: 35(12): 2900-2910.

Krebs, C.J. 1989. Ecological methodology. New York: Harper Collins Publishers, Inc. 654 p.

Laudenslayer, W.F., Jr. 2002. Cavity-nesting bird use of snags in eastside pine forests of northeastern California. Gen. Tech. Rep. PSW-181. Fresno, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 223-236.

Ligon, J.D. 1973. Foraging behavior of the white-headed woodpecker in Idaho. Auk. 90: 862-869.

Mariani, J.M.; Manuwal, D.A. 1990. Factors influencing brown creeper (*Certhia americana*) abundance patterns in the southern Washington Cascade Range. Studies in Avian Biology. 13: 53-57.

Mellen, K.; Marcot, B.G.; Ohmann, J.L.; Waddell, K.; Livingston, S.A.; Willhite, E.A.; Hostetler, B.B.; Ogden, C.; Dreisbach, T. 2006. DecAID, the decayed wood advisor for managing snags, partially dead trees, and down wood for biodiversity in forests of Washington and Oregon. Version 2.0. Portland, OR: USDA Forest Service, Pacific Northwest Region and Pacific Northwest Research Station; USDI Fish and Wildlife Service, Oregon State Office. http://wwwnotes.fs.fed.us:81/pnw/DecAID/DecAID.nsf. (April 4, 2008).

Morrison, M.L.; Dedon, M.F.; Raphael, M.G.; Yoder-Williams, M.P. 1986. Snag requirements of cavity-nesting birds: Are USDA Forest Service guidelines being met? Western Journal of Applied Forestry. 1: 38-40.

Ormsbee, P.C.; McComb, W.C. 1998. Selection of day roosts by female longlegged myotis in the central Oregon Cascade Range. Journal of Wildlife Management. 62(2): 596-603.

Otvos, I.S. 1979. The effects of insectivorous bird activities in forest ecosystems: an evaluation. In: Dickson, J.G.; Connor, R.N.; Fleet, R.R. [and others], eds. The role of insectivorous birds in forest ecosystems. New York: Academic Press: 341-374.

Parks, C.G.; Bull, E.L.; Torgersen, T.R. 1997. Field guide for the identification of snags and logs in the interior Columbia River basin. Gen. Tech. Rep. PNW-390. Portland, OR: U.S. Department Agriculture, Forest Service, Pacific Northwest Research Station. 40 p.

Quigley, T.M.; Haynes, R.W.; Graham, R.T., tech. eds. 1996. Integrated scientific assessment for ecosystem management in the interior Columbia basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-382. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 303 p.

Rose, C.L.; Marcot, B.G.; Mellen, T.K.; Ohmann, J.L.; Waddell, K.L.; Lindley, D.L.; Schreiber, B. 2001. Decaying wood in Pacific Northwest forests: concepts and tools for habitat management. In: Johnson, D.H.; O'Neil, T.A., eds. Wildlife-habitat relationships in Oregon and Washington. Corvallis, OR: Oregon State University Press: 580-623. http://www.nwhi.org/inc/data/GISdata/docs/chapter24.pdf. (April 2007)

Saab, V.; 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-11. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 17 p.

Sokal, R.R.; Rohlf, F.J. 1981. Biometry. New York: W.H. Freeman and Company. 859 p.

Styskel, E.W. 1983. Problems in snag management implementation–a case study. In: Snag habitat management: proceedings of the symposium. Gen. Tech. Rep. RM-99. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 24-27.

Swihart, R.K.; Slade, N.A. 1985. Testing for independence of observations in animal movements. Ecology. 66(4): 1176-1184.

Thomas, J.W.; Anderson, R.G.; Maser, C.; Bull, E.L. 1979. Snags. In: Wildlife habitats in managed forests--the Blue Mountains of Oregon and Washington. Agric. Handb. 553. Washington, DC: U.S. Department of Agriculture: 60-77

U.S. Department of Agriculture, Forest Service. 1991. Field procedures guide: stand examination program. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region.

Wiegert, R.G. 1962. The selection of an optimum quadrat size for sampling the standing crop of grasses and forbs. Ecology. 43(1): 125-129.

Wisdom, M.J.; Bate, L.J. 2008. Snag density varies with intensity of timber harvest and human access. Forest Ecology and Management. 255(7): 2085-2093.

Appendix 1: General Snag and Tree Sampling Guidelines

1. Sampling objectives.
    a. What snag (tree) size(s) will be surveyed (diameter and height)?
    b. What condition (decay class) of snags (trees) will be surveyed?
    c. How will the data be used? Baseline data? Compliance data? This often dictates answers to the following questions.
    d. How precise does the estimate need to be?
    e. Is snag/tree species important? If so, why?
    f. Will signs of wildlife use be recorded (for example, woodpecker foraging, cavities?)?
    g. Are estimates for separate areas needed?
2. Landscape definition and selection.
    a. Define the landscape, or area of interest, by delineating the boundaries. This area is the sampling frame, within which a random sample is drawn for the purpose of making inferences to the entire area.
3. Landscape stratification.
    a. Visit the survey area first, if it is unfamiliar, with a map delineating the boundaries. What differences/similarities are visible in regard to snag/tree abundance and/or vegetative structure across the landscape?
    b. Obtain reference maps for field use, such as geographic information system maps or U.S. Geological Survey orthoquad maps, or both. Always request metadata (data definitions) for the polygon data. Maps should display the following information:
        i. Road system with difference in road type and maintenance level displayed.
        ii. Stand, polygon, or vegetation units and their respective unique numeric identifiers.
        iii. Current seral stage of vegetation at a scale of 1:31,680 or better resolution. Keep in mind that scale is a ratio or fraction, so polygons mapped at 1:24,000 scale will appear larger than they do in the 1:31,680-scale map. This information may be on one or more maps.
    c. Query the polygon database for information about forest type (low versus high elevation, dry versus moist), management activities, seral stage, disturbance history (wind, fire, insects, and disease), and any other factors that may affect snag/tree abundance. Ensure that the report includes types of management activities, such as harvest method used, slash and burn prescriptions, thinning, and snag/tree retention.

  d. Check the map and information from the polygon database for general agreement with features that can be viewed with aerial photographs of the area. The degree to which the map and database information appear similar to what is shown on the aerial photographs provides a good indication about how much field reconnaissance will be needed for accurate landscape stratification.
  e. Revisit the survey area with the field maps. Plan to spend at least one day to validate the information on the map(s) and in the report from the query. Assign each polygon to a stratum. Estimate the number of acres (ha) within each polygon or stratum.
4. Establishing transects
  a. There are two options for establishing transects: the single-stratum landscape method, and the stratified method. For the single-stratum landscape method, follow these steps to establish transects within a single polygon or a nonstratified landscape:
    i. Randomly place a grid over the area.
    ii. Randomly select 10 grid points for sampling.
    iii. Randomly select compass bearings for each of the 10 transect starting points.
  b. For the stratified method on heterogeneous landscapes composed of numerous polygons or units, it may be more efficient to randomly select polygons for sampling. To do this:
    i. Select polygons for sampling by randomly picking polygon unit numbers from the complete list of polygons within that stratum.
    ii. Place a grid over the polygon.
    iii. Randomly pick two grid points within each polygon.
    iv. Randomly pick compass bearings for each point.
5. Plot size selection
  a. Based on information gathered during the stratification process, it may be beneficial to preselect a plot size for sampling. Wide plots work best in areas of low snag densities, unlimited visibility, and easy travel conditions. Narrower plots (66 ft or 20 m wide) work best in areas of higher densities or clumped distributions or where visibility is limited. The smallest plots work best in extremely high-density areas.
  b. Postpilot sampling plot size selection. In most forested conditions, the optimal plot size for sampling is unknown until the density and distribution of the snags or trees can be evaluated. In these situations:
    i. Use pilot sample data to determine which plot sizes minimize sampling

        effort to obtain your desired objectives. See “Optimal Plot Size Analysis” section for details.
        ii. Use optimal plot size for remainder of survey.

6. Field surveying techniques
    a. Use an engineer’s surveying or measuring tape to establish transects, starting each transect from the randomly selected points (described above).
    b. Assign a unique numeric identifier to each transect, delineating the subsegment lengths (50 ft [or 12.5 m]) as you walk along the transect (400 ft or 100 m).
    c. Number each transect’s subsegments 1 through 8.
    d. Conduct a complete count of all qualifying snags or trees out to 66 ft (20 m), using the tape as centerline. A snag or tree is “in” if its midpoint is <66 ft (20 m) from the centerline (tape).
7. Data entry
    a. Open the **Snag_Tutorial_Data.xls** file.
    b. Activate the **Data Entry** sheet.
    c. Click on **Move or Copy Sheet** under the **Edit** menu.
    d. Check the box **Create a copy**.
    e. Under **To book** click on (**new book**).
    f. Rename the new file, and then use this sheet to make hard copies for fieldwork.
8. To save the entered data as a CSV file:
    a. Activate the data entry sheet.
    b. Select **Save As** from the **File** menu.
    c. Scroll to find CSV (comma delimited) (*.**csv**).
    d. Click **Save**.

Appendix 2: Field Form Explanations

1. Stratum: Enter the stratum number: 1, 2, 3, or 4.
2. Location: Enter the polygon number or the geographic coordinates where the transect originates.
3. Transect: Assign a unique numeric identifier to indicate which 100-m or 400-ft transect length is being surveyed (for example, 1, 2, 3...). No two transects within a survey area should be the same number regardless of the stratum.
4. Subsegment: Assign a unique numeric identifier (1 through 8) to indicate which 12.5-m or 50-ft-long subsegment is being surveyed. The first subsegment of each transect should start with "1." This allows SnagPRO to join consecutive subsegments.
5. Distance: Enter the distance between the midpoint of the qualifying snag or tree and the center of the transect line to the nearest foot (nearest meter). If no snag is encountered within the entire subsegment, enter "9999" under distance. It is critical to measure distances accurately. If the midpoint of a snag or tree falls directly on the boundary, include the first one, exclude the second one, and so on. If a plot width has already been selected, enter "1" for distance.
6. Species: SnagPRO can accommodate either alpha (six characters) or numeric data. Listed below are the standardized numeric species codes taken from Stand Exam Program in the Pacific Northwest Region [USDA Forest Service 1991]. Customize for your own use:

Douglas-fir/redwoods:	
Douglas-fir (*Pseudotsuga menziesii (*Mirb.) Franco)	202
Redwood (*Sequoia sempervirens* (D. Don) Endl.)	211
True firs:	
Pacific silver fir (*Abies amabilis* Dougl. ex Forbes)	011
White fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.)	015
Grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.)	017
Subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.)	019
California red fir (*Abies magnifica* A. Murray var. *magnifica*)	020
Shasta red fir (*Abies magnifica* A. Murray var. *shastensis* Lemmon)	021
Noble fir (*Abies procera* Rehd.)	022
Cedars:	
Port-Orford-cedar (*Chamaecyparis lawsoniana* (A. Murr.) Parl.)	041
Alaska-cedar (*Chamaecyparis nootkatensis* (D. Don) Spach)	042
Incense-cedar (*Calocedrus decurrens* (Torr.) Florin)	081

Western redcedar (*Thuja plicata* Donn ex. D. Don)	242
Larch:	
Western larch (*Larix occidentalis* Nutt.)	073
Spruce:	
Brewer spruce (*Picea breweriana* Wats.)	092
Engelmann spruce (*Picea engelmannii* Parry ex Engelm.)	093
Sitka spruce (*Picea sitchensis* (Bong.) Carr.)	098
Pines:	
Lodgepole pine (*Pinus contorta* Dougl ex. Loud)	108
Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.)	116
Sugar pine (*Pinus lambertiana* Dougl.)	117
Western white pine (*Pinus monticola* Dougl. ex D. Don)	119
Ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.)	122
Hemlock:	
Western hemlock (*Tsuga heterophylla* (Raf.) Sarg.)	263
Mountain hemlock (*Tsuga mertensiana* (Bong.) Carr.)	264
Hardwoods:	
Bigleaf maple (*Acer macrophyllum* Pursh)	312
Red alder (*Alnus rubra* Bong.)	351
Western paper birch (*Betula papyrifera* Marsh.)	376
Pacific madrone (*Arbutus menziesii* Pursh)	361
Golden chinkapin (*Castanopsis chrysophylla* (Dougl.) A. DC.)	431
Oregon ash (*Fraxinus latifolia* Benth.)	542
Tanoak (*Lithocarpus densiflorus* (Hook. & Arn.) Rehd.)	631
Quaking aspen (*Populus tremuloides* Michx.)	746
Black cottonwood (*Populus trichocarpa* Torr. & Gray)	747
Oregon white oak (*Quercus garryana* Dougl. ex Hook.)	815
California black oak (*Quercus kelloggii* Newb.)	818
Oregon myrtle (*Umbellularia californica* (Hook. & Arn.) Nutt.)	981
Other conifers:	
Subalpine larch (*Larix lyallii* Parl.)	072
Cypress (*Cupressus* L.)	050

All junipers (*Juniperus* L.)	060
Pacific yew (*Taxus brevifolia* Nutt.)	231
Knobcone pine (*Pinus attenuata* Lemm.)	103
Limber pine (*Pinus flexilis* James)	113
Whitebark pine (*Pinus albicaulis* Engelm.)	101

7. Class: Enter the numeric code for the appropriate decay or structural class of the snag or tree encountered. Snag data should be collected on a data form separate from large trees. For snags, the numeric value should increase with increasing amounts of decay. For example, Parks and others (1997) have categorized snags into three structural classes.

A. Snag classes

1. Snags that have recently died.
2. Snags that have been dead several years and have lost some branches and bark.
3. Snags that have been dead more than several years and lack branches and bark (except grand fir and Douglas-fir, which tend to retain bark).

By contrast, numeric codes for the structural class of trees should decrease with increasing amounts of decay. For example:

B. Tree classes

1. Hollow
2. Some decay evidence (broken branch or top, fungi, wildlife signs)
3. Broomed trees
4. Sound

Refer to Bull and others (1997) for detailed information on establishing categories and identifying trees useful to wildlife in the field.

8. D.B.H.: Enter the diameter at breast height of the snag or tree encountered measured with a d.b.h. stick or tape, to the nearest inch (cm).

9. Height: Enter the height of the snag or tree to the nearest foot (m).

10. Cavity: Enter the appropriate numeric code to indicate any nesting use of the snag or tree under consideration. In cases where it is not possible to determine whether any cavities are present, leave the Cavity field blank so that the snag is not included in the availability total.

0. No cavities.
1. New cavity indicated by one or all of following: fresh wood chips on ground below hole, light-colored wood around entrance, bird occupying cavity (excavated or natural).
2. Old cavity: gray-colored chips on ground below hole, gray-colored wood around entrance, no sign of bird occupying cavity (excavated or natural).

3. Both old and new cavities
4. Other wildlife use.

11. Foraging: Enter the appropriate numeric code to indicate any foraging use of the snag or tree under consideration.

1. New foraging indicated by light-colored wood around foraging sign, recent scaling.
2. Old foraging indicated by gray-colored wood around foraging sign.
3. Both old and new foraging.
4. No foraging signs.

Appendix 3: General Computer Instructions for Snag or Large-Tree Analyses Within a Single Stratum

1. To get started:
   a. Double click on **SnagPRO.exe**.
   b. Click on **Snags or Trees** button under **Habitat Component**.
   c. From the **Measurement** menu, select **Metric** or **English**.
   d. Open your data file by clicking on **Open** under the **File** menu.
   e. Highlight the name of your comma-separated value (CSV) file and click **Open**.
2. To apply formula:
   a. Notice Segment and Section fields have been added and numbers computed for each column.
   b. Notice that four Width columns have been added.
   c. Click on **Multiple** tab in bottom left of screen for analyses with multiple species included; click **Single** for analysis of only one species.
   d. Enter minimum diameter at breast height in message box labeled "D.B.H."
   e. Enter minimum height of snags or trees to be considered in message box labeled "Height"; enter "0" if all heights will be considered or heights were not measured.
   f. Enter maximum value for decay or structural class in message box labeled "Decay Class."
   g. Enter numeric code of snag or tree species you would like to exclude (to include, if Single button was clicked) in box labeled "Species."
   h. From the **View** menu, decide upon a cost code for each stratum prior to initiating next section.
3. Summarize statistics:
   a. Click on **Optimal** tab at the top of the screen.
   b. Click on the first of the Optimal pages (**Optimal 1**).
   c. Check desired level of precision and t-value; if different values are desired, enter them and repeat steps 2c through 2g.
   d. Enter brief description of stratum and snag/tree characteristics for your records.
   e. In section labeled "Stratum to Process" highlight the **Single** circle.
   f. Click the **Compute** button.
   g. Examine Optimal pages for statistics, estimated sample size required, sample area required, lowest product and total cost values.
   h. Print copy of page if desired by selecting **Print Preview** from the **File** menu, then clicking tab labeled **Print**.

4. Conduct serial correlation test:
    a. Switch to Summary Statistics page.
    b. Click on **Correlation** button.
    c. Enter optimal transect length (section, segment, or subsegment) into “Correlation Length” input box.
    d. Enter optimal transect width into “Correlation Width” input box.
    e. Determine whether chosen plot size can be considered independent.
5. Density estimate:
    a. Click on **Densities** tab.
    b. Check to ensure t-value is correct for the analysis.
    c. Select **Simple-Random Sampling Equation** tab.
    d. Examine Densities sheet for estimated parameters and current level of precision to decide whether an adequate number of samples have been taken. Refer back to the Optimal page for additional number of samples needed to achieve desired level of precision.
6. Statistical test:
    a. Enter the target density into the “Target Value” box.
    b. Enter the estimated density into the “Estimated Value” box.
    c. Enter the bound of the estimated density.
    d. If target value (red line) falls within the bounds (green lines) of the estimated value (blue line), accept the null hypothesis that there is no difference between the estimated and target values for the given variable; otherwise, reject the null hypothesis.
    e. For borderline cases, consider additional sampling effort.

Appendix 4: General Computer Instructions for Snag or Large-Tree Analyses on a Stratified Landscape

1. To get started:
   a. Double click on **SnagPRO.exe**.
   b. Click on **Snags or Trees** button under **Habitat Component**.
   c. From the **Measurement** menu, select **Metric** or **English**.
   d. Open your data file by clicking on **Open** under the **File** menu.
   e. Highlight the name of your comma-separated value (CSV) file and click **Open**.
   f. Note that Segment and Section fields have been added and computed for each column.
   g. Note that four Width columns have been added.
2. To apply formula to all strata:
   a. Click on **Stratum 1** tab.
   b. Click on **Multiple** tab in bottom left of screen for analyses with multiple species; click **Single** for analysis of only one species.
   c. Enter minimum diameter at breast height in message box labeled "D.B.H."
   d. Enter minimum height of snags or trees to be considered in message box labeled "Height"; enter "0" if all heights will be considered or heights were not measured.
   e. Enter maximum value for decay or structural class in message box labeled "Decay Class."
   f. Enter numeric code of snag or tree species you would like to exclude (to include, if Single button was clicked) in box labeled "Species."
   g. From the **View** menu, decide upon a cost code for each stratum prior to initiating next section.
3. Summarize statistics:
   a. Click on **Optimal** tab at the top of the screen.
   b. Click on the first of the Optimal pages (**Optimal 1**).
   c. Check desired level of precision and t-value; if different values are desired, enter them and repeat steps 2b through 2f.
   d. Enter brief description of stratum and snag/tree characteristics for your records.
   e. In section labeled "Stratum to Process" highlight the **Stratum 1** circle.
   f. Click the **Compute** button.
   g. Enter a numeric value for total number of strata in this analysis in the "Number of Strata" input box.

   h. Enter one of six available cost codes into "General Cost per Sample" for the first stratum.
   i. Enter the area (in hectares or acres) of each of your strata.
   j. Examine Optimal pages for statistics, estimated sample size required, sample area required, lowest product and total cost values.
   k. Print copy of page if desired by selecting **Print Preview** from the **File** menu, then clicking **Print** tab.
4. Conduct serial correlation test:
   a. Switch to Summary Statistics page.
   b. Click on **Correlation** button.
   c. Enter optimal transect length (section, segment, or subsegment) into "Correlation Length" input box.
   d. Enter optimal transect width into "Correlation Width" input box.
   e. Determine whether chosen plot size can be considered independent.
   f. Repeat 3d through 3k and 4a through 4e for all strata.
5. Density estimate:
   a. Click on **Densities** tab.
   b. Check to ensure t-value is correct for the analysis.
   c. Select **Stratified-Random Sampling Equation** tab.
   d. Click **Calculate Stratified Values** button.
   e. Examine Densities sheet for estimated parameters and current level of precision to decide whether an adequate number of samples have been collected.
6. Sample size required:
   a. Click on **Sample Size** tab.
   b. Examine Optimal and Proportional sections for estimated sample sizes required within each stratum. Refer to the "Parameter Estimates for a Stratified Landscape" section on differences between two allocation methods.
7. Statistical test:
   a. Enter the target density into the "Target Value" box.
   b. Enter the estimated density into the "Estimated Value" box.
   c. Enter the bound of the estimated density.
   d. If target value (red line) falls within the bounds (green lines) of the estimated value (blue line), accept the null hypothesis that there is no difference between the estimated and target values for the given variable; otherwise, reject the null hypothesis.
   e. For borderline cases, consider additional sampling effort.

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Effects of Stand-Replacement Fire and Salvage Logging on a Cavity-Nesting Bird Community in Eastern Cascades, Washington

Abstract

We monitored the response of cavity-nesting species to three snag density treatments (high = 37-80 snags/ha, medium = 15-35 snags/ha, and low = 0-12 snags/ha) during two breeding seasons 4-5 yr post-fire and logging in Douglas-fir- ponderosa pine forests in the eastern Cascades, Washington. Snag surveys were used to describe habitat, and both breeding bird surveys and nest surveys were used to characterize the bird community. Stands with the medium snag density treatment had the highest abundance, species richness, and nesting population of cavity nesters. The reasons for this may be: 1) snags were not evenly distributed within a stand such that both clumped and dispersed snag density habitats were interspersed in this treatment, and 2) a greater proportion of ponderosa pine snags in medium density treatments may have attracted species that prefer ponderosa pine for nesting and foraging. Ponderosa pine was preferred for nest sites and large snags (> 48 cm dbh) provided nesting habitat for more species than smaller snags. However, smaller snags were used for nesting and foraging by some species.

Introduction

Snag abundance is a limiting factor for primary cavity excavators because they excavate a nest cavity each year (McClelland et al. 1979, Zarnowitz and Manuwal 1985, Bull et al. 1990). Primary cavity excavators are important members of forest ecosystems because the cavities they excavate may be used by secondary cavity nesters, including bats, American marten (*Martes americana*), many owl species, and other birds (Bevis 1994, Strangel 1994, Bull et al. 1997), and because they influence insect numbers (Mannan et al. 1980).

The snags retained during salvage logging following a fire can strongly influence the bird community (Blake 1982, Medin 1985). Designing salvage logging for snag retention is especially important in areas where there has been a stand-replacement fire. Snags in large burned areas have greater exposure to wind, causing them to fall at high rates. Morrison and Raphael (1993) found an 85% decrease in snag density (from 31.4 to 4.6 snags/ha) 18 to 23 yr following a fire in the Sierra Nevada.

The relationship between snag density and bird populations in areas of stand-replacement fires has not been well studied. Our objectives in this study were to:

(1) measure the abundance and species richness of cavity-nesting birds at different levels of snag retention, and (2) characterize nest trees, nest sites, and estimate the number of nests by primary and secondary cavity nesting birds.

Study Area

The study area was located on the eastern side of the Cascade Mountains in the Wenatchee National Forest. The study took place in areas that were affected by the high intensity stand-replacement Rat Creek fire of 1994 that burned on the Leavenworth Ranger District and adjacent private land. The landscape was dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) plant associations (Lillybridge et al. 1995). Elevations ranged from 650 m to 1300 m. Average annual precipitation is from 25 to 50 cm, falling mostly as snow.

Methods

Stand Selection

Six stands (two replicates of three treatments) that were salvaged logged in 1996 were selected for monitoring of cavity-nesting bird communities (Figure 1). These stands were selected based upon those available following logging and those that met snag retention and site characteristic criteria.

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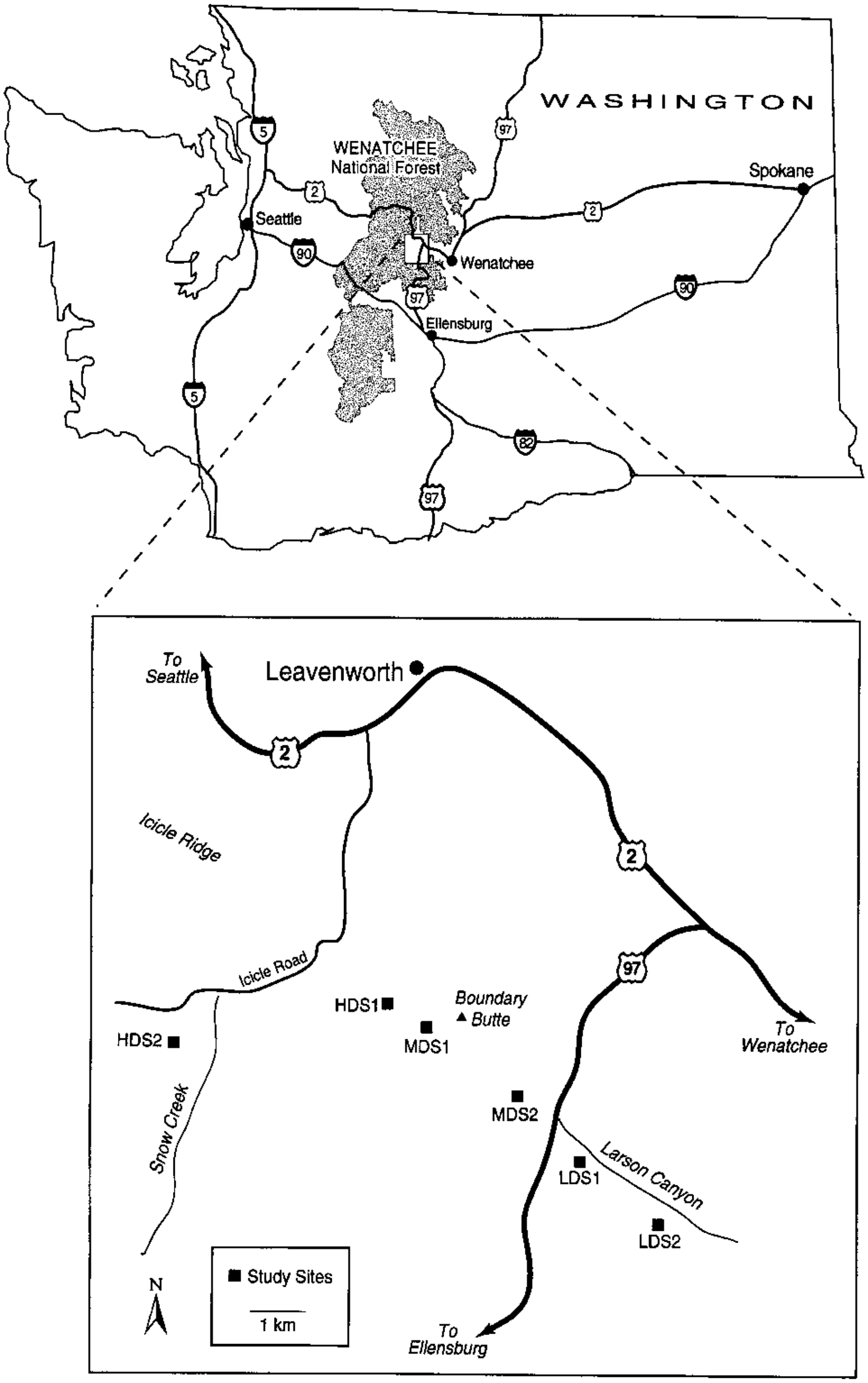

Figure 1. Location of study sites on the Wenatchee National Forest (Low Density = LD, Medium Density = MD, and High Density = HD).

Retention in the low treatment was 0-12 snags/ha. Medium retention treatment contained 15-35 snags/ha and high was characterized by 37-80 snags/ha. The six treated stands were in areas that burned with high-intensity stand-replacment fire. Each stand had a northerly aspect and was about 36.5 ha, with corrections made for topographic features. Slope averaged 40%, with elevations ranging 579-732 m.

Distance between stands was maximized to the extent possible. All but the low snag density treatments were in different drainages to increase the probability of sampling different bird populations. In the low snag density treatments, one stand was picked at the mouth of a canyon and the other near the top end of the same canyon to maximize the distance between them.

Snag Surveys

Snag density was recorded at each point-count station in two snag plots. One plot was established uphill and to the right and the other plot downhill and to the left of station center. Plot size varied with snag treatment: plots in low density measured 48 x 48 m; those in medium density measured 30 x 67 m; and those in high density measured 30 x 30 m. For each snag, tree species, dbh (diameter breast height), height, and decay stage (Cline et al. 1977) was recorded.

Breeding Bird Surveys

Seven fixed radius point-count stations were set up in each of the study stands, except for one low-density stand that only fit six stations, for a total of 41 stations. The point-count method was chosen because it is efficient and is the preferred method in rough terrain (Ralph et al. 1993). The radius for all point counts was 75 m and point-count stations were established using a systematic sampling design. The first station center was established 100 m from the edge of the treatment stand and subsequent centers were placed along a designated compass direction across slope with corrections for topography. The station centers were a minimum 225 m apart (Hutto et al. 1986, Ralph et al. 1993). In one high density and one low density stand a 100-m buffer was placed around small (<0.1 ha) green islands.

Bird abundance was surveyed using a fixed-radius counting method to provide a relative index of abundance of cavity-nesting species (Ralph et al. 1993). Bird counts were conducted during the breeding season mid-April–June, 1998 and 1999. Surveys were conducted within 30 min of sunrise and up to 5 hr following sunrise. Surveys were not conducted during periods of heavy rains or high winds. Surveys were a minimum of one week apart. All bird surveyors spent two weeks prior to initiation of point counts learning to identify birds in the study stands. Surveyors were trained to recognize birds visually, by vocalizations and drumming patterns, and flight characteristics.

Detection of woodpeckers during 10-min intervals at each count station were based on visual observation and calls (Hutto et al. 1986). For each point count, birds were classified as those within the 75-m circle, those beyond the 75-m circle, and those detected while walking between points (Manuwal and Carey 1991). Birds originally detected outside the 75-m circle that moved inside the 75-m circle were counted as being within the 75-m circle. Each point-count station was visited six times during the breeding season.

Nest surveys

Cavity nests were located through nest searches. Transects were walked through the entire stand and each snag ≥25 cm dbh was examined for evidence of a cavity. The smallest nest tree reported to be used by any cavity nester was 25 cm dbh (Scott 1978, Schroeder 1983). Fresh wood chips at the base of a cavity, and incubating behavior of adults or evidence of nestlings, were used as indicators of an active nest and were used to detect nest cavities (Zarnowitz and Manuwal 1985).

Four attributes of cavity snags were recorded: tree species, decay class (Cline et al. 1977), tree size, and surrounding spatial distribution. We compared the nest tree with other trees on paired plots. One plot was centered on the nest tree and the other plot was centered on a tree similar to the nest tree, but without a cavity, and at least 75 m away. The direction and distance from plot center, dbh, decay stage, and species of each snag within a 25-m radius of plot center were measured.

Data Analyses

All significance levels were set at $P = 0.05$. A one-way ANOVA followed by a Tukey test (Zar 1996) was used to evaluate snag densities by size class among treatments as a way of validating the implementation of the treatments. Snag size classes used for analyses were: <15 cm, 16-24 cm, and ≥25 cm. These sizes classes encompassed a range of nesting and foraging habitats used by cavity nesters (Bull et al. 1997). A G-test (Zar 1996) was used to compare the percentage of Douglas-fir and ponderosa pine snags in the different treatment areas.

A one-way ANOVA followed by a Tukey test (Zar 1996) was used to assess bird abundance among treatments. A G-test (Zar 1996) was used to determine whether cavity nesters were nesting in ponderosa pine snags in proportion to their availability on the landscape. Paired t-tests (Zar 1996) were used to compare site characteristics of nest trees with non-nest trees in paired plots. Paired t-tests were conducted for the following variables: % ponderosa pine and % Douglas-fir trees within 25 m of tree; mean dbh of surrounding trees out to 10 m; mean dbh of trees from 10-25 m; number of trees < and ≥ 25 cm within 10 m of plot center; number of trees < and ≥ 25 cm within 10-25 m of plot center; and number of trees out to 25 m at the same orientation as the nest cavity.

Results

Stand Treatment Characteristics

The number of snags ≥ 25 cm dbh and snags 16-24 cm dbh differed significantly ($P < 0.001$ and $P = 0.0058$) among the three treatment areas (Table 1). High standard error values reflect the heterogeneity of snags on the landscape.

TABLE 1. Snag densities in each treatment (low, medium, high) following stand replacement fire and salvage logging, Wenatchee National Forest, 1998.

Treatment and Plot Number	<15 cm dbh Mean	<15 cm dbh SE	16-24 cm dbh Mean	16-24 cm dbh SE	≥ 25 cm dbh Mean	≥ 25 cm dbh SE
Low Density						
LDS1	118.9	2.85	9.0	0.72	9.0	1.58
LDS2	65.6	0.78	8.2	0.44	7.0	0.65
Medium Density						
MDS1	220.4	3.34	39.8	1.84	28.9	0.62
MDS2	138.7	2.90	28.2	1.37	29.3	0.71
High Density						
HDS1	282.4	2.31	74.1	2.49	82.8	1.13
HDS2	130.6	1.95	28.9	1.31	75.5	1.13

The distribution of size classes was similar among treatment areas, although very large snags (>100 cm dbh) occurred only in high-density stands. The mean dbh of trees >15 cm dbh in low-density was 31.56 cm ± 17.68 cm; in medium-density 30.77 cm ± 7.04 cm; in high-density 37.55 cm ± 23.63 cm. Tree height distribution varied between treatments with the tallest trees occurring in stands with the highest densities of snags. The mean height of trees >15 cm dbh in low-density was 16.30 m ± 9.75 m; medium-density 17.58 m ± 7.97 m; high-density 21.28 m ± 8.90 m.

The predominant tree species in the study area were ponderosa pine and Douglas-fir. The frequency of ponderosa pine and Douglas-fir was significantly different among treatments for snags ≥ 25 cm dbh (G = 40.24, DF= 2, $P < 0.001$), and snags between 16 and 24 cm dbh (G = 114.6, DF= 2, $P < 0.001$). The percentage of ponderosa pine snags was lower than the percentage of Douglas-fir snags in all sites for snags >15 cm dbh. High density had 10% ponderosa pine and 89% Douglas-fir, medium density had 35% ponderosa pine and 62% Douglas-fir, and low density had 13% ponderosa pine and 87% Douglas-fir (Table 2).

Stage of snag decay did not vary among treatments since most trees were killed during the fire and case-hardened. Snags in stages 3 or 4 would have been present before the fire; all others were classified as decay stage 2.

Bird Abundance and Species Richness

Thirteen cavity-nesting species were observed on the study plots and species composition

TABLE 2. Relative frequencies of snag species in each study stand, WNF, 1998. Other trees include alder and big leaf maple. PP = ponderosa pine, DF = Douglas-fir.

Treatment and Plot Number	16-24 cm dbh %PP	16-24 cm dbh %DF	16-24 cm dbh %Other	≥ 25 cm dbh %PP	≥ 25 cm dbh %DF	≥ 25 cm dbh %Other
Low Density						
LDS1	10	90	0	12	88	0
LDS2	8	92	0	23	77	0
Medium Density						
MDS1	35	53	12	53	47	0
MDS2	28	72	0	24	76	0
High Density						
HDS1	4	95	1	7	93	0
HDS2	7	90	3	23	77	0

TABLE 3. The mean number of birds/point count station in each of the treatments, Wenatchee National Forest, (1998-1999 averaged). Numbers in bold show significant differences (P = 0.05). P = primary cavity nester, S = secondary cavity nester.

	Low Density		Medium Density		High Density	
Species	LDS1	LDS2	MDS1	MDS2	HDS1	HDS2
White-headed woodpecker (*Picoides albolarvatus*) - P	0.09	0	0	0.17	0	0
Lewis' woodpecker (*Melanerpes lewis*) - P	1.84	1.59	0	1.42	0	0
Black-backed woodpecker (*Picoides arcticus*) - P	0	0	0.25	0.75	1.34	0.17
Hairy woodpecker (*Picoides villosus*) - P	0.75	2.09	2.50	2.50	3.09	3.83
Northern flicker (*Colaptes auratus*) - P	1.75	2.84	4.25	3.92	3.00	2.00
Western bluebird (*Sialia mexicana*) - S	**1.57**	**1.92**	0.75	1.50	0.67	0
Mountain bluebird (*Sialia currucoides*) - S	2.33	1.59	**3.25**	**3.67**	2.25	0.67
Red-breasted nuthatch (*Sitta canadensis*) - S	0.09	0.17	0.34	0.75	0.25	0
House wren (*Troglodytes aedon*) - S	0.92	1.50	2.00	1.92	1.25	0.42
European starling (*Sturnus vulgaris*) - S	0.09	0.92	0	4.17	0	0.17
American kestrel (*Falco sparverius*) - S	0.34	1.09	0.50	1.84	0	0.17
Brown preeper (*Certhia americana*)	0	0	0.09	0	0	1.25
Northern Pygmy owl (*Glaucidium gnoma*)	0	0	0	0	0	0.09

varied with snag density (Table 3). Species such as Lewis' woodpecker and western bluebird were most abundant in stands with low snag density. Northern flicker and mountain bluebird occurred in highest numbers in stands with medium density. Black-backed and hairy woodpecker were most common in stands with high snag density. Species richness of cavity-nesting species was highest in the medium-density treatment.

The mean number of cavity nesters was significantly different among treatments in 1998 (F = 5.08, $P = 0.033$), with the mean number of cavity nesters in medium snag density sites being higher than in the other sites. The number of cavity nesters did not differ significantly among the treatment areas in 1999 (F = 3.75, $P = 0.065$) although the trend was the same, with the highest number of birds in the medium-density treatment (Figure 2). In terms of individual species, the

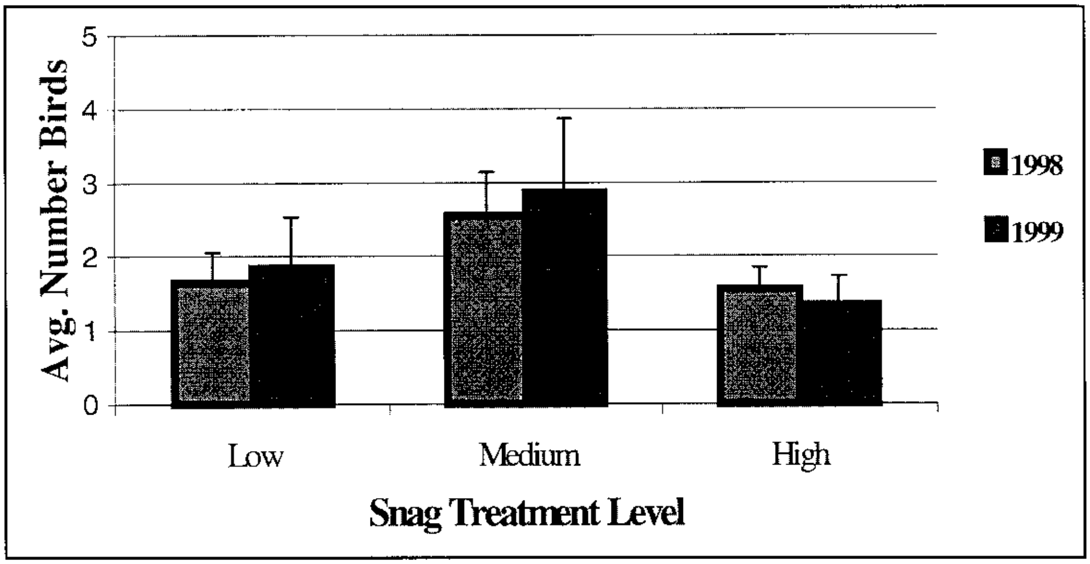

Figure 2. Mean number of cavity nesting birds in the three treatment areas: low, medium, and high snag density.

western bluebird was more abundant in low density treatments in 1998 (F = 4.08, $P = 0.03$) and the mountain bluebird occurred most often in stands with medium snag density in 1999 (F = 6.92, $P < 0.005$). No other significant differences in individual species abundance between treatments were observed.

Nest–Site Characteristics

We found a total of 114 nests. Northern flicker were the most common primary cavity excavators, while mountain bluebird were the most common secondary cavity nesters to nest on all sites. Northern flicker, western bluebird, mountain bluebird and American kestrel nested in all treatment areas, but had the most nests in stands with medium snag density. Hairy woodpecker also nested in all treatment areas, but had fewest nests in the low snag density. Black-backed woodpecker and house wren nested only in treatments with medium and high snag densities. Lewis' woodpecker was the only species to nest most often in low density treatments and did not nest in treatments with the high snag densities. Overall, medium snag density had the highest number of nests with eight species and 56 total nests (Table 4).

Approximately 66% of all nests were in ponderosa pine snags (Table 5). Cavity nesters used ponderosa pine snags for nesting significantly more often than expected (G = 216, DF = 1, $P < 0.001$). Secondary cavity nesters had a higher proportion of nests in Douglas-fir than the primary cavity excavators (woodpecker species).

TABLE 4. Number of active cavity nests of bird species in the treatments, WNF. 1998-1999 combined.

Species	Tree Density		
	Low	Medium	High
Northern flicker	4	12	6
Hairy woodpecker	3	6	6
Black-backed woodpecker	0	1	1
Lewis' woodpecker	10	4	0
Western bluebird	5	7	2
Mountain bluebird	7	12	5
House wren	0	12	7
American kestrel	1	2	1
TOTAL	30	56	28

The mean dbh of nest trees was smaller for secondary cavity nesters than for primary cavity excavators. Northern flicker nested in large trees (mean dbh of 64 cm) and American kestrel nested in small trees (mean dbh of 24 cm).

The height of nest trees and cavities ranged widely for all species. Mean tree height was 20.0 m (range = 2.2-37.5 m) and mean nest cavity height was 8.8 m (range = 1.5-25 m) for all nests. Sixty-nine percent of the nest trees had a broken top. Sample sizes were adequate to compare tree densities surrounding the nest tree with those of non-nest paired plots for the northern flicker, hairy woodpecker, and Lewis' woodpecker. Plots around northern flicker nest trees (N = 17) contained larger trees (mean dbh = 19.5 cm) within 10 m than in the plots surrounding the non-nest trees (mean dbh = 10 cm, $P = 0.011$). Areas within 10 m of northern flicker nest trees had more trees ≥ 25

TABLE 5. Characteristics of snags used as nest sites by seven species of cavity-nesting birds in ponderosa pine/Douglas-fir forests, WNF. Other (O) nest trees include cottonwood and alder.

Species	# Snags[1]	Tree species (no.)			Snag dbh (cm)		Broken top	Tree height (m)		Cavity height (m)	
		PP	DF	O	Mean	Range	(%)	Mean	Range	Mean	Range
Northern flicker	17	14	3		64	38-117	65	19.3	2.4-37.2	9.2	1.8-23.9
Hairy woodpecker	13	11	1	1	56	28-86	62	19.8	7.3-37.5	5.9	2.0-13.7
Lewis' woodpecker	10	6	3	1	58	38-99	80	18.3	8.2-27.4	9.3	7.6-20.4
Black-backed woodpecker	2	2			50	48-51	50	21.3	19.2-23.5	3.0	2.3-3.7
Mountain bluebird	20	12	8		36	15-71	90	14.5	2.1-35.7	7.0	1.5-11.6
Western bluebird	11	5	6		42	18-74	64	19.1	4.6-33.5	10.2	1.5-15.2
House wren	18	8	9	1	42	18-93	67	18.9	4.3-35.4	8.5	1.5-25.0
American kestrel	4	2	1	1	24	18-30	75	29.0	23.0-36.0	17.5	5.2-23.9
Total or Grand Mean	95	60	31	4	46.5	15-117	69	20.0	2.1-37.5	8.8	1.5-25.0

[1] Numbers may not agree with those in Table 4 because some nest trees were used more than once.

cm dbh (mean = 2.5) than in non-nest plots (mean = 0.6, $P = 0.008$). The number of trees 16-24 cm dbh at a distance of 10-25 m from hairy woodpecker nest trees (mean = 52.4) was greater than for paired trees (mean = 28.2, $P = 0.034$). No significant differences between nest trees and paired non-nest trees were found for the number of trees < 25 cm within 10 m, the mean dbh of surrounding trees, the number of trees > 25 cm dbh 10-25 m from the tree, or the percentage of ponderosa pine or Douglas-fir within a 25 m radii.

We examined nests of woodpecker species and found no significant difference between the density of trees in the same orientation as the cavity opening and the density of trees elsewhere around the nest tree for northern flicker, hairy and Lewis' woodpecker. Hairy woodpecker nest trees had significantly more snags from 10-25 m in the same orientation as the cavity than the paired tree ($t = 2.99$, DF= 12, $P = 0.005$). There were no significant differences between the nest tree and the paired tree for Lewis' woodpecker ($t = 1.13$, DF= 9, $P = 0.145$) or northern flicker ($t = 0$, DF= 16, $P = 0.5$).

Mountain bluebird nested in cavities previously occupied by hairy woodpecker (4), northern flicker (4), and black-backed woodpecker (1). In three known cases, mountain bluebird nested in the same cavity in both 1998 and 1999. Western bluebird nested in cavities previously occupied by hairy woodpecker (5), northern flicker (1), mountain bluebird (3), and house wren (1). In only one instance did western bluebird occupy the same nest in both 1998 and 1999. House wren nested in hairy woodpecker (7) and black-backed woodpecker cavities (2). Two house wren pairs occupied the same cavities in both 1998 and 1999. American kestrel nested in old northern flicker cavities (2). In two instances, a cavity had three occupants in the two years. The occupants of one cavity changed from black-backed woodpecker to mountain bluebird to house wren. The occupants of the other cavity changed from hairy woodpecker to western bluebird to house wren.

Discussion

Species composition varied among treatments (Tables 3,4). Specifically, Lewis' woodpeckers occurred primarily in stands with low snag density, but also occurred in stands with a medium snag density. Black-backed woodpecker and brown creeper occurred in both high and medium snag densities, but neither was present in the low density. Mountain bluebird, northern flicker, and house wren occurred throughout, but were in greatest abundance in the stands with medium snag densities. Saab and Dudley (1998) studied stand-replacement fire and salvage logging and also reported changing species composition in stands with varying snag densities, however, overall densities of cavity-nesting birds were similar.

The density of cavity-nesting birds was not positively associated with the number of snags ≥ 25 cm dbh in this system. Evidence from other studies suggests that cavity-nesting birds select for more than the snag tree itself and avian assemblages may change in relation to the structure of the stand of snags (Raphael and White 1984, Shackelford and Conner 1997). For example, hairy woodpecker were closely associated with the presence of large-diameter snags and logs, while northern flicker were associated with increasing numbers of small-diameter snags (Shackelford and Conner 1997). Both small snags (<15 cm dbh) and large snags are important for foraging (Horton and Mannan 1988). Extensive foraging on small snags occurred in all treatment areas in this study.

The spatial structure of snags can also influence bird communities. Bird species composition may be determined by the degree of openness of a habitat. Logging can cause changes in bird species composition (Hagar 1960) because the openness presents new opportunities for aerial foragers (Franzreb and Ohmart 1978). Fire-altered habitat also leads to an increase in species that forage in low brush or open ground (Bock and Lynch 1970).

The size and distribution of snags in medium density treatments were conducive to a higher abundance of cavity-nesting species (Table 4). Medium density also had a greater number of ponderosa pine snags than the other treatments. The high proportion of ponderosa pine compared to Douglas-fir may have influenced the presence of species that prefer ponderosa pine for both nesting and foraging. However, the occurrence of unused large ponderosa pine snags in both low and high density treatments suggests that the presence of ponderosa pine snags alone may not result in an increased abundance of cavity-nesting birds.

The number of nests was highest in the medium snag density treatment (Table 4). We speculate that medium snag densities provided habitat for species that prefer open nesting habitat and those that prefer to nest in higher tree densities. For example, Lewis' woodpecker and American kestrel nest in open or semi-open sites; hairy woodpecker, black-backed woodpecker, and house wren nest in more dense tree stands (McClelland et al. 1979, Saab and Dudley 1998). Yet, all species were observed in the medium snag density treatments. Ponderosa pine was the preferred tree species for primary cavity nesters. Nests in Douglas-fir were most often occupied by secondary nesters and many Douglas-fir nest snags had old cavities or were dead before the fire.

In burned forests, large snags are often used for nesting (Raphael and White 1984, Hutto 1995, Saab and Dudley 1998) and cavity nesters chose large nest snags in this study as well (Table 5). Primary cavity-excavator species used larger-diameter nest trees than secondary cavity nesters. The difference might be because secondary cavity nesters typically nested in Douglas-fir snags, which had a smaller mean dbh than the ponderosa pine snags. The high number of small Douglas-fir trees in this study was due to several decades of fire exclusion (Harrod et al. 1999).

In this study, as in several others (McClelland et al. 1979, Mannan et al. 1980, Zarnowitz and Manuwal 1985, Welsh and Capen 1992, Bevis 1994, Hutto 1995, Saab and Dudley 1998), most nests were in broken top snags (Table 5). A broken top provides an avenue for heart-rotting fungi, which makes the snag more suitable for cavity excavation (McClelland et al. 1979). Broken top trees are especially important in burns because they provide nest sites for the first few years following a high-intensity fire when other trees are not easily excavated due to case-hardening (Saab and Dudley 1998).

Factors other than the suitability of the tree itself may play important roles in selection of the nest site (Welsh and Capen 1992, Vierling 1997). Forest stand characteristics are sometimes a better predictor of bird use than individual snag characteristics (Swallow et al. 1986). Saab and Dudley (1997) showed that the density of trees surrounding the nest tree of cavity-nesting birds was higher than the density of trees at random sites. In contrast, Lewis' woodpeckers avoid nesting in dense tree stands (Vierling 1997). In this study northern flickers chose nest trees with at least two large-diameter snags (≥ 25 cm) within 10 m, and hairy woodpeckers chose nest trees surrounded by a high density of small snags.

We found a number of unoccupied cavities suggesting that the availability of suitable nest sites may not be the only factor limiting populations of cavity-nesting birds (Ingold and Ingold 1984, Peterson and Gauthier 1985, Rendell and Robertson 1989). Cavities can remain unused if they fall within the territory of another individual or breeding pair that defends more than one cavity for roost sites (Peterson and Gauthier 1985, Rendell and Robertson 1989). Unused cavities may also be the result of a surplus of cavities rather than the result of interspecific or intraspecific competition (Ingold and Ingold 1984). In addition, the cavity entrance size or volume may make it unsuitable for some secondary-nesting species (Rendell and Robertson 1989). Even though there were unoccupied cavities, some nest cavities changed occupants three times during two years of observation. This pattern may be the result of temporal breeding differences between species and not the result of interspecific competition for nest sites (Ingold and Ingold 1984). For example, mountain bluebirds commonly used old hairy woodpecker cavities after hairy woodpecker young were fledged. It would be worthwhile to investigate this trend further by measuring specific cavity parameters and habitat variables between used and unused sites.

Management Implications

The results of this study have management implications for retaining snag habitat during salvage logging in dry forests on the east side of the Cascades following stand-replacement fire. Snag densities of 15-35 snags ≥ 25 cm dbh per hectare provided the highest abundance, species richness, and nesting populations of cavity nesters. Snags > 48 cm dbh provided nesting habitat for more species. An average of 21 snags > 48 cm dbh per hectare yielded highest nesting populations, supported multiple cavities, and were important for foraging. Smaller snags provided foraging and nesting habitat for some species. The treatment with highest bird abundance had 34 snags/ha in 16-24 cm dbh size class and 180 snags/ha in the < 15 cm dbh size class. Treatments with snags distributed in clumps and individually dispersed had the highest abundance and species richness of cavity nesting species.

This study also poses new questions. The highest bird abundance occurred in medium density stands, which were not only different in snag density, but also had the highest percentage of ponderosa pine, a favored nest tree. Future studies designed to look at the relationship between the availability of ponderosa pine and the breeding bird population would help to separate snag density effects and tree species effects. In addition, long term monitoring is necessary to understand the dynamics between snag deterioration rates and changes in bird assemblages following stand-replacement fires and salvage logging.

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Literature Cited

Bevis, K. R. 1994. Primary cavity excavators in grand fir forests of Washington's East Cascades. M.S. Thesis. Central Washington University, Ellensburg, Washington.

Blake, J. G. 1982. Influence of fire and logging on nonbreeding bird communities of Ponderosa pine forests. Journal of Wildlife Management 46:405-415.

Bock, C. E., and J. F. Lynch. 1970. Breeding bird populations of burned and unburned conifer forest in the Sierra Nevada. Condor. 72:182-189.

Bull, E. L., R. S. Holthausen and D. B. Marx. 1990. How to determine snag density. Western Journal of Applied Forestry 5:56-58.

Bull, E. L., C. G. Parks, and T. R. Torgerson. 1997. Trees and logs important to wildlife in the interior Columbia River basin. USDA Forest Service General Technical Report PNW-GTR-391, Pacific Northwest Research Station, Portland, Oregon.

Cline, S. P., A. B. Berg and H. M. Wight. 1977. Snag characteristics and dynamics in Douglas-fir forests, western Oregon. Journal of Wildlife Management 44:773-786.

Franzreb, K. E., and R. D. Ohmart. 1978. The effects of timber harvesting on breeding birds in a mixed-coniferous forest. Condor 80:431-441.

Hagar, D. C. 1960. The interrelationships of logging, birds, and timber regeneration in the Douglas-fir region of northwestern California. Ecology 41:116-125.

Harrod, R. J., B. H. McRae, and W. E. Hartl. 1999. Historical stand reconstruction in ponderosa pine forests to guide silvicultural prescriptions. Forest Ecology and Management 114:433-446.

Horton, S. P., and R. W. Mannan. 1988. Effects of prescribed fire on snags and cavity-nesting birds in southeastern Arizona pine forests. Wildlife Society Bulletin 16:37-44.

Hutto, R. L. 1995. Composition of bird communities following stand-replacement fires in northern Rocky Mountain (U.S.A.) conifer forests. Conservation Biology 9:1041-1057.

Hutto, R. L., S. M. Pletschet, and P. Hendricks. 1986. A fixed-radius point count method for nonbreeding and breeding season use. The Auk 103:593-602.

Ingold, D. J., and D. A. Ingold. 1984. A study of possible niche preferences of cavity-nesting birds in the Colorado Rockies. New Mexico Ornithological Society Bulletin 12:1-9.

Lillybridge, T.R., B.L. Kovalchik, C.K. Williams, and B.G. Smith. 1995. Field guide for forested plant associations of the Wentachee National Forest. USDA Forest Service, General Technical Report PNW-GTR- 359, Pacific Northwest Research Station, Portland, Oregon.

Mannan, R. W., E. C. Meslow, and H. M. Wight. 1980. Use of snags by birds in Douglas-fir forests, western Oregon. Journal of Wildlife Management 44:787-797.

Manuwal, D. A., and A. B. Carey. 1991. Methods for measuring populations of small diurnal forest birds. USDA Forest Service, General Technical Report PNW-GTR-275, Pacific Northwest Research Station, Portland, Oregon.

McClelland, B. R., S. S. Frissell, W. C. Fischer, and C. H. Halvorson. 1979. Habitat management for hole-nesting birds in forests of western larch and Douglas-fir. Journal of Forestry. 77:480-483.

Medin, D.E. 1985. Breeding bird responses to diameter-cut logging in west-central Idaho. USDA Forest Service Research Paper INT-355, Intermountain Research Station, Ogden, Utah.

Morrison, M. L., and M. G. Raphael. 1993. Modeling the dynamics of snags. Ecological Applications 3:322-330.

Peterson, B., and G. Gauthier. 1985. Nest site use by cavity-nesting birds of the Cariboo Parkland, British Columbia. Wilson Bulletin 97:319-331.

Ralph, C. J., G. R. Geupal, P. Pyle, T. E. Martin, and D. F. DeSante. 1993. Handbook of field methods for monitoring landbirds. USDA Forest Service, General Technical Report PSW-GTR-144, Pacific Southwest Station, Berkley, California.

Raphael, M. G., and M. White. 1984. Use of snags by cavity nesting birds in the Sierra Nevada. Wildlife Monographs 86:1-66.

Rendell, W. B., and R. J. Robertson. 1989. Nest-site characteristics, reproductive success and cavity availability for tree swallows breeding in natural cavities. Condor 91:875-885.

Saab, V. A., and J. G. Dudley. 1998. Responses of cavity-nesting birds to stand-replacement fire and salvage logging in ponderosa pine/Douglas-fir forests of southwestern Idaho. USDA Forest Service, Research Paper RMRS-RP-11, Rocky Mountain Research Station, Denver, Colorado.

Schroeder, R. L. 1983. Habitat suitability index models: downy woodpecker. USDI Fish and Wildlife Service, FWS/OBS- 82/10.38.

Scott, V. E. 1978. Characteristics of ponderosa pine snags used by cavity-nesting birds in Arizona. Journal of Forestry 77:26-28.

Shackelford, C. E., and R. N. Conner. 1997. Woodpecker abundance and habitat use in three forest types in eastern Texas. Wilson Bulletin 109:614-629.

Strangel, P. 1994. Woodpeckers and nesting cavities. Wildbird 8:28-31.

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Swallow, S. K., R. J. Gutierrez, and R. A. Howard, Jr. 1986. Primary cavity-site selection by birds. Journal of Wildlife Management 50:576-583.

Vierling, K. T. 1997. Habitat selection of the Lewis' woodpeckers in southeastern Colorado. Wilson Bulletin 109:121-130.

Welsh, C. J. E., and D. E. Capen. 1992. Availability of nesting sites as a limit to woodpecker populations. Forest Ecology and Management 48:31-41.

Zar, J. H. 1996. Biostatistical Analysis. Prentice-Hall, Englewood Cliffs, New Jersey.

Zarnowitz, J. E. and D. A. Manuwal. 1985. The effects of forest management on cavity nesting birds in northwestern Washington. Journal of Wildlife Management 49:225-263.

Chapter 11

Conservation Status of Boreal Owls in the United States

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INTRODUCTION

Previous chapters outlined the biology and ecology of boreal owls as well as the ecology of important vegetation communities based on literature from North America and Europe. That technical review provides the basis to assess the current conservation status of boreal owls in the United States. By conservation status, we mean the demographic condition of the species as it relates to the likelihood of local and national persistence of wild populations over the long term. Are populations of boreal owls in the United States currently threatened? Are current land management practices likely to lead toward the peril of local or regional populations?

Like any scientific story, our understanding of boreal owl ecology is incomplete. In the face of incomplete knowledge, I will evaluate the status of boreal owls by asking a series of critical questions about the species and its habitat. My goal is to synthesize evidence necessary to build a case for one of the following conclusions: 1) populations of boreal owls in the United States are secure and will likely remain so given current land management practices; 2) populations of boreal owls are in peril (declining or experiencing some demographic trauma) or are likely to be in peril in the future given current land management practices; or 3) there is insufficient evidence to determine the species' conservation status.

Populations of boreal owls differ in biology and ecology depending on geographic setting (Korpimaki 1986, Hayward *et al.* 1993). Therefore, for this assessment, when answering the critical questions, I rely first on investigations from North America and use European studies to a lesser extent. A minimum of references are presented here as the literature was thoroughly reviewed in the previous chapters.

Is the Distribution and Abundance of the Boreal Owl Declining in All or Part of Its Range?

Distribution

The boreal owl is broadly distributed in North America, and its distribution likely has remained the same over the past few decades. The extensive geographic range of the species contributes toward species persistence.

During the past 15 years, numerous published reports have extended the recognized range of boreal owls in western North America. In 1980, the southern extent of the species' breeding range was thought to end in Canada. Today, evidence exists for breeding populations throughout the Rocky Mountains south to southwestern Colorado and northern New Mexico. Breeding boreal owls have also been documented in northern Minnesota. Do these records indicate an extension of the species' range?

The weight of evidence suggests that the actual distribution of boreal owls has not changed recently; rather, our knowledge of the species has changed radically. Several indirect lines of evidence support the contention that the extension of the species' recognized range stems from an increase in survey effort. First, historical records indicate that boreal owls were recorded in the western United States but not recognized as breeding. A close look at the literature indicates that boreal owls were documented as far south as Colorado for nearly 100 years (see Ryder *et al.* 1987). Historical records of boreal owls in Wyoming, Idaho, and Colorado were thought to represent nonbreeding "visitors." Despite the occurrence of boreal owls in the western United States, checklists and field guides did not list the species, even after breeding populations were documented in 1983.

Second, human use of boreal owl habitat has increased recently, raising the probability of documenting existing breeding populations. Winter recreation in high mountain lands has increased since the 1970's, bringing more people into boreal owl habitats during the owls' most vocal period. Coincident with increased interest in winter sports (cross country and downhill skiing) has been an increase in roads in high mountain areas. Furthermore, biologists working with land management agencies have conducted surveys directed toward finding boreal owls. An increase in roads accessing high elevation forests and interest in the owl have facilitated location of breeding owls. In 1984 alone, during the first extensive surveys in the Northern Rocky Mountains, agency personnel found boreal owls on nine western national forests where the owl was not recognized previously.

Third, biologists in Europe have also located new populations of boreal owls and attributed these to increased interest in the species. Cramp (1977) describes extensions of the recognized range in Europe but does not believe the species has actually broadened its distribution.

Abundance

Local and regional trends in boreal owl abundance cannot be assessed with available data. Breeding populations of boreal owls were only recently documented throughout most of the species' range in the United States. In most cases, estimates of density or an index to abundance have not been made, precluding any assessments of trend in the near future. I am aware of only two populations (one in Idaho and one in Montana) that have been sampled using methods that will facilitate assessment of trend within the next 5 years (see Hayward *et al.* 1992).

Do Habitats Vary in Their Capacity to Support Boreal Owl Populations or to Support Particular Activities of the Owl? What Are the Important Characteristics of Those Habitats?

Study of boreal owl habitat use is limited. Investigators in Europe who have studied boreal owls for 2 decades have not focused on habitat use. In North America, only three studies have intensively examined habitat use. Despite this limited knowledge, the evidence supports the contention that boreal owls favor particular habitat characteristics at a variety of geographic scales. Consistently occupied habitat generally is mature or old spruce or spruce-fir forest.

The combined results of three, multiyear studies of boreal owls in North America indicate that boreal owls choose sites for nesting, roosting, and foraging nonrandomly (Bondrup-Nielsen 1978, Palmer 1986, and Hayward *et al.* 1993). Knowledge of habitat use in North America stems largely from these studies. Studies from Europe corroborate the conclusion that boreal owls choose specific habitats at a variety of spatial scales. In general, habitat studies were observational, rather than experimental, and suffered from small sample sizes. Furthermore, none of the North American studies compared the relative fitness or productivity of individuals using various habitats. Despite these shortcomings (which are the norm in vertebrate ecology) these investigations were sound mensurative studies that showed boreal owls use habitats differentially for important life functions.

Regional

At the regional scale, knowledge of boreal owl distribution indicates particular habitat associations. Boreal owls occur only in subalpine forest habitats in the western United States (e.g., Hayward *et al.* 1993). Breeding populations have not been found more than 100 m below the spruce-fir zone in the Rocky Mountains. East of the Rocky Mountains, boreal owls do not occur south of boreal and transition boreal-temperate forests. These distributional boundaries suggest strong physiological, behavioral, or ecological barriers limiting the boreal owl. What characteristics of these forests are important in determining the broad distribution pattern of boreal owls is unknown but reasonable hypotheses were outlined in Chapters 9 and 10.

Landscape

At the landscape and home range scales, limited evidence indicates boreal owls use sites with particular forest characteristics (e.g.,Sonerud 1986, Korpimaki 1988, Hayward *et al.* 1993). In both Europe and North America, quality foraging habitat is characterized as mature and older spruce or spruce-fir forest. During prey population declines in Finland, owl home ranges with a high proportion of spruce forest are consistently occupied while other ranges are only used during prey peaks. The regularly occupied home ranges also produce more fledglings than other sites, indicating a match between preferred habitat and productivity (Korpimaki 1988). In Idaho, nest sites of radio-marked owls occurred in the lowest elevation portion of home ranges (edge of elliptical home ranges) indicating the spatial segregation of habitats used for nesting vs. foraging and roosting (Hayward *et al.* 1993). Nest sites occurred in old aspen and old, mixed-conifer stands while roost and foraging sites were often in mature and older spruce-fir forest.

Microhabitat

At the microhabitat scale, boreal owls appear to use a nonrandom subset of sites for nesting, roosting, and foraging. The strength of evidence for habitat choice varies among investigations, each of which was conducted in a very different geographic setting. In some studies, selection was not demonstrated, while in others, used habitat was simply described.

Boreal owls are obligate cavity nesters. One study suggests that boreal owls select among available nest sites when a range of sites is available. In an experimental study in Idaho, suitable nest sites in lodgepole pine forest were not used when alternates were available in the old mixed-confer forest (Hayward *et al.* 1993). In the same study, an analysis comparing 28 nest sites and 101 random sites indicated the owls used forests with multiple canopy layers, large diameter trees, and high basal area.

Although boreal owls have been shown to choose particular forest habitats for nesting, the species will accept a broad range of nest sites. Simple descriptive studies demonstrate this range. Nests have been found almost exclusively in aspen in Canada and Minnesota, in spruce and lodgepole in Colorado, and lodgepole and spruce-fir forest in Montana. Nest boxes placed in clearcuts in Idaho and Sweden have been used. In these cases, spruce or spruce-fir forest occurred nearby.

The importance of specific roosting habitat seems to vary depending upon the threat of predation and degree of thermal stress. In Canada, owls did not select particular sites for roosting. In Idaho, boreal owls exhibited symptoms of summer heat stress and were shown to choose cool microsites for roosting. Mature and old spruce-fir forest was chosen for summer roosts. These stands had higher basal areas, higher crown closures, and higher tree densities than random sites. During winter, these same owls were less selective in roost choice.

Microhabitat characteristics of boreal owl foraging habitat have not been studied. Therefore, despite evidence for foraging habitat choice at broader scales, the microhabitat characteristics of quality foraging habitat have not been identified.

Do Habitats Vary in Their Capacity to Support Principal Prey Species?

Primary prey of boreal owls in North America include red-backed voles (*Clethrionomys* spp.), field voles (*Microtus* spp.), deer mice, shrews, flying squirrels (*Glaucomys sabrinus*), and pocket gophers (*Thomomys talpoides*). Most important among these are red-backed voles and field voles, both of which occur in specific habitats. In the western United States and Canada, red-backed voles are most abundant in old spruce-fir forests and rarely occur in unforested habitats. Major foods of red-backed voles in the western United States are scarce in young forest stands. In contrast, red-backed voles in the eastern United States occur in a variety of forest age classes but (similar to western areas) are most common in mesic forest. Field voles rarely occur in forest stands and are most abundant in mesic meadows. Forest management practices significantly influence the abundance of these and other small mammal prey species (Campbell and Clark 1980, Ramirez and Hornocker 1981, Halvorson 1982, Scrivner and Smith 1984). Although the outcome of particular management practices is poorly understood, stand replacement treatments (e.g., clear cut harvests) lead to the most dramatic changes.

If the Boreal Owl or Its Prey Select Particular Habitats, Are These Habitats Declining or Being Stressed by Current Management?

Studies from a few geographic areas indicate boreal owls and their prey demonstrate selection for particular habitats. The paucity of research on this owl and its prey makes the geographic extent of this pattern unclear. Furthermore, the characteristics of high quality habitat for the owl's prey are not known sufficiently to set management guidelines.

The available evidence (see Chapter 9 concerning nesting, roosting, and foraging habitat) does suggest that mature and older forest in the spruce-fir zone provides the highest quality habitat for boreal owls and their prey. These forests occur as the upper forested zone on mountains in the western United States. As such, if global climate change shifts lifezones upward in elevation (as is predicted to happen), these habitats will decline (see Chapter 10).

Climate change portends consequences beyond a potential future change in the elevation of life zones. The mature and older forests used by boreal owls today became established centuries ago, under different climatic circumstances. As pointed out by Knight (Chapter 10), "Conceivably, climatic conditions now are less favorable for seedling establishment than they were two or three centuries ago, when the harvested forest became established. At high elevations it may not be possible to count on clearcut or burned forests eventually growing back to the kind of old forests that boreal owls currently use." If timber harvest and other land management practices are accelerating the rate of stand replacement and changing the distribution of forest age classes, the abundance of old forest stands may be declining faster than under a natural disturbance regime dominated by gap forming disturbance such as disease and blowdown.

Compared to recent historic times, old spruce-fir forests are likely less abundant (Chapter 10). While fire suppression has promoted an increase in older successional stages, timber harvest, using even-aged methods, has reduced the area of old forest. A long-term consequence of fire suppression, however, is fuel build-up that may lead to larger, more intensive fires, ultimately reducing the area of old forest. Overall, in the western United States, where most is known of boreal owl biology, the area of high quality habitat is likely declining and will continue to decline as forest management is carried out as currently outlined in forest plans. A review of 14 National Forest plans from Regions 1 and 4 indicated a reluctance to initiate uneven-aged management in many spruce-fir stands (Hayward *et al.* 1993). Our interpretation of 14 plans in 1989 indicated even-aged management would dominate on all but one forest.

The quality of habitats used by the owls and primary prey is likely declining as well as the area. Alexander (1987) indicated that spruce now leads all species except ponderosa pine in annual volume cut in the central and southern Rocky Mountains. Current knowledge is not sufficient to quantify the rate or extent of habitat decline. Patterns of subalpine forest dynamics described in Chapter 10 indicate some potential consequences of timber harvest dominated by large clearcuts. While insects and wind were the most frequent disturbance agents in subalpine forests prior to European settlement, the effects of clearcuts are similar to fire, which was a less common disturbance agent. Tree mortality due to insects and wind lead to gap processes that support the boreal owl food web. Natural disturbance patterns also resulted in a more heterogeneous forest than occurs with prolonged fire suppression and clearcut harvesting. The mosaic forest would support a variety of small mammal species and abundant red-backed voles (Chapter 9). The loss of large snags and large downed logs associated with standard forest practices likely lowers habitat quality for the owl and its prey. Forest practices that reduce arboreal lichen, particularly *Bryoria* spp., also likely reduce habitat quality.

Do the Life History and Ecology of the Boreal Owl Suggest That Populations Are Vulnerable to Habitat Change?

Cavity Nesting

Boreal owls require large tree cavities or artificial nest structures to breed. This is the most obvious habitat requirement of the species and one that has important consequences. Unless artificial structures are provided, boreal owls will not persist in landscapes where trees are too small to produce the large cavities required by the owl <u>or</u> where primary cavity excavators are missing. Natural tree cavities (produced by branch loss or other breakage) are used occasionally by boreal owls but unlikely to be common enough to support a population of owls. Rotations of 70-120 years will not produce the size class of trees necessary for natural nest sites.

Information is not available indicating in what geographic areas boreal owls may be cavity limited. Owl populations in regions south of the breeding range for pileated woodpeckers (*Dryocopus pileatus*) are more likely candidates for cavity limitation.

Changes in cavity availability have likely occurred during the past century due to forest management. The extent of these changes and their consequences have not been documented. Timber harvest prescriptions that removed all trees, or all large trees and snags, have eliminated existing cavities and precluded new cavities on the site for up to two centuries depending on tree growth. Harvest rotations that prevent the development of snags >38 cm dbh permanently preclude nesting from the site. In contrast to the consequences of timber management, in the short term fire suppression has likely increased the availability of large cavities by reducing the loss of old forest through fire. The long-term consequence may be different, however, if fire suppression leads to larger, higher intensity fires that burn stands on mesic and moist microsites that were less likely to burn under the natural fire regime.

Changes in forest conditions that lead to reductions in large diameter snags or large live trees with heart-rot will lead to cavity limitation. Similar consequences will occur with changes that reduce habitat quality for primary cavity nesters.

Productivity

Boreal owls in the western United States exhibit variable year-to-year productivity and appear to have relatively low average clutch sizes. These factors have been associated with decreased probabilities of population persistence (Goodman 1987, Pimm, *et al.* 1988). Variable productivity in boreal owls stems largely from year-to-year variation in available prey. In Europe, extreme variation in the number of breeding pairs and clutch size have been documented. In North America few investigators have documented productivity over multiple years but variation due to changing prey populations has been reported. Variation in winter and spring weather may also lead to variation in productivity. Small, isolated populations of owls would be most susceptible to a series of years with extremely low reproduction.

The average and maximum productivity of boreal owls recorded in the western United States are much lower than records from Europe. This suggests that populations studied in the United States may produce fewer surplus individuals even in good breeding years. The ability of source populations to supplement less productive populations therefore may be less than in Europe. Our understanding of the comparative demography of boreal owls is not sufficient to assess the influence of productivity on the relative stability of various populations.

The degree to which productivity is density dependent, especially at low population densities, is important in assessing boreal owl demography. We do not know to what extent productivity is density dependent or whether boreal owls are likely to experience an Allee affect (Allee 1931) at modest population sizes. A strong Allee affect could result if boreal owls experience difficulty in locating mates at low population densities.

Survival

Limited information on boreal owl annual survival gives an unclear picture of the impact this life history parameter may have on population growth. The few estimates of adult and juvenile survival have potential for bias and are imprecise. Estimates range from about 45 to 80% adult survival and 20-50% for juveniles. Both year-to-year variability and average survival rates are important in assessing the status of boreal owl demography. Furthermore, the degree of inverse density dependence in this parameter is important. Neither is known for any population. In populations where average survival is high and not variable, concern over low relative productivity is reduced. Low survival rates, however, would lead to greater concern over the relatively low clutch sizes recorded in the western United States.

The environment occupied by boreal owls is variable and harsh. Therefore, the probability for catastrophic events leading to increased mortality may be high. The nomadic nature of boreal owls is a testament to this variability (Andersson 1980). During periods of environmental stress, boreal owls move to new locations. These movements could contribute to periodic extinction within local habitats. It is unknown, however, whether nomadism increases or decreases persistence among linked small populations.

Home Range Size

Home range sizes of boreal owls in the western United States are large; winter and summer ranges both average over 1,000 ha and home ranges as large as 3,390 ha have been estimated. These areas are large for a medium size predator. Boreal owl home ranges are comparable to those used by the much larger spotted owl (*Strix occidentalis*). Whether large ranges are the norm for the species in the United States is unknown.

Several factors likely contribute to large boreal owl home ranges. In some regions, no single vegetation type provides optimum nesting, roosting, and foraging habitat, and these vegetation types are geographically disjunct. Therefore, geographic features may lead to a broad dispersion of resources, forcing the owls to move long distances to fulfill life requirements. In addition, low productivity of small mammals may also contribute to large owl ranges. Lindstedt *et al.* (1986) showed that home range size among carnivores is related to prey production.

In any case, large home ranges lead to high energy expenditure during daily movements. Boreal owls in some areas appear to need large areas to meet seasonal needs. Hirons (1985) has shown that, at least for the tawny owl (*Strix aluco*), clutch size is limited by energy available to the female prior to laying. The large home ranges and low clutch sizes observed in boreal owls in the western United States seem to fit this pattern.

The use of large home ranges by boreal owls is a conservation concern for two reasons. Populations of individuals requiring large ranges may be energetically stressed and less resilient to further stress. Also, land management must provide habitat within large areas to meet individual as well as population needs.

Trophic Position

Boreal owls are likely the most important avian predator of small mammals in subalpine forests in the western United States As such, they rely on the integrity of 2-3 trophic levels. As described in Chapter 9, the boreal owl's food web in subalpine forests is linked strongly to the detritus system and involves many direct and indirect linkages among trees, insects, pathogens, fungi, and vertebrates. This system appears to support larger prey biomass in older forests (Hayward *et al.* 1993). The food web is poorly understood but the boreal owl certainly occupies a top trophic position. The probability for persistence of species at higher trophic levels is thought to be less than for primary producers or primary consumers.

Metapopulation Structure

Boreal owls in western North America occur in relatively small, semi-isolated populations (see figure 1 in Chapter 9) and therefore, individual populations are vulnerable to extinction due to demographic and environmental stochasticity (Pimm *et al.* 1988). Therefore, the natural distribution pattern of the species south of the boreal forest places individual populations at risk due to their relative small size. Why is this the case?

Boreal owls are the only Strigiform in the western United States that occurs almost exclusively in subalpine forest. Because these forests occur only in high mountain areas, populations exist in patches limited by the extent of subalpine forest, separated from other patches by montane forest and nonforested habitats (see figure 1, Chapter 9 for an example). For example, within USDA Forest Service Region 1, ~9% of the forested land supports spruce-fir forest 15 cm dbh or larger (J. W. Laux, pers. comm.). On seven forests in Idaho south of the Salmon River, spruce-fir forest covers ~7% of the forested landscape (H. A. Cheatham, pers. commun.). These figures demonstrate the limited extent of boreal owl populations despite their broad geographic range. Demographic linkage among patches likely depends on long-distance juvenile dispersal and adult emigration. The nomadic nature of boreal owls should facilitate this linkage. The degree of connectivity, characteristics of the demographic relationships, and processes that control the connectivity, however, are not known.

Small isolated populations of any organism are expected to experience lower persistence probabilities than larger or more linked populations (MacArthur and Wilson 1967, Pimm *et al.* 1988). This is hypothesized to occur for several reasons. Small populations may drift to extinction due to random demographic events (demographic stochasticity; e.g., Shaffer 1981). Similarly, an environmental catastrophe affecting a relatively small area (e.g., stand replacement fire) is more likely to influence a large proportion of individuals in a small, rather than large, population. Isolated populations are also less likely to experience demographic rescue than connected groups (Brown and Kodric-Brown 1977). Therefore, compared to other owl species, individual boreal owl populations may have lower persistence probabilities due specifically to natural distribution patterns.

Any environmental change that reduces the average size of habitat islands occupied by boreal owls is likely to decrease the probability of population persistence in the larger boreal owl metapopulation. An increase in fire frequency in subalpine forests, or reduction in forest area through timber harvest, could lead to reduced habitat area. Our understanding of owl-habitat and prey-habitat relationships is not sufficient to adequately predict what range of habitat alterations (e.g., silvicultural prescriptions) lead to reduced habitat area; however, short rotation, even-age management will clearly be detrimental. We also do not understand how large boreal owl populations must be before stochastic events become less of a concern.

Similarly, environmental change that reduces the linkage among populations is likely to decrease the probability of population persistence in the larger boreal owl metapopulation (due to demographic and genetic problems). Habitat conditions in the matrix surrounding occupied owl habitat will influence the probability of successful dispersal among populations. Furthermore, the productivity of individual populations will influence the number of individuals dispersing to other groups. What factors control successful dispersal and how those factors interact, however, is unknown.

WHAT IS THE CURRENT AND PROJECTED CONSERVATION STATUS OF THE BOREAL OWL?

The current conservation status of boreal owls is unknown. Knowledge of the species in North America is far from sufficient to adequately assess the species' status. Fifteen years ago, the owl was not recognized as a breeding resident of the lower 48 states. To date, only three studies have examined the species' habitat and/or demography; each of these were small, short-term investigations. Based on existing information, however, I tentatively conclude that except in local situations, boreal owls are currently secure but are likely to be in peril in the future given current land management practices. Therefore, I suggest that a conservation strategy is needed for boreal owls. These conclusions are based on the following points.

- Boreal owls currently are well distributed across a large geographic range and therefore the species is not in any immediate peril in the United States or worldwide. Furthermore, based on the only two populations where nest occupancy and productivity is being monitored in the United States, nest occupancy and productivity are remaining constant.

- Populations of boreal owls in Fennoscandia have persisted and appear to be demographically vigorous despite human disturbance and a long history of forest management. These populations rely on artificial nest structures and exist in a very different ecological setting (boreal forest) than most populations in the United States. Still, the persistence of these populations suggests that, with proper forest management, boreal owls can co-occur with resource development including timber harvest. This statement must not be taken to indicate that all populations of boreal owls will respond similarly to environmental change. Boreal owls use habitat differently and have different demographic characteristics, throughout their range. Therefore, the response to forest management must be expected to differ in different ecological settings. Populations in the southern portion of the species' range have lower productivity and appear to use old forest habitats that are declining in aerial extent.
- In productive forest habitats, boreal owls have a high potential rate of population growth based on their dramatic numeric and functional response to changing prey populations. Therefore, populations can recover following declines if habitat is intact and prey are abundant.
- Conservation concerns for boreal owls arise when one considers long term persistence and/or focuses on individual populations. These concerns are based on the available information on boreal owl habitat use, the dynamics of those habitats, trends in forest management, and the species' life history. Specifically:
  - — The available data indicate that boreal owls, in at least some populations in the United States, use mature and older forest for foraging, roosting, and nesting. Primary prey of boreal owls also are more abundant in mature and older forests. The relative abundance of mature and older forest declines under traditional forest management programs as demonstrated in the Pacific Northwest and Northern Rockies. Therefore, habitat that provides necessary life requisites (food, thermal cover, and nesting substrate) has been declining and will continue to do so under current management plans, jeopardizing the long-term persistence of boreal owls in the United States.
  - — Habitats used by boreal owls develop slowly after deforestation due to the short growing season in subalpine environments. Therefore, forest stands may require several centuries to become quality foraging or nesting habitat after stand removal; recovery of degraded habitat will be an extremely long-term process. As outlined in National Forest plans, clearcutting is the dominant management direction in subalpine forests in the northern Rockies. Development of mature and old forest from this management is questionable and, to date, has not been observed (see Chapter 10). If timber harvest decreases in the Pacific Northwest and shifts to the northern Rockies the threat to boreal owl habitat will increase.
  - — Populations of boreal owls studied in the western United States are less productive than most of those documented in Europe. Further reductions in productivity due to declines in habitat quality will reduce the average persistence time for populations.
  - — Individual populations of boreal owls in the United States are relatively small and dispersed due to the naturally patchy distribution of subalpine and boreal forest habitats. The populations have lower individual persistence probabilities than would larger populations. The persistence of individual populations, then, will be influenced by relatively small land management activities.
  - — Because of the high temporal variability in boreal owl productivity and the nomadic nature of the species, persistence of individual populations may rely heavily on neighboring populations. Due to this metapopulation structure, the persistence of individual populations and (potentially) large segments of the metapopulation could rest on particular key populations that provide surplus dispersing individuals or act as stepping stones for exchange among populations. The identity or even existence of such key populations is unknown and therefore their protection is not assured.
  - — Land management in the matrix of habitat surrounding subalpine forests will influence the success of dispersing owls. Therefore management outside the species' primary habitat will have consequences for the owl.

WHAT ARE THE MANAGEMENT CONSIDERATIONS?

A thorough discussion of management considerations of this assessment would be largely redundant with previous sections. A brief outline of the most relevant considerations follows.

Boreal owls occupy forest habitat. The future condition of forest structure will influence populations of this avian carnivore. The link between forest structure and composition and the status of boreal owl populations is strong but indirect. Forest structure influences the availability of suitable cavities, the quality of roost sites, the foraging movements of individual owls, and prey availability. Landscapes without forest cannot support boreal owls; in forested lands, forest structure influences owl population status. Management of forest structure from the stand to landscape scale in subalpine and boreal to boreal-transition forest, then, will influence the long-term conservation status of this species. Because forest succession is slow in spruce-fir and boreal forests, management must acknowledge that clearcut sites will remain unsuitable for roosting or foraging for a century or more and new nest trees will not develop in some situations for two centuries or longer. Scientific understanding of boreal owl habitat relationships has not advanced sufficiently to devise sophisticated habitat management guidelines for any region. It is clear, however, that large clearcuts eliminate boreal owl habitat for many years and that clearcutting does not mimic the dominant natural disturbance agents in this system. Modification of these forests in ways that remove characteristics of mature and old forests should be done with caution. Monitoring the consequences of forest change at the stand and broader scales will be important in improving management (i.e., adaptive management involving researchers and managers).

Management of forest structure for any single life requisite (nesting, roosting, or foraging) will not assure suitable habitat to conserve boreal owl populations. Therefore, management should not be framed in terms of "management for nesting habitat" or another life requisite. Instead, management for boreal owls will be most successful if placed in an ecosystem context. The boreal owl depends on a plethora of other forest organisms: primary cavity nesters, small mammals, fungi, lichen, insects, and the dominant forest trees to name just a few (see figure 5, Chapter 9). Therefore, boreal owl management is a component of ecosystem management in subalpine forests in the western United States and boreal/boreal-transition forests in the east.

Conservation of boreal owls will require a regional approach to habitat management. Because populations of boreal owls in the United States likely occur in a complex metapopulation structure, the status of any single population is determined in part by many other populations. Management of individual populations outside the context of the larger metapopulation ignores the fact that most boreal owl populations are small and therefore have low probability of persistence in isolation. Even if high quality habitat remains within any small owl population, the population is likely to become extinct without dispersal from other groups. Identification and maintenance of source populations within a region will be a key to boreal owl management.

Finally, the knowledge necessary to build a conservation strategy is lacking and without a conservation strategy, persistence of this owl over the long term is questionable. Many key aspects of boreal owl demography, habitat use, and the owl's relationship with the forest system (primary cavity nesters, prey, predators, etc.) have not been investigated for any population. Most of the links represented in figure 5 (Chapter 9) are inferred and not backed by direct empirical information. Much of what we do know results from investigations in Europe. How this knowledge relates to particular populations in North America is unknown. Therefore, anything but the most general analysis of management impacts will not be possible without further knowledge.

REFERENCES

Alexander, R. R. 1987. Ecology, silviculture, and management of the Engelmann spruce-subalpine fir type in the central and southern Rocky Mountains. United States Forest Service Agricultural Handbook 659. Washington, DC, USA.

Allee, W. C. 1931. Animal aggregations. A study in general sociology. University of Chicago Press, Chicago, Illinois, USA.

Andersson, M. 1980. Nomadism and site tenacity as alternative reproductive tactics in birds. Journal of Animal Ecology 49:175-184.

Bondrup-Nielsen, S. 1978. Vocalizations, nesting, and habitat preferences of the Boreal Owl (*Aegolius funereus*). Thesis. University of Toronto, Toronto, Ontario, Canada.

Brown, J. H., and A. Kodric-Brown. 1977. Turnover rates in insular biogeography: effects of immigration on extinction. Ecology 58:445-449.

Campbell, T. M., and T. W. Clark. 1980. Short-term effects of logging on red-backed voles and deer mice. Great Basin Naturalist 40:183-189.

Cheatham, H. A. Timber Management Planner, U. S. Forest Service, Region 4, Ogden, UT. [Personal communication]. Autumn 1988.

Cramp, S., editor. 1977. Handbook of the birds of Europe, the Middle East and Europe; the birds of the western palearctic. Volume IV. Oxford University Press, London, England.

Goodman, D. 1987. The demography of chance extinction. Pages 11-34 *in* M. E. Soule, editor. Viable populations for conservation. Cambridge University Press.

Halvorson, C. H. 1982. Rodent occurrence, habitat disturbance, and seed fall in a larch-fir forest. Ecology 63:423-433.

Hayward, G. H., P. H. Hayward, E. O. Garton., and R. Escano. 1987. Revised breeding distribution of the Boreal Owl in the northern Rocky Mountains. Condor 89:431-432.

Hayward, G. D., P. H. Hayward, and E. O. Garton. 1993. Ecology of Boreal Owls in the northern Rocky Mountains, USA. Wildlife Monographs 124:1-59.

Hayward, G. D., R. K. Steinhorst, and P. H. Hayward. 1992. Monitoring Boreal Owl populations with nest boxes: sample size and cost. Journal of Wildlife Management 56:776-784.

Hirons, G. J. M. 1985. The importance of body reserves for successful reproduction in the Tawny Owl (*Strix aluco*). Journal of Zoology (London) Series B. 1:1-20.

Korpimaki, E. 1986. Gradients in population fluctuations of Tengmalm's Owl *Aegolius funereus* in Europe. Oecologia 69:195-201.

Korpimaki, E. 1988. Effects of territory quality on occupancy, breeding performance, and breeding dispersal in Tengmalm's Owl. Journal of Animal Ecology 57:97-108.

Laux, J. W. Timber Management Planner, U. S. Forest Service, Region 1, Missoula, MT. [Personal communication]. Autumn 1988.

Lindstedt, S. L., B. J. Miller, and S. W. Buskirk. 1986. Home range, time and body size in mammals. Ecology 67:413-418.

MacArthur, R. H., and E. O. Wilson. 1967. The theory of island biogeography. Princeton University Press, Princeton, New Jersey, USA.

Palmer, D. A. 1986. Habitat selection, movements and activity of Boreal and Saw-whet Owls. Thesis. Colorado State University, Fort Collins, Colorado, USA.

Pimm, S. L., H. L. Jones, and J. Diamond. 1988. On the risk of extinction. American Naturalist 132:757-785.

Ramirez, P., and M. Hornocker. 1981 Small mammal populations in different-aged clearcuts in northwestern Montana. Journal of Mammalogy 62:400-403.

Ryder, R. A., D. A. Palmer, and J. J. Rawinski. 1987. Distribution and status of the Boreal Owl in Colorado. Pages 169-174 *in* R. W. Nero, C. R. Knapton, and R. H. Hamre, editors. Biology and conservation of northern forest owls: symposium proceedings. United States Forest Service General Technical Report RM-142.

Scrivner, J. H., and H. D. Smith. 1984. Relative abundance of small mammals in four successional stages of spruce-fir forest in Idaho. Northwest Science 58:171-176.

Shaffer, M. L. 1981. Minimum population sizes for species conservation. BioScience 31:131-134.

Sonerud, G. A. 1986. Effect of snow cover on seasonal changes in diet, habitat, and regional distribution of raptors that prey on small mammals in boreal zones of Fennoscandia. Holarctic Ecology 9:33-47.

Chapter 9

Review of Technical Knowledge: Boreal Owls

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INTRODUCTION

The boreal owl (*Aegolius funereus*), known as Tengmalm's owl in Eurasia, occurs throughout the holarctic in boreal climatic zones. This medium-size owl (100-170 g) occupies boreal and subalpine forests in an almost continuous circumboreal distribution that extends from Scandinavia eastward across the northern forests of Siberia and from Alaska across Canada to the Atlantic (Dement'ev and Gladkov 1954). On each continent, disjunct populations occur in mountains south of the broad transcontinental boreal forest populations (Cramp 1977, Voous 1988). Boreal owls in the mountain regions of Europe and Asia have long been recognized as isolated resident breeding populations, whereas in North America, breeding status was only recently documented in the mountains of the western United States (Hayward and Garton 1983, Palmer and Ryder 1984, Hayward *et al.* 1987a, Whelton 1989).

In-depth study of boreal owl biology and ecology in North America is limited to four, short-term investigations (Bondrup-Nielsen 1978, Meehan 1980, Palmer 1986, and Hayward *et al.* 1993). As an example of the lack of attention paid this species, prior to 1979 the USDI Fish and Wildlife Service had no records for banded boreal owls west of the Mississippi (W. Martin, pers. comm.). Knowledge of the species' biology and ecology comes mostly from Fennoscandia where *Aegolius funereus* may be the most studied owl. Many investigations in Europe are long-term efforts. Franz *et al.* (1984), Sonerud (1989), Schelper (1989), and Korpimaki (1992) each report studies lasting over 15 years. Korpimaki, who initiated investigations in 1966, continues work on the same sites today. Ecologists in Fennoscandia and eastern Europe have emphasized study of breeding biology, productivity, movements, food habits, and relationship with prey populations. These studies stem largely from examination of populations that breed almost exclusively in nest boxes. Results from studies in the Old World indicate that the biology and ecology of boreal owls vary geographically and are strongly related to local forest conditions and prey populations.

In contrast with studies in Europe, habitat use has been emphasized in the few investigations in North America. Studies on the two continents have generated few data with which to contrast the biology of the species between continents. Therefore, the basis for inferring North American biology and ecology based on European results is unclear. The variability witnessed in Europe suggests caution. However, to the degree that variation in Europe follows geographic, climatic, or habitat gradients, a more sound basis upon which to build inferences for North America is possible.

The paucity of scientific knowledge from North America necessitates reliance on the extensive knowledge accumulated in Europe for portions of the assessment. Ignoring that knowledge would be careless. However, we cannot directly infer ecological patterns in North America based on the European knowledge. Therefore, I have been careful to point out the geographical source of knowledge, and where appropriate, describe ecological patterns for Europe that have been related to environmental gradients. By doing so, I seek to describe patterns recognized in Europe that may relate to populations in North America.

Note: Throughout this paper, measures of variation are 95% bounds on estimates unless otherwise indicated.

SYSTEMATICS

Ford (1967) associated the genus *Aegolius* with *Surnia* and *Ninox* (northern and southern hawk owl genera) based on osteology of 75 owl species. Aside from the boreal owl, the genus *Aegolius* includes three species: the northern saw-whet owl (*A. acadicus*), unspotted saw-whet owl (*A. ridgwayi*), and buff-fronted owl (*A. harrisii*), which all occur only in the New World. The largest species of the genus, *A. funereus* occurs north of the others and is more widely distributed. Norberg (1987) speculates that the genus originated in the New World and only the boreal owl expanded its range beyond the Americas. The more northern distribution and larger size of *A. funereus* likely facilitated range expansion via the Bering Strait.

Boreal owls in North America represent a homogenous taxonomic group and are recognized as a single subspecies, *A. funereus richardsoni*. Six subspecies are recognized in Eurasia. Abrupt distinctions are apparent in only *A. f. beickianus* and *caucasicus*, which are southern, more isolated populations. Otherwise, *A. f. funereus* — north and central Europe; *A. f. sibiricus* — north and central Asia; *A. f. magnus* — northeast Siberia; and *A. f. pallens* — west and central Siberia vary as a cline across Eurasia (Dement'ev and Gladkov 1954). Generally the largest and lightest forms are found in northeast Siberia, with a size reduction and darkening westward and southward (Dement'ev and Gladkov 1954). *A. f. richardsoni* is among the darkest forms.

DISTRIBUTION

Species Range

Boreal owls occupy boreal forests throughout the northern hemisphere forming an almost continuous band across North America and Eurasia. In Europe, scattered populations extend south of the circumboreal range in the Pyrenees, Alps, and Caucasus mountains and in Asia along Tarbagatai, Tien Shan, and Zervshan ranges (see maps in Dement'ev and Gladkov 1954:436 and Cramp 1977:607, 608 for worldwide distribution). Similar southern populations occur in North America as described below.

Recently the species' documented range has expanded in Europe like in North America. Most new records are from mountainous locales (see Cramp 1977:607 and Hayward *et al.* 1987a). Rather than a recent range expansion, these records likely represent increased interest in owls and increased human recreation in mountain areas during winter.

North America

Within North America, boreal owls occur in a continuous band concurrent with the boreal forests of Alaska and Canada (see Johnsgard 1988 for continental distribution). The breeding range extends from northern treeline southward in forested regions of Canada to the extreme northern United States in Minnesota (Eckert and Savaloja 1979, Lane 1988) and likely Wisconsin (Erdman 1979), Michigan, and Maine (Catling 1972). East of the Rocky Mountains, breeding has been confirmed only in Minnesota. In western North America the species' range extends southward beyond 38° N latitude (Map 2). South of the continuous transcontinental band, populations are restricted to subalpine forests in the Rocky Mountains, Blue Mountains, and Cascade Ranges (Palmer and Ryder 1984, Hayward *et al.* 1987a, Whelton 1989). The southernmost records occur in mountains of northwestern New Mexico (Stahlecker and Rawinski 1990).

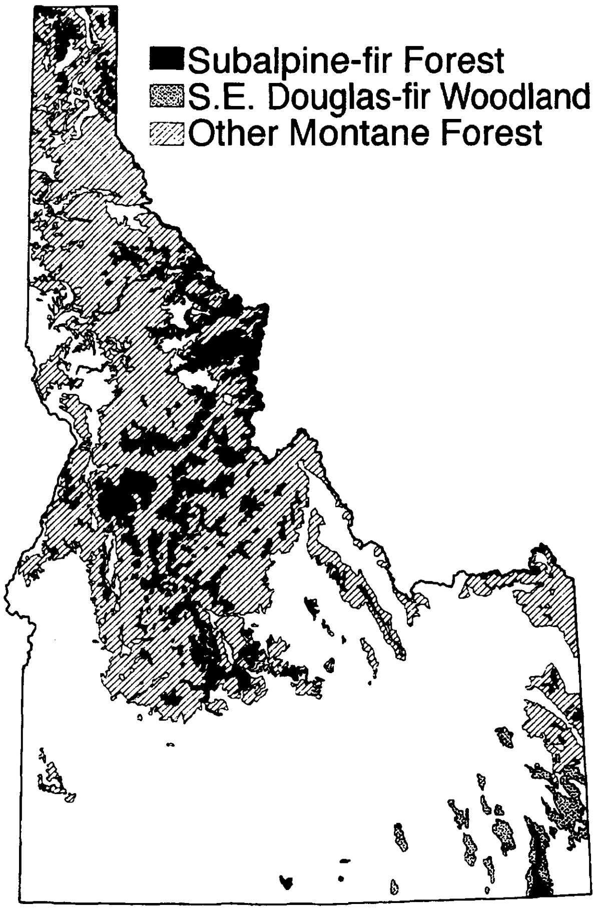


Figure 1.—Example of the patchy nature of boreal owl distribution in the western United States based on the species' estimated distribution in Idaho. Owl distribution inferred from distribution of forest vegetation types. Potential habitat is defined as forested sites in the subalpine-fir zone throughout the state and Douglas-fir woodland in southeastern Idaho. Other montane forests are not considered potential habitat. Data taken from Idaho gap analysis project (adapted from Hayward *et al.* 1993).

Due to the species' association with high elevation forests in the western United States (discussed in-depth under Habitat Use), populations may occur as geographic isolates dispersed throughout the western mountains (for an example see figure 1). As a result of the naturally fragmented nature of boreal owl habitat in the western mountains, the species is distributed in North America in two contrasting patterns. In the north, populations of interacting individuals may extend for hundreds of miles, while in the south, numerous breeding populations occur as islands of habitat linked only through long-distance dispersal through extensive areas without breeding habitat.

Although boreal owls are thought to breed in much of the forested portion of Alaska, surveys have been conducted in few portions of the state (see Gabrielson and Lincoln 1959, Armstrong 1980). Literature documentation for boreal owls in Alaska extends from the Brooks Range (Campbell 1969), to the Pribilof Islands (Evermann 1913), and to the north Gulf Coast (Isleib and Kessel 1973). Recent surveys document singing boreal owls in southeast Alaska on the mainland and a number of islands (draft agency report, Suring 1993; see Map 2 in sleeve of this book).

The recognized distribution of boreal owls has changed yearly since 1979 as interest in the owl developed and efforts to locate breeding populations increased. Prior to 1979, breeding populations of boreal owls were not thought to occur south of Canada. The 1983 American Ornithological Union checklist of North American birds described the southern extent of western boreal owl populations as south-central Canada, although it also recorded breeding populations in Colorado and northwestern Wyoming. In 1985, Idaho, Washington, and Montana were added (data reported in Hayward *et al.* 1987a) but populations were recognized in only isolated locales in each state. By 1987, biologists realized that populations occurred throughout the northern Rockies in high elevation conifer forests south to northern New Mexico. I expect the documented range to continue to expand as previously unsurveyed regions receive attention. In Idaho and Montana, where surveys have been conducted for over a decade, our understanding of boreal owl distribution will become more refined. In regions where few surveys were conducted in the past, such as Utah, Alaska, northern Wisconsin, northern Michigan, and northern New England, I expect significant changes in the recognized distribution.

Map 2 depicts the estimated breeding range of the species based on reports from the literature and recent surveys conducted largely by state and federal agencies. Reports from the technical literature are acknowledged separately from agency surveys, because these records have undergone greater scrutiny. I recognize that individual records may be suspect. Some surveys were conducted by inexperienced persons and the level of training and experience of personnel conducting surveys varied. Because the majority of survey personnel received some training and discussed their observations with owl experts, however, I believe the estimated distribution to be reliable.

Species Status and Trend

Direct measures of population status or trend are not available for populations in North America. In contrast with Europe, investigations of boreal owls in North America have been short term and have not emphasized study of productivity or demography. Due to the paucity of historical information, direct estimates of status and trend will be difficult in the near future. Currently, I am aware of only one effort, begun in 1988, to intensively monitor population trend in North America (Hayward *et al.* 1992).

The boreal owl's range in North America is extensive. In northern Canada, it occurs in many areas where land management currently does not alter natural vegetation patterns. Recent surveys indicate the species also occupies an extensive geographic range south of Canada. Populations in this region occur on lands where human impact is greater. The potential influence of land management on owls across these lands will be discussed later in this document. Since direct measures of trend are not available, and the species occupies a large geographic area, any inferences to population trend must be inferred indirectly by linking the species' ecology and observed patterns of landscape change.

In Fennoscandian forests, boreal owls are considered the most abundant Strigiform (Merikallio 1958, cited by Korpimaki 1984). Despite long-term investigation of the species, however, reliable indication of long-term trends are unavailable due to the difficulty in surveying and censusing nocturnal owls (Lundberg 1978). Short-term fluctuations in breeding populations are evident from nest box surveys (e.g. Franz *et al.* 1984, Lofgren *et al.* 1986, Schelper 1989, Sonerud 1989, and Korpimaki 1992), but status and long-term trends have not been reported. Significant reduction in natural breeding cavities in Scandinavia resulting from removal of old forest (Korpimaki 1981 and others) would imply reduced populations and potentially restricted distribution.

MOVEMENTS: ANNUAL, SEASONAL, AND DAILY

Annual Movements and Site Tenacity of Adults

Annual movement patterns of boreal owls are poorly understood in North America but have received considerable attention in Fennoscandia and Germany. Trapping stations at Whitefish Point, Michigan, and Hawk Ridge Research Station, Minnesota, and records of owl sightings by birders represent the majority of data on boreal owl movements in North America (Kelley and Roberts 1971, Catling 1972, Evans and Rosenfield 1977 and references therein). Trapping observations are difficult to interpret, and conclusions drawn from these observations must be regarded as hypotheses.

Based on the periodic sightings of boreal owls (1922, 1954, 1962, 1965, 1968) south of the species' range in eastern North America, winter irruptions have been hypothesized by Catling (1972) and Evans and Rosenfield (1977). Reported irruptions extend from Maine through Michigan and Minnesota (Catling 1972). Periodic observations of boreal owls have been documented in Illinois (Coale 1914, Wyman 1915), Minnesota (Evans and Rosenfield 1977), Wisconsin (Erdman 1979), and New York (Yunick 1979) and frequently coincide with increased observations of northern saw-whet, great gray (*Strix nebulosa*), and northern hawk owls (*Surnia ulula*). Sightings and captures are concentrated in autumn (late October-mid November) and late winter (February-April). The age and sex composition of the irruptive populations are poorly understood. Furthermore, whether individuals observed during these irruptions attempt to breed in southern areas, return to breeding areas in the north, or represent a population sink, is unknown. Catling (1972:223) suggests that a return flight occurs in April and May. Speculation concerning direction of movements appears to be based on little empirical evidence.

In Idaho, during a single week in February 1986, two radio-marked males left home ranges occupied for more than a year (a third male died during the same period). One male was relocated in May 80 km away. Three radio-marked females in Idaho left their former home ranges within 2 weeks of ceasing brooding young in July. One moved ~17 km while the others moved greater distances and could not be relocated (Hayward *et al.* 1987b). Although these owls were documented making nomadic-like movements, other radio-marked owls in the study remained sedentary. These observations are very limited but suggest nomadic behavior.

In contrast with limited information in North America, extensive European studies suggest a complex pattern of nomadism and site tenacity that varies geographically and differs among sex and age classes. In general, the species is characterized as nomadic, at times exhibiting year-round residence within a stable home range but dispersing in years of poor prey populations (Mysterud 1970, Wallin and Andersson 1981, Lofgren *et al.* 1986, Korpimaki *et al.* 1987, Sonerud *et al.* 1988, Schelper 1989). Korpimaki (1986b) recognized a trend of increased population fluctuations in more northern populations associated with a greater degree of nomadism. He related the pattern to winter snow depth and range of prey available to the owls in winter.

In Scandinavia where year-to-year movements were studied using band recoveries from long-term site specific studies employing nest boxes, a unique pattern of residency and nomadism was first recognized by Mysterud (1970). Mysterud (1970) suggested that nomadic behavior in the Fennoscandian population is adapted to the 3-4 year microtine cycle and regional variability in microtine abundance. Lundberg (1979) refined the model and hypothesized that the conflicting pressures of food stress favoring nomadism and nest site scarcity favoring site tenacity result in different movement patterns in males and females; females exhibit nomadism while males exhibit greater site tenacity. Lofgren *et al.* (1986), Korpimaki *et al.* (1987), and Sonerud *et al.* (1988) confirmed the mixed pattern of male residency and irregular female dispersal in adult Tengmalm's owls. Korpimaki's review (1986b) further refined the understanding of nomadism in the species, suggesting that sexual differences in residency vary geographically. In central Europe both sexes appear to be largely site tenacious, but young owls are nomadic (Franz *et al.* 1984). In southern Fennoscandia males are resident and females and juveniles nomadic. In northern Sweden, both adults and juveniles exhibit nomadism (Korpimaki 1986b).

In addition to the influence of snow conditions, geographic setting, and prey conditions mentioned above, nest predation and nesting success have been shown to influence dispersal in adult female boreal owls (Sonerud *et al.* 1988). Adult females whose nests are unsuccessful have an increased probability of dispersing long distances. Predation of nestlings further increases the probability of long dispersal (figure 2).

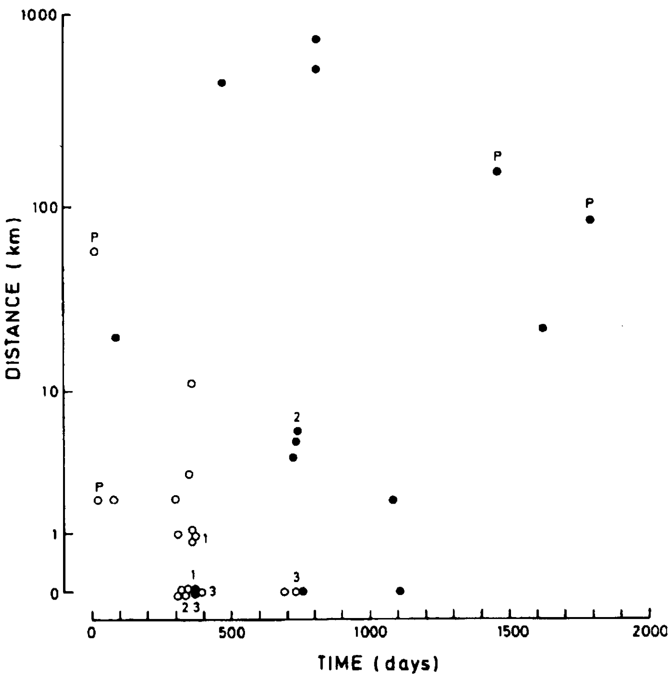


Figure 2.—Distance moved and time elapsed between ringing and recovery of female Tengmalm's owls ringed in Norway while breeding. Open circles denote dispersals occurring within a microtine peak (high prey availability), and filled circles denote dispersals involving a microtine decline (low prey availability). Dispersals following nest predation are indicated by a P. Dispersals made by the same female are indicated by numbers (from Sonerud *et al.* 1988).

Dispersal Frequency

Interpopulation movements are extremely important in metapopulation (population of populations) dynamics. Therefore it is important to determine rates of immigration and emigration among component populations. Estimating the portion of a population involved in nomadic or dispersal movements is difficult, however. For instance, most recoveries of banded birds are from nesting birds retrapped by the original bander. Therefore, estimates of dispersal will be biased toward documenting site tenacity or short-distance movements.

Despite these shortcomings, studies of boreal owls in Fennoscandia and Germany have estimated emigration rates that follow the north-south geographic gradient described earlier (more nomadic movements in northern populations). After successfully nesting, 0% and 8% of adult females dispersed farther than 20 km from two populations in Germany (central Europe) (Franz *et al.* 1984 and Schwerdtfeger 1984, both according to Sonerud *et al.* 1988). Corresponding proportions for central Norway, Finland, and northern Sweden were 14%, 31%, and 33%, respectively (Lofgren *et al.* 1986, Korpimaki *et al.* 1987, and Sonerud *et al.* 1988). The proportion of adult females dispersing farther than 100 km in central Norway, Finland, and northern Sweden were 13%, 17%, and 17%. In all cases dispersal over 100 km took place between microtine peaks.

Proportions of males dispersing is more poorly documented because of the greater difficulty in trapping nesting males. In one Finnish study (Korpimaki *et al.* 1987), all retrapped males ($n = 23$) were caught within 5 km of the original banding site. Of 170 males recovered in Finland, only two have been recovered far from their original breeding site (97 and 180 km).

Based on these patterns, I suggest that boreal owls in the United States likely occur in a metapopulation structure. The nomadic nature of the species, frequent movements by adults and young, and the ability of individuals to disperse long distances indicate the species' behavior facilitates a metapopulation distribution. Furthermore, suitable habitat in the United States occurs in numerous patches separated by tens to hundreds of km (figure 1, also see Movements as Related to Demography and Metapopulation Structure later in this chapter). The habitat distribution, then, provides a landscape that will support small populations each separated by distances greater than the normal daily movement and normal yearly movement distances of individual owls. Linkage among populations, then, results from the nomadic movement of adults or exceptional long-distance dispersal of some young owls.

Dispersal Distances

Adults who disperse over 20 km from a breeding site may frequently move long distances as nomads. Documenting long movements is difficult, however. Lofgren *et al.* (1986) reported females breeding 550, 308, 289, 220, 70, and 70 km from their original breeding site in northern Sweden. In the same study, Lofgren *et al.* (1986) reported males breeding 21 and 115 km from their original breeding site. Sonerud *et al.* (1988) reported dispersal distances for breeding adult females first banded in southeastern or central Norway (figure 2) while Korpimaki *et al.* (1987) summarized dispersal distances for Finland (figure 3). In Germany, based on owls banded at nest boxes, Franz *et al.* (1984) found 5% of females nesting in the same box as the previous year and that the shortest 93% (left side of the distribution) of all dispersal movements averaged 9.3 km. Of the 2% of females who dispersed long distances, the maximum was 194 km. Other maximum distances include 728 km for Norway (Sonerud *et al.* 1988), 550 km for Sweden (Lofgren *et al.* 1986), and 550 km for Finland (Korpimaki *et al.* 1987).

Annual Movements and Site Tenacity of Juveniles

Young boreal owls frequently disperse long distances from natal sites but have been recorded breeding within 0.5 km of their natal site (Hayward, G. D. and P. H. Hayward unpublished data from Idaho). Korpimaki *et al.* (1987) reported median distances of 88 and 21 km between juvenile male and juvenile female banding sites and later at two breeding sites in Finland (figures 3 and 4). In Norway, 3 males banded as juveniles were recaptured breeding 5-11 km from the natal site while 9 females had moved 2-239 km (Sonerud *et al.* 1988). Twenty percent of recoveries for owls marked as nestlings exceed 100 km in West Germany (Franz *et al.* 1984) and 51% in Finland (Korpimaki *et al.* 1987).

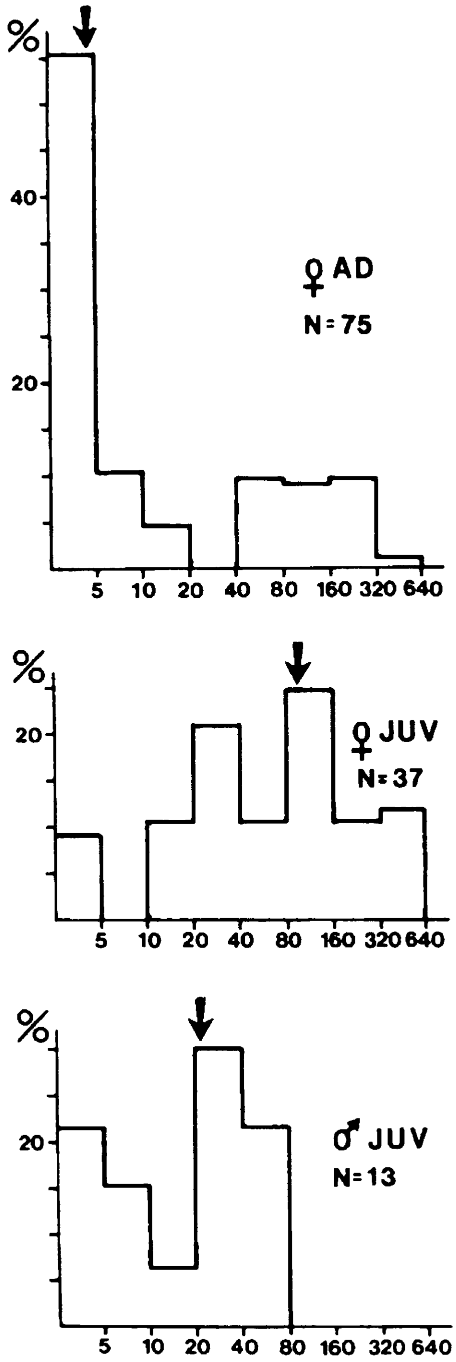


Figure 3.—Dispersal distances (km, log scale) between ringing and recovery sites of Tengmalm's owls ringed as breeding females (upper chart) or nestlings (lower charts) and retrapped in later years when breeding. Medians (4 km, 88 km, and 21 km) are indicated by arrows. N = number of recoveries (from Korpimaki *et al.* 1987).

Seasonal Movements

Patterns of movements associated with seasonal cycles have been studied in only one locale (Hayward *et al.* 1993). Patterns observed during this study in the wilderness of central Idaho may be unique to the geographic characteristics of the study area. Winter and summer home ranges of individual owls overlapped extensively but centers of activity for 12 radio-marked owls shifted. Average elevation of roosts used by the owls was 186 (±105) m lower in winter than summer. Despite this shift, areas used in winter had complete snow cover exceeding 0.5 m each winter and the owls frequently used areas with 1.5-2.0 m of snow accumulation. Snow-free slopes occurred within 2 km of most owls' ranges during most winters, but owls were not observed using these areas.

Movements Within the Home Range

Burt (1943:351) defined home range as the area traversed by an individual in its normal activities of food gathering, mating, and caring for young. For boreal owls, these movements define how individuals use space during periods when they are not nomadic or dispersing. Except during periods of nomadism, boreal owls are resident within and between years.

Boreal owls studied in the western United States use large home ranges. In Colorado, home ranges of two males located on daytime roosts (>8 locations for each owl spanning 252 and 173 days) encompassed 1,395 and 1,576 ha and overlapped one another by >90% (Palmer 1986). In central Idaho, nest sites occurred in lower portions of home ranges (few cavities were found at higher elevations) while roosting and foraging occurred throughout the range. Winter ranges covered 1,451 ha (±522; n = 13, range 320-3390 ha), and summer ranges covered 1,182 ha (±334; n = 15, range 229-2386 ha). These estimates of home range size are based on modest sample sizes and therefore should be considered minimum use areas. Harmonic mean estimates (which were used in this case) tend to be biased low with small sample size (E. O. Garton; pers. comm.).

Boreal owls are very mobile predators; the owls frequently traverse much of their home range in the course of 2-3 days or weeks (Hayward *et al.* 1987b). In spruce-fir forests of Colorado, roosts used on consecutive days averaged 708 m apart (n = 113) (Palmer 1986). In Idaho, distance between consecutive roosts of 14 owls (150 locations of consecutive roosts) averaged 1,540 m (±446) in winter and 934 m (±348) in summer (Hayward *et al.* 1993).

Daily Movements

Diurnal Period

Boreal owls move little during the day; they generally remain within the same forest stand during daylight. These owls frequently change roost trees but rarely fly over 40 m when changing roosts (Hayward *et al.* 1993). Based on studies in Idaho (Hayward *et al.* 1993), during daylight boreal owls perch quietly with eyes closed a majority of the time (77% based on 46 hours of observation on 16 days). Periods of sleep rarely exceed 40 minutes and are broken by 2-5 minute periods of preening (6% of time) and looking about (10% of time). Eating (4%), daytime hunting (1%), and moving among roost perches (<1%) are other important daily activities. I observed owls hunt during daylight in winter at 2.9% of roost locations (n = 448) and in summer at 7.4% (n = 446) of roost locations (Hayward 1989).

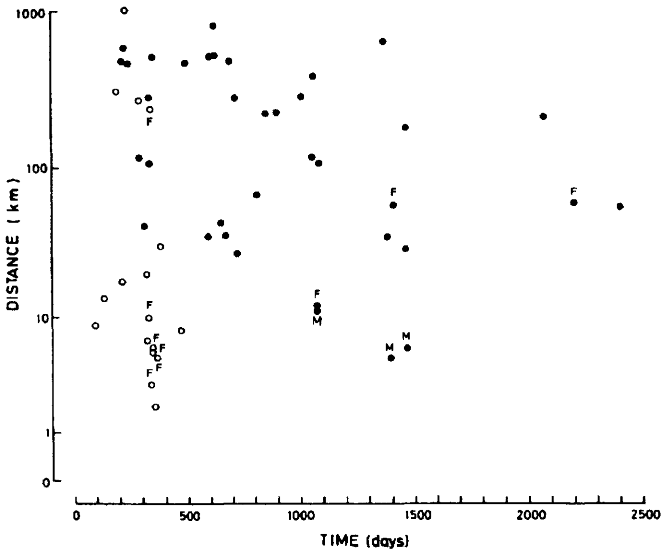


Figure 4.—Distance moved and time elapsed between ringing and recovery of female Tengmalm's owls ringed as nestlings in Norway. Open circles denote dispersals occurring within a microtine peak (high prey availability), while filled circles denote dispersals involving a microtine decline (low prey availability). Males recaptured while breeding are indicated by M, females recaptured while breeding by F (from Sonerud *et al.* 1988).

Nocturnal Period

Nocturnal activity is poorly studied, especially outside the breeding season. Boreal owl foraging activity is concentrated after dark except in northern latitudes during summer. During periods of 24-hour light, foraging is concentrated between sunset and sunrise.

Event recorders have been employed at nest cavities to infer foraging activity patterns of male owls provisioning nests. In Finland, during the incubation period, prey deliveries generally began 1 hour 14 minutes after sunset and ended 49 minutes before sunrise based on records for 6 years (Korpimaki 1981). Depending upon latitude and phase of nesting cycle, night-time activity follows a bimodal pattern with peaks in nest deliveries during the first hours after sunset and again prior to sunrise (Klaus *et al.* 1975, Korpimaki 1981, Hayward 1983). This pattern is most apparent in southern latitudes (i.e., East Germany, Idaho) and early in nesting. In northern Scandinavia, a bimodal pattern appears early in incubation but the two peaks fuse as daylength increases and night-time foraging period decreases (Korpimaki 1981). The foraging activity period also varies depending on phase of the vole cycle. In peak vole years, activity lasted longer each night, the peaks in activity were more pronounced, and prey deliveries after sunrise were more frequent (Korpimaki 1981).

Night-time foraging can be very intense, especially when nestlings near fledging. In Idaho, records from four nests suggest that females leave the nest once each night during incubation (for evacuation) and usually twice after the young hatch (Hayward, G. D. and P. H. Hayward, unpubl. data). Assuming that all records other than for the female's evacuation were prey deliveries, deliveries averaged 3.5 (± 0.33 SD, n = 84, range = 0-9) during incubation and 5.0 (± 0.61; SD, n = 6, range = 0-12) during brooding. Clutches at the four nests were 2, 2, 3, and 3; each fledged two young. In Finland, Korpimaki (1981) estimated 9.8 deliveries/night during brooding period and 8.0 after the female left the nest.

Norberg (1970), Bye *et al.* (1992), and Hayward *et al.* (1993) documented hunting movements of boreal owls. Based on these observations, the owls can be classified as sit-and-wait predators or searchers (as opposed to pursuers) but are very active while hunting. During a foraging bout, the birds move through the forest in an irregular or zigzag pattern, flying short distances between perches (Hayward 1987). They spend a majority of time perched; little time is spent actively pursuing prey. While perched, the owl constantly looks about with rapid head movements, apparently responding to forest sounds.

When foraging, owls usually fly 10 to 30 m between hunting perches (Norberg 1970, Hayward 1987). In Idaho, over 75% of all flights were 25 m or less. Although the pattern of flights varied, owls observed foraging in Idaho doubled back frequently and, thus, covered a relatively small area within several forest stands rather than a long narrow path. While searching for prey, boreal owls perch on low branches. Perches used during foraging observations in Idaho averaged 4 ± 0.6 m high ($n = 114$). Similarly, average perch height for 17 owls monitored in Norway ranged from 1.7 to 8.7 m (Bye *et al.* 1992).

Boreal owls may traverse several km during a nocturnal foraging bout. Because daytime roosts appear to represent the end of nighttime foraging bouts, locations of consecutive daytime roosts suggest the magnitude of minimum travel distances (Hayward *et al.* 1987b). Distances between consecutive day roosts of 14 owls (7 females and 7 males) on 150 occasions over 4 years in Idaho ranged from 0-6935 m. Mean distances did not differ significantly between winter and summer (winter 1540 [±446] m, summer 934 [±348] m). During nesting, five males roosted over 1000 m from their nests 85% of the time (average 1729 [±831] 200-5600 m) (Hayward *et al.* 1993).

HABITAT USE

Broad Habitat Use Patterns

As year-round residents, boreal owls use similar habitats during all seasons. They occur only in forested landscapes where they nest exclusively in tree cavities or artificial nest structures (Mikkola 1983). The few studies documenting nesting habitat indicate the species uses a range of vegetation types depending on geographic region (e.g., Bondrup-Nielsen 1978, Eckert and Savaloja 1979, Palmer 1986, Korpimaki 1988a, Hayward *et al.* 1993). In northern portions of their range in North America (Alaska and Canada) the owls breed in boreal forest characterized by black and white spruce (*Picea mariana, P. glauca*), aspen (*Populus tremuloides*), poplar (*P. balsamea*), birch (*Betula papyrifera*), and balsam fir (*Abies balsamea*) (Bondrup-Nielsen 1978, Meehan and Ritchie 1982). In northern Minnesota and Michigan, singing sites and nests have been documented in old aspen and mixed-forest sites (Eckert and Savaloja 1979, Lane 1988). In the southern portions of their range in North America (Rocky Mountains, Blue Mountains, and Cascades) published research documents boreal owls in subalpine forest habitats characterized largely by subalpine fir (*Abies lasiocarpa*) and Engelmann spruce (*Picea engelmannii*) and transition forests within 100 m of this elevation (Palmer 1986, Hayward *et al.* 1987a). Because of changes in life zones with latitude, an elevation range cannot be specified for the entire western region of the United States. However, extensive surveys in Idaho and Montana in 1984 and 1985 found no owls below 1,292 m elevation, and 75% of the locations were above 1,584 m (Hayward *et al.* 1987a). Less extensive surveys in northern Colorado found most locations above 3050 m (Palmer 1986).

In USDA Forest Service Regions 1, 2, 4, 6, 9, and 10, biologists have documented boreal owls occurring (but not confirmed breeding) on 26 National Forests and confirmed breeding on 11 other Forests (Chapter 8, table 2). We asked these biologists for an indication of the forest types where boreal owls have occurred. Spruce-fir forest was reported more than any other type (45% of forests with documented breeding). Other forest types in decreasing order of frequency were lodgepole pine (*Pinus contorta*), mixed-conifer, Douglas-fir (*Pseudotsuga menziesii*), aspen (*Populus tremuloides*), black spruce (*Picea mariana*), red-fir (*Abies magnifica*), and western hemlock (*Tsuga heterophylla*).

In Europe, descriptions of breeding habitats have included conifer and deciduous forest types. In Scandinavia, studies report nests in artificial structures hung in pine (*Pinus* spp.), spruce (*Picea* spp.), and birch (*Betula* spp.) forest (Norberg 1964, Korpimaki 1981, Solheim 1983*a*). In France, "mountain pine" (*Pinus uncinata* and *P. sylvestris*) forest and old forest stands with beech (*Fagus* spp.) were used by owls located by Dejaifve *et al.* (1990:267) and Joneniaux and Durand (1987), respectively. In Germany, conifer forest with old trees were used for nesting (Konig 1969, Jorlitschka 1988).

Landscape Scale Habitat Use

Published accounts of boreal owl habitat use from North America do not directly address patterns of habitat use at the landscape scale. Studies have not compared density, productivity, frequency of breeding attempts, or other measures indexing habitat suitability among landscapes with different mixes of forest cover. Neither have studies directly examined patterns of foraging habitat use across landscapes.

Indirect evidence from Europe and North America does suggest that boreal owls differentiate among forest habitats at the landscape scale. Evidence presented below supports the general statement by Konig (1969) that "in certain parts of [the study] area the density of Tengmalm's owls was rather high, while in other ... forests no *Aegolius* existed."

Studies by Hayward *et al.* (1993) in Idaho provide some indirect information on landscape scale habitat use patterns. Nest sites and singing sites (considered representative breeding habitat) were not distributed randomly throughout the study area. Nesting was concentrated in mixed-conifer and aspen forests with no nesting in lodgepole pine forest and infrequent nesting in spruce-fir forests. In contrast, summer roost sites and foraging sites were concentrated in spruce-fir forests. Due to the natural segregation of forest types used for nesting and those used for roosting and foraging, habitat used for different ecological functions was segregated in the landscape. All the resources used by the owls were not provided by any single vegetation type leading to a complex pattern of habitat use. Spruce-fir forest in this study area had few potential nest cavities but small mammal sampling documented that this type supported the most abundant prey populations. In contrast, nest cavities were abundant in mixed conifer forest that supported few prey. Our discussion of microhabitat later provides some insight into landscape patterns through the examination of differences in habitat quality at the stand scale.

Korpimaki (1988a) provides a more direct examination of differences in habitat quality at the landscape scale from his studies in Finland. He rated territory quality of 104 nest sites based on frequency of use over 10 years. Territory occupancy varied from 0 to 9 nestings in 10 years. Poor territories (never occupied) occurred in extensive, uniform forests dominated by pine (*Pinus sylvestris*). These territories had little spruce forest and a high proportion of marshland. The proportion of pine forest decreased and the proportion of spruce forest (*Picea abies*) and agriculture land increased with increasing grade of territories (those with more frequent nesting). The conclusion that territories with spruce forest and agricultural land (in small patches) were the highest quality habitat was corroborated by evidence beyond the frequency of nesting. High quality sites supported breeding during prey crashes, mean clutch size was higher ($P < 0.05$), and number of fledglings was generally greater ($P < 0.05$) than other sites. Poor territories (occupied 0, 1, or 2 times) supported breeding only during peaks in the well-documented vole cycle (Korpimaki 1988a and references therein).

Korpimaki (1988a) explained this pattern based on variation in the abundance and stability of small mammal populations across the vegetation categories. Spring and fall densities of *Clethrionomys glareolus*, a major small mammal prey, were three times higher in spruce than pine forests (Korpimaki 1981). The mean densities of small birds, important alternative prey, were also higher in spruce forests than in pine (331 versus 260 pairs/km^2) (Korpimaki 1981). Furthermore, small mammal populations were more stable in the spruce forests than other types (Korpimaki 1988a). Korpimaki (1988a) pointed out that the pattern of habitat occupancy (virtually all habitats used in vole peaks but only "good" territories occupied during cyclic lows) fit the Fretwell and Lucas (1969) "ideal free" model of habitat use. Thus the size of the breeding population strongly influenced the pattern of habitat use at the landscape scale.

Home Range Scale Habitat Use

Home range size and movements within boreal owl home ranges was discussed in the Movements Within the Home Range section. Research in North America has not directly examined patterns of habitat use within individual home ranges except at the microhabitat scale which is discussed below. Patricia Hayward and I are currently analyzing data collected during the study reported in Hayward *et al.* (1993) at the home range scale but results are not yet available.

Sonerud *et al.* (1986) provide some data at this scale based on observations of a single radio-marked male owl followed on five nights. Because the results stem from observations of a single bird during a single week, general patterns cannot be inferred. The results are important, however, because they are the only data currently available and the pattern observed corroborates results reported throughout this section. The owl used an area of 205 ha during the 5 nights (based on 107 nocturnal locations using a minimum convex polygon estimator). Nightly use areas ranged from 40-78 ha and the maximum distance between foraging areas and the nest for each night varied from 1030-1320 m. While foraging, the owl favored old forest and avoided clear cuts and young plantations in spite of lower prey densities in the former (Sonerud *et al.* 1986:105).

Microhabitat

Nest Sites

A majority of nest site locations described in the literature have not resulted from efforts designed to survey a range of habitats to determine both habitats used and those not used. The results, then, cannot be interpreted as an indication of selection but rather to describe some subset of used habitats. In Alaska, eight nests located near Fairbanks occurred in closed-canopy deciduous or mixed forest; none occurred in uniform conifer forest (Meehan and Ritchie 1982). Of five nests in natural cavities, four occupied flicker holes and one a natural cavity. In Canada, Bondrup-Nielsen (1978) located 6 nests, all in aspen—3 in live trees, 3 in snags. Minnesota nests have been documented in old aspen clones intermixed with conifers (Eckert and Savaloja 1979 and Lane 1988). In a more extensive investigation involving 9 National Forests in Montana and Idaho, 76% of 49 boreal calling sites (recognized as potential breeding sites) occurred in mature and older forest stands (Hayward *et al.* 1993). The exceptions were locations in lodgepole pine (*Pinus contorta*) stands in drainages where lodgepole was the only forest type. The majority (88% of 49 observations) of owls were located in stands on subalpine-fir habitat types. Proportions for other habitat types included Engelmann spruce (3%), Douglas-fir (6%), and western hemlock (3%).

During 4 years of study in the wilderness of central Idaho, Hayward *et al.* (1993) documented nests in stands of old mixed-conifer (11), old Engelmann spruce (7), old aspen (5), and old Douglas-fir (5) forest. A nest box experiment in the same area suggested that owls avoided nesting in forests lacking the structural features of mature and old forest when alternate sites in old forest were available (Hayward *et al.* 1993). This study did examine available forest structure and compared used sites with a sample of available sites. Forest structure at nest sites differed from the random sample (101 sites) of available forest. Used sites occurred in more complex forest, with higher basal area, more large trees, and less understory development than available sites. The forest immediately around nest trees had an open structure. Density of trees 2.5 to 23-cm-dbh (diameter at breast height) in a 0.01-ha plot around the nest tree averaged 398 ± 162/ha (range 0-1,482). The density of trees at nests was three times lower than the average at winter roost sites. Nest sites averaged 57 (±16) trees/ha over 38 cm dbh, 17.8 (±3.1) m^2/ha basal area, and 30% (±4.3) overstory (> 8 m above ground) canopy cover (this is not total canopy cover but cover of upper canopy). Stands used for nesting supported an average of 9 (±6.0) snags per ha over 38 cm dbh. Size of the stand containing the nest ranged from 0.8-1.3 ha in aspen and 1.6-14 ha in conifer forest.

The range of sites used by boreal owls is quite broad despite the evidence that the species chooses particular forest structures when a variety of nest sites are available. In Idaho and Norway nest boxes in clearcuts have been used (see Sonerud 1989 and Hayward *et al.* 1992). The use of these sites, however, does not indicate that this is high quality habitat.

Nest Tree and Cavity Characteristics

Boreal owls are secondary cavity nesters and nest primarily in cavities excavated by pileated woodpecker (*Dryocopus pileatus*) and northern flicker (*Colaptes auratus*) in North America, and black woodpecker (*Dryocopus martius*) cavities or nest boxes in Europe. In central Idaho 18 of 19 nests were attributed to pileated woodpeckers; a northern flicker probably excavated the other. Cavity dimensions averaged 31 cm (±7.61; n=19, range 7-50) deep and 9 cm (±2.11 range 15-26) horizontally. Cavity entrances measured 102 mm (±12.41 range 64-150) high and 95 mm (±11.89 range 56-148) wide (Hayward *et al.* 1993).

Nests located in Idaho were generally in large trees or snags. Tree diameter at the cavity averaged 41 ± 5.21 cm (range 26-61 cm) and tree dbh averaged 64 ± 11.02 cm (range 33-112 cm). The smallest of these were all aspen and, therefore, still larger trees grew in the nest stand. Ten (of 19) nests occupied snags, including eight ponderosa pine, one aspen, and one Douglas-fir. Snag condition included 3 old branchless snags >11 m tall, 2 hard snags with sloughing bark and only large branches remaining, and 5 young snags with bark and complete limbs (Hayward *et al.* 1993).

In contrast with nest conditions in the United States, over 90% of some Scandinavian populations nest in artificial structures. This pattern is attributed to the scarcity of primary cavity nesters and paucity of large old trees (Korpimaki 1981, 1985).

Roost Sites

Three studies in North America addressed roosting habitat: one in Canada by Bondrup-Nielsen (1978), one in Colorado by Palmer (1986), and one in Idaho by Hayward and Garton (1984) and Hayward *et al.* (1993). These studies demonstrate that, unlike many forest owls, individual boreal owls roost at many different sites and choose roosts dispersed widely throughout their home range. The available evidence suggests that under some circumstances (see below) the owls select particular forest conditions for roosting but much of the time are unselective.

In Canada, 30% of 30 roosts located in spring and summer were in aspen or birch; the remainder were in conifers (Bondrup-Nielsen 1978). Based on comparison with paired random sites, Bondrup-Nielsen (1978) concluded that the owls were not selective in roost choice. In Colorado, 174 roosts located in winter and summer did not differ significantly between seasons although low statistical power may have led to this conclusion. It was not clear from the analysis whether forest structure at roosts differed from paired random sites. Combining seasons, roost sites averaged 14.7 trees/ha >39 cm dbh, 6 snags/ha, and 44% canopy cover (Palmer 1986). Average species composition of roost stands were 42% Engelmann spruce, 42% subalpine fir, and 6% lodgepole pine suggesting that the owls choose late successional stands for roosting.

In Idaho, based on habitat measurements from 430 roosts used by 24 radio-marked owls, habitat type (as defined by Steele *et al.* 1981) and forest structure differed between roosts used in winter and summer ($P < 0.001$, Hayward *et al.* 1993). Forest stands used for winter roosts averaged 58% canopy cover, 26 m^2/ha basal area, 1,620 trees/ha with 2.5-23 cm dbh, and 165 trees/ha over 23.1 cm dbh. Summer roosts averaged 63% canopy cover, 30 m^2/ha basal area, 2,618 trees/ha with 2.5-23 cm dbh, and 208 trees/ha over 23.1 cm dbh. Winter and summer roosts differed in all aspects of forest structure measured. All roosts (n=882) were in conifers; the owls were never observed roosting in cavities as is reported in Europe (Korpimaki 1981).

In the same study, roost sites were compared with paired random sites using a paired Hotelling's T^2 (189 winter, 241 summer sites). The results provided strong evidence for selection in summer, but results for winter also suggested selection (winter $P = 0.021$; summer $P < 0.0001$). Summer roosts occurred at cool microsites with higher canopy cover, higher basal area, and greater tree density than paired random sites (Hayward *et al.* 1993). When the authors compared temperature at the roost and in the nearest opening (both temperatures taken in the shade while the owl was roosting), roost sites where significantly cooler when ambient temperatures exceeded 4° C ($P < 0.001$). The difference in temperature increased with increasing ambient temperature and the owls gullar fluttered when temperatures were as mild as 20° C. The authors concluded that in summer, the owls chose cool microsites for roosting to avoid heat stress. In winter, the owls did not appear to be thermally stressed and used a wider variety of roost conditions.

Foraging Sites

The nocturnal foraging pattern of boreal owls has hampered attempts to study foraging habitat use (Hayward 1987). Therefore, the inferences concerning foraging habitat are largely based on indirect evidence.

Studies in Idaho (Hayward 1987, Hayward *et al.* 1993) based on roost locations (assumed to represent the end of a foraging bout) suggest that mature and older spruce-fir forests were important for foraging. Owls were observed successfully foraging in these forests and the locations of radio-tagged birds also indicated male owls were hunting in these forests while feeding young at nests located at lower elevations. Data on prey distribution and food habits further supported this contention (Hayward *et al.* 1993). Palmer's (1986) observations in Colorado also indicated older spruce-fir forest was used for hunting.

Studies in Norway also noted the importance of mature spruce forest for foraging (Sonerud 1986, Sonerud *et al.* 1986). Direct observations and diet indicated that during winter and summer the owls foraged primarily in older forest sites. In early spring, immediately following snowmelt, owls hunted clearcuts for a short period until lush vegetation developed. Owls favored mature forest during winter because snow conditions (uncrusted snow) facilitated access to prey. In summer, mature forest sites had less herbaceous cover than open sites that allowed greater access to prey. Following spring thaw, before herbaceous vegetation became dense, owls shifted to openings where densities of voles exceeded densities in forested stands.

In his 1987 address to the Northern Owl Symposium, Norberg (1987) highlighted the morphological adaptations of *Aegolius funereus* that facilitate foraging in forest stands at night. He noted the extreme skeletal asymmetry that facilitates ocular prey detection and localization under dark forest conditions. The short, broad, rounded wings of the boreal owl facilitate silent, agile flight in tight forest conditions. These morphological characteristics open up possibilities for exploiting habitat types unavailable to species lacking the traits. He also noted that the light wing-loading of boreal owls allow individuals to efficiently forage among habitat patches dispersed throughout their home ranges without expending excessive energy commuting between patches. Furthermore, the light wing-loading reduces the cost of foraging at distant sites and transporting prey back to the nest. This line of reasoning corroborates the limited observations that suggest that small, dispersed patches of high quality foraging habitat (high prey availability) are hunted by boreal owls who use large home ranges.

Morphology, of course, is not the only potential explanation for observed habitat use patterns. Predators and competitors may also influence foraging habitat use.

FOOD HABITS

Feeding

Foraging Movements

Boreal owls hunt primarily after dark except in northern regions without summer darkness (Norberg 1970, Mikkola 1983). In southern areas the species exhibits a biphasic rhythm with peaks of activity 2000-2200 h and 0200-0500 h (Mikkola 1983). Prey deliveries at monitored nests in Idaho (Hayward, G. D. and P. H. Hayward, unpubl. data) never occurred between sunrise and sunset; however, owls observed on daytime roosts (n = 882) occasionally hunted in daylight (13 observations in winter, 33 observations in summer) (Hayward *et al.* 1993). On 10 occasions the author observed owls capture prey from daytime roosts.

Boreal owls forage using sit and wait tactics (as opposed to pursuit). Four owls observed foraging on 13 occasions in Idaho moved through the forest in a zig-zag pattern, flying short distances ($\bar{x}$ = 25 ±8 m; n = 123) between perches. Perch heights averaged 4 (±0.6 n = 114) m, and owls watched for prey for less than 5 minutes on 75% of 150 perches (Hayward *et al.* 1993). Norberg (1970) recorded perch heights averaging 1.7 (0.5-8) m (n = 154) and flight distances of 17 (2-128) m, and Bye *et al.* (1992) recorded similar observations.

Prey Capture

Boreal owls observed in Idaho usually attacked prey within 10 m of their hunting perch (Hayward *et al.* 1993). In Norway, Bye *et al.* (1992) reported attack distances (direct distance between the owl and the prey) from 2.2 to 12.6 m. Successful attacks averaged 5.3 m (n = 10) and unsuccessful attacks 6.1 m (n = 10). Norberg (1970) describes pouncing and killing behaviors in detail. He (Norberg 1970, 1987) notes observations of boreal owls capturing prey either under the snow surface (plunge diving) or obscured by vegetation. The ability to locate prey aurally is attributed to the extreme asymmetry of the owl's skull (Norberg 1978, 1987), which permits localization of sounds in vertical, as well as horizontal, directions.

Diet

In North America, usual prey species are voles, particularly red-back voles (*Clethrionomys gapperi*), heather voles (*Phenacomys intermedius*), northern bog lemming (*Synaptomys borealis*), and *Microtus* spp.; mice, including deer mice (*Peromyscus* spp.) and jumping mice (*Zapus princeps*); shrews, (*Sorex* spp.); northern pocket gophers (*Thomomys talpoides*); squirrels, including northern flying squirrels (*Glaucomys sabrinus*) and chipmunks (*Tamias* spp.); birds, especially thrushes (*Catharus* spp.), warblers, dark-eyed junco (*Junco hyemalis*), red crossbill (*Loxia curvirostra*), American robin (*Turdus migratorius*), mountain chickadee (*Parus gambeli*), common redpoll (*Carduelis flammeus*), kinglets, and woodpeckers; and insects, especially crickets (Bondrup-Nielsen 1978, Palmer 1986, Hayward and Garton 1988, Hayward *et al.* 1993). Weasel (*Mustela* spp.), woodrat (*Neotoma cinerea*), juvenile snowshoe hare (*Lepus americanus*), and pica (*Ochotona princeps*) represent unusual prey.

Within North America, little difference in diet is apparent between studies in Alaska (T. Swem, pers. comm.), Canada (Bondrup-Nielsen 1978), and the Rocky Mountains (Palmer 1986, Hayward *et al.* 1993). In each locale, red-backed voles (*Clethrionomys* spp.) and *Microtus* spp. were dominant prey.

Boreal owl food habits have been studied more thoroughly in Europe; for a summary see Cramp (1977). The results are surprisingly similar to North America. *Clethrionomys* sp. and *Microtus* sp. dominate the diet in most cases. Results suggest, however, that in Scandinavia, boreal owls consume more voles associated with open habitats than are recorded in the Rocky Mountains of North America. This could be due to differences in habitat characteristics in particular study areas, in the owls foraging behavior, in predation risks, or in competitive interactions.

Quantitative Analysis

Most samples of boreal owl prey in North America are small. Bondrup-Nielsen (1978) reported 58 individual prey from his two study sites in Canada, Palmer (1986) recorded 72 prey found in 4 years in Colorado, and Hayward *et al.* (1993) reported 914 prey identified from 4 years in Idaho (table 1). These data are not sufficient to make in-depth comparisons between geographic areas, examine functional or numeric responses to changes in prey populations, or predict changes in diet or owl demography in response to changes in prey populations. As a group, however, these investigations cover a broad geographic area and provide a sound basis for generalizations concerning boreal owl diet in North America.

The breadth of prey represented in the boreal owl diet contrasts with the narrow range of prey taken frequently. The data suggest that the boreal owls are vole specialists under most circumstances. *Microtus* and *Clethrionomys* constituted 45 and 31% (by frequency) of prey identified from the two study sites in Canada (Bondrup-Nielsen 1978). In Colorado, *Clethrionomys* and *Microtus* were 54 and 25% of the diet (Palmer 1986). In Idaho, red-backed voles were the most frequent prey in summer (35% by frequency) and winter (49% by frequency) (32 owls over 4 years). In terms of prey biomass, red-backed voles accounted for 37% of the annual prey. Northern pocket gophers (26%) and *Microtus* spp. (11%) were the only other species accounting for over 10% of the annual prey biomass (Hayward *et al.* 1993). Northern flying squirrels were captured by female owls in winter and accounted for 45% of winter prey biomass. Overall, small mammals accounted for 79% of prey (95% of estimated biomass).

Table 1. – Diet of boreal owls in Idaho (Hayward et al. 1993), Colorado (Palmer 1986), and Canada (Bondrup-Nielsen 1978) based on pellets and prey identified from nests.

Prey	Idaho		Colorado	Canada
	% of prey items	Biomass[1] (%)	% of prey items	% of prey items
Mammals				
Red-backed vole (*Clethrionomys spp.*)	36	37	54.2	31
Northern pocket gopher (*Thomomys talpoides*)	10	26		
Unidentified shrews (*Sorex* spp.)	11	3	5.5	2
Unidentified voles (*Microtus* spp.)	9	11	25	45
Deer mouse (*Peromyscus maniculatus*)	6	5	1.4	10
Heather vole (*Phenacomys intermedius*)	4	3		
Northern flying squirrel (*Glaucomys sabrinus*)	1	7		
Chipmunk (*Tamias spp.*)	2	3	1.4	
Jummping mouse (*Zapus princeps*)	2	1		
Woodland jumping mouse (*Napaeozapus insignis*)				3
Pica (*Ochotona princeps*)	tr[2]	tr		
Woodrat (*Neotoma cinerea*)	tr	tr		
Unidentified weasel (*Mustela* spp.)	tr	tr		
Water vole (*Microtus richardsoni*)	tr	tr		
Birds	5	3	7	5
Insects	13	1		
Total count	914	-	72	58

[1] *Biomass calculated using values from Hayward et al. (1993).*

[2] *tr indicates <1%.*

Quantitative results from Europe demonstrate a similar pattern. *Microtus* and *Clethrionomys* dominate the diet in most locales but a more varied diet is evident in more southern populations (Korpimaki 1986b). In an 8 year study documenting contents of 67 owl nests in central Finland, Jäderholm (1987) found *Clethrionomys* spp. and *Microtus agrestis* together accounted for 80% of Tengmalm's owl prey biomass. Shrews were the next most important prey, accounting for 18% of individual prey and 8% of the biomass. Korpimaki's (1986c, 1988b, Korpimaki and Norrdahl 1989) work in western Finland reveals a similar pattern. *Microtus* spp. were the most abundant prey in nests (45% by frequency), followed by *Clethrionomys* spp. (32%), shrews (15%), and birds (5%). Values for prey identified from pellets differed in that shrews dominated the sample (33% by frequency), followed by *Microtus* spp. (27%), *Clethrionomys* spp. (24%), and birds (12%).

In Czechoslovakia, mice (especially *Apodemus* spp. and *Muscardinus avellanarius,* together 18% of prey biomass) were more important in the diet than in more northern populations and the diet included more species of mammals (24 species) (Kloubec and Vačik 1990). *Microtus* spp., *Sorex* spp., and *Clethrionomys* spp. were still major prey, together accounting for 39% of prey biomass. This study summarized information from 11 sites distributed throughout Czechoslovakia. Schelper (1989) summarized information from another southern population, in Germany. *Apodemus* spp. dominated the prey (39%) followed by *Microtus* spp. (25%), *Clethrionomys* spp. (14%), *Sorex* spp. (12%), and birds (6%).

Marti *et al.* (1993) summarized results of 20 papers from Europe and 4 from North America and found the geometric mean weight of prey for 4 regions in Europe, moving northward, to be 14.7, 17.6, 15.0, and 19.9 g; and 19.2 and 22.2 g for the Rocky Mountains and Alaska, respectively. In Europe, food-niche breadth declined from southern to northern populations while in North America food-niche expanded in northern populations (Marti *et al.* 1993).

Seasonal Variation

Boreal owl diets differ from winter to summer due to the natural variation in availability of prey due to snow cover and the hibernation of some small mammal prey. In Idaho, northern pocket gophers (one of the most frequent summer prey), western jumping mice, and yellow-pine chipmunks were all unavailable in winter. The owls relied on southern red-backed voles for nearly 50% of winter prey. Flying squirrels were captured far more frequently in winter than summer. Of 12 recorded flying squirrel prey, 11 were captured during winter, 10 of these by females. The squirrels represented 45% of prey biomass recorded for female owls during winter, indicating the importance of these prey when other prey are less available. During summer, southern red-backed voles continued to be the most frequent prey and accounted for 31% of prey individuals. The owl summer diet was diverse compared to winter with the addition of chipmunks, jumping mice, and crickets. The relative importance of birds in the diet did not change between seasons (5% by frequency).

In Finland, the owl's diet shows a marked seasonal pattern that varies depending on the stage of the multi-year vole cycle (tables 2 and 3). This study covered the period January-June from 1973-1985 and included four peak vole phases. In all years, birds were important from January through mid-March (24-24% of diet by frequency) and in late May and June (8 - 27%) while shrews increased in the diet as they matured in late April. In good vole years, *Microtus* spp. were taken most frequently in late March and April (74-84% of diet) and formed 35-49% of the diet in other months. During the high vole years, *Clethrionomys* captures increased in late April as *Microtus* became less important. In poor vole years the frequency of *Clethrionomys* in the diet increased earlier in March, when they accounted for 51% of the diet; *Clethrionomys* captures remained high through May.

Yearly Variation

In Idaho, *Clethrionomys gapperi* varied from 26 to 45% of the annual diet (by frequency) over 4 years (Hayward *et al.* 1993). Deer mice, pocket gophers, and heather voles (*Phenacomys intermedius*) increased in years when *Clethrionomys* was less frequent. Years with a low proportion of *Clethrionomys* were poor breeding years for the owl. The frequency of *Microtus* spp. remained relatively constant during this study and averaged 11%. The frequency of shrews and birds also remained relatively constant.

In Finland, the owl's diet varied sharply among years in response to the well documented (e.g., Hansson and Henttonen 1985) vole cycle (Korpimaki 1988b). The proportion of *Microtus* in diet correlated positively (Spearman rank correlation: $r_s = 0.86$, $P < 0.001$) with the abundance of these voles in spring trapping samples and varied from 6 to 71% of the diet (Korpimaki 1988b). Proportions of shrews and birds in the diet varied inversely with the numbers of *Microtus.* The proportion of *Clethrionomys* in the diet correlated positively with the proportion of *Microtus* ($r_s = 0.46$, $P < 0.10$) and varied from 3 to 45% of prey.

Table 2.—The seasonal changes in the food composition (as percentages by number) of the Tengmalm's owl during the first half of the year in peak vole years (pooled data from 1973, 1977, 1982, and 1985). The statistical significance of the differences between consecutive time periods was examined using chi-square tests. From Korpimaki (1986c).

	Time periods							Total
Prey groups	1 Jan.-15 March	16-31 March	1-15 April	16-30 April	1-15 May	16-31 May	1-30 June	
Shrews	17.3	2.4	4.6	15.9	10.5	15.8	20.0	11.8
Water vole	-	-	-	-	0.2	-	-	0.1
Bank vole	17.3	11.2	15.5	34.0	34.1	27.2	20.0	27.7
Microtus spp.	34.7	84.0	74.0	46.6	49.0	45.1	48.0	52.4
Murids	6.7	2.4	5.5	2.8	5.6	-	-	4.1
Birds, adults	24.0	-	0.5	0.7	0.7	12.0	8.0	3.9
nestlings and young		-	-	-	-	-	4.0	0.1
total birds	24.0	-	0.5	0.7	0.7	12.0	12.0	4.0
No. of prey items	150	125	219	427	602	184	25	1732
Diet width	9.04	2.49	3.12	4.02	3.98	5.15	4.25	
χ^2	45.80	26.38	65.39	13.64	54.80	5.49		
df	4	4	5	5	5	4		
P<	0.001	0.001	0.001	0.05	0.001	ns		

Table 3.—The seasonal changes in the food composition (as percentages by number) of the Tengmalm's owl during the first half of the year when vole populations were not at a peak (pooled data from 1974-76, 1978-81, and 1983-84). Statistical analysis same as in table 1. From Korpimaki (1986c).

Prey groups	1 Jan.-15 March	16 March-15 April	16-30 April	1-15 May	16-31 May	1-30 June	Total
Shrews	11.9	17.1	26.0	24.0	36.8	45.8	25.2
Red squirrel	-	0.3	-	-	-	-	0.0
Water vole	-	0.3	0.3	0.2	-	0.6	0.2
Bank vole	9.9	51.6	46.5	42.3	19.6	14.5	35.4
Microtus spp.	42.9	25.1	23.1	18.8	7.7	4.2	21.8
Murids	1.4	4.8	3.2	3.6	4.2	-	3.2
Birds, adults	34.0	0.9	1.0	11.2	26.7	18.7	12.3
nestlings and young	-	-	-	-	4.9	16.3	1.9
total birds	34.0	0.9	1.0	11.2	31.6	34.9	14.2
No. of prey items	294	351	624	421	285	166	2141
Diet width	8.18	3.05	3.24	3.95	5.20	4.13	
χ^2	222.10	9.02	50.47	84.56	6.28		
df	5	5	5	5	4		
P <	0.001	0.10	0.001	0.001	ns		

Energetics

During the nestling period, young owls in Finland consume an average of 21 g per bird per day and captive adults 65 g/day (Korpimaki 1981). Jäderholm (1987) calculated that during nesting, young boreal owls are provided about 650 g of prey, which is similar to Korpimaki's estimate for a 29-30 day nestling period. Prey biomass provided for each nestling changed little for broods from 2-7 nestlings but was higher when only one nestling was present (about 1,600 g) (Jäderholm 1987:Fig 3).

As an indirect measure of prey consumed in Idaho, Patricia Hayward and I monitored four nests with mechanical event recorders triggered by a perch mounted at the cavity. These records suggested that the female left the nest once each night during incubation (for gut clearing) and usually twice after the young had hatched. Assuming that all records other than female gut clearing were prey deliveries, deliveries averaged 3.5 (±0.33; n = 84 nights of records; range 0-9) during incubation and 5.0 (±0.61; n = 76 nights; range 0-12) during brooding. Clutches at the four nests were 2, 2, 3, and 3; each fledged two young. Korpimaki (1981) estimated 9.8 deliveries/night during brooding period and 8.0 after the female left the nest.

Temperature Regulation

No data have been published on thermal neutral zone, basal metabolic rate, and metabolism while active. Winter and summer roost characteristics indicate boreal owls in central Idaho were not stressed by winter conditions but chose roosts to reduce summer heat stress (Hayward *et al.* 1993). Gullar fluttering was noted only in summer but occurred when temperatures at roosts were as mild as 18°C and 23°C, suggesting the owls are easily heat stressed.

Food Caches

Immediately prior to nesting (1-2 weeks) and during nesting, prey are cached in the nest cavities (Norberg 1987). In Finland (13-year study, Korpimaki 1987a), the size of nest caches was related to phase of the vole cycle. During peak phase, caches averaged 6.9 items/nest weighing 89.3 g; in low phase, 1.5 items/nest weighing 19.6 g. *Clethrionomys glareolus* were the most common cached prey. Prey are also cached at roosts. In Idaho, owls were observed retrieving cached prey or caches were observed near roosting owls at 17% of summer and 4% of winter roost locations (n = 882).

ECOLOGY OF PRINCIPAL PREY

Forest dwelling small mammals dominate boreal owl diets in most regions (see previous Food Habits section). In North America, important species include red-backed voles, flying squirrels, deer mice, shrews, and pocket gophers. Microtine voles are also important throughout the species' range and seem to increase in importance in more northern latitudes. In this section I will briefly review the ecology of several prey species that occurred frequently in boreal owl diets in the United States: red-backed vole, deer mouse, flying squirrel, and other voles (Palmer 1986, Hayward *et al.* 1993). This review is intended only to give the reader a preliminary understanding of small mammal prey as a background for the remainder of the conservation assessment. I concentrate on habitat use and food habits of the selected prey species.

Red-Backed Vole

The genus *Clethrionomys*, or red-backed voles, occurs throughout the range of boreal owls and represents an important prey genus in all populations studied. These 20-30 g voles are active year-round and their circadian activity pattern includes periods of foraging throughout the 24 hour cycle (Stebbins 1984). Red-backed voles do not form colonies but nest singly or in family groups in natural cavities, abandoned holes, or nests of other small mammals near the ground surface. During winter they spend most of their time at the snow-ground interface.

The genus occurs almost exclusively in forest habitats although Whitney and Feist (1984) describe populations occurring in grassland habitats in Alaska. Merritt (1981:4) characterizes their habitat as "chiefly mesic habitats in coniferous, deciduous, and mixed forests with abundant litter of stumps, rotting logs, and exposed roots." In Idaho, red-backed voles were most abundant in mature and older spruce-fir forest where they were the most abundant small mammal (Hayward *et al.* 1993). The relationship between forest successional stage and red-backed vole abundance appears to vary geographically. In the western and northeastern portions of North America, red-backed voles are most abundant in mesic, mature conifer forest, particularly spruce-fir forests (Brown 1967, Scrivner and Smith 1984, Millar *et al.* 1985, Raphael 1988). In these regions red-backed voles decline sharply after clearcutting (Campbell and Clark 1980, Ramirez and Hornocker 1981, Halvorson 1982, Martell 1983a, Medin 1986). Martell (1983b) showed that the loss of red-backed voles from clearcuts may lag 2-3 years, but the voles were still rare after 13 years. In contrast to the radical population changes observed after clearcutting, red-backed vole populations remained abundant after patch cutting (3 acre clearcuts) and selection harvest in several locales (Campbell and Clark 1980, Ramirez and Hornocker 1981, Scott *et al.* 1982, Martell 1983b). Wywialowski (1985), using voles caught in Utah and placed in an artificial experimental arena, showed that the voles preferred areas with greater overstory cover.

Observations in the central and southeastern portion of the species' range suggest a more varied pattern of habitat use (see references in Merritt 1981). In Minnesota, Michigan, Maine, and Nova Scotia red-backed voles were common, or in some cases, most abundant in clearcut sites or sapling stages following cutting (Swan *et al.* 1984, Probst and Rakstad 1987, Clough 1987). The pattern seems to be associated with moist deciduous forests where sites remain mesic after deforestation.

Food habits of red-backed voles fit their association with forest habitats. Hypogeous ectomycorrhizal and surface fruiting fungi are dominant foods in many regions (see references in Merritt 1981 and Ure and Maser 1982). These fungi are associated with tree roots, rotting logs, and litter on the forest floor in mesic forest stands. Fruticose lichen, particularly the arboreal *Bryoria* spp., are important food across the species' range, especially in winter (Martell and Macaulay 1981, Ure and Maser 1982). In Ontario, lichen and fungi together formed 80-89% of the diet across four study sites (Martell and Macaulay 1981). Ure and Maser (1982) noted that lichen is especially important to voles at higher elevations where the fruiting season for fungi is brief. Other foods include green vegetation (e.g. leaves of *Vaccinium* spp.), seeds, berries, and some insects in summer and autumn.

Non-Forest Voles

Voles in the genus *Microtus* are consumed by boreal owls throughout the owl's range in North America (table 1). Predation on *Microtus* is especially significant because these 25-40 g rodents occur most commonly in nonforested habitats. *Microtus* are active year round; they nest on the ground surface in grass nests and live at the snow-ground interface during winter. *Microtus* feed almost exclusively on leafy vegetation and the inner bark of small trees and shrubs (Vaughan 1974). Numerous studies demonstrate that, aside from dispersing individuals, these voles do not occur in forest stands (see references in Johnson and Johnson 1983). Populations will occur in small (several acres) grassland or shrub openings in otherwise forested landscapes.

Deer Mouse

Deer mice (*Peromyscus* spp.) are eaten frequently by boreal owls throughout North America but never are the dominant prey. These 20-30 g mice are highly nocturnal (Stebbins 1984) and active year-round. Deer mice are partially arboreal (Getz and Ginsberg 1968). Their diet is omnivorous, being dominated by seeds (Martell and Macaulay 1981). Compared with other small rodents their population densities are relatively stable (Van Horne 1982).

Deer mice occupy both forested and open habitats from desert to temperate rain forest. Within the geographic range and life zone used by boreal owls, deer mice occupy most habitats. In the mountains of Colorado deer mice were captured in a wider variety of montane habitats than other rodents (Williams 1955). In Idaho, deer mice were captured in spruce-fir forests, Douglas-fir forests, lodgepole pine forests, ponderosa pine forests, and sagebrush-bunchgrass openings. Wet meadows were the only habitats where the mice did not occur (Hayward *et al.* 1993). In most locales these mice increase or remain equally abundant with disturbance or deforestation (Campbell and Clark 1980, Ramirez and Hornocker 1981, Van Horne 1981, Halvorson 1982, Martell 1983a, Buckner and Shure 1985, Medin 1986). Deer mice tend to be more abundant than red-backed voles in drier, rockier, forested habitats that are dominated by pines rather than spruce or firs (Millar *et al.* 1985, Raphael 1988).

Northern Flying Squirrel

Northern flying squirrels have been identified as important prey in only a single study in North America (Hayward *et al.* 1993); however, in this study, northern flying squirrels represented 45% of the prey biomass for female owls during winter. These ~140 g squirrels are highly nocturnal and active year-round (Wells-Gosling and Heaney 1984). Their diet is poorly understood, but fungi and lichens are thought to be the major foods in areas without substantial mast crops. Other foods include buds, catkins, fruits, tree sap, and insects (Wells-Gosling and Heaney 1984). Lichen is also important to the squirrels as a winter nesting material (Hayward and Rosentretter 1994). As with diet, habitat relationships are poorly understood. Across their extensive range, northern flying squirrels are found in conifer, hardwood, and mixed forests (Wells-Gosling and Heaney 1984). Squirrel densities in Douglas-fir forests of the Oregon Cascade Range were not correlated with habitat characteristics (Rosenberg and Anthony 1992). The only substantial published study linking flying squirrels with older forest has been questioned (see Rosenberg *et al.* in press concerning Carey *et al.* 1992). It is therefore interesting that mature and older forests provide necessary foods such as fungi, lichen, and large mast crops that do not occur commonly in younger forests.

BREEDING BIOLOGY

Phenology of Courtship and Breeding

Data on the phenology of courtship and breeding for populations in North America stem from a handful of studies that were not designed to address this topic per se.

Courtship

In Colorado, singing began by mid-February, early March, late March, and mid-April in 4 years. Courtship singing by individual owls lasted up to 102 days with an average of 26 (4-59, n = 4) days for successful males (Palmer 1986). In Idaho, during 3 good breeding years, males were heard on 27 January, 30 January, and 16 February (each within 2 days of beginning field-work). During a poor breeding year, calling was first heard 9 February, 16 days after field-work was begun. At a similar latitude in Europe (Germany), singing begins around the first of January (Schelper 1989). In Sweden, Carlsson (1991) found individual males began singing on some successful territories over 2 months after the first males began singing. Late singers tended to be younger and may have immigrated into the area. Daylength, prey availability, and nightly minimum temperatures (Bondrup-Nielsen 1978, Korpimaki 1981) are purported to determine onset of the courtship period. The variation observed in courtship activity suggests that prey availability, weather conditions, and resident status interact to modify the influence of daylength, which likely acts as the primary factor.

During courtship, displays are limited to flights by the male between perches near the female and a potential nest cavity, accompanied by vocalization of the "prolonged song" or extended singing from the nest cavity. Courtship feeding may begin 1-3 months prior to nesting. The female occupies the nest up to 19 days and usually 1 week prior to laying (Hayward 1989) and is fed nightly by the male.

Nest Occupancy

Courting owls rendezvous nightly at the potential nest site toward the end of the courtship period where the male displays and presents food. Late in the courtship season, prior to laying, the female occupies the cavity day and night for 1-19 (usually ~6) days where she is fed by her mate. Over 4 years, known first day of occupancy ranged from 13-30 April for seven owls in Idaho (Hayward 1989).

Egg Laying

In Minnesota, clutches were initiated by 30 March and 12 April (Lane 1988). In Colorado, laying dates were estimated from 17 April to 1 June with half the known nests being initiated by 10 May (R. Ryder, Colo. State Univ., Ft Collins Co). In the central Idaho wilderness, initiation dates extended from 12 April to 24 May with half the nests begun by 1 May (5 years, 13 nests; Hayward 1989). Near Anchorage, Alaska, nests located in nest boxes were initiated from 27 March to 5 May with a median date of 10 April (T. Swem, U.S. Fish & Wildl. Serv., Fairbanks, AK). A population in Germany began laying as early as February in good vole years but more often in April (Schelper 1989). Finnish nests were initiated from 8 March to 15 May with over half begun before 10 April; nests were initiated earlier in good prey years (12 years; Korpimaki 1981). Studies in Norway suggest that second clutches of biandrous females were laid 50-64 days following the first (Solheim 1983a).

Fledging

The nestling period extends from 28-36 days (average 31.7) (Korpimaki 1981). First-hatched young stay in the nest an average of 2.3 days longer than the last hatched because adults feed young in the nest less when siblings beg outside the nest. In Idaho, the older nestlings left 27-32 days after hatching (Hayward 1989).

Mating System and Sex Ratio

The boreal owl's mating system has not been studied thoroughly in North America. Therefore, the differences in mating systems described for the New and Old Worlds may be artifacts of research emphasis rather than true biological differences. Boreal owls are considered monogamous for the duration of a breeding season in North America. The pair bond lasts only a single season; most individuals nest with a new mate each year. Extra-pair copulations have not been observed. In Europe, polygymy has been observed in most regions and is recognized as an important aspect of the species' mating system (Solheim 1983a, Schelper 1989, Korpimaki 1991). In Scandinavia and Germany, bigyny (male mated to two females), trigyny (male mated to three females), and biandry (female mated to two males) coincide with vole peaks (Solheim 1983a, Schelper 1989, Korpimaki 1991). An estimated 10-67% of males are polygynous in good years but polygamy was never recorded in poor years (Carlsson *et al.* 1987). In two good vole years, bigynous males reared an average of 7.8 and 9.5 fledglings compared to 4.2 and 5.1 for monogamous males. Males achieve polygamy through polyterritorial behavior, advertising at multiple (up to 5) cavities within the home range (Carlsson 1991). Primary and secondary females were separated by an average of 1,050 m (median, n = 17) (Korpimaki 1991). Bigynous males feed primary and secondary females equally during laying but favor primary females during the brooding period (Carlsson *et al.* 1987). Secondary females produce fewer young than their primary counterparts (2.8 vs. 5.1 and 3.3 vs. 6.2 in two years, Carlsson *et al.* 1987). Biandrous females (multiple broods with the same male not recorded) cease caring for the first brood about three weeks after the young hatch (normal end of brooding) and may begin a second clutch with a new mate prior to departure of the first brood (Schelper 1989). The interval between clutches ranges from 50-64 days, the distance between nest sites ranges 0.5-10 km, and there is no significant difference in the number of eggs or mortality of young for biandrous vs. monogamous females (Solheim 1983a).

The sex ratio of adult boreal owls has not been estimated in North America. In Europe, where long-term studies of population ecology are more common, sex ratio of breeding individuals was estimated as 8:10, 0:10, 5:10, and 4.3:10 (females to males) during 4 years in northern Sweden using autumn playback and mist-net trapping (Carlsson 1991). These estimates may be biased, however, by sexual differences in response to playback of primary song.

Nest Site

Nest

Boreal owls nest exclusively in secondary tree cavities—in North America primarily pileated woodpecker, common flicker, or natural tree cavities or in artificial nest boxes. Boreal owl populations are likely limited in portions of their range by availability of cavities.

Maintenance or Re-Use of Nests

In Colorado, 2 (of 6 observed) nests were used 2 years in succession (R. Ryder, Colo. State Univ., Ft. Collins, CO). Both instances were in natural cavities in lodgepole pine. The owls were not captured so whether the same or different birds used the nests was unknown. Natural nest cavities were never used 2 years in succession in Idaho and rarely used again by the same individual (Hayward and Hayward 1993). Nest cavities may be reused by different individuals but generally after a "rest" period of more than one year. Nest boxes in Idaho have been occupied in successive years but only by new individuals and after the box was cleaned. In Europe, where cavities are more limited, repeated use of nest boxes is more frequent (Sonerud 1985, Korpimaki 1988a).

Nesting

Egg Laying and Care of Young

Clutches in Idaho were begun 1-19 (usually about 6) days after the female occupied the cavity. In Finland, eggs were laid at intervals of 48 hours but varied from 0.3 to 0.7 eggs per day (Korpimaki 1981). The female does all incubation. There are no reports of egg dumping.

Brooding is performed exclusively by the female beginning immediately after hatching and lasting until the oldest nestling reaches 20-24 days. During the first 3 weeks the male brings all food to the nest for the female and young and the female feeds the young. The male continues to provide for the young throughout the nestling stage and the female supplies food to nestlings after ceasing brooding at some nests. After fledging the young are dependent on the adults for food for over a month.

Growth and Development

Variation in clutch size is reported under Demography, below. During the nestling period, which lasts 30 days for most young, nestlings gain about 5.2 g per day with the greatest absolute gains from 8-13 days (10 g/day). Young reach adult mass by 14-17 days; at 30 days nestlings average 156 (±21.3; n = 5; range 130-174 g) (Hayward and Hayward 1993).

DEMOGRAPHY

Life History Characteristics

Age of First Reproduction

Banding records in the northern Rocky Mountains indicate that boreal owls breed the year after hatching. More intensive study in Finland indicates that, except in years of reduced food availability, both sexes can breed the year after hatching, but a larger proportion of females than males breed their first year (Korpimaki 1988c). Over an 8-year period, 16% of first-year males and 65% of second-year males bred (Korpimaki 1992). Both sexes are capable of breeding each year, but prey availability determines individual status year-to-year (Korpimaki 1988c). Second broods are not reported in North America;

see the Mating System and Sex Ratio section under Breeding Biology.

Clutch

Variation in clutch size is one of the most studied aspects of the species' biology, particularly in Europe. These studies have established that the number of eggs laid by boreal owls varies in relation to environmental conditions, particularly prey availability. Clutch size varies among geographic regions, among years, and among individuals within years. In both Europe and North America, northern populations that prey on fluctuating vole populations display the greatest variation in clutch size and have the largest potential clutches (Bondrup-Nielsen 1978, Korpimaki 1986a, Hayward *et al.* 1993) (table 4). Over 17 years, mean clutch size varied from 4.3 to 6.7 in western Finland (Korpimaki and Hakkarainen 1991) and over 4 years in Idaho from 2.5 to 3.5 (Hayward *et al.* 1993). The dramatic variation in clutch size within and among years is further shown in table 4.

Korpimaki (1987b, 1989) and Hornfeldt and Eklund (1990), using experimental and observational studies, demonstrated the direct link between vole abundance and clutch size in Finland. Further support for this pattern comes from observational studies in Norway (Lofgren *et al.* 1986), Sweden (Sonerud 1988), Germany (Schelper 1989), France (Joneniaux and Durand 1987), and Idaho (Hayward *et al.* 1993). Each of these studies reported larger clutches in years when indices of small mammal abundance, based on snap or live-trapping, were high.

To further demonstrate the variation in clutch size that has been observed I provide additional summary statistics from a sample of studies. Clutch size for separate populations in Idaho averaged 3.25 (± 0.42 SD, n = 11, range = 2-4) and 3.57 (± 0.34 SD, n = 31, range = 2-5) (Hayward *et al.* 1993, Hayward, G. D. and P. H. Hayward, unpubl. data) (table 4). From a similar latitude in Europe (Germany), Schelper (1989) reported clutches of 3-4 eggs with larger clutches in years when voles dominated the diet. An earlier study in Germany reported 34 nests averaged 3.8 eggs (Konig 1969). In Finland, pooling results from 2 areas over 12 years shows clutches averaging 5.6 (± 0.13 SD, n = 412, range = 1-10) (Korpimaki 1987b).

Fledging Success and Population Productivity

Patterns of fledging success reported for boreal owls in Europe and North America reflect the patterns reported for clutch size. Experimental and observational results strongly support the contention that prey availability influences fledging success and overall population productivity (e.g., Korpimaki 1987b, 1989, Hornfeldt and Eklund 1990). Therefore this section will not repeat results that simply duplicate those reported but will note important differences. Representative fledging rates include: 2.3 (± 0.54 SD, n = 6, range = 2-3) fledglings/successful nest in Idaho (Hayward 1989); 3.4 young/nest in Germany (Konig 1969); and 3.2 fledglings/nest and 3.9 fledglings/successful nest over a 14-year period in Finland (Korpimaki 1987b).

Korpimaki's (1988d) studies in western Finland suggest that fledging success is more strongly influenced by prey availability during decrease and low phases of the vole cycle. Clutch size, in contrast, is more sensitive to prey availability during the increase phase.

Korpimaki (1988c) has also shown that breeding performance in Tengmalm's owl is dependent on the experience of both members of the breeding pair; pairs of older birds experience the highest productivity. These data suggest that annual reproductive success increases over time, within individuals.

Fledging success is usually reported as the mean

Table 4.—Summary of reproductive statistics for boreal owls from sites in North America and Europe.

Location[1]	Median laying date	Range laying dates	Mean clutch size	Mean no. young fledged[2]
Colorado	10 May	17 Apr-1 Jun		
Idaho	1 May	12 Apr-24 May	3.25	2.3
Minnesota		30 Mar-12 Apr		
Alaska	10 Apr	27 Mar-5 May		
Finland	3 Apr	23 Feb-7 Jun	5.6	3.9
Germany			3.8	3.4

[1] Sources of information: Colorado (Palmer 1986), Idaho (Hayward 1989), Minnesota (Lane 1988), Alaska (T. Swem, U.S. Fish & Wildl. Serv., Fairbanks, AK), Finland (Korpimaki 1987b), and Germany (Konig 1969).
[2] Calculated only for successful nests.

number of fledglings per successful nest. Productivity, however, is strongly influenced by nesting success (rate of unsuccessful nests). In some years, the small proportion of the population breeding has a greater impact on productivity than reduced clutch size or fledging success.

In central Idaho, 10 of 16 nests produced no young in a study where all but one nest was a natural cavity (Hayward 1989). In Norway, during a 13-year study employing nest boxes, 48% of 101 clutches were lost to predation (Sonerud 1985). A nest box study in Finland reported 85% of eggs hatched and 53% of the eggs laid (n = 890) produced a fledgling, averaging 3.2 fledglings/nest and 3.9 fledglings/successful nest over 14-year period (Korpimaki 1987b).

The influence of owl density on reproduction has not been directly addressed in the literature. The patterns described above do not suggest strong inverse density dependent reproduction. Clutch sizes and fledging rates tend to be highest in years when prey is abundant and the greatest number of owls are breeding. These results, however, do not preclude the potential for density dependent limitation of population growth. Perhaps density dependence is determined by the number of adult owls breeding per 1000 voles per km^2. Because prey availability is a primary factor influencing reproduction, and boreal owls consume up to 17% of available *Microtus* (Korpimaki and Norrdahl 1989), a feedback loop is available to self-limit population growth to some degree. As discussed below, however, territoriality is not likely to be a mechanism for density dependent self-limitation.

Lifetime Reproductive Success

Lifetime reproductive success is difficult to study in any mobile vertebrate. No studies in North America have examined this topic. Based on 11 years of data, lifetime reproduction (LR) of 141 males in Finland varied from 0-26 fledglings (mean 5.2); 21% of males reared 50% of all fledglings (Korpimaki 1992). Among males hatched in a given year, 5% produced 50% of fledglings in the next generation. Offspring survival from egg to fledgling, lifespan of individual, clutch size of nests, and phase of the vole cycle at which an individual entered the populations were important components of LR for individual males. Offspring survival (as represented by the number of fledglings per nest) varied from 0 to 7. Most males breed for only a single season but the number of seasons ranged from 1-7 years ($\bar{x} = 1.5$). Clutch sizes varied from 2-8. Finally, the temporal variation in habitat quality due to fluctuating vole abundance was the most important environmental determinant of LR. Males entering the population in the low and increase phases of the cycle had larger LR than those entering in decrease or peak (individuals raised in the low and increase phases had better food conditions in their first 1-2 years of *breeding*). The extreme variation among individuals in lifetime reproductive success is expected because prey availability varies greatly among years and within years among breeding sites in Finland. Other vertebrates exhibit similar patterns (e.g., Clutton-Brock *et al.* 1982, Grant and Grant 1989).

Proportion of Population Breeding

Sound estimates of the number of non breeding individuals are not available. Indirect evidence from North America and Europe, however, demonstrates extreme yearly variation in breeding attempts; e.g., in Sweden, nest box occupancy in one area varied from 0.8% to 40.2% in 1980-81, and 39.4%, 0.8%, and 23.8% in 1982-84 (Lofgren *et al.* 1986). In Idaho, the number of calling males heard per kilometer surveyed varied from 0.02 to 0.24 from 1984 to 1987 and some radio-marked individuals did not breed even in good breeding years (Hayward 1989).

The most direct estimates come from Korpimaki's studies based on 10 years of monitoring his smaller study area (100 km^2, Korpimaki and Norrdahl 1989). The number of non breeding males (based on singing males who did not nest) varied from 0 to 66% of the population and averaged 47%.

Survivorship

In Idaho, adult annual survival estimated from 25 radio-marked birds was 46% (95% confidence interval 23-91%) (Hayward *et al.* 1993). In Finland, based on 281 banding recoveries, first-year male annual survival was 50% (95% confidence interval 43-57%) and adult male annual survival was 67% (95% confidence interval 61-75%). Based on retrapping birds for 11 years in an intensive study area, 78% of fledgling males died before their first breeding attempt (Korpimaki 1992). In Germany, results of a long-term banding study in an area with natural and artificial nest sites suggested juvenile survival of 20% and adult survival of 72% (Franz *et al.* 1984). In Norway, Sonerud *et al.* (1988) estimated 62% adult annual survival.

Breeding males remain in the breeding population an average of 1.5 (range = 1-7) years (Korpimaki 1988c) with an average life span of 3.5 (range = 2-11) years (E. Korpimaki, pers. comm.). In Germany, females in a nest box study were documented living 8 (n = 6), 9 (n = 5), and 10 (n = 1) years (Franz *et al.* 1984).

Ecological factors influencing survival have not been explored in any detail. Korpimaki (1992) established that owls in his population survived in the breeding population longer during increase than decrease phases of the vole cycle. Although starvation is often presumed to be a major mortality factor, direct and indirect causes of mortality have not been identified for any populations.

Movements as Related to Demography and Metapopulation Structure

As described earlier, boreal owls usually remain resident within a multiannual home range but are capable of moving long distances between breeding sites. In Sweden, young females that bred the year after fledging moved 24 km (median) from their natal territory while males moved less far (median 4.5 km). In Finland, adult females disperse up to 580 km (median 4 km) between successive breeding seasons while males rarely move more than 5 km (median 1 km; Korpimaki *et al.* 1987). During prey declines, more than half of females in Sweden were nomadic (Lofgren *et al.* 1986). Adult nomadism occurs in response to prey shortage, which may be more acute and regular in northern geographic areas. Juvenile boreal owls frequently remain within the same breeding population but also have been documented moving long distances. Research methods are biased toward detecting residency, however, so movements between populations may be quite common. Both adult and juvenile movements have not been studied carefully in North America so inferences concerning the influence of movements on demography stem from European studies.

The nomadic life history of boreal owls and the capacity for juveniles to disperse long distances may result in a strong metapopulation structure within North America. Suitable habitat in the western United States occurs in numerous patches separated by tens to hundreds of km (figure 1). The habitat distribution, then, provides a landscape that will support small populations each separated by distances greater than the normal daily movement and normal yearly movement distances of individual owls. Linkage among populations, then, results from the nomadic movement of adults or exceptional long distance dispersal of some young owls. Subpopulations of boreal owls that occur in disjunct locales may be linked through nomadic movements and juvenile dispersal. These movements are potentially important in the species' population dynamics. Individual populations may act alternately as sources and sinks depending on the status of prey, cavity availability, weather events, predators, and competitors. The long-term persistence of individual populations may be determined in large part by the rescue effect (Brown and Kodric-Brown 1977) resulting from interpopulation movements of owls, particularly experienced breeding adults.

Local Densities

There are no reliable estimates of population density for boreal owls in North America. Estimates from Europe all refer to breeding season populations, rarely include estimates of non-breeding individuals (Korpimaki and Norrdahl 1989), and most frequently refer to calling male owls. Korpimaki and Norrdahl (1989) for the period 1977-1987 reported a minimum of 1 breeding pair and 2 non-breeding males, and a maximum of 26 breeding pairs and 8 non-breeding males within a 100 km^2 study area in western Finland.

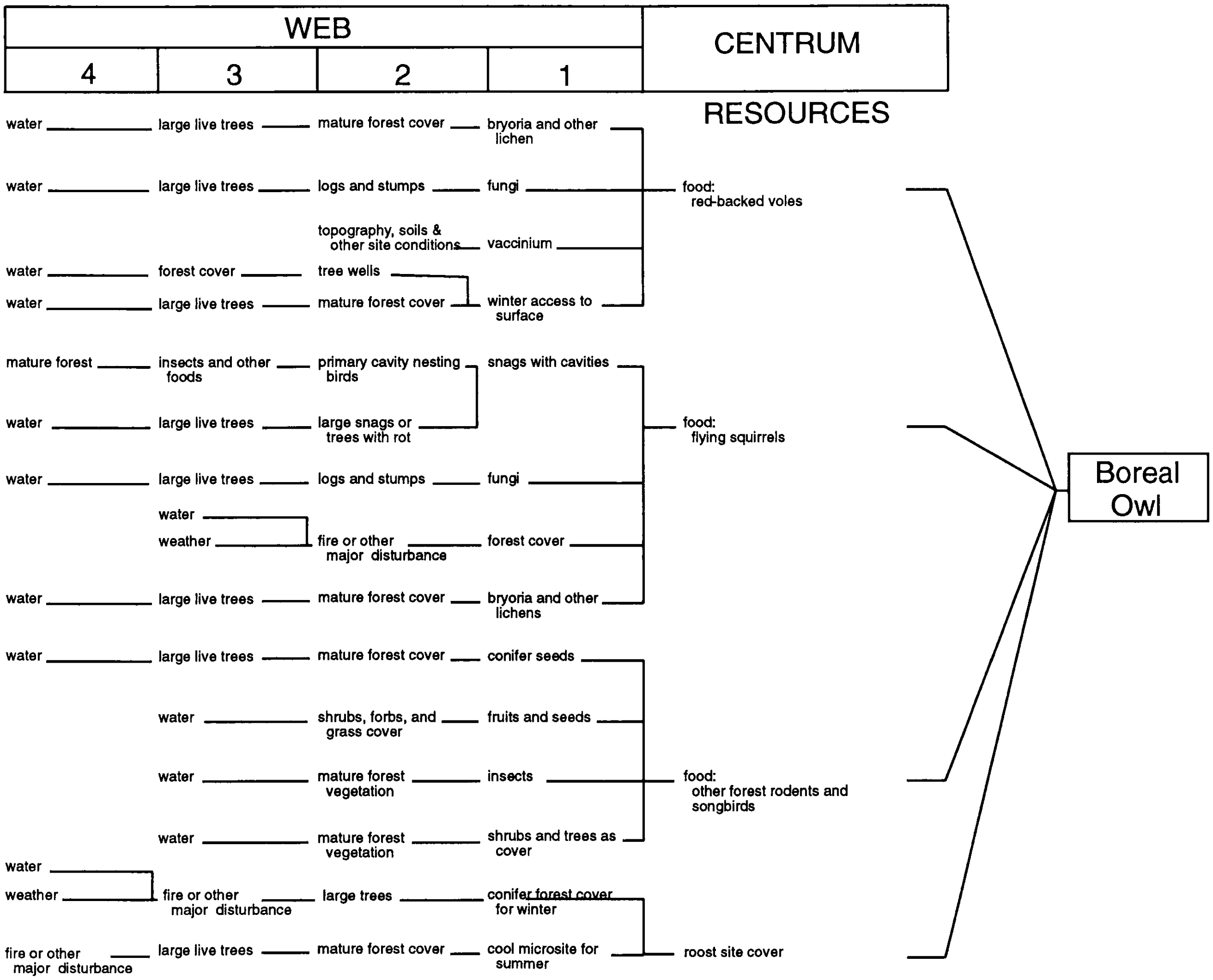
Indices of density based on calling surveys or number of active nests exhibit extreme yearly variation that corresponds with fluctuating indices of rodent abundance. Density estimates include: 0.6-1.3 nests/km^2, averaging 0.25/km^2 in France (Joneniaux and Durand 1987); 0.05-0.46/km^2 with some small areas as high as 4/km^2 in Southern Lower Saxony (Schelper 1989); and 0.19 to 0.48/km^2 in Sweden (Källander 1964).

Spacing and Population Regulation

Behavioral interactions, particularly territoriality, function to limit population size in many bird species (e.g., Hensley and Cop 1951, Krebs 1971, Watson and Moss 1980). Studies in Europe and North America suggest that under most circumstances, territoriality has no influence on abundance of boreal owls. The direct effects of prey abundance and cavity abundance are the most likely factors influencing population size; however, the links between these and other proximate factors are not established. Figure 5 displays the array of environmental factors thought to affect boreal owls based on the studies discussed in this report.

Spacing

Individuals, including mated pairs, are seldom found together except during courtship rendezvous at the nest site. Five mated pairs radio-marked prior to nesting in Idaho roosted within 150 m of one another on 7 occasions (n = 121) (Hayward *et al.* 1993); 1 pair accounted for 4 of these observations. Locations where paired individuals roosted together oc-



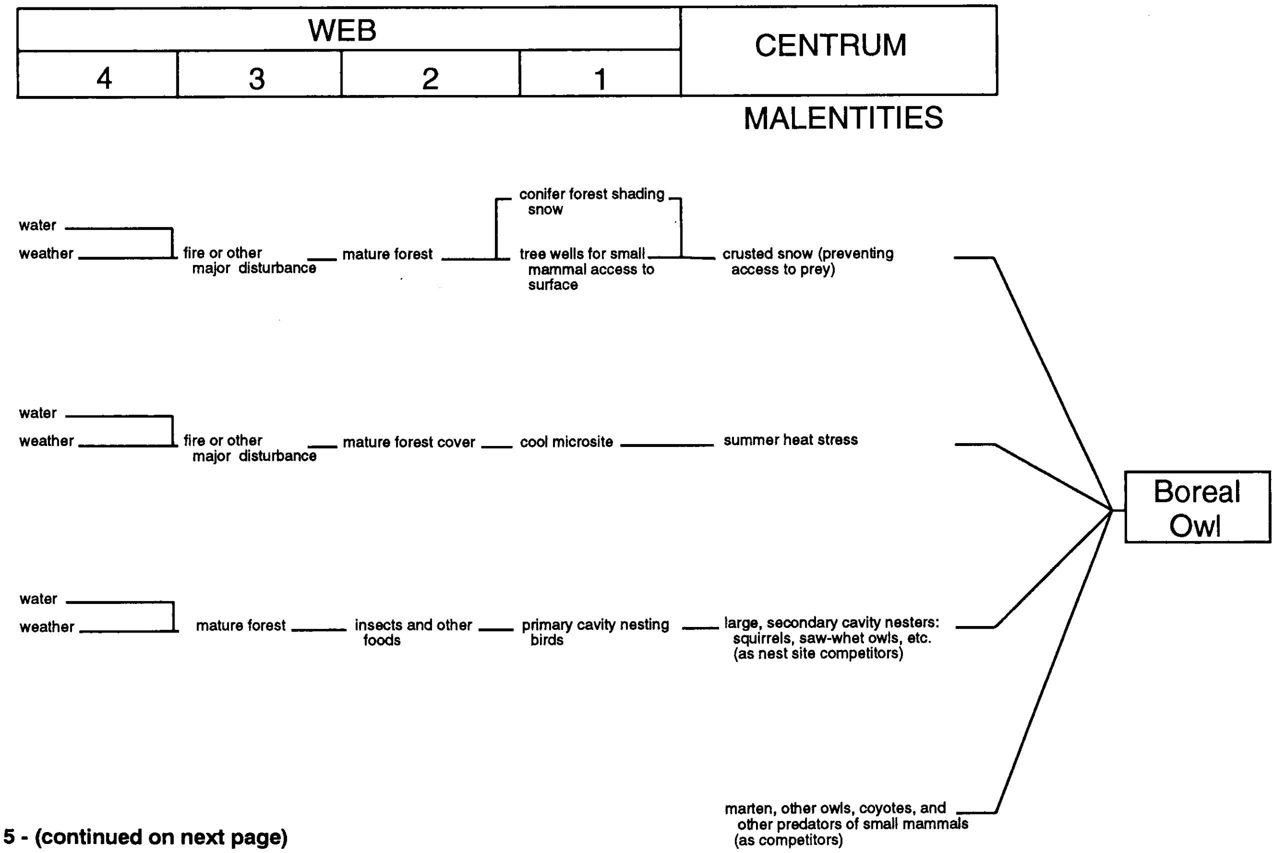


Figure 5 - (continued on next page)

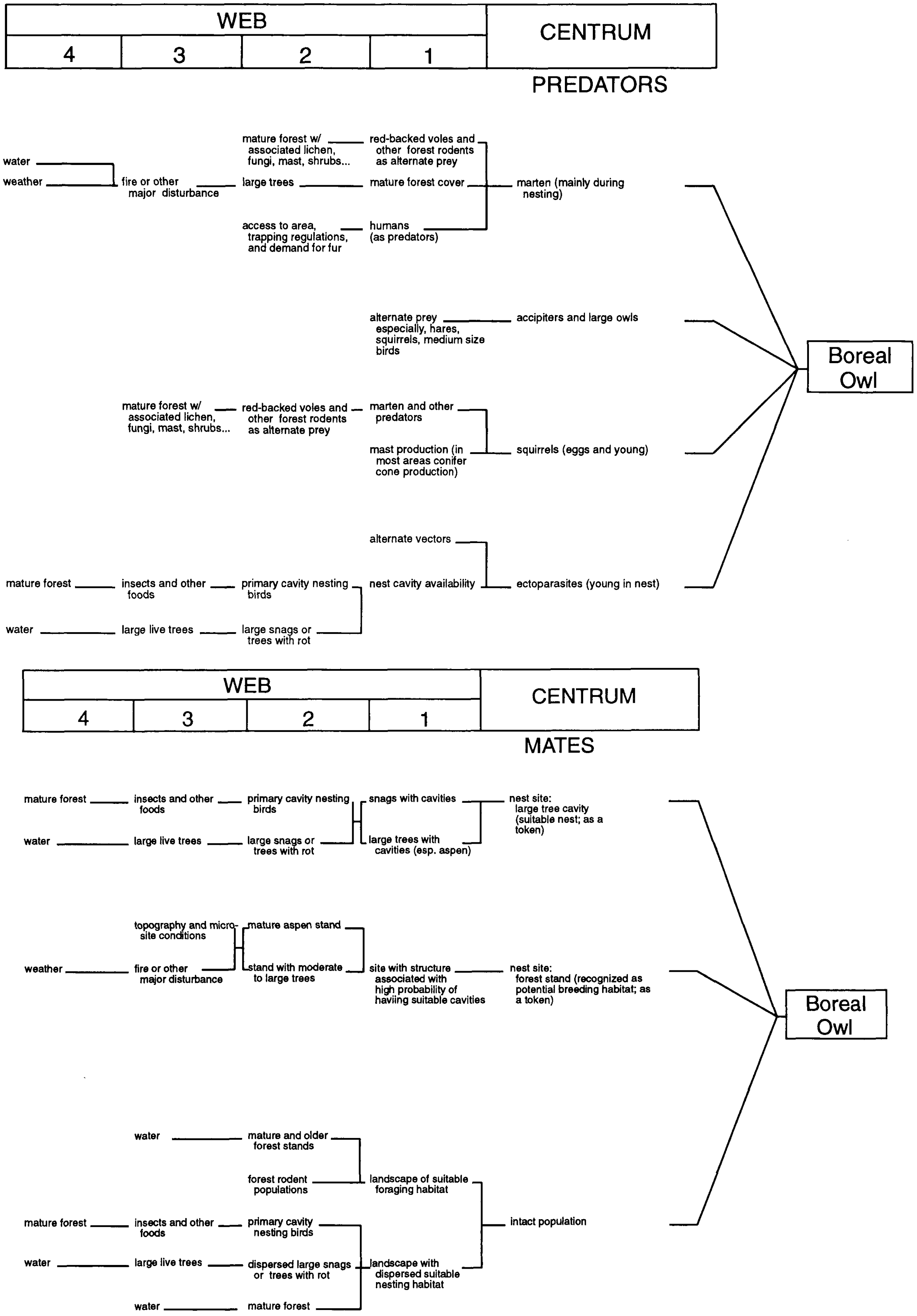


Figure 5.—Envirogram representing the web of linkages between boreal owls in the western United States and the forest ecosystem they occur in. This web should be viewed as a series of hypotheses based on the ecology of boreal owl as described throughout this assessment. For more on the application of envirograms in conservation biology see Andrewartha and Birch (1984) and Van Horne and Wiens (1991).

curred up to 6.5 km from the nest and never at the nest (Hayward *et al.* 1993). Unmated owls were located within 150 m of one another on 2 occasions: 2 males in May and an unmated male and a female caring for young in June.

Although individual owls rarely interact closely, home ranges of individuals living in the same drainage overlap extensively. In Colorado, Palmer (1986) observed > 90% overlap in ranges of two males. In Idaho, ranges of 13 owls monitored in two adjacent drainages overlapped another owl's by at least 50% and the degree of overlap was not dependent upon sex (Hayward *et al.* 1987b and Hayward *et al.* 1993).

Territoriality

Boreal owls do not exhibit strong territorial behavior. Males sing to maintain a territory only in the immediate vicinity of potential nest cavities. Territory defense is confined to the nest site and seems to include less than a 100-m radius around the nest (Mikkola 1983). Carlsson (1991) reports a male calling within 200 m of another male's nest. The paired male flew within 50 m of the calling bird and uttered a "screech" call but did not pursue the caller. Minimum distances reported between nests were 100 m (Mikkola 1983) and 0.5 km (Solheim 1983b). The distance between territories depended on prey abundance (Schelper 1989, Korpimaki and Norrdahl 1989). How factors other than prey abundance (e.g., cavity availability, habitat structure) influence territorial spacing has not been studied.

Territorial behavior is thought to be confined to the courtship and breeding period (January - July), but Kampfer-Lauenstein (1991) reported warning calls and direct flight attacks (suggesting territorial behavior) in response to playback from August-November. This suggests that the autumn territory is within the year-round home range but may not coincide with breeding territory.

Population Regulation

The availability of nest cavities and prey are the most likely environmental factors to limit populations of boreal owls (when populations are limited). The role of prey availability in observed nomadic movement patterns and the yearly variation in productivity suggests that food may regulate boreal owl abundance at times in some locales. The mechanism of limitation by food is not completely understood; but prey available to the female prior to nesting may be a critical factor in laying date and clutch size. Prey availability during the nestling period strongly influences the number of young fledged (Korpimaki 1989, Hornfeldt and Eklund 1990). Large clutches have been shown to produce more young leaving the nest (Korpimaki 1989). In his 1989 paper, Korpimaki reported an experiment in which he manipulated the abundance of food available to females prior to laying during a peak in the vole cycle. Despite the abundance of natural prey, females provided additional prey laid earlier, laid larger clutches, and fledged more young than control individuals.

Other investigators, using nonexperimental approaches, have concluded that prey availability has a direct positive correlation with boreal owl productivity (e.g. Lofgren *et al.* 1986, Sonerud *et al.* 1988, Hayward *et al.* 1993). The number of owls nesting (Lofgren *et al.* 1986), laying date (Hörnfeldt and Eklund 1990), clutch size (Lofgren *et al.* 1986), nest abandonment (Hayward *et al.* 1993), number of fledglings (Hayward *et al.* 1993), and movements of individuals following nesting (Sonerud *et al.* 1988) have all been linked with abundance of small mammals. Prey limitation leads to nomadic movements and likely results in higher mortality.

These demographic data have not been incorporated into a model (verbal or quantitative) describing population growth. Whether the absolute abundance or changes in prey populations is more important has not been pursued. Neither have the links between prey availability and changes in other environmental features been explored. And finally, the role of stochastic events in the pattern of population change has not been addressed.

In some areas of Europe, natural cavity availability is thought to limit population size and distribution (Korpimaki 1981, Franz *et al.* 1984). In North America, in regions with few (or no) pileated woodpecker or flicker cavities, nest site availability may limit boreal owl abundance. Within the geographic range of pileated woodpeckers, the absence of the woodpeckers at higher elevations may limit abundance (Hayward *et al.* 1993).

Cavity availability and abundance of prey likely interact to influence boreal owl population growth. Tree cavities occur nonrandomly across the landscape as do small mammal populations. The spatial arrangement of cavities and prey (in relation to one another) are important in determining boreal owl abundance.

Other factors potentially play a role in boreal owl population growth but research has not addressed these possibilities. Indirect evidence suggests that the owl's southern distribution and its lower elevation range in montane areas may be related to summer heat stress (Hayward *et al.* 1993). Boreal owls are easily heat stressed and seek cool roost locations in summer. The owl's physiological response to heat stress has not been measured, however.

COMMUNITY ECOLOGY

Predation on Boreal Owls

Marten (*Martes* spp.) are the most important predator of owlets and adult females at the nest site. Over 13 years , 48% of clutches were preyed upon in Norway, most by marten (Sonerud 1985). In Idaho, loss of nests are also most frequently attributed to marten; red squirrel (*Tamiasciurus hudsonicus*) predation upon eggs is also suspected (Hayward, G. D. and P. H. Hayward, unpubl. data). Aside from predation by marten at the nest, Cooper's hawk (*Accipiter cooperi*), northern goshawk (*Accipiter gentilis*), great-horned owl (*Bubo virginianus* or *Bubo bubo*), Ural owl (*Strix uralensis*), and tawny owl (*Strix aluco*) are the most important predators of young and adults (Herrera and Hiraldo 1976, Mikkola 1983, Reynolds *et al.* 1990). Research has not examined the impact of predation away from nests on population dynamics.

Relationship With Prey Populations

As described earlier, small mammal abundance has a direct and significant impact on boreal owl movements, reproduction, and survival. This relationship between prey abundance and boreal owl demography has been studied by several scientists in Europe (e.g., Korpimaki 1984, 1987a, Lofgren *et al.* 1986, Sonerud 1986) and to a lesser extent in North America (Hayward *et al.* 1993). In contrast, the influence of boreal owls on the dynamics of their prey populations has not been studied.

Korpimaki and Norrdahl (1989) provide the only focused discussion of this topic based on a 10 year study in western Finland. This work combined monitoring of owl breeding activity, owl breeding success, owl diet, and small mammal abundance in a 100 km^2 area. The results suggested that Tengmalm's owls had a direct effect on *Microtus* and to a lesser extent *Clethrionomys* populations. Predation by boreal owls likely dampens fluctuations in vole populations through the combined influence of the numerical and functional response of the owls to changing vole abundance.

Korpimaki and Norrdahl (1989) reported up to a 21-fold year-to-year variation in the number of Tengmalm's owls. Breeding population size was correlated with vole abundance ($r = 0.80$, $P < 0.01$). The nomadic nature of the owls in Finland, their potential to produce large clutches, and a breeding system that promotes bygamy and biandry in good prey years accounted for the dramatic numeric response that showed no time lag with the vole fluctuations. The owls exhibited a type 1 linear functional response with respect to vole (*Microtus* and *Clethrionomys*) abundance with no leveling off in capture rate even at the highest vole densities. The proportion of *Microtus* in the diet varied from 0-89%.

Combining the observed numeric and functional response of the owl population revealed that the proportion of available *Microtus* and *Clethrionomys* captured was higher in years when voles were most abundant (11 and 8% of the respective mammal populations) than in other years (4 and 5% of the respective mammal populations). Korpimaki's argument that Tengmalm's owl directly impacts the dynamics of its primary prey stems from his data on the owls' demographics and behavior. As a nomadic vole specialist, which can rapidly switch prey, the owl responds rapidly to changes in vole abundance. The owl's functional response indicates a lack of satiation at high vole densities which, when combined with the numeric response, leads to increased predation with increased prey abundance.

South of Finland, Tengmalm's owl is characterized as a resident-generalist, rather than a nomadic-specialist. Korpimaki and Norrdahl (1989) argue that these two life histories lead to similar impacts on fluctuating prey. Therefore, although the results can not be directly generalized to other regions, the evidence suggests that boreal owls may influence prey populations elsewhere.

Competitors

The influence of competitors on boreal owl populations has not been studied. Hayward and Garton (1988) described the pattern of resource partitioning among montane forest owls in central Idaho, and Korpimaki (1987c) described community dynamics in Finland.

Korpimaki (1987c) indicated that boreals were the most numerous species in a spruce forest in locations where populations of Ural owl were scarce. He suggested that Ural owls may limit the density and distribution of boreal owls. In North America, in sympatric situations, there is a potential for exploitative competition (when prey is limited) with saw-whet owls (Hayward and Garton 1988), great gray owls, and maybe most important, American marten (*Martes americana*). The degree to which this competition limits the distribution or abundance of boreal owls is unknown.

Potential competition for nest cavities may have the most direct influence on boreal owl distribution and abundance. Northern flying squirrel, roosting pileated woodpeckers, northern hawk owl, and saw-whet owl are the most likely competitors. Again these relationships have not been examined.

BOREAL OWL RESPONSE TO FOREST CHANGE

Individual and population response of boreal owls to forest change has not been studied directly using either experimental or observational studies. Below I interpret the results of studies examining habitat use and population dynamics as they relate to this question. Because much of the knowledge necessary to infer the owl's response has been described in earlier sections, this section is brief in relation to its importance.

Nesting Habitat

As an obligate cavity nester, boreal owl populations may be influenced by changes in cavity availability resulting from changes in snag abundance or woodpecker populations. The strength of the relationship is dependent on the relative abundance of nest sites. Changes in forest structure that reduce the number and dispersion of trees larger than ~45 cm dbh could limit the owls. Similarly, changes in forest structure that alter woodpecker prey availability or the foraging ability of flickers and pileated woodpeckers will affect boreal owl nest site availability. Finally, changes in tree species composition, regardless of tree size class, could influence nest site availability as tree species differ in their longevity as a snag and in suitability for cavities (McClelland 1977).

Because nest cavities are a species requirement, the function relating cavity availability to boreal owl breeding population density is likely a complex curve. In landscapes where nest sites are not limiting, a steady linear reduction in cavities may initially have no impact on the owls. As cavities become less abundant, breeding owls may decline initially, not due to an absolute lack of nest sites but due to the imperfect ability of the owls to locate suitable cavities or due to the juxtaposition of cavities and foraging habitat and their dispersion. As cavities become still more scarce, breeding owl abundance will decline in direct, linear response to the decline in cavity abundance.

Franz *et al.* (1984) demonstrated that cavities were limiting for boreal owls on their study site in Germany. Nest box studies in Sweden, Norway, and Finland also suggest that natural cavities were limiting. Biologists suggest that the long history of forest management that has removed old forest and large trees from Fennoscandia has led to significant natural nest site limitation. Up to 90% of the owls in these studies rely on nest boxes for nesting structures.

Changes in forest structure may also impact aspects of nest quality rather than nest site availability. Nests may become more vulnerable to predation or owls may have more difficulty locating suitable cavities under various forest structures. Results of a small nest box experiment in Idaho (Hayward *et al.* 1993) suggested that the owls prefer old forest sites for nesting. The results were not conclusive, however, and other studies of nesting habitat have been strictly observational (Bondrup-Nielsen 1978, Palmer 1986). The pattern of nest site use does indicate that older forest sites are used for nesting by these owls and therefore nesting opportunities may decline if the distribution of forests change toward younger age classes.

Roosting Habitat

The elimination of forest from a portion of an individual owl's home range will reduce roosting opportunities. The impacts of less dramatic changes in forest structure are not so clear. Observational studies of roosting habitat in Canada and Idaho led to different conclusions regarding the potential impact of forest change. A small sample of roosts and paired random forest sites in Canada did not differ from one another, implying the owls were not selective among the range of available sites (Bondrup-Nielsen 1978). In Idaho, owls did select forest with particular structural features, especially during summer (Hayward *et al.* 1993). Results from this study suggest that a reduction in the abundance or distribution of mature and old spruce-fir forest sites could limit roost sites during summer. Because cool roost locations dispersed throughout the home range may be important in boreal owl thermoregulation, a reduction in the quality of roost sites may influence owl survival rates.

Forest change involving type conversion (shift in tree species composition) could similarly influence roosting habitat. Old spruce-fir forest would provide a greater degree of microhabitat amelioration than old lodgepole pine forest.

Foraging Habitat

Changes in forest structure and/or species composition will influence boreal owls by changing prey abundance or availability. Prey availability will be influenced by changing the dispersion of hunting perches or the owls' access to prey. Because boreal owls hunt from perches, forest removal affecting patches larger than several hectares will always eliminate foraging habitat even if prey populations are increased. Dense ground vegetation or crusted snow will reduce access to prey.

Sonerud (1986) described the importance of old spruce forest as foraging habitat for boreal owls in Norway despite the lower abundance of small mammals in this habitat. In winter, uncrusted snow facilitated the movement of prey to the snow surface providing the owls access to prey. In summer, the lack of dense forest-floor vegetation provided the owls clear access to small mammals. These results stress the importance of conifer canopy cover in maintaining small mammal availability.

Red-backed voles represent important prey for boreal owls in much of North America (Bondrup-Nielsen 1978, Palmer 1986, Hayward *et al.* 1993). Changes in forest structure or composition that influence red-back vole populations will likely influence boreal owl populations. The effect of forest structure and composition on red-backed vole population dynamics is not well known aside from the decline in red-backed vole populations usually observed following forest removal. Similar knowledge for other prey species (northern flying squirrels, northern pocket gophers, heather voles, etc.) is also lacking.

Broad-Scale Habitat Change

As the reader can well imagine, the influence of regional changes in habitat conditions on boreal owl populations is unknown. Changes at this scale will influence metapopulation structure through dispersal and local extinction. Changes in the size of subpopulations, distance between neighboring subpopulations, changes in productivity of source populations, and characteristics of habitat separating subpopulations likely influence metapopulation stability and would be important to manage on a regional scale.

BOREAL OWL RESPONSE TO HUMAN OR MECHANICAL DISTURBANCE

Boreal owls tolerate human and machine noise. In Colorado, owls have nested within 30 m of a major highway (R. A. Ryder; pers. comm.). In Europe, nests have been located within farmsteads and are associated with agriculture (Korpimaki 1981). Owls tolerate frequent (every 4-5 days) direct nest inspection (except during laying) and will deliver prey to the nest while humans observe from several meters away. There is no evidence that disturbance is an important factor in nest loss or owl movements.

ASSESSMENT OF SCIENTIFIC BASIS FOR PARTICULAR MANAGEMENT TOOLS

Monitoring

Intensive management of wildlife populations, particularly threatened, endangered, and sensitive species, requires information on population trend of the target species and on habitat trend. Monitoring regional trends in boreal owl populations may be approached intensively or extensively. An intensive approach involves tracking a measure of abundance for sample populations within the target region over time. An extensive approach tracks presence / absence for a large sample of populations over time.

These approaches differ in method and objective. The intensive approach facilitates examination of environmental features associated with trends in individual populations but requires a large field effort, as described below. The extensive approach costs less and tracks the "winking" on and off of populations throughout the region, but it provides no insights into the causes of population changes.

Methods for monitoring boreal owl populations have received little attention. Playback surveys have been used extensively to determine the geographic distribution of the species (Palmer and Ryder 1984, Hayward *et al.* 1987a) and have been promoted as a promising monitoring technique for other owls (Johnson *et al.* 1981, Forsman 1983, Smith *et al.* 1987). Playback surveys cannot be considered the best technique to assess trends in boreal owl populations, however, because many factors influence calling rate. Lundberg (1978) suggested that the number of boreal owls singing may be inversely related to breeding success. He found that "territorial and breeding pairs were more silent than non-territorial individuals" and concluded that "censuses made at roadside stops give unacceptable results for population studies of both the Ural owl and Tengmalm's [*boreal*] owl" (Lundberg 1978:171).

Although Lundberg's (1978) results suggest that playback surveys should not be used for intensive population monitoring, playback could be useful in developing methods of presence / absence monitoring. Playback methods seem to be the most efficient method to determine the occurrence of boreal owls in an area. These provide the basic data necessary in a presence/absence sampling design. Research to date has not explored the potential of these techniques for monitoring owls on a regional basis. These methods would fit well into a scheme designed to approach management in a metapopulation framework.

Some work has been done to develop more intensive population level monitoring. Hayward *et al.* (1992) examined the sampling efficiency of employing nest boxes to monitor response of boreal owls to changes in foraging habitat. The results suggest that when boreal owls are moderately abundant (nest box occupancy >7%), modest changes in clutch size and occupancy rate could be detected with a system of 350 nest boxes. When owls are less abundant, the number of nest boxes necessary to detect modest changes would be prohibitively large. Research has not addressed the underlying assumptions of the methods suggested in this study (Hayward *et al.* 1992).

An understanding of boreal owl vocalizations is necessary in designing surveys to determine distribution or to develop a presence/absence monitoring program. Difficulties observing behaviors associated with vocalizing boreals and problems interpreting phonic representations of calls have led to some confusion in describing the array of sounds produced and the function of various vocalizations. Authors within the United States and in Europe have used a variety of names to describe vocalizations and no one set of names is preferable. Meehan (1980) and Bondrup-Nielsen (1984) provide the most complete vocal analysis for the boreal owl. Cramp (1977) and Johnsgard (1988:221) summarize information for North America and Eurasia. Throughout this discussion I refer to Meehan (1980) as RHM and Bondrup-Nielsen (1984) as SBN.

The call most important in terms of management is the primary song (staccato song--SBN, song--RHM). This is the call that can be attributed most certainly to boreal owls and is the call most frequently elicited in springtime playback surveys. The primary song is uttered loudly only by males from a perch near a potential nest cavity, is not commonly used outside the breeding season, and isn't used during antagonistic encounters among individuals. It is presumed to function in mate attraction as a long distance advertisement song. The call is a loud vocalization uttered as a series of trills consisting of 11-23 notes at ~0.74 kHz that increase in volume during a trill lasting 1.8 (1.32-2.32) seconds (SBN). The trill is repeated after a silence of 1 to several seconds; singing bouts frequently last 20 minutes but may extend 2-3 hours with infrequent pauses of several minutes. The song is frequently heard by humans over 1.5 km and up to 3.5 km.

Singing in Idaho began by 20 January, reached greatest intensity by late March, and became uncommon by late April (Hayward 1989). In Colorado, Palmer (1987) reported singing 18 February - 21 June; singing peaked in late April and a lull followed in early May with renewed frequency late May through June. Palmer (1987) speculated that calling in June resulted from first-time breeders and unmated males. In Alaska, singing peaked by mid-February to March (RHM).

See Hayward and Hayward (1993) for a summary of the characteristics of other songs.

Viability Analysis

Biologists working with land management agencies are often asked to evaluate the impact of management activities on sensitive plants and animals. Biologists must document their judgments about whether or not a proposed management action will increase the likelihood of sensitive species becoming threatened or endangered. The basis for the "determination of effect" necessarily involves some kind of population viability analysis (PVA). Gilpin and Soule (1986) described PVA as a complex process of considering all factors that affect the processes of species extinction or persistence while Boyce (1992) discussed both theoretical and practical aspects of PVA.

Tools necessary to conduct PVA for boreal owls are not available. Neither mathematical nor word models linking the relevant factors have been developed. Furthermore, the ecological understanding of the owl's ecology in North America has not reached the level of maturity necessary to conduct formal viability analyses. The biological and ecological information summarized in this chapter, however, could provide the background necessary to structure assessments for individual impact analyses until more general guidelines for PVA are developed. Further ecological research will be necessary, though, before developing any formal analysis tools.

Effects Criteria Identification

Although PVA is an important tool for impact analysis, the identification of criteria upon which to base statements of effects is important in most environmental assessments. Therefore, guidelines from which to build effects criteria are important for resource managers. These types of guidelines are not currently available to managers. The paucity of information on boreal owl ecology and life history specific to different management regions precludes development of elaborate criteria. Based on the ecological relationships depicted in figure 5, however, some basic guidelines can be outlined. These will be stated generally here but could be elaborated for particular regions:

(1) Large trees are required for nesting boreal owls.

(2) Primary cavity nesters (e.g., pileated woodpeckers, common flickers) provide a majority of nesting sites in most areas and the status of populations of these birds is important to the productivity of boreal owls.

(3) The availability of small mammals limits populations of boreal owls in many areas; therefore, factors that influence small mammal abundance and availability will directly influence the abundance of boreal owls.

a) Red-backed voles are important prey for boreal owls everywhere the owl has been studied. In the western United States the abundance of red-backed voles is related, at least in part, to forest age, fungi abundance, and lichen abundance.

b) Prey availability is related to forest structure characteristics as the structure influences mobility of boreal owls. Dense shrub cover or high tree density will limit the access of boreal owls to small mammals. Conditions that promote snow crusting (large openings) will also reduce small mammal availability.

(4) In the western United States, boreal owl distribution may be limited, in part, by warm summer temperatures. Cool microsites for daytime roosts may be important in determining the species' current distribution. In Idaho, old forest sites provided cool microsites used for roosting (Hayward *et al.* 1992).

Stand and Watershed Scale Silviculture Prescriptions

Guidelines with which to develop specific stand and watershed scale silviculture prescriptions are not published. Knowledge of boreal owl ecology and habitat choice limits the specificity of any guidelines. As shown above, some general statements can be made with certainty.

Current understanding of boreal owl habitat use suggests that the maintenance of forested landscapes is required for boreal owls. Furthermore, silvicultural prescriptions must provide for large diameter trees well dispersed over space and time. The roosting, nesting, and foraging ecology of boreal owls in the western United States also suggests that mature and older forest must be well represented in the landscape to support a productive boreal owl population. In most cases, uneven-aged management or other silvicultural practices that maintain canopy structure and forest floor moisture will maintain boreal owl nesting, roosting, and foraging habitat. Forest clearcuts provide little or no habitat for boreal owls for two to several decades after disturbance and may not provide high quality habitat for one to two centuries.

REFERENCES

Andrewartha, H. G., and L. C. Birch. 1984. The ecological web: more on the distribution and abundance of animals. University of Chicago Press, Chicago, Illinois, USA.

Armstrong, R. H. 1980. Guide to the birds of Alaska. Alaska Northwest Publishing Company, Edmonds, Washington, USA.

Bondrup-Nielsen, S. 1978. Vocalizations, nesting, and habitat preferences of the Boreal Owl (*Aegolius funereus*). Thesis. University of Toronto, Toronto, Ontario, Canada.

Bondrup-Nielsen, S. 1984. Vocalizations of the Boreal Owl, *Aegolius funereus richardsoni*, in North America. Canadian Field-Naturalist 98:191-197.

Boyce, M. S. 1992. Population viability analysis. Annual Review of Ecology and Systematics 23:481-506.

Brown, J. H., and A. Kodric-Brown. 1977. Turnover rates in insular biogeography: effects of immigration on extinction. Ecology 58:445-449.

Brown, L. N. 1967. Ecological distribution of mice in the Medicine Bow Mountains of Wyoming. Ecology 48:677-680.

Buckner, C. A., and D. J. Shure. 1985. The response of *Peromyscus* to forest opening size in the southern Appalachian Mountains. Journal of Mammalogy 66:299-307.

Burt, W. H. 1943. Territoriality and home range concepts as applied to mammals. Journal of Mammalogy 24:346-352.

Bye, F. N., B. V. Jacobsen, and G. A. Sonerud. 1992. Auditory prey location in a pause-travel predator: search height, search time and attack range of Tengmalm's Owl. (*Aegolius funereus*). Behavioral Ecology 3:266-276.

Campbell, J. M. 1969. The owls of the central Brooks Range, Alaska. Auk 86:565-568.

Campbell, T. M., and T. W. Clark, 1980. Short-term effects of logging on red-backed voles and deer mice. Great Basin Naturalist 40:183-189.

Carey, A. B., S. P. Horton, and B. L. Biswell. 1992. Northern Spotted Owls: influence of prey base and landscape characteristics. Ecological Monographs 62:223-250.

Carlsson, B-G., B. Hornfeldt, and O. Lofgren. 1987. Bigyny in Tengmalm's Owl *Aegolius funereus* effect of mating strategy on breeding success. Ornis Scandinavica 18:237-243.

Carlsson, B-G. 1991. Recruitment of mates and deceptive behavior by male Tengmalm's Owls. Behavioral Ecology and Sociobiology 28:321-328.

Catling, P. M. 1972. A study of the Boreal Owl in southern Ontario with particular reference to the irruption of 1968-69. Canadian Field-Naturalist 86:223-232.

Clough, G. C. 1987. Relations of small mammals to forest management in northern Maine. Canadian Field-Naturalist 101:40-48.

Clutton-Brock, T. H., F. E. Guinness, and S. D. Albon. 1982. Red deer: behavior and ecology of two sexes. University of Chicago Press, Chicago, Illinois, USA.

Coale, H. K. 1914. Richardson's Owl in northeast Illinois. Auk 31:536.

Cramp, S., editor. 1977. Handbook of the birds of Europe and the Middle East: the birds of the western palearctic. Volume 4. Oxford University Press, London, England.

Daniel, T. W., J. A. Helms, and F. S. Baker. 1979. Principles of silviculture. McGraw-Hill, New York, New York, USA.

Dejaifve, P. A., C. Novoa, and R. Prodon. 1990. Habitat et densite de la Chouette de Tengmalm *Aegolius funereus* a l'extremite orientale des Pyrenees. Alauda 58:267-273.

Dement'ev, G. P., and N. A.Gladkov. 1954. Birds of the Soviet Union. Volume I. State Publishing House, Moscow, Russia. (English translation 1966.)

Eckert, K. R., and T. L. Savaloja. 1979. First documented nesting of the Boreal Owl south of Canada. American Birds 33:135-137.

Emslie, S. D. 1981. Birds and prehistoric agriculture: the New Mexican pueblos. Human Ecology 9:305-329.

Erdman, T. C. 1979. Boreal Owl in northeastern Wisconsin. Passenger Pigeon 41:58-59.

Erkinaro, E. 1973. Seasonal variation of the dimensions of pellets in Tengmalm's Owl *Aegolius funereus* and the Short-Eared Owl *Asio flammeus*. Aquilo Ser Zoologica 14:84-88.

Evans, D. L., and R. N. Rosenfield. 1977. Fall migration of Boreal Owls. Loon 49:165-167.

Evermann, B. W. 1913. Eighteen species of birds new to the Pribilof Islands, including four new to North America. Auk 30:15-18.

Ford, N. L. 1967. A systematic study of the owls based on comparative osteology. Ph.D. diss., Univ. of Michigan, Ann Arbor.

Forsman, E. D. 1983. Methods and materials for locating and Studying Spotted Owls. United States Department of Agriculture Forest Service Pacific Northwest Forest and Range Experiment Station, Portland, Oregon, USA. General Technical Report PNW-162.

Franz, A., T. Mebs, and E. Seibt. 1984. Zur populationsbiologi des Rauhfusskauzes (*Aegolius funereus*) im sudliehen Westfalen und in angrenzenden Gebieten anhand von Beringungsergebnissen [Several aspects of population biology in the Tengmalm's Owl (*Aegolius funereus*) in southern Westphalia and adjacent areas on the base of ringing results.]. Die Vogelwarte 32:260-269.

Fretwell, F. D., and H. L. Lucas. 1969. On territorial behavior and other factors influencing habitat distribution in birds. I. Theoretical development. Acta Biotheor. 19:16-36.

Gabrielson, I. N., and F. C. Lincoln. 1959. The birds of Alaska. Stackpole Company, Harrisburg, Pennsylvania, USA.

Garton, E. O. Professor, Department of Fish and Wildlife Resources, University of Idaho, Moscow, ID. [Personal communication]. Summer 1990.

Getz, L. L., and V. Ginsberg. 1968. Arboreal behaviour of the red-back vole, *Clethrionomys gapperi*. Animal Behaviour 16:418-424.

Grant, B. R., and P. R. Grant. 1989. Evolutionary dynamics of a natural population: the large Cactus Finch of the Galapagos. University of Chicago Press, Chicago, Illinois, USA.

Gilpin, M. E., and M. E. Soule. 1986. Minimum viable populations: Process of species extinction. Pages 19-34 *in* M. E. Soule, editor. Conservation biology, the science of scarcity and diversity. Sinauer Associates, Saunderland, Massachusetts.

Halvorson, C. H. 1982. Rodent occurrence, habitat disturbance, and seed fall in a larch-fir forest. Ecology 63:423-433.

Hansson, L., and H. Henttonen. 1985. Gradients in density variations of small rodents: the importance of latitude and snow cover. Oecologia 67:394-402.

Hayward, G. D. 1983. Resource partitioning among six forest owls in the River of No Return Wilderness, Idaho. Thesis. University of Idaho, Moscow, Idaho, USA.

Hayward, G. D. 1987. Betalights: an aid in the nocturnal study of owl foraging habitat and behavior. Journal of Raptor Research 21:98-102.

Hayward, G. D. 1989. Habitat use and population biology of Boreal Owls in the northern Rocky Mountains, USA. Dissertation. University of Idaho, Moscow, Idaho, USA.

Hayward, G. D., and J. W. Deal. 1993. An optic device to inspect nest boxes from the ground. Journal of Field Ornithology 64:179-187.

Hayward, G. D., and E. O. Garton. 1983. First nesting record for boreal owl in Idaho. Condor 85:501.

Hayward, G. D., and E. O. Garton. 1984. Roost habitat selection by three small forest owls. Wilson Bulletin 94:690-692.

Hayward, G. D., and E. O. Garton. 1988. Resource partitioning among forest owls in the River of No Return Wilderness, Idaho. Oecologia 75:253-265.

Hayward, G. D., and P. H. Hayward. 1991. Body measurements of Boreal Owls in Idaho and a discriminant model to determine sex of live specimens. Wilson Bulletin 103:497-500.

Hayward, G. D., and P. H. Hayward. 1993. Boreal Owl. In A. Poole,P. Stettenheim, and F. Gill, editors. Birds of North America. Philadelphia Academy of Natural Sciences, Washington, DC, USA.

Hayward, G. D., P. H. Hayward, E. O. Garton, and R. Escano. 1987a. Revised breeding distribution of the Boreal Owl in the northern Rocky Mountains. Condor 89:431-432.

Hayward, G. D., P. H. Hayward, and E. O. Garton. 1987b. Movements and home range use by Boreal Owls in central Idaho. Pages 175-184 *in* R. W. Nero, C. R. Knapton, and R. H. Hamre, editors. Biology and conservation of northern forest owls. Symposium proceedings, Winnipeg, Manitoba, February 1986. United States Department of Agriculture Forest Service Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado, USA. General Technical Report RM-142.

Hayward, G. D., P. H. Hayward, and E. O. Garton. 1993. Ecology of Boreal Owls in the northern Rocky Mountains, USA. Wildlife Monographs 124:1-59.

Hayward, G. D., and R. Rosentretter. 1994. Lichen as nesting material for northern flying squirrels in the northern Rocky Mountains. Journal of Mammalogy 75:663-673.

Hayward, G. D., R. K. Steinhorst, and P. H. Hayward. 1992. Monitoring Boreal Owl populations with nest boxes: sample size and cost. Journal of Wildlife Management 56:776-784.

Hensley, M. M., and J. B. Cop. 1951. Further data on removal and repopulation of the breeding birds in a spruce-fir forest community. Auk 68:483-493.

Herrera, C. M., and F. Hiraldo. 1976. Food-niche and trophic relationships among European owls. Ornis Scandinavica 7:29-41.

Hornfeldt, B., B. Carlsson, and A. Nordstrom. 1988. Molt of primaries and age determination in Tengmalm's Owl (*Aegolius funereus*). Auk 105:783-789.

Hornfeldt, B., and U. Eklund. 1990. The effect of food on laying date and clutch-size in Tengmalm's Owl *Aegolius funereus*. Ibis 132:395-406.

Howard, H. 1931. *Cryptoglaux funerea* in New Mexico. Condor 33:216.

Isleib, M. E., and B. Kessel. 1973. Birds of the north gulf coast—Prince William Sound region, Alaska. Biological Papers Number 14, University of Alaska, Fairbanks, Alaska, USA.

Jaderholm, K. 1987. Diets of the Tengmalm's Owl *Aegolius funereus* and the Ural Owl *Strix uralensis* in central Finland. Ornis Fennica 64:149-153.

Johnsgard, P. A. 1988. North American owls: biology and natural history. Smithsonian Institution Press, Washington, DC, USA.

Johnson, M. L., and S. Johnson, 1983. Voles; *Microtus* species. Pages 326-354 *in* J. A. Chapman and G. A. Feldhamer, editors. Wild mammals of North America: biology, management, and economics. Johns Hopkins University Press, Baltimore, Maryland, USA.

Johnson, R. R., B. T. Brown, L. T. Haigh, and J. M. Simpson. 1981. Playback recordings as a special avian censusing technique. Pages 68-75 *in* C. J. Ralph and J. M. Scott, editors. Estimating numbers of terrestrial birds. Studies in Avian Biology, Number 6.

Joneniaux, A., and G. Durand, 1987. Forest management and population ecology of Tengmalm's Owl. Revue d'Ecologie la Terre et la Vie, Supl. 4:84-87.

Jorlitschka, W. 1988. Untersuchung zur habitatstruktur von Revieren des Rauhfusskauzes *Aegolius funereus* im Nordschwarzwald. Die Vogelwelt 109:152-155.

Källander, H. 1964. Irruption in 1958 of Tengmalm's Owl (*Aegolius funereus*) into Central Sweden and some aspects on the distribution of the species in Sweden. Vår Fågelvärld 23:134-135.

Kampfer-Lauenstein, A. 1991. Intraspecific territorial behavior of Tengmalm's Owl (*Aegolius funereus*) in autumn. Ecology of Birds 13:111-120.

Kelley, A. H., and J. O. L. Roberts. 1971. Spring migration of owls at Whitefish Point. Jack-Pine Warbler 49:65-70.

Klaus, S., H. Mikkola, and J. Wiesner. 1975. Activity and food of Tengmalm's owl, *Aegolius funereus* (L.), during the breeding season. Zoologische Jahrbuecher Abteilung fuer Systematik Oekologie und Geographie der Tiere 102:293-294.

Kloubec, B., and R. Vacik. 1990. Outline of food ecology of Tengmalm's owl (*Aegolius funereus* L.) in Czechoslovakia. Tichodroma 3:103-125.

Konig, C. 1969. Six year observations of a population of Tengmalm's Owl *Aegolius funereus*. Journal fuer Ornithologie 110:133-147.

Korpimaki, E. Research Professor, Department of Zoology University of Oulu, Oulu, Finland. [Personal communication]. February 1993.

Korpimaki, E. 1981. On the ecology and biology of Tengmalm's Owl (*Aegolius funereus*) in southern Ostrobothnia and Suomenselka, western Finland. Acta Universitatis Ouluensis Series A Scientiae Rerum Naturalium Biologica 118. 13:1-84.

Korpimaki, E. 1984. Clutch size and breeding success of Tengmalm's Owl *Aegolius funereus* in natural cavities and nest-boxes. Ornis Fennica 61:80-83.

Korpimaki, E. 1985. Clutch size and breeding success in relation to nest box size in Tengmalm's Owl *Aegolius funereus*. Holarctic Ecology 8:175-180.

Korpimaki, E. 1986a. Gradients in population fluctuations of Tengmalm's Owl *Aegolius funereus* in Europe. Oecologia 69:195-201.

Korpimaki, E. 1986b. Reversed size dimorphism in birds of prey, especially in Tengmalm's Owl *Aegolius funereus*: a test of the "starvation hypothesis." Ornis Scandinavica 17:326-332.

Korpimaki, E. 1986c. Seasonal changes in the food of the Tengmalm's Owl *Aegolius funereus* in western Finland. Annales Zoologici Fennici 23:339-344.

Korpimaki, E. 1987a. Clutch size, breeding success, and brood size experiments in Tengmalm's Owl *Aegolius funereus*: a test of hypothesis. Ornis Scandinavica 18:277-284.

Korpimaki, E. 1987b. Composition of the owl communities in four areas in western Finland. importance of habitats and interspecific competition. Proceedings of the Fifth Nordic Ornithological Congress.

Korpimaki, E. 1987c. Prey caching of breeding Tengmalm's Owls *Aegolius funereus* as a buffer against temporary food shortage. Ibis 129:499-510.

Korpimaki, E. 1988a. Costs of reproduction and success of manipulated broods under varying food conditions in Tengmalm's Owl. Journal of Animal Ecology 57:1027-1039.

Korpimaki, E. 1988b. Diet of breeding Tengmalm's Owls *Aegolius funereus*: long-term changes and year-to-year variation under cyclic conditions. Ornis Fennica 65:21-30.

Korpimaki, E. 1988c. Effects of age on breeding performance of Tengmalm's Owl *Aegolius funereus* in western Finland. Ornis Scandinavica 19:21-26.

Korpimaki, E. 1988d. Effects of territory quality on occupancy, breeding performance, and breeding dispersal in Tengmalm's Owl. Journal of Animal Ecology 57:97-108.

Korpimaki, E. 1989. Breeding performance of Tengmalm's Owl *Aegolius funereus*: effects of supplementary feeding in a peak vole year. Ibis 131:51-56.

Korpimaki, E. 1991. Poor reproductive success of polygynously mated female Tengmalm's Owls: are better options available? Animal Behaviour 41:37-47.

Korpimaki, E. 1992. Fluctuating food abundance determines the lifetime reproductive success of male Tengmalm's Owls. Journal of Animal Ecology 61:103-111.

Korpimaki, E., and H. Hakkarainen. 1991. Fluctuating food supply affects the clutch size of Tengmalm's Owl independent of laying date. Oecologia 85:543-552.

Korpimaki, E., M. Lagerstrom, and P. Saurola. 1987. Field evidence for nomadism in Tengmalm's Owl *Aegolius funereus*. Ornis Scandinavica 18:1-4.

Korpimaki, E., and K. Norrdahl. 1989. Predation of Tengmalm's Owls: numerical responses, functional responses and dampening impact on population fluctuations of microtines. Oikos 54:154-164.

Krebs, J. R. 1971. Territory and breeding density in the Great Tit, *Parus major* L. Ecology 52:2-22.

Lane, W. H. 1988. 1988 Boreal Owl survey in Cook County. Loon 60:99-104.

Lofgren, O., B. Hornfeldt, and G. Carlsson. 1986.

Site tenacity and nomadism in Tengmalm's Owl (*Aegolius funereus*) in relation to cyclic food production. Oecologia 69:321-326.

Lundberg, A. 1978. Census methods for the Ural Owl *Strix uralensis* and the Tengmalm's Owl *Aegolius funereus*. Anser, Supplement 3:171-175.

Lundberg, A. 1979. Residency, migration and a compromise: adaptations to nest-site scarcity and food specialization in three Fennoscandian owl species. Oecologia 41:273-281.

Martell, A. M. 1981. Food habits of southern red-backed voles (*Clethrionomys gapperi*) in northern Ontario. Canadian Field-Naturalist 95:325-328.

Martell, A. M. 1983a. Changes in small mammal communities after logging in north-central Ontario. Canadian Journal of Zoology 61:970-980.

Martell, A. M. 1983b. Demography of southern red-backed voles (*Clethrionoys gapperi*) and deer mice (*Peromyscus maniculatus*) after logging in north-central Idaho. Canadian Journal of Zoology 61:970-980.

Martell, A. M., and A. L. Macaulay. 1981. Food habits of deer mice (*Peromyscus maniculatus*) in northern Ontario. Canadian Field-Naturalist 95:319-324.

Marti, C. D., E. Korpimaki, and F. M. Jaksic. 1993. Trophic structure of raptor communities: a three continent comparison and synthesis. Current Ornithology 10.

Martin, W. Research Scientist,U.S. Fish and Wildlife Service, Laurel, MD. [Personal communication]. October 1982.

McClelland, B. R. 1977. Relationships between hole-nesting birds, forest snags, and decay in western larch/Douglas-fir forests of the northern Rocky Mountains. Dissertation. University of Montana, Missoula, Montana, USA.

Medin, D. E. 1986. Small mammal responses to diameter-cut logging in an Idaho Douglas-fir forest. United States Department of Agriculture Forest Service Intermountain Forest and Range Experiment Station, Ogden, Utah, USA. Research Note INT-36.

Meehan, R. H. 1980. Behavioral significance of Boreal Owl vocalizations during breeding season. Thesis. University of Alaska, Fairbanks, Alaska, USA.

Meehan, R. H., and R. J. Ritchie. 1982. Habitat requirements of Boreal and Hawk Owls in interior Alaska. Pages 188-196 *in* W. N. Ladd and P. F. Schempgf, editors. Raptor management and biology in Alaska and Western Canada. United States Fish and Wildlife Service, Anchorage, Alaska, USA.

Merikallio, E. 1958. Finnish birds: their distribution and numbers. Fauna Fennica 5:1-181.

Merritt, J. F. 1981. *Clethrionomys gapperi*. Mammalian Species Number 146:1-9.

Mikkola, M. 1983. The owls of Europe. Buteo Books, Vermillion, South Dakota, USA.

Millar, J. S., D. G. L. Innes, and V. A. Loewen. 1985. Habitat use by non-hibernating small mammals of the Kananaskis Valley, Alberta. Canadian Field-Naturalist 99:196-204.

Mysterud, I. 1970. Hypotheses concerning characteristics and causes of population movements in Tengmalm's Owl *Aegolius funereus*. Nytt Magasin for Zoologi (Oslo) 18:49-74.

Norberg, A. 1964. Studies on the ecology and ethology of Tengmalm's Owl (*Aegolius funereus*). Var Fagelvarld 23:228-234.

Norberg, A. 1970. Hunting techniques of Tengmalm's Owl *Aegolius funereus*. Ornis Scandinavica 1:51-64.

Norberg, A. 1978. Skull asymmetry, ear structure and function, and auditory localization in Tengmalm's Owl, *Aegolius funereus* (Linne). Philosophical Transactions of the Royal Society of London B Biological Sciences 282:325-410.

Norberg, A. 1987. Evolution, structure, and ecology of northern forest owls. Pages 9-43 *in* R. W. Nero, C. R. Knapton, and R. H. Hamre, editors. Biology and conservation of northern forest owls. Symposium proceedings. United States Department of Agriculture Forest and Range Experiment Station, Fort Collins, Colorado, USA. General Technical Report RM-142.

Palmer, D. A. 1986. Habitat selection, movements and activity of Boreal and Saw-whet Owls. Thesis. Colorado State University, Fort Collins, Colorado, USA.

Palmer, D. A. 1987. Annual, seasonal, and nightly variation in calling activity of Boreal and Northern Saw-whet Owls. Pages 162-168 *in* R. W. Nero, C. R. Knapton, and R. J. Hamre, editors. Biology and conservation of northern forest owls. Symposium proceedings. United States Department of Agriculture Forest Service Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado, USA. General Technical Report RM-142.

Palmer, D. A., and R. A. Ryder. 1984. The first documented breeding of Boreal Owl in Colorado. Condor 86:251-217.

Probst, J. R., and D. S. Rakstad. 1987. Small mammal communities in three aspen stand-age classes. Canadian Field-Naturalist 101:362-368.

Ramirez, P., and M. Hornocker. 1981. Small mammal populations in different-aged clear-cuts in northwestern Montana. Journal of Mammalogy 62:400-403.

Raphael, M. G. 1988. Habitat associations of small mammals in a subalpine forest, southeastern Wyoming. Pages 359-367 *in* R. C. Szaro, K. E. Severson, and D. R. Patton, editors. Management of amphibians, reptiles and small mammals in North America. United States Department of Agriculture Forest Service Forest and Range Experiment Station, Fort Collins, Colorado, USA. General Technical Report RM-166.

Reynolds, R. T., S. Joy, and T. B. Mears. 1990. Predation and observation records of Boreal Owls in western Colorado. Colorado Field Ornithology Journal 24:99-101.

Rosenberg, D. K., and R. G. Anthony. 1992. Characteristics of northern flying squirrel populations in young second- and old-growth forests in western Oregon. Canadian Journal of Zoology 70:161-166.

Rosenberg, D. K., C. J. Zabel, B. R. Noon, and E. C. Meslow. *In press* . Northern Spotted Owls: influence of prey base—a comment. Ecology 75.

Ryder, R. A. Professor Emeritus, Department of Fisheries and Wildlife Biology, Colorado State University, Fort Collins, CO. [Personal communication]. August 1986.

Schelper, V. W. 1989. Zur Brutbiologie, Ernahrung und Populationsdynamik des Rauhfusskauzes *Aegolius funereus* im Kaufunger Wald (Sudniedersachsen). Vogelkundliche Berichte aus Niedersachsen 21:33-53.

Schwerdtfeger, O. 1984. Verhalten und Populationsdynamik des Rauhfusskauzes (*Aegoloius funereus*). Vogelwarte 32:183-200.

Schwerdtfeger, O. 1988. Analysis of the stored prey in the nest-holes of the Tengmalm's Owl (*Aegolius funereus*). Vogelwelt 109:176-181.

Scott, V. E., G. L. Crouch, and J. A. Whelan. 1982. Responses of birds and small mammals to clearcutting in a subalpine forest in central Colorado. United States Department of Agriculture Forest Service Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado, USA. Research Note RM-422.

Scrivner, J. H., and H. D. Smith. 1984. Relative abundance of small mammals in four successional stages of spruce-fir forest in Idaho. Northwest Science 58:171-197.

Smith, D. G., A. Devine, and D. Walsh. 1987. Censusing Screech Owls in southern Connecticut. Pages 255-267 *in* R. W. Nero, R. J. Clark, R. J. Knapton, and R. H. Hamre, editors. Biology and conservation of northern forest owls. Symposium proceedings. United States Department of Agriculture Forest Service Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado, USA. General Technical Report RM-142.

Solheim, R. 1983a. Bigyny and biandry in the Tengmalm's Owl. Ornis Scandinavica 14:51-57.

Solheim, R. 1983b. Breeding frequency of Tengmalm's Owl *Aegolius funereus* in three localities in 1974-1978. Proceedings of the Third Nordic Congress of Ornithologists 1981:79-84.

Sonerud, G. A. 1985. Nest hole shift in Tengmalm's Owl *Aegolius funereus* as defense against nest predation involving long-term memory in the predator. Journal of Animal Ecology 54:179-192.

Sonerud, G. A. 1986. Effect of snow cover on seasonal changes in diet, habitat, and regional distribution of raptors that prey on small mammals in boreal zones of Fennoscandia. Holarctic Ecology 9:33-47.

Sonerud, G. A. 1989. Reduced predation by pine martens on nests of Tengmalm's Owl in relocated boxes. Animal Behaviour 37:332-333.

Sonerud, G. A., R. Solheim, and B. V. Jacobsen. 1986. Home-range use and habitat selection during hunting in a male Tengmalm's Owl *Aegolius funereus*. Fauna Norvegica Series C Cinclus 9:100-106.

Sonerud, G., R. Solheim, and K. Prestrud. 1988. Dispersal of Tengmalm's Owl *Aegolius funereus* in relation to prey availability and nesting success. Ornis Scandinavica 19:175-181.

Stahlecker, D. W., and J. W. Rawinski. 1990. First records for the Boreal Owl in New Mexico. Condor 92:517-519.

Stebbins, L. L. 1984. Overwintering activity of *Peromyscus maniculatus*, *Clethrionomys gapperi*, *C. rutilus*, *Eutamias amoenus*, and *Microtus pennsylvanicus*. Pages 301-314 *in* J. F. Merritt, editor. Winter ecology of small mammals. Carnegie Museum of Natural History, Pittsburgh, Pennsylvania, USA. Special Publication Number 10.

Steele, R., R. D. Pfister, R. A. Ryker, and J. A. Kittams. 1981. Forest habitat types of central Idaho. United States Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah, USA. Gen. Tech. Rep. INT-114. 138pp.

Swan, D., B. Freedman, and T. Dilworth. 1984. Effects of various hardwood forest management practices on small mammals in central Nova Scotia. Canadian Field-Naturalist 98:362-364.

Swem, T. Wildlife Biologist, U. S. Fish & Wildlife Service, Fairbanks, AK. [Personal communication]. March 1993.

Ure, D. C., and C. Maser. 1982. Mycophagy of red-backed voles in Oregon and Washington. Canadian Journal of Zoology 60:3307-3315.

Van Horne, B. 1981. Demography of *Peromyscus maniculatus* populations in seral stages of coastal coniferous forest in southeast Alaska. Canadian Journal of Zoology 59:1045-1061.

Van Horne, B. 1982. Niches of adult and juvenile deer mice (*Peromyscus maniculatus*) in seral stages of coniferous forest. Ecology 63:992-1003.

Van Horne, B., and J. A. Wiens. 1991. Forest bird habitat suitability models and the development of general habitat models. United States Fish and Wildlife Service Fish and Wildlife Research 8.

Vaughan, T. A. 1974. Resource allocation in some sympatric subalpine rodents. Journal of Mammalogy 55:764-795.

Voous, K. H. 1988. Owls of the northern hemisphere. Massachusetts Institute of Technology Press, Cambridge, MS.

Wallin, K., and M. Andersson. 1981. Adult nomadism in Tengmalm's Owl *Aegolius funereus*. Ornis Scandinavica 12:125-126.

Watson, A., and R. Moss. 1980. Advances in our understanding of the population dynamics of Red Grouse from a recent fluctuation in numbers. Ardea 68:103-111.

Wells-Gosling, N., and L. R. Heaney. 1984. *Glaucomys sabrinus*. Mammalian Species 229:1-8.

Whelton, B. D. 1989. Distribution of the Boreal Owl in eastern Washington and Oregon. Condor 91:712-716.

Whitney, P., and D. Feist. 1984. Abundance and survival of *Clethrionomys rutilus* in relation to snow cover in a forested habitat near College, Alaska. Pages 113-119 *in* J. F. Merritt, editor. Winter ecology of small mammals. Carnegie Museum of Natural History, Pittsburgh, Pennsylvania, USA. Special Publication 10.

Williams, O. 1955. Distribution of mice and shrews in a Colorado montane forest. Journal of Mammalogy 36:221-231.

Wyman, L. E. 1915. Richardson's Owl in Illinois. Auk 32:101.

Wywialowski, A. P. 1985. Preferences of red-backed voles and deer mice for differing structural habitats. Dissertation. Utah State University, Logan, Utah, USA.

Yunick, R. P. 1979. A review of New York State Boreal Owl records. Kingbird 29:181-189.

RESPONSE OF BOREAL OWL TO EPIDEMIC MOUNTAIN PINE BEETLE-CAUSED MORTALITY UNDER A NO-ACTION ALTERNATIVE

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INTRODUCTION

Boreal owls (*Aegolius funereus*) occur throughout the boreal forests of Canada and Alaska and in subalpine forests of the Rocky Mountains north of central New Mexico (Hayward and Hayward 1993). Throughout this range, they occupy an array of forest types but in the central Rocky Mountains are restricted to subalpine aspen or conifer forests. In Colorado and Wyoming the owl is restricted to forest habitats within the forested life-zone supporting lodgepole pine and above. Quality boreal owl habitat is characterized by landscapes of mature and older forest stands – forest structure influences availability of suitable cavities for nesting, quality of roost sites especially during summer heat, foraging movement of the owls, and prey availability (Hayward et al 1993, Hayward and Varner 1994). Predicting the response of individual owls and populations to extensive forest mortality from mountain pine beetles requires an understanding of limiting factors and the ecological relationship between boreal owls and habitat conditions. In this brief introduction I review understanding of habitat associations of boreal owls at multiple spatial scales, examine limiting factors, and describe the consequences of changes that will occur following pandemic beetle induced mortality in lodgepole pine forest.

Nesting Habitat. Boreal owls are the largest secondary cavity nesting bird in subalpine forests of Colorado and Wyoming. Large aspen as well as conifer trees are used for nesting by boreal owls. Suitable nest cavities are rare; cavity entrances excavated by woodpeckers smaller than northern flicker (*Calaptes auratus*) must be enlarged by squirrels, or other means, to be used by boreal owls. The distribution and abundance of suitable cavities likely limits populations of boreal owls throughout much of this Region (Hayward 1997). Therefore, factors that increase or decrease the distribution and abundance of suitable cavities will influence both distribution and abundance of boreal owls (see section below on prey populations for more complete understanding). Evidence from experiments in Idaho suggest that nest cavities in forest stands with mature and old forest structure are preferred by boreal owls but successful nesting can occur when cavities are available in younger forests (Hayward et al 1993). Thus, watersheds that do not support snags or dead-top trees large enough for a cavity (approximately 13” dbh) are unlikely to provide habitat for boreal owls.

Home Range. The spatial distribution of suitable cavities will influence broad scale abundance of boreal owls differently than Strix owls such as spotted owls (*Strix occidentalis*). Boreal owls have large home ranges (summer home range $\bar{x}$ = 1,182 ha in Idaho) but do not defend a large territory (Hayward and Hayward 1993). Therefore, several individuals may nest within one km^2, and occupy overlapping territories in the surrounding landscape. Although little empirical evidence is available to determine densities, long-term studies in Finland found maximum densities of 26 nests /100 km^2. This figure likely exceeds densities in the Rockies by 2-3 times because of the influence of vole cycles on *Aegolius* abundance and reproduction

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(Hayward 1997). Furthermore, any suitable foraging habitat further than 5 km from suitable nest sites is unlikely to be used.

Foraging Habitat. Boreal owls are forest hunting owls. Southern red-backed voles (*Clethrionomys gapperi*) are the primary prey for boreal owls throughout the central Rocky Mountains. Investigations in a variety of settings demonstrate that red-backed voles (and small mammal biomass), is highest in mature and old forest stands within the subalpine zone (Hayward and Hayward 1995, Hayward et al. 1999). Mature and old forest habitats are an important component of foraging habitat but are not necessarily a habitat requirement (see definition Peek 83:1986). Observations of foraging boreal owls demonstrate that they forage in a range of forest structures. Seedling, sapling, and dense pole stands are generally unsuitable for foraging and observations suggest the owls forage most effectively in mature and older forests. Maybe most important, forest canopy may be critical in mid to late winter for suitable foraging habitat. These mid-size forest owls occupy landscapes with winter snow depths over 1 m and often much deeper. Meadows and forest openings will develop crusted snow during the course of the winter. Therefore, during mid and late winter, boreal owls favor foraging in forests where canopy conditions lead to less snow crusting and greater access to prey (prey moving to the surface and the opportunity for shallow plunge diving) (Sonerud 1986).

Broad-scale Habitat. The extent of mature and old forest habitat providing quality foraging habitat likely interacts with nest cavity abundance to determine the abundance of boreal owls at spatial extents of 10,000's of ha. Scott (2004) examined nesting success of boreal owls in landscapes that varied significantly in the arrangement and amount of forest in different seral stages. Her results suggest that the arrangement of forest patches (e.g., patch size, amount of edge, and patch location) was not important. However, landscapes with less than 23% mature and old forest were unlikely to support successful nesting boreal owls. This information along with an understanding of boreal owl nesting behavior suggests that boreal owls are likely to occupy subalpine forest watersheds if land units of approximately 2,000 ha support both large nest cavities and at least 20-25% of the area is occupied by living, mature and older forest.

The response of boreal owls to broad scale disturbance such as fire and beetle kill has not been investigated. However, unpublished monitoring results from a system of over 450 nest boxes in central Idaho provides some insights. Boreal owl breeding abundance was monitored from 1988 – 2000 using the system of nest boxes (see initial pattern in Hayward et al. 1992) in a setting of mixed subalpine forest types (aspen, lodgepole, and spruce/fir) that experienced a large fire in 1994. The populations exhibited a delayed response – the number of breeding owls declined by over 50% two years later and by almost 95% after 3 years in response to approximately 50% of the landscape being burned. During the next 3 years owl nesting abundance appeared to stabilize at about half the pre-fire level.

Assumptions Regarding Tree Mortality

This document currently assumes that lodgepole pine mortality will be close to 100% for trees over 6 " DBH. Lodgepole pine mortality is assumed to extend from central Colorado to northern Wyoming (Figure 1[2]). Nearly 1.6 million acres of forest formerly dominated by lodgepole pine will begin various succession pathways as a result of the disturbance. Throughout the paper I also assume that spruce beetle will not expand, and therefore I address mortality in lodgepole pine alone. The paper also assumes that wildfire will occur in some locations but the spatial extent and location of those fires can not be predicted and will be strongly influenced by weather events. Therefore, in terms of predictions regarding the no-action-alternative, I assume that fire will not occur in project areas being evaluated and the immediate area. When fires occur,

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[2] The figure information will be provided in next draft.

depending on extent and intensity, a changed condition should be assumed and analyses adjust to those specific changes.

Assumptions Regarding Climate Change

At this time I am unaware of climate change response information providing predictions of future distribution and composition of forests in the current subalpine life zone within Region 2. Therefore, predictions outlined below assume that the match between climate and regeneration niches for lodgepole, aspen, Englemann spruce, and subalpine fir will be similar during regeneration after the pandemic as in the previous sere.

EXPECTED RESPONSE TO NO-ACTION ENVIRONMENT

This section describes the predicted response of individuals and populations of boreal owls to beetle induced lodgepole pine tree mortality. Empirical investigations provide little insight into the pattern of response; therefore, predictions are based largely on ecological insights based on owl life history and an understanding of the pattern of forest change described in …[3].

Short-term Effects (1-5 years)

Fine-scale: Pure Lodgepole. (*Gradual decline in habitat quality especially after 3 years).* Landscapes without significant spruce-fir forest will support limited numbers of boreal owls prior to, and immediately following forest mortality. The response by boreal owls to beetle infestation during the initial stage of mortality will be gradual. Individual owls will likely continue to occupy the same home ranges the first 3 years, adult owl mortality will remain similar to pre-disturbance levels the first 3 years, and reproduction will begin to decline about 3 years after widespread tree death. Winter foraging conditions will begin to decline dramatically after 3 years and changes in winter home ranges may be dramatic or individuals will focus foraging on those stands with sufficient canopy to shade the snow late in the winter when snow crusting becomes an important factor influencing prey availability. Breeding cavities will gradually increase in abundance over time but this increase in availability will not lead to increased owl numbers because winter prey will become increasingly limited. *[High certainty]*

Prey population levels will likely remain at pre-disturbance levels until 3 years following initial mortality. At this point, seed rain will have declined significantly and the forest floor will begin to become more xeric. Access to prey will change dramatically in winter as canopy shading declines with loss of dead needles and fine limbs. This loss of shading will allow more snow crusting to occur in late winter. *[Moderate certainty]*

Fine-scale: Lodgepole with Spruce-Fir. (*Potential improvement in habitat quality).* The response of boreal owls to forest mortality in landscapes with lodgepole pine mixed with spruce-fir will be directly related to the proportional representation of spruce-fir forest. In areas where lodgepole pine occupies less than 25% of a watershed, the short-term response will likely be immeasurable. In landscapes with less spruce-fir forest, negative influences on boreal owls will increase in proportion to the representation of spruce-fir forest. *[High certainty]*

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[3] Plan to cite Regan et al (2008).

In mixed landscapes, lodgepole pine represents secondary foraging habitat. If patches of lodgepole pine are small (approximately 4 ha), the results of beetle mortality will be similar to patch cutting and could represent POSITIVE effects for the owls if diversity of prey increases in small openings. *[Moderate certainty]*

Before and after lodgepole mortality, most nesting will occur in spruce or aspen. Therefore the disturbance is unlikely to significantly influence nest site availability. Similarly, boreal owls will roost in remaining spruce-fir (thus little negative effect). Effects then, will result largely from foraging habitat loss and will be proportional to the loss of foraging habitat, in this case the inferior, secondary habitat represented by lodgepole pine. *[High certainty]*

Broad-scale. *(Variable: stationary population to decline depending on amount of spruce-fir forest).* As described above, the influence of lodgepole mortality on boreal owls depends critically on the proportion of the landscape supporting spruce-fir and aspen forest. Because, lodgepole pine is secondary nesting, foraging, and roosting habitat for boreal owls, landscapes over 10,000 acres with over 50% spruce-fir forest will experience little change in boreal owl abundance following bark beetle mortality over the short-term. The remaining spruce-fir will continue to provide adequate, quality suitable habitat in these areas. *[High certainty]*

In contrast, landscapes dominated by lodgepole pine will begin to experience declines in abundance and distribution of boreal owls by the end of 5 years, post beetle-induced mortality. Major changes in owl abundance should not be expected for the first 3 years while trees retain needles (providing shade on winter snow). However, after needle drop, rapid decline, especially in winter foraging habitat quality, is likely and the suitability of these lodgepole pine landscapes will decline. Aspen provides potential nesting habitat but, like dead lodgepole does not provide winter foraging. Therefore, landscapes dominated by lodgepole pine and aspen will experience similar response by boreal owls in winter. Because the owls are year- round residents, the change in winter foraging habitat influences breeding densities, and ultimately population size. Since these habitats are secondary in quality and do not support high populations of owls, overall large scale population losses are not expected to be significant. *[Modest certainty]*

Region-wide. (*Decline in abundance and distribution)*. Within the area currently influenced by mountain pine beetle, lodgepole pine represents about 40% of the landscape in the subalpine zone. Throughout much of the region, lodgepole pine occurs in large patches immediately below spruce-fir forest. Near this transition, lodgepole pine and spruce-fir forest are intermixed. Even where lodgepole pine dominates watersheds, small inclusions of spruce-fir forest occurs. Therefore, although lodgepole pine is marginal (rather than high quality) habitat for boreal owls, these forests provide some resources for the owl. As the bark beetle pandemic moves through the region in a wave from south to north, measurable but small declines in boreal owl abundance are likely.

Mid-term Effects (6-20 years)

Fine-scale: Pure lodgepole. (*Decline in habitat quality).* Within 6 years of forest mortality, boreal owls are likely to cease using lodgepole forest for winter foraging. Because trees will be available for perching, owls are likely to use these forests for summer foraging habitat through 10-15 years post-mortality. However, by the end of 20 years, the loss of branches and the falling of snags will reduce summer foraging habitat to near zero except along the edges of spruce-fir or aspen stands. Nesting cavity availability will increase through the first 20 years however the extent of use by woodpeckers will depend on rot characteristics. In any case, the in crease in nesting habitat is unlikely to be used because foraging habitat will be limiting. *[Moderate to high certainty]*

Fine-scale: Lodgepole with Spruce-Fir or Aspen. (*Potential improvement in areas dominated by spruce-fir).* As described for short-term effects, the response of boreal owls to lodgepole pine mortality in landscapes with spruce-fir will be directly related to the proportional representation of spruce-fir forest. In areas where lodgepole pine occupies less than 25% of a home range, the mid-term response will likely be immeasurable or could be a positive response due to the high quality mature spruce-fir and aspen remaining. Increased herbaceous cover is likely to result in small-mammal population changes in pine mortality stands. This change will add to prey diversity (beta diversity when considering neighboring spruce-fir and aspen forests). Spruce-fir forest will provide nesting habitat, foraging perches, and winter foraging habitat. In landscapes with less spruce-fir forest, negative impacts on foraging habitat quality will dominate because lodgepole pine forest represented a large proportion of the initial foraging habitat. *[Moderate to high certainty]*

Regeneration of aspen during the first 20 years will have little influence on boreal owls. Regenerating aspen will begin to provide foraging structure toward the end of this period but is unlikely to influence habitat quality significantly. *[High certainty]*

Broad-scale. (*Depends on proportion of other forest types).* In landscapes dominated by lodgepole pine, significant declines in boreal owl habitat will be observed from 6-20 years post-mortality [because of spatial extent of LP and virtual complete loss of function as habitat]. By the end of this period, large patches of former lodgepole pine will be non-habitat for any life function and both distribution and abundance of boreal owls will decline. [High certainty]. In landscapes with higher proportions of spruce-fir and aspen forest, the decline in boreal owls will be proportionally less and at the extreme, when lodgepole pine represented patches of forest within a larger matrix of spruce-fir forest, boreal owl populations may be more stable as a result of greater diversity of prey. *[Moderate to high certainty]*

Region-wide. (*Decline in distribution and abundance).* By the end of 20 years post-mortality, a majority of the 1.6 million acres of lodgepole pine forest in beetle affected portions of Colorado and Wyoming will be non-habitat for boreal owls. The primary exception to this is along the edges of mature spruce-fir stands where owls will persist and will forage in the edges of the openings created by the dead LPP. Assuming that at least 75% of the spruce-fir forest and at least 60% of the mature aspen forest in this region is alive, boreal owls will decline by approximately 10-15%.[4] This is based on the assumption that a unit area of lodgepole pine forest provides 25% of the value of a unit area of spruce-fir forest for foraging. Boreal owls will be absent from areas larger than 2000 ha that formerly were 85% or more lodgepole pine.

Long-term Effects (21-80 years)

Fine-scale: Pure lodgepole. (*Non-habitat through 40 years with gradual increase in foraging habitat quality thereafter).* Seedling, sapling, and dense pole stands will provide no nesting habitat, limited roosting habitat especially during summer, and very limited foraging habitat. As stands begin to self-thin, or in stands that are more open grown with space between crowns, boreal owls may begin to forage during summer. However stands are unlikely to provide measurable foraging habitat value prior to 50 years post-mortality and the baseline foraging value is unlikely to be restored till 80-100 years post-mortality (and recall that the baseline foraging value in lodgepole pine is not high compared to spruce-fir forest). *[Low to moderate certainty because these conditions have not been examined.]*

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[4] Note that these ## are simply an educated guess based on the assumptions set up in this paper. We will modify this estimate in the winter if we can exercise a model for the region.

Fine-scale: Lodgepole with Spruce-Fir or Aspen. (*Declining conditions through 40-50 years post-mortality and improving conditions thereafter for home ranges dominated by spruce-fir*). In areas where lodgepole pine occupies less than 25% of a home range, foraging habitat conditions will decline from 21-40 or 50 years as the lodgepole stands become dense stands of regeneration. As these stands self-thin, or in open stands, foraging conditions will improve from 40/50 years through 80 years. The lodgepole component will not provide nesting habitat or quality roosting habitat through this time period. *[Moderate to high certainty]*

Regeneration of aspen on stands formerly dominated by lodgepole pine will provide limited summer foraging habitat, no winter foraging habitat, no roosting habitat, and no nesting habitat through 40 years post-mortality. From 40 – 80 years summer foraging habitat will increase in quality but winter foraging habitat remain poor because of snow crusting. Aspen may begin to provide nesting habitat from 60-80 years post-disturbance. *[Moderate certainty]*

Broad-scale. (*Depends on proportion of other forest types).* In landscapes dominated by lodgepole pine, boreal owl habitat will not be available through 40/50 years post-mortality. Foraging and roosting habitat will gradually improve from 50 – 80 years returning to pre-disturbance levels somewhere between 80 and 120 years. However, nesting habitat will be limited during this period. *[Moderate to high certainty]* In landscapes with higher proportions of spruce-fir and aspen forest, the condition for boreal owls will depend on the proportion of lodgepole pine in the former forest. Stands of primarily LPP will remain secondary habitat. In all cases, however, habitat quality will improve from 40/50 years through 80 years. *[Moderate to high certainty]*

Regeneration of aspen on stands formerly dominated by lodgepole pine will provide beta diversity and therefore potentially greater stability in prey populations compared to pre-disturbance conditions. Regenerating aspen will provide improving summer foraging habitat throughout this period but little winter foraging, roosting, or nesting habitat. From 60 to 80 years post pandemic, aspen will begin to provide nesting habitat. *[Moderate certainty]*

Region-wide. (*Stable habitat and populations with improvement in habitat and increases in populations toward end of period – populations below pre-pandemic levels).* Through a majority of the period from 20-80 years post-pandemic, boreal owl populations are likely to remain at reduced distribution and abundance similar to levels experienced following the loss of canopy within a decade of beetle attack.[5] By the end of this period, however, regenerated aspen and lodgepole stands will be providing foraging habitat and contributing to recovery of regional boreal owl populations. *[Moderate certainty]*

Very Long-term Effects (81-200 years)

During this timeframe, in the absence of widespread disturbance events such as fire or mountain pine beetle, or if climate change has not substantially changed the distribution of lodgepole pine, habitat conditions will become similar to those experienced immediately prior to the mountain pine beetle pandemic as the landscape develops more advanced seral stages.

Fine-scale: Pure lodgepole. (*Foraging habitat quality will continue to improve and nesting habitat develop – conditions will begin to approach the quality of mature spruce-fir forest).* Foraging habitat quality will improve throughout the period as stands become more mesic, fine-scale diversity increases through gap

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[5] Would like to add some quantitative statement here regarding the % reduction since pre-beetle, but will need to wait to conduct some modeling.

processes, multi-storied stands develop, and a variety of course woody debris conditions develop. Trees will reach size and stage to facilitate suitable cavity development providing breeding opportunities. For the later third of this period, habitat conditions are likely better than prior to stand mortality. *[Moderate certainty.]*

Fine-scale: Lodgepole with Spruce-Fir or Aspen. (*Steady improvement and continuing excellent habitat conditions).* These stands will have supported high quality habitat since the mountain pine beetle epidemic and will continue to support excellent habitat. Furthermore, habitat quality will continue to improve as stands become older. *[Moderate to high certainty]*

Broad-scale. (*Foraging, roosting and nesting habitat improve and become very high quality).* In landscapes dominated by lodepole pine, boreal owl habitat will be present at 80 years and develop to very high quality by 150 years. This will include foraging, roosting and nesting habitat and conditions will be superior to pre-pandemic conditions (assuming these stands are older and more diverse than pre- pandemic stands). *[Moderate to high certainty]* In landscapes with higher proportions of spruce-fir and aspen forest, the condition for boreal owls will also improve but the degree of improvement will be less than in lodgepole dominated landscapes because there is less room for improvement. In any case, these landscapes will provide superior habitat. *[Moderate to high certainty]*

Regeneration of aspen on stands formerly dominated by lodgepole pine will provide beta diversity and conditions superior to those experienced prior to forest mortality. These stands will provide nesting, foraging and roosting habitat as a result of the mixed deciduous/conifer characteristics. Toward the end of this period (150-200 years), aspen may be lost from the stands reducing their quality for boreal owls because of the loss of habitat diversity and loss of cavities represented by the aspen. If aspen snags persist, nesting habitat will be improved. *[Moderate certainty]*

Region-wide. (*Habitat quality and boreal population abundance will likely exceed those experienced prior forest mortality (because of aspen)).* Through the first 50 to 70 years of this period, owl habitat will be high quality but still improving. Habitat quality will remain high through the remainder of the period with the potential for slight declines in quality in areas with aspen, if the aspen begins to decline significantly. Owl populations will be at or exceed pre-pandemic levels by 120 to 150 years post-mortality, and remain high throughout. *[Moderate certainty]*

SUMMARY

Short-term Response (1-5 years)

Boreal owl populations will exhibit a delayed response to forest mortality. Population declines are unlikely to be observed until 4-5 years after stand mortality, 1-2 years after red-needle fall. Region-wide, populations will decline only slightly within the first 5 years.

Mid-term Response (6-20 years)

The most important declines in boreal owl populations will be observed during this period. Owl populations will decline rapidly beginning 4-6 years post mortality and continue to decline through the period. However, because spruce-fir forest represents the most important boreal owl habitat, declines will be limited to areas dominated by lodgepole pine and Region-wide the decline will not be large. Boreal owls will become absent from home ranges and watersheds that were largely lodgepole pine forest but persist at levels similar to before the beetle pandemic in areas dominated by spruce-fir forest. *[Need modeling to give a sense of overall decline]*

Long-term Response (21-80 years)

Boreal owl populations in areas dominated by lodgepole pine will remain low through the first half of this period and begin to increase gradually there-after. The most rapid changes will occur in landscapes formerly dominated by lodgepole pine where aspen regenerate mixed with conifer. Throughout the region, increases will be slow throughout the last 1/3 of this period. *[Need modeling to give a sense of overall decline]*

Very Long-term Response (81-200 years)

Habitat conditions will continue to improve during this period becoming superior to those present prior to the pandemic as old forest conditions develop and aspen regeneration provides increased nesting habitat.

Unique Circumstances and Dominant Uncertainties

Predictions regarding the response of boreal owl to widespread mortality of lodgepole pine are based largely on intensive studies of boreal owl ecology in central Idaho. Limited investigations on the Arapahoe and Grand Mesa National Forests provide more local information that corroborates the investigations from Idaho. The response of boreal owls to regenerating aspen forest is particularly difficult to predict.

The influence of climate change on patterns of vegetation development, on small mammal populations, and on boreal owls is also unknown. Boreal owls are easily heat stressed. Therefore, rising summer temperatures may force the owls to higher elevations or to rely on cool forest micro-sites to a greater extent. Extending the snow-free period may reduce small mammal populations. Red-backed voles seek the sheltered climate conditions in the subnivian environment during winter and some evidence suggests the voles experience increased mortality during years of late snowfall. Finally, the influence of climate change on seed production, fungi production, and *Vaccinium,* all important foods for red-backed voles, is unknown.

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LITERATURE CITED

Hayward, G. D. 1997. Forest management and conservation of boreal owls in North America. J. Raptor Research 31:114-124.

Hayward, G. D. and P. H. Hayward. 1993. Boreal owl *in*: A. Poole, P. Stettenheim, and F. Gill, eds., Birds of North America. No. 63. Philadelphia Academy of Natural Sciences, Washington, D.C., 20pp.

Hayward, G. D., P. H. Hayward, and E. O. Garton. 1993. Ecology of boreal owls in the northern Rocky Mountains, USA. Wildlife Monographs 59 pp.

Hayward, G. D. and P. H. Hayward. 1995. Relative abundance and habitat associations of small mammals in Chamberlain Basin, central Idaho. Northwest Science. 69:114-125

Hayward, G. D., S. H. Henry, and L. Ruggiero. 1999. Response of red-backed voles to recent patch cutting in subalpine forest. Conservation Biology 13:168-176

Hayward, G. D., R. K. Steinhorst, and P. H. Hayward. 1992. Monitoring boreal owl populations with nest boxes: Sample size and cost. J. Wildl. Manage. 56:777-785

Hayward, G. D. and J. Verner (eds). 1994. Flammulated, boreal, and great gray owls in the United States: A technical conservation assessment. U. S. Forest Service, Gen Tech Rep. RM-253. 214 pp

Peak, J. M. 1986. A review of wildlife management. Prentice-Hall. 486 pp.

Sonerud, G. A. 1986. Effect of snow cover on seasonal changes in diet, habitat, and regional distribution of raptors that prey on small mammals in boreal zones of Fennoscandia. Holarctic Ecology 9:33-47.

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by

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FRONTISPIECE. Radio-marked male boreal owl (*Aegolius funereus*) in spruce–fir forest of central Idaho (photo by Patricia H. Hayward). The owl holds a recently captured southern red-backed vole (*Clethrionomys gapperi*), which was the most frequent prey species for both male and female owls.

ECOLOGY OF BOREAL OWLS IN THE NORTHERN ROCKY MOUNTAINS, USA

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Abstract: We studied habitat use by boreal owls (*Aegolius funereus*) in the northern Rocky Mountains from January through August during 1984–88. Habitat use was examined at several spatial scales. The geographic distribution and range of life zones used by boreal owls were documented in western Montana, Idaho, and northwestern Wyoming. Habitat use, at the level of the home range, and microhabitats used for nesting, roosting, and foraging were observed in the Frank Church River of No Return Wilderness (RNRW) of central Idaho.

Boreal owls inhabited forests within the spruce–fir (*Picea* spp.-*Abies* spp.) life zone throughout the mountains of Montana, Idaho, and northern Wyoming. Nearly 90% of breeding territories located throughout this region were in subalpine fir (*Abies lasiocarpa*) habitat types (based on Steele et al. 1981), and no owls were detected below 1,292 m. Within the RNRW, owl breeding sites occurred (n = 28) in mixed-conifer (39%), spruce–fir (25%), Douglas-fir (*Pseudotsuga menziesii*) (18%), and aspen (*Populus tremuloides*) (18%) stands. Lodgepole-pine (*Pinus contorta*) forest, which was the most common vegetation type in the area, was not used for nesting. Nest sites were restricted to mature and old forest stands with complex physical structure. Roosting habitat differed between winter and summer. Winter roosts differed little from available forest cover whereas summer roosts had greater canopy cover, higher tree density, and higher basal area than paired random sites. During summer, the owls used cool microsites for roosting; during hot weather, boreal owls frequently exhibited symptoms of heat stress by gullar fluttering and perching with wings lowered. The best foraging habitat was associated with older spruce–fir stands. These sites had prey populations 2–10 times greater than other sites and provided open forest structure that facilitated hunting. Because of the wide geographic dispersion of suitable nesting, roosting, and foraging habitat, the owls used large home ranges; ranges averaged 1,451 ± 552 ha in winter (n = 13) and 1,182 ± 335 ha in summer (n = 15).

Boreal owls at our intensive study site fed primarily on small mammals in both winter and summer. During both seasons, southern red-backed voles (*Clethrionomys gapperi*) were the most frequent prey, composing 36% of all prey items. In winter, northern flying squirrels (*Glaucomys sabrinus*) were captured by females (14% of prey items) but not males. During snow-free seasons, boreal owls captured northern pocket gophers (*Thomomys talpoides*), yellow-pine chipmunks (*Tamias amoenus*), and western jumping mice (*Zapus princeps*) that were unavailable when the ground was snow covered.

The size of breeding populations and breeding success varied from year to year. During 1 of 4 years, few owls attempted to breed, and we knew of none that raised young. Estimates of demographic characteristics of boreal owls suggest that the population declined during our investigation. We estimated annual adult survival as 46% (95% CI = 23–91%), and production by nesting females averaged 2.3 (±0.542) young per successful nest. We documented long distance movements by adult owls, which support the contention that immigration of nomadic owls may help maintain populations that would otherwise face local extinction.

These results suggest that conservation of boreal owls will require forest management that maintains the distribution and abundance of mature and older forest stands. Because boreal owls in the northern Rocky Mountains occur in a narrow life zone, populations exist in isolated patches that cover a relatively small portion of the landscape. To maintain quality habitat at any given site will require snag retention and timber harvest practices that retain forest structure. A shift to uneven-age management or modifications of even-age systems that retain particular old forest characteristics in spruce–fir forest should meet the owls' needs for nesting structures and roosting sites while maintaining prey populations.

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CONTENTS

INTRODUCTION

Boreal owls (*Aegolius funereus*), referred to as Tengmalm's owl outside North America, occur throughout the holarctic in boreal climatic zones. On each continent, disjunct populations occur in mountains south of the broad transcontinental boreal forest populations, which extend from Scandinavia eastward across Asia and from Alaska through Canada to the Atlantic (Cramp 1977, Voous 1988). Boreal owls located in mountain regions of Europe and Asia have long been recognized as isolated resident breeding populations, whereas in North America, breeding status was only recently documented in the mountains of the western United States (Hayward and Garton 1983, Palmer and Ryder 1984, Hayward et al. 1987*a*, Whelton 1989). Studies of this species, mainly in Scandinavia, Germany, and France, indicate that the biology and ecology of boreal owls varies geographically and is strongly related to local forest conditions and prey populations.

Boreal owls inhabit forested habitats where they nest exclusively in tree cavities or artificial nest structures (Mikkola 1983). Nesting habitat includes a range of vegetation types depending on geographic region. In Scandinavia, studies report nests in artificial structures hung in pine (*Pinus* spp.), spruce (*Picea* spp.), and birch (*Betula* spp.) forest (Norberg 1964, Korpimaki 1981, Solheim 1983*a*). In France, "mountain pine" (*Pinus uncinata* and *P. sylvestris*) forest and old forest stands with beech (*Fagus* spp.) trees were used by owls located by Dejaifve et al. (1990:267) and Joneniaux and Durand (1987), respectively. In Germany, conifer forest with old trees was used for nesting (Konig 1969, Jorlitschka 1988). Nest sites in Canada and Colorado have occurred mainly in spruce, aspen (*Populus tremuloides*), and mixed forests (Bondrup-Nielsen 1978, Palmer 1986). Although foraging habitat has not been studied extensively, observations in Scandinavia suggest that old spruce forest is important, especially during winter. Clearcuts and cultivated fields are used for hunting only in early spring before vegetation becomes dense (Sonerud 1986, Korpimaki 1988*a*).

Small mammals, particularly voles (*Clethrionomys* spp. and *Microtus* spp.), are primary prey of boreal owls throughout their geographic range. Voles frequently account for over 75% of prey items identified in pellets and prey remains at nests (Klaus et al. 1975, Korpimaki 1981). Other prey include shrews, mice, birds, and insects (Cramp 1977). The range of prey consumed increases in southern regions of Europe where the owls are considered generalist predators (Solheim 1983*a*; Lofgren et al. 1986; Korpimaki 1986, 1987*a*). In northern regions with pronounced vole cycles, boreal owls function as microtine specialists and exhibit extreme fluctuations in breeding parameters (Korpimaki 1986). Breeding population size, laying date, clutch size, fledging success, and the frequency of bigyny and biandry all vary among years and geographically, corresponding to variation in prey availability (Carlsson et al. 1987, Korpimaki 1987*b*). Local breeding densities during good prey years exceed 4 pairs/km^2 whereas in poor years few owls initiate nesting (Lofgren et al. 1986, Schelper 1989). These studies demonstrate how geographic variation in prey populations influence local boreal owl populations.

The influence of prey populations on boreal owl ecology extends from population dynamics, mating system, and habitat use to yearly movement patterns (Mysterud 1970, Lundberg 1979, Lofgren et al. 1986). Korpimaki (1986) hypothesized that conflicting selective pressures of nest site scarcity favoring site tenacity and fluctuating prey availability favoring nomadism have resulted in a variety of movement patterns in boreal owl populations. Where boreal owls consume a varied diet and prey availability fluctuates little, populations are resident. Under conditions of reduced prey diversity and moderate prey fluctuations, populations exhibit a strategy of partial migration (males are resident and females nomadic) whereas in areas with pronounced vole cycles, adults are nomadic (Korpimaki 1986, Lofgren et al. 1986). Migratory status in North America has not been systematically investigated. In eastern and central North America, irruptions are evident (Catling 1972, Evans and Rosenfield 1977) but the phenomenon has not been noted in the West.

Based on the variety of ecological patterns described for boreal owls in Europe, it is difficult to predict the ecological characteristics of populations in western North America. The relationship among boreal owls, forest habitat, and small mammal populations, however, suggests that understanding patterns of habitat use and prey relationships will be central to the formulation of conservation programs and to understanding the owl's ecology (Scott et al. 1987).

In this study, we investigated habitat use by boreal owls in the northern Rocky Mountains of the United States. We approached the study of habitat from a broad perspective by investigating natural history and by examining habitat use on several scales (i.e., several levels of resolution from a regional to microhabitat scale). Prior to our investigation, information on boreal owl habitat use in North America was limited to a study in Canada (Bondrup-Nielsen 1978) and a concurrent investigation in Colorado (Palmer 1986). Established breeding populations of boreal owls had been observed in central Idaho and Colorado (Hayward and Garton 1983, Palmer and Ryder 1984); otherwise, the extent of the species' distribution south of Canada was completely unknown.

Our goal was to describe habitat use by boreal owls at several scales emphasizing regional distribution and microhabitat use. We also sought to describe population characteristics (e.g., population trend, productivity), both to place the habitat data in an ecological context and to assess the species' current status. To meet these goals we pursued 5 specific objectives: (1) to estimate the geographic extent of boreal owl populations in the northern Rocky Mountains within the United States, (2) to estimate breeding habitat associations and life zone of boreal owls in this region, (3) to describe habitat use for nesting, roosting, and foraging in 1 local population of boreal owls, (4) to estimate seasonal and yearly movement patterns of boreal owls in 1 local population, and (5) to describe demographic characteristics of boreal owls in 1 local population.

Acknowledgments.—We appreciate the assistance of L. Flaccus, A. L. Wright, and D. Zebley who worked tirelessly collecting data. R. Ryker helped secure initial funding. R. J. Gutiérrez, R. L. Hutto, M. G. Raphael, and A. L. Wright graciously reviewed the manuscript and offered insights. We thank the biologists who cooperated in regional surveys for owls in Montana, Idaho, Wyoming, and Utah. The U.S. Forest Service Intermountain and Rocky Mountain Forest and Range Experiment Stations, Idaho Department of Fish and Game, University of Idaho Wilderness Research Center, University of Idaho Forest, Wildlife, and Range Experiment Station (contribution no. 666), Max McGraw Wildlife Foundation, North American Bluebird Society, TDK Electronics Corporation, Duracell Corporation, Richland Washington Audubon Society, Susan Hurd family, and the Robert Hayward family provided funding and support; we thank each of them.

STUDY AREA

To examine habitat use at multiple geographic scales, we examined broad patterns through extensive field studies at a regional scale (covering several states in the northern Rocky Mountains) and examined fine scale patterns on a local scale (including 2 study areas, each encompassing over 6,500 ha, in the wilderness of central Idaho). We allocated our effort disproportionately, emphasizing field work at the intensive study site.

Our broad scale surveys for boreal owls included portions of the northern Rocky Mountains from northern Idaho and Montana to northern Wyoming and Utah. Cooperators throughout this region surveyed for owls on 13 national forests, including the Idaho Panhandle, Kootenai, Flathead, Lolo, Lewis and Clark, Beaverhead, Nez Perce, Payette, Salmon, Boise, Caribou, Bridger-Teton, and Fishlake national forests. Owl surveys were restricted to coniferous forest but extended from lower conifer tree line (Ponderosa pine, *Pinus ponderosa*, forests) to upper timberline. Our investigation in this broad geographic region was limited to areas that had roads providing winter access to the higher elevation forests.

Our intensive investigation was in the mountains of central Idaho in the northern portion of the 956,515-ha Frank Church River of No Return Wilderness (RNRW) (Fig. 1). The study area extended from 1,580 to 2,400 m elevation in a high, dissected plateau surrounded on 3 sides by deep, rocky canyons that plunge to 700 m elevation along the Salmon River. The entire area is mountainous but lacks steep jagged peaks; few slopes exceed 50%.

Climate in the RNRW has a strong Pacific coastal influence during winter but follows continental patterns in summer (Finklin 1988). At 1,700 m elevation, Chamberlain Basin Guard Station receives 76 cm of precipitation per year (50% as snowfall). Snow depths reach 50–90 cm at

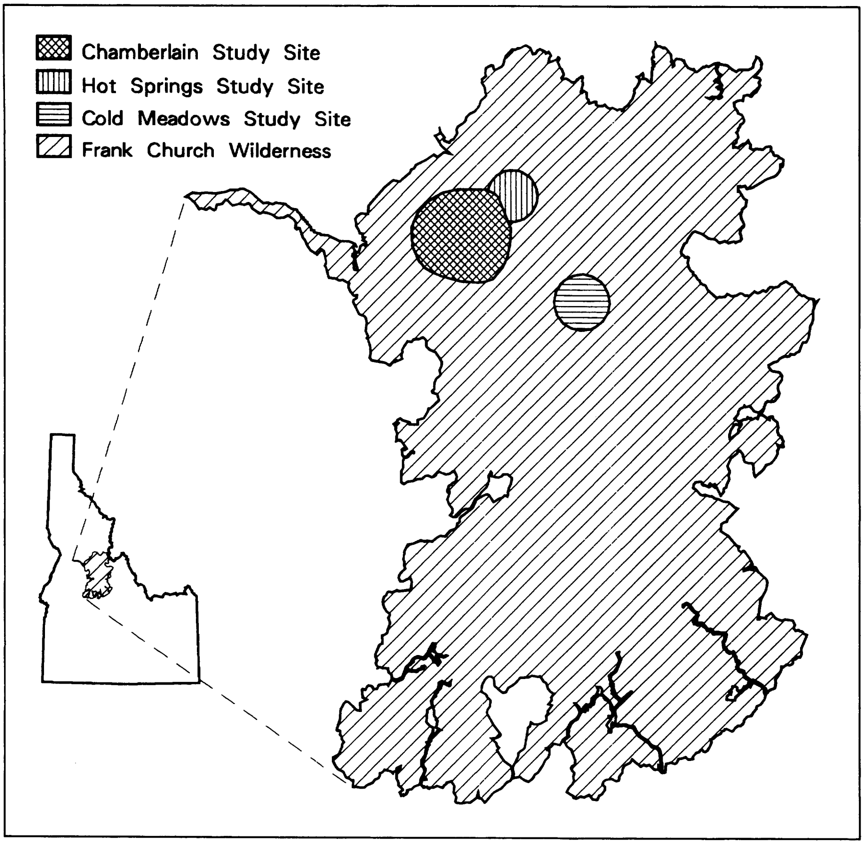

Fig. 1. Location of Chamberlain, Cold Meadows, and Hot Springs Meadow study sites in the 956,500-ha, Frank Church River of No Return Wilderness in central Idaho.

this elevation. At 2,150 m, 70% of precipitation is snowfall, and snow depths exceed 150 cm. The frost-free period at 1,700 m lasts 35–40 days. In July, maximum temperatures averaged 28 C with average minimums of 5 C; in February, daily temperature extremes averaged 5 C maximum and −20 C minimum at 1,700 m.

We established 3 study sites in the northern mountains of the RNRW—hereafter referred to as Chamberlain, Cold Meadows, and Hot Springs Meadow (Fig. 1). The Chamberlain site, where 90% of our field effort was centered, included a 35,000-ha portion of Chamberlain Basin, particularly the basin upstream from the junction of the West Fork and Chamberlain Creek. Cold Meadows included about 9,000 ha within 5.5 km of Cold Meadows Guard Station. This site is 30 km east of Chamberlain Basin Guard Station at 2,130 m elevation and provided access to lodgepole-pine (*Pinus contorta*) and spruce–fir forests. Hot Springs Meadow site, 8 km northeast of Chamberlain Basin Guard Station, included about 6,000 ha surrounding the Hot Springs Meadow cabin at 2,160 m and also provided access to spruce–fir forest.

The landscape of all 3 study sites within the wilderness study area was dominated by coniferous forest (Fig. 2). Prior to the initiation of the study, most forest stands in the area exceeded 125 years old except for a 2,225-ha area that burned in 1966. Lodgepole pine dominated the forest, especially in cold-air drainages where monospecific stands of this species covered areas exceeding 300 ha. Southern slopes below 1,825 m supported open forest of 60–130-

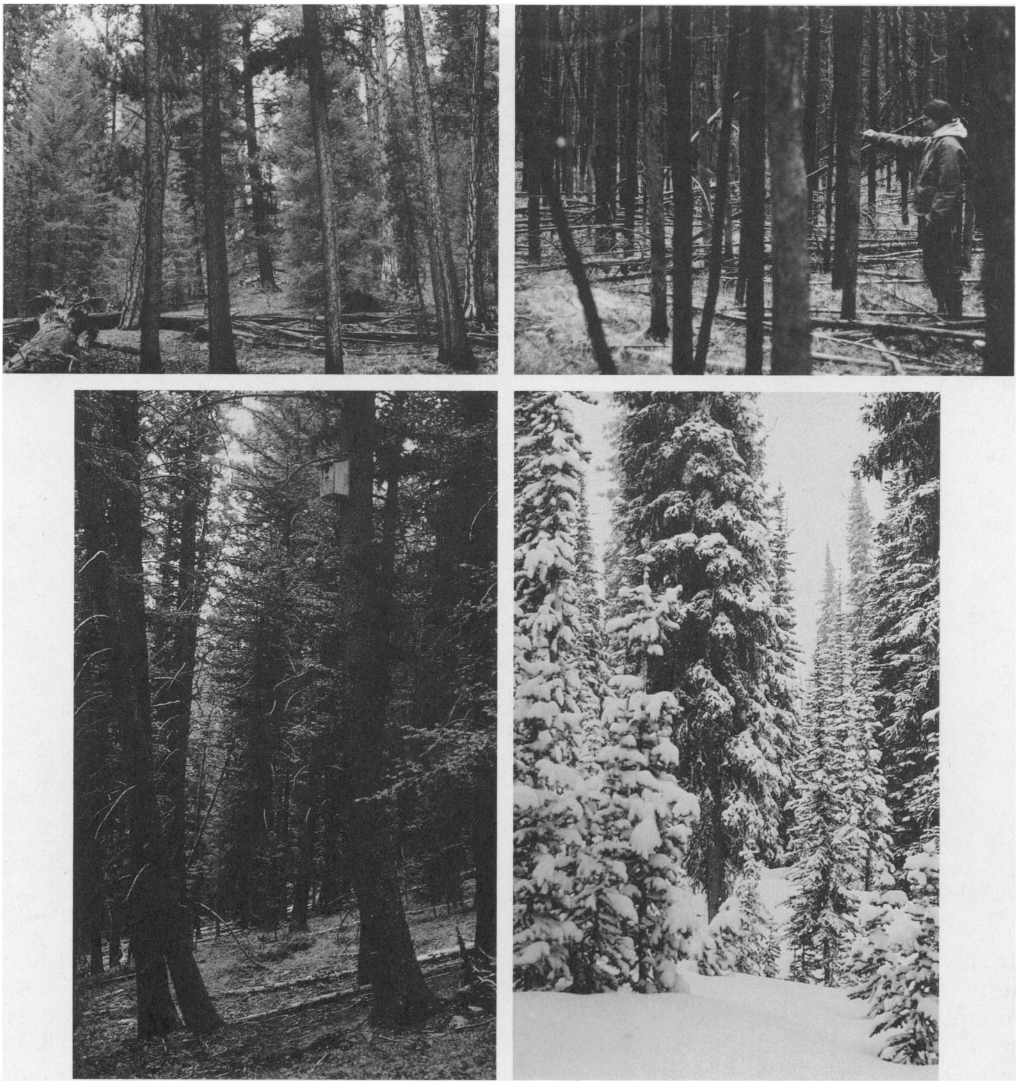

Fig. 2. Contrasting structure of 4 major conifer vegetation types that dominate the landscape of the Frank Church River of No Return Wilderness study sites. Stands shown here include (upper left) old mixed-conifer forest dominated by ponderosa pine on southeast exposure of Flossie Creek drainage; (lower left) old Douglas-fir forest on northeast exposure of Hotzel Creek drainage; (upper right) mature lodgepole-pine forest in frost pocket along Chamberlain Creek; and (lower right) mature spruce–fir forest on west exposure near top of Trout Creek drainage.

cm Ponderosa pine and Douglas-fir (*Pseudotsuga menziesii*). These forests are called mixed conifer throughout this text. On cooler aspects up to 2,000 m, Douglas-fir dominated unless recent fire had produced a lodgepole-pine forest. Douglas-fir forests were mixed with subalpine fir (*Abies lasiocarpa*) and Engelmann spruce (*Picea engelmannii*) up to 2,200 m. Above 2,000 m, spruce–fir forest occupied sites without recent fire.

Meadows and open shrub fields covered <3% of the area. Sagebrush–bunchgrass (*Artemisia* spp.–*Festuca idahoensis*) slopes were the most common open habitat below 1,850 m whereas *Carex* meadows and willow (*Salix* spp.) bogs occurred at higher elevations. Aspen occurred in small clumps (usually <0.5 ha) below 2,100 m and covered <1% of the landscape.

For the Chamberlain study site, we classified the landscape by dominant overstory vegetation using color 1:5800 aerial photographs. Rank order of landscape coverage from most to least dominant were lodgepole pine, Douglas-fir, mixed conifer, spruce–fir, meadow and shrub-field openings, and aspen. Lodgepole-pine forest covered over 50% of the Chamberlain site.

Other species of owls that were heard calling each year in the RNRW included northern pygmy owl (*Glaucidium gnoma*), northern saw-whet owl (*Aegolius acadicus*), great horned owl (*Bubo virginianus*), and great gray owl (*Strix nebulosa*). Barred owls (*S. varia*) called each year after 1984. Long-eared owls (*Asio otus*) were seen occasionally but may not nest in the area, whereas flammulated owls (*Otus flammeolus*) were heard during 3 years in the mixed-conifer stands. Two important primary excavators, pileated woodpecker (*Dryocopus pileatus*) and northern flicker (*Colaptes auratus*), breed in the area but are most abundant in the lower elevations.

DEFINITIONS

Several terms used loosely in the ecological literature are defined below to clarify our use of the terms:

1. A forest stand is a homogeneous portion of forest that can be differentiated from surrounding units by age, composition, structure, or geography (Daniel et al. 1979).
2. Microhabitat refers to conditions within an area smaller than a forest stand—a site where an individual owl performs a single activity such as roosting, nesting, or foraging.
3. Macrohabitat refers to characteristics of the environment in an area large enough to include an individual's seasonal home range. For boreal owls, forest stands are convenient elements to measure for macrohabitat description because home ranges are composed of numerous forest stands. Macrohabitat, then, generally includes several vegetation types and land forms and is quantified by the proportion of area covered by each.

We believed that existing definitions of forest age and structure, especially definitions of old growth, were not adequate for the forests of central Idaho (Hayward 1991). Here we apply the following working definitions of stand age (*see* Hayward 1991 for more details):

1. Young forest—forest with few or no seed-producing trees, where seedling establishment is common and leaf area is increasing. Structurally, a young forest stand is dominated by small trees that are growing vigorously.
2. Aggradation stage forest—a forest stand in which tree establishment is significantly reduced and competition has resulted in tree mortality, but stand structure is primarily a result of the major disturbance that created the stand. Aggradation stage forest, then, has trees of a single age class, new snags, and few seedlings.
3. Mature forest—a stand that has developed long enough since catastrophic disturbance that mortality and regeneration are prominent processes and regeneration results from parent trees in the stand. The mature stand has treefall gaps created after stand establishment and an uneven-age tree diameter distribution.
4. Old forest—a stand whose age and physical structure is currently influenced by processes within the stand rather than the last catastrophic disturbance. Old forest will have a wide variety of tree sizes and ages and a patchy structure resulting from tree mortality and regeneration.

FIELD METHODS

We conducted field work from 26 January to 28 September 1984, 26 January to 28 August 1985, 14 January to 12 October 1986, 16 February to 23 August 1987 and for 2 weeks in both February and May 1988.

Locating and Capturing Owls

Extensive Surveys in the Northern Rockies.—We estimated the regional distribution of boreal owls from winter surveys conducted in the northern Rockies (the Rocky Mountains south of Canada and north of 42°N). We divided the region included in these surveys into 2 groups based on differences in climate that result in different forest structures. The northern group included the Flathead, Idaho Panhandle, and Lolo national forests. This region experiences abundant rain and snowfall and moderate temperatures from maritime air masses (Arno 1979). Stands on the Beaverhead, Salmon, Caribou, and Payette national forests experience a continental climate with cooler winter temperatures and less summer moisture. These more southern forests support smaller trees and a less developed shrub layer.

During February–April 1984–86, personnel from 13 national forests, Grand Teton National Park, Wyoming Game and Fish, and the Garnett District of the Bureau of Land Management searched for boreal owls by foot, car, and snowmobile using playback of tape-recorded boreal owl staccato song (Bondrup-Nielsen 1978). Biologists were recruited for the survey with the understanding that our objective was to estimate the geographic extent and elevation range of the species in the northern Rockies. Therefore, biologists were not instructed to concentrate surveys in certain habitats or particular geographic locations. Surveys were conducted from 500 to 3,050 m elevation with over 80% being over 1,280 m. No explicit effort was made to repeat survey routes during any year; however, biologists did resurvey some areas.

Playback surveys began in evening when darkness permitted viewing the first stars. Biologists traveled forest roads stopping every 1–2 km, depending on terrain, to call for owls. At each calling stop, the biologist played 3 series of owl calls of 2-minute duration and listened for a response for 2 minutes after each series. Playback tapes for the geographic survey were compiled by G. D. Hayward using 3 source recordings, and duplicates were distributed to 60 cooperators. The master recording included boreal owl songs recorded in Alberta, Canada, a recording from Cornell Laboratory of Sound, and a recording from the Soviet Union supplied by R. A. Ryder (Colo. State Univ.). These 3 recordings were segregated on the tape into 3 segments approximately 6 minutes long. Songs recorded from 1 of these geographic localities were broadcast at each playback station. Cooperators were advised to conduct surveys during good listening conditions (low wind, no rain, no heavy snowfall).

Locating Potential Nest Sites in the RNRW.—We used similar playback surveys to estimate the elevational distribution of boreal owls and to locate individual breeding sites in the RNRW. All wilderness surveys were conducted on skis from January through April in 1984–87. Beginning when the first stars became visible, we played tape recordings of the boreal owl staccato song at 0.5–1-km intervals along trails and ridge lines. We remained at each calling station 10–12 minutes playing 3 series of staccato song with 2 minutes of silence after each series. In addition to time spent listening for owls at each calling station, we paused for 1 minute at least once between stations. We sampled available habitat along survey routes radiating 10 km from field lodgings at Chamberlain and about 5 km from Cold Meadows and Hot Springs Meadow. Some survey routes followed existing trails radiating from each study site whereas others followed ridge lines and stream courses. Routes at Chamberlain and Cold Meadows were laid out to access at least 80% of the landscape within 5 km of the camp. At Chamberlain, routes extended up to 10 km from the camp, but coverage of the landscape in the outer 5-km "donut" was less complete. Because Hot Springs Meadow was surveyed less intensively and in only 1 year, coverage was limited to about 50% of the area within 5 km of the camp.

At Chamberlain and Cold Meadows, each route was surveyed 2–5 times a year. When we heard owls during a survey, we skied toward the singing male in an attempt to locate the potential nest site. Locations were not considered potential breeding sites for habitat analysis unless they met 1 of 3 criteria: (1) we located an active nest, (2) we heard both a male and female boreal owl courting at the site (Bondrup-Nielsen 1978), or (3) a male was heard singing at the site more than once during a year.

Capture and Radio Tagging.—We captured and radio tagged boreal owls at the Chamberlain site to study movements, habitat use, food habits, and survival. We trapped owls during January–May at potential breeding sites located during playback surveys. After locating a site with a singing-male boreal owl, we returned to the site 1–15 nights later and set 10 bal-chatri traps and a mist net within 100 m of the singing location (Bull 1987). We hung the mist net in an aisle in the forest along the contour. Bal-chatri traps baited with a live vole (*Microtus* spp.) or deer mouse (*Peromyscus maniculatus*) were placed 5–60 m from the mist net. Beginning at dark, a recorded staccato song was played for 20 minutes each hour from beneath a tree about 3 m from the net. Captured owls were retrieved immediately from the net and traps were checked each hour. This method was effective for capturing both male and female boreal owls.

Captured owls were weighed and measured to determine sex (Hayward and Hayward 1991), and, beginning in 1986, wing-molt pattern was recorded to determine age (R. Solheim, Zool. Inst., Oslo, Norway, pers. commun.). We radio marked the owls with 6-g, backpack-mounted radio transmitters (Wildlife Materials, Inc., Carbondale, Ill., model MPB-1220-LD) with an expected battery life of 150 days (Hayward 1987). The backpack harness that held the transmitter on the owl's back was made of 4-mm-wide elastic webbing. The elastic webbing was sewn together with upholstery thread to assure permanent attachment. Each harness was individually fitted so that the tip of a person's little finger could slip between the owl's back and the transmitter.

Radio-marked owls were recaptured and outfitted with a new transmitter after approximately 140 days or when abnormalities in the radio signal indicated potential transmitter failure. We recaptured owls from their daytime roosts with a dip net on a 5.5-m fiberglass-extension pole or a miniature mist net (2.5 m long and 1.5 m tall) as we described in Bull (1987).

Microhabitat Measurements

Breeding Habitat.—We characterized the structure of forest stands at confirmed nest sites and potential breeding sites (male-singing locations) by measuring forest vegetation, topographic features, and habitat type on a plot within the nest or singing stand. We considered singing sites that met the criteria defined above for potential breeding sites as representative breeding habitat. Our observations of radio-marked owls and observations by Bondrup-Nielsen (1978) and Palmer (1986) all indicate that boreal owls do not sing from sites throughout their home range. Rather, they consistently sing within 100 m of a potential nest cavity.

Forest structure was measured in a 30-by-125-m plot centered on the nest or singing location but within a single forest stand—the plot was not bisected by any abrupt ecotone. In cases where a nest or calling site lay near an ecotone, we placed the plot in the stand in which the nest or calling tree was rooted. If we had measured a combination of both forest stands at an ecotone, the resulting vegetation description would not have typified either forest stand, but rather some average stand that did not exist (Mueller-Dombois and Ellenberg 1974). Within the forest stand, structure was quantified by density of trees and by percent cover of trees, shrubs, grasses, and forbs.

Cover of trees and shrubs was sampled along 8 30.5-m-parallel transects placed perpendicular to the slope within the plot (Bonham 1989). Transects were randomly spaced 6–30 m apart. Intercept measurements along these transects provided shrub cover estimates (all shrubs were <1 m high) and tree cover estimates in 5 height categories (canopy cover layers) (0–1 m, 1.1–2 m, 2.1–4 m, 4.1–8 m, >8 m) (Bauer 1943, Borman and Buell 1964, Hayward and

Garton 1988). We recorded the dominant tree species and number of trees and snags in 6 diameter-at-breast-height (dbh) classes (2.5–7.6 cm, 7.7–15 cm, 15.1–23 cm, 23.1–38 cm, 38.1–53 cm, and >53 cm) within 16 systematically located 83-m^2-circular plots, 2 per transect (modified from James 1971, Hayward and Garton 1988). Cover of forbs, grasses, and subshrubs (any woody plant species commonly under 15 cm tall) was estimated on 16 0.1-m^2-rectangular quadrats (Daubenmire 1959), 2 quadrats per transect.

Because the small (83-m^2), circular tree plots would underestimate numbers of large diameter trees, variable-radius-plot tree samples were measured from 4 points—1 point along each of transects 2, 4, 6, and 8. We used a relaskop (Spiegel, Inc., Germany) with a 10-factor prism and 30-cm-dbh lower limit to count trees. Densities and basal area were calculated as described by Avery (1975:170).

In addition to measuring forest structure, we recorded topographic variables (aspect, slope, topographic position, and distance to water) and classified the forest vegetation within the nest stand or potential breeding site by habitat type (to the level of phase) (Steele et al. 1981). "Habitat type" as used by Steele et al. (1981) refers to a particular method of vegetation classification based on potential forest vegetation and environmental conditions. Habitat type differs from vegetation type in that vegetation type is defined by vegetation currently on a site and habitat type is determined by vegetation potential. Throughout this text "habitat type" will refer to forests classified using this method.

We measured forest structure using these procedures at owl locations in the RNRW and sites located throughout the northern Rockies. For sites outside the RNRW, however, we used only 4 transects and measured all variables with half as many subsamples.

Available Breeding Habitats.—Four cover types (lodgepole pine, spruce–fir, mixed conifer, and Douglas-fir) together represented over 99% of the forest cover in the study area. We chose 25 random points in each of the 4 types from throughout the study area (26 in mixed conifer by accident) using a random-number table and map-grid overlay. Random sites were classified into 1 of the 4 types based on overstory species composition. At each site we measured forest structure to define the physiognomy of the stand. Methods followed those described above for breeding habitat except that we used only 4 line-intercept transects and all variables were measured with half as many subsamples.

We chose to select a stratified random sample because a strictly random sample would have been dominated by lodgepole-pine forest, a vegetation type in which the owls did not nest. Instead we sought to obtain a sample that would adequately describe the full range of forest physiognomy of each cover type occupying 5% or more of the study area. Sample size was determined using data from our earlier studies (Hayward and Garton 1988). We calculated the minimum adequate sample necessary to reduce the coefficient of variation (CV) to 30–70% for measurements of tree density and cover in Douglas-fir forest and used this sample size for all vegetation types. The sample for each forest cover type also was similar in magnitude to our sample of owl nest locations.

Nest Tree and Site Characteristics.—At each nest site, we recorded characteristics of the nest cavity, nest tree, and forest immediately around the nest. Nest cavity measurements included entrance diameter (vertical and horizontal), cavity depth (vertical from base of cavity to bottom of entrance hole), cavity width, tree diameter at cavity, cavity aspect, and cavity height above the ground.

Forest structure immediately adjacent to the nest was described by measuring tree density, canopy cover, and basal area. We collected tree density data within a 5.2-m-circular plot and an outer donut (5.3–11.4 m) both centered on the nest tree. Trees were counted in 7 diameter classes: 2.5–7.6-cm, 7.7–15-cm, 15.1–23-cm, 23.1–38-cm, 38.1–53-cm, 53.1–68-cm, and >68-cm dbh (modified after James 1971). We also counted the number of snags in the same diameter classes on the combined plots and estimated the height of the tallest tree in each plot. We measured overstory canopy cover using a modified, concave, forest densiometer (Lindsey 1956, Strickler 1959). Holding the densiometer at waist height, the number of 17 points obscured by foliage was counted 5 paces from the nest tree in each of 4 cardinal directions and while facing the tree 3 paces to the north. Percent canopy cover was calculated as the proportion of 85 points obscured by foliage. We measured basal area of trees >15-cm dbh using a relaskop and 20 factor angle centered at the nest tree. The nest tree was excluded from basal area calculations. We also recorded dominant tree species, canopy height, distance to nearest opening, distance to ecotone, and habitat type to phase (Steele et al. 1981).

Roosting Habitat.—We located owls on their daytime roosts through radio tracking. We tried to locate each owl twice each week during our field seasons. Measurements taken at each roost were designed to quantify the degree of cover provided by the roost tree and surrounding vegetation, to record site characteristics that would influence microclimate, and to classify the forest stand by habitat type (Steele et al. 1981).

While observing the roosting owl, we recorded compass aspect, slope (using a clinometer), topographic position, snow depth, dominant tree species (i.e., the most abundant species), and height to forest canopy. For the roost tree, we recorded roost tree species, dbh (measured with a reach stick [James 1971]), height of lowest foliage, tree crown diameter, tree height, perch height, and roost tree structure (open, closed, dense, or witch's broom [abnormally dense growth of foliage]). In addition, we recorded aspect of perch from bole, aspect the bird was facing, direction of wind (using a hand-held compass), wind speed at roost and in the open (using a hand-held anemometer), temperature at the roost and in the closest opening (both recorded 1.3 m above ground level, in shade, using a pocket thermometer accurate at 0.5 C), distance from the owl to the bole, distance to nearest foliated branch above, below, and to the side of the owl, percent of cloud cover, and percent of snow cover in the open and under the canopy. Unless otherwise indicated, measurements were ocular estimates by trained observers (crew leaders worked with new technicians on at least 5 sites). We marked the tree so that it could be located for further habitat measurements.

In summer we returned to approximately 67% of the roost trees located during winter and summer to measure vegetation structure. To gather a sample from approximately 200 winter and 200 summer roosts, we systematically eliminated every third roost (ordered chronologically) for structural measurements. We collected tree density data within 2 concentric circles centered on the roost tree as described above (*see* Nest Tree and Site Characteristics). Tree size classes were modified by changing the 2 largest classes to 38.1–68 cm and >68 cm.

At 120 winter roosts and 123 summer roosts, we collected the same forest structure, topographic, and floristic data for a paired random tree. To locate the paired tree, we first spun a compass dial while looking away to determine the number of paces to travel (constrained to 30–180). A second compass spin indicated direction. After pacing off the distance in the designated direction, the closest tree >5-cm dbh became the new plot center.

Nest Box Experiment

We examined the relative importance of vegetation structure and cavity availability to nest site selection by boreal owls with a nest box experiment. Forty-five wooden nest boxes hung by the authors in July 1984 within a 9-km^2 portion of the Chamberlain study area provided suitable nest structures in all forest vegetation types other than riparian forest stands (15 in mixed conifer, 11 in Douglas-fir, and 19 in lodgepole pine). Nest boxes (constructed from 2-cm-thick lumber) measured 44-cm high, 25-cm wide, 18-cm deep, and had a 9-cm-diameter entrance (modified from

R. A. Ryder, Colo. State Univ., pers. commun.). The boxes were hung in a rectangular grid with 500 m between grid points. Boxes were paired at each grid intersection to increase the probability that 1 box at each grid would be suitable for nesting each year. The paired boxes were placed 100 m apart and hung 4–15 m high. Boxes were placed on north or east sides of the tree bole unless overstory canopy would shade the box from afternoon sunshine. We checked and cleaned boxes of debris each year during June.

Three broad forest categories dominated the experimental area: old mixed-conifer forest (dominated by ponderosa pine), mature and older Douglas-fir, and mature lodgepole pine. Availability of natural cavities suitable for boreal owls varied in these types (G. D. and P. H. Hayward, pers. obs.) and in physiognomy. At Chamberlain, during 5 years of field work, we never saw a large cavity in lodgepole-pine forest and saw <10 in Douglas-fir stands. The paucity of cavities in Douglas-fir may result from the pattern of decay in Douglas-fir trees in this region (McClelland 1977). In contrast, a single hectare of mixed-conifer forest contained over 9 pileated woodpecker cavities in 1 instance. Our playback surveys throughout the experimental area also indicated significant differences in boreal owl nesting across the 3 forest vegetation types. Within the experimental area, we never observed nesting or a potential nesting attempt by boreal owls in lodgepole-pine forest and observed only a single nest site in Douglas-fir forest. In contrast nest trees and calling sites were located 6 times in mixed-conifer forests.

By hanging nest boxes in these 3 forest vegetation types (which differed in forest structure and cavity availability), we sought to differentiate the influence of cavity availability from forest structure in the choice of nest sites by boreal owls. If forest structure dominated the choice of nest site, we expected nest-box use to differ between lodgepole pine and the other forest vegetation types. If cavity availability (regardless of forest structure) was important to the owls, we expected to observe similar use of nest boxes among the forest vegetation types.

In our experimental design, each forest vegetation type represented a treatment and each nest box was considered an independent observation repeated over a 4-year period (1985–88). An optimum level of interspersion among treatments was not possible because of the natural distribution of vegetation within the study area. Each forest vegetation type was distributed across the study area in large blocks. Therefore, nest boxes within a treatment (e.g., lodgepole-pine forest) were not completely interspersed with boxes in other treatments (this is a problem in any field experiment of this type) (Hurlbert 1984).

Food Habits

We sampled prey being captured by boreal owls at Chamberlain each year during winter and summer by collecting regurgitated pellets at daytime roosts, identifying prey cached with roosting owls, and by identification of prey at nest sites. We thoroughly searched for regurgitated pellets under each roost tree while the bird was on the roost and again when we visited the site for vegetation measurements. Because boreal owls used different roosts each day, regurgitated pellets found beneath roost trees located through radio tracking could be assigned to individual owls and specific dates. Similarly, prey identified from nests while the female was incubating or brooding were attributed to the male. When the female ceased nest attendance, we used radiotelemetry to determine whether she was delivering prey to the nest and thereby assigned the prey to the appropriate owl.

Prey samples were recorded from nest sites by visiting nests weekly to identify fresh prey and collect prey remains. Fresh prey were identified at the nest and left for the owls to consume. Nest debris that included regurgitated bones, fur, feather, and insect chitin was collected during the visit for later identification. After air drying, the debris was carefully sorted by hand and all bones and feathers removed for identification. Mammals were identified by dentition using reference collections of mammals captured on the study area and specimens at the University of Idaho, Department of Zoology, Museum. To avoid duplicate counting of fresh prey and debris, prey identified fresh was subtracted from those identified in the laboratory from debris before recording the total. Prey from pellets collected at roost sites were enumerated by dissecting the pellets and identifying prey as described for nest contents.

Frequency of occurrence for particular taxa in the diet of male and female owls was tallied within seasons by combining samples across years and owls using collections from roosts and nest sites. We examined differences among years and among individuals, but present the results elsewhere (Hayward 1989). To estimate the proportional biomass represented by each prey taxa, we multiplied prey frequency by estimated prey mass. Representative prey mass was determined by calculating the average mass of each mammal species captured during our small mammal investigations at Chamberlain. Estimates for northern pocket gophers (*Thomomys talpoides*) and northern flying squirrels (*Glaucomys sabrinus*) were taken from Hall (1946).

Small Mammal Habitat Use and Relative Abundance

To estimate the relative abundance and species composition of small mammals and to discern any trend in small mammal abundance between years, we sampled small mammals at Chamberlain from 1984 to 1988 using methods similar to those employed in long-term studies of boreal owls in Scandinavia (Korpimaki 1984, Lofgren et al. 1986, Korpimaki and Norrdahl 1991). Samples consisted of snap-trapping grids run each spring and pit-trap sets established during spring 1985 and monitored through August 1988. The methodology was chosen to minimize time invested but provide an index of year-to-year population trends and an indication of habitats used by each small mammal species.

Rectangular 9 × 10 trapping grids of museum special traps spaced 15 m apart and baited with peanut butter and rolled oats were checked daily for 7 days each spring. The first trap grid was begun 19 May in 1984 and 1985, 26 May 1986, 22 May 1987, and 23 May 1988. In 1984, 1 grid was placed in each of 4 vegetation types: sagebrush–bunchgrass, *Carex* wet meadow, lodgepole-pine forest, and old mixed-conifer forest. We placed trapping grids in forest stands at least 30 m from an ecotone. In 1985, the same sites were trapped in the same order with 1 exception. The *Carex* wet meadow (a pasture used by U.S. Forest Service stock) was replaced with an adjacent, less disturbed meadow that was trapped each of the next 3 years. A second lodgepole-pine stand and a mature Douglas-fir forest site were added in 1985. All sites trapped in 1985 also were trapped in 1986 plus an additional old spruce–fir stand. These same 7 sites were trapped in 1987.

Pit-trap sets established on 18 sites in 1985 and 2 additional sets established in 1986 permitted sampling of numerous plots with a minimum investment of time. Each pit-trap set consisted of 4 cone-shaped, 40-cm-deep pits placed along a 3-m-long, 15-cm-high, sheet metal, drift fence (Williams and Braun 1983). One pit was located at the ends of the fence and 1 on either side at the center. Each pit was covered with sheet metal held 5 cm above the ground by corner stays. This cover prevented sticks, rain, and snow from entering the pit. Each metal pit contained 8 cm of 50% ethanol topped with mineral oil to preserve captured specimens. Traps were checked once each month from May to August and left functional through the winter.

Sites for pit-trap sets were chosen subjectively to distribute the traps throughout the Chamberlain study area and to sample 6 vegetation types: wet meadow, sagebrush–bunchgrass, subalpine-fir forest, lodgepole-pine forest, Douglas-fir forest, and mixed-conifer forest. Pit-trap sets sampled 3 stands in each of the 6 vegetation categories, except we placed 5 sets in spruce–fir forest (2 in wet, old-spruce bottoms and 3 in mesic spruce–fir). Vegetation was measured at each site using methods described for owl breeding habitat. One pit-trap set was paired with each of the 6 snap-trap grids. The paired pit-trap sets were placed 30–70 m from the snap-trap grid within the same vegetation type.

STATISTICAL ANALYSIS

Throughout our analysis, we divided each year into 2 periods (snow free and snow covered), which we refer to as summer and winter. The period of snow cover each year was defined as the period from January (when we began field work) until over 50% bare ground was exposed on level ground at 1,800 m (about 1 May). When reporting averages, confidence limits (CL) represent the 95% margin of error unless otherwise specified, and for hypothesis tests $P < 0.05$ was considered significant unless otherwise noted.

We examined the pattern of population growth at the Chamberlain study site during 1984–87 using simple matrix projection models (Leslie 1945, Caswell 1989:15–26). Parameters for the models were estimated using vital rates from nests and radio-marked owls at Chamberlain.

Boreal Owl Habitat Analysis

Nesting Habitat RNRW.—We compared the structure of forest vegetation in 101 stratified random forest stands with 33 boreal owl nesting or calling sites using Principal Component Analysis (PCA) (Stauffer and Peterson 1985, Digby and Kempton 1987). Prior to analysis, we plotted the frequency distributions of each structural variable at random sites and boreal owl use sites separately. Few approached a normal distribution and transformations of several variables failed to achieve normality. We calculated principal components without transforming variables, however, because Johnson and Wichern (1982:362) indicated that ordination by PCA does not require a multivariate normal assumption. Because we measured different characteristics of forest structure in different units (e.g., cover in %, density in no./ha), we transformed all variables to Z scores prior to analysis (Pielou 1984). We entered the 21 habitat variables measured on 101 stratified random forest sites into PCA. This analysis defined the principal component axes describing primary gradients of the forest habitats of the Chamberlain site. We then used the eigenvectors from this analysis to calculate principal component scores for boreal owl use sites. Finally, we plotted random and use sites along the gradients defined by the original PCA. The pattern was analyzed descriptively comparing the distribution of random sites and owl locations.

Roost Habitat.—To test for differences in forest structure between winter and summer roosts, we cast a 2-way MANOVA, blocking by owl. Through this analysis we controlled for the effect of individuals and could test for interactions among seasons and owls.

To test whether boreal owls chose roost sites with different microhabitat structure than forests in the vicinity of roosts, we used a multivariate paired-T test (Hotelling's T^2, Mendenhall et al. 1971, Johnson and Wichern 1982). The test was calculated separately for winter and summer. To do so, we subtracted the value for each of 20 structural variables measured at the roost from the corresponding values at the paired random site. We then tested whether the resulting mean vector differed significantly from a vector of zeroes. A significant result was used as justification to explore which individual structural variables differed between roosts and random sites (Johnson and Wichern 1982).

Home Range and Owl Movements

Seasonal home range size and boundaries were calculated by the harmonic mean method using the program HOME RANGE (Samuel et al. 1985). We also calculated home range areas using a more conventional kernel method (Worton 1989), which employed an adaptive kernel estimator (Silverman 1986). Although the adaptive kernel has desirable properties relative to the harmonic mean method (Worton 1989), it did not perform well with multimodal distributions. We considered the adaptive kernel estimates for several owls unrealistically large based on plots (over an order of magnitude greater than harmonic mean estimates) and therefore base our interpretations on the harmonic mean estimates. However, means for both estimators are reported.

Prior to home range analysis, we examined the input data for outliers (e.g., observations representing distant excursions from an owl's normal activity areas) (Samuel et al. 1985) and tested the influence of sample size on area estimates (*see* Hayward et al. 1987*b*). Extreme observations inevitably plague home range studies (Schoener 1981). Outliers in a bivariate test (Samuel et al. 1985), defined as those points with bivariate normal weights lower than 0.6, were considered for removal. If the point represented a movement to an area over 2 km from the owl's normal use area and was used for <3 days, the point was removed.

To describe daily movements and variation in seasonal movements, we calculated the distance between roost sites used on consecutive days and roost-to-nest distances. The sampling units were mean distances for individual owls during a season. The measurements of distances between roosts were considered subsamples, and mean distance between consecutive roosts (within seasons) was calculated from these values.

RESULTS

Regional Habitat Use

Outside the RNRW, 49 boreal owl nests or singing male owls were found during 130 surveys covering nearly 1,300 km of transect from 500 to 3,050 m elevation. Forest cover in survey areas ranged from ponderosa pine at the lower elevations to lodgepole pine and spruce–fir at the highest sites. All owl locations were in subalpine-fir, Engelmann spruce, western hemlock (*Tsuga heterophylla*), and Douglas-fir habitat types (Steele et al. 1981) within the subalpine-fir life zone or within 100 m elevation of the subalpine-fir zone (Hayward et al. 1987*a*). Forest cover at owl locations outside the RNRW included lodgepole pine, Douglas-fir, western hemlock–western larch–subalpine fir, Engelmann spruce, and mixed spruce–fir.

The majority (88% of 49 observations) of owls were located in stands on subalpine-fir habitat types. Proportions for other habitat types included Engelmann spruce (3%), Douglas-fir (6%), and western hemlock (3%). Boreal owls were not heard below 1,292 m elevation, and 75% of the locations were above 1,584 m. Topographic position of owl locations ranged from bottoms to upper slopes.

Biologists classified 76% of the sites, where they heard boreal owls and could locate the stand the owl was using, as mature or older. The exceptions were lodgepole-pine stands on the Beaverhead National Forest in drainages where lodgepole was the only coniferous vegetation type. These lodgepole sites supported the largest trees in the area. On 1 lodgepole site, 404 trees/ha exceeded 23.1-cm dbh and 15/ha exceeded 53-cm dbh. Boreal owls were never heard singing in even-aged, young forest stands in drainages where mature forest stands were available. Over 50% of the boreal owl locations we visited outside the RNRW were marked for timber sale.

Regional Microhabitat Use.—We measured stand structure at 21 owl sites located by cooperating biologists in the region-wide survey. The sample included all sites located prior to August 1985 where the singing site could be assigned to a specific forest stand. Stands used by boreal owls in forests within the northern portion of the survey region had higher basal area and more large trees than stands used in the southern forests (Table 1). In both northern and southern areas, owls occurred predominantly in multistoried, old forest stands. On the Beaverhead National Forest where multistoried forest was not available, owls sang in even-age lodgepole-pine

Table 1. Characteristics of forest structure measured at boreal owl singing sites located throughout the northern Rockies, 1984–85. We summarized northern forests (Maritime) separately from southern, dryer sites (Continental). Canopy cover is recorded in height categories above ground.

Structural feature	Maritime (n = 12)				Continental (n = 9)			
	$\bar{x}$	±95% CL	Range	CV (%)	$\bar{x}$	±95% CL	Range	CV (%)
Tree density (No./ha)								
2.5–7.6-cm dbh	740	476.5	30–2,543	102	387	164.7	150–793	56
7.7–15-cm dbh	298	149.7	15–853	80	284	59.7	164–419	28
15.1–23-cm dbh	101	52.3	0–314	82	204	104.6	75–478	68
23.1–38-cm dbh	131	54.2	45–329	66	176	87.5	0–374	66
38.1–53-cm dbh	62	19.9	25–125	51	43	23.4	6–94	72
>53-cm dbh	43	27.3	0–137	102	11	12.3	0–49	145
2.5–38-cm-dbh snags	118	96.7	0–464	129	111	79.8	0–314	98
>38-cm-dbh snags	35	20.3	0–112	92	13	9.9	0–80	205
Basal area (m^2/ha)								
Trees >30.5-cm dbh	29.7	11.35	7.3–68.5	61	14.7	6.57	3.7–34.2	59
Tree canopy cover by height category (%)								
0–1 m	16	3.8	4–37	52	8	6.3	0–21	107
1.1–2 m	16	3.2	5–33	47	8	4.5	1–17	75
2.1–4 m	19	2.6	8–34	32	12	5.4	2–22	57
4.1–8 m	25	3.3	7–45	30	22	5.5	11–32	33
>8 m	30	6.0	7–52	46	28	12.7	12–65	58
Shrub canopy cover (%)	43	16.6	0–76	61	14	12.7	0–44	118
Ground cover (%)								
Forb	27	12.4	2–65	73	7	4.9	0–21	95
Grass	8	7.6	0–31	134	14	8.0	1–28	75
Subshrub	2	1.9	0–9	185	7	11.6	0–46	205

forest. These lodgepole stands were open and included large trees (>38-cm dbh). Density of trees over 23-cm dbh on 3 lodgepole-pine sites averaged 260/ha with 778 trees ≤23-cm dbh/ha. Aside from these lodgepole-pine sites, boreal owls were heard calling only from stands with components of old forest (more than 1 canopy layer, large trees, or more than 2 size classes of trees).

Nesting Habitat Use in the RNRW

Available Forest Structure.—Prior to sampling stand structure, we subjectively classified forest vegetation within the study area into 4 cover types based on dominant overstory tree species—lodgepole pine, Douglas-fir, mixed conifer, and spruce–fir. Each type exhibited some distinct structural characteristics, whereas other structural features are not correlated with the overstory species (Table 2). For instance, overstory canopy cover did not vary across cover types, but basal area differed between lodgepole pine and other cover types. Thus, sites could not be adequately described simply through classification by overstory (Table 2). These results indicate that an analysis that relied only on classification of habitat (Alldredge and Ratti 1986) would ignore important variation.

Principal components analysis (PCA) of 101 stratified random stands produced a description of the structural gradients available in the study area (Table 3). From 21 simple structure variables, PCA defined 6 components with eigenvalues >1, accounting for 75% of the overall variance. The seventh component accounted for less variance (4.75%) than would be expected for any one of the original variables.

We interpreted the principal components as gradients in vegetation structure and plotted sample sites along these axes (Fig. 3).

1. Component 1 describes a complex gradient from highly structured, mature and older forest with many canopy lay-

Table 2. Structural characteristics of 4 forest vegetation types in Chamberlain Basin, Idaho, measured on 25 randomly selected macroplots for each vegetation type, 1984–86. Vegetations were defined by dominant overstory trees.

Structural feature	Forest type							
	Lodgepole pine		Mixed conifer		Douglas-fir		Spruce–fir	
	$\bar{x}$	±95% CL	$\bar{x}$	±95% CL	$\bar{x}$	±95% CL	$\bar{x}$	±95% CL
Tree canopy cover by height category (%)								
0–1 m	2	1.2	3	0.7	6	1.4	16	3.5
1.1-2 m	2	1.3	6	1.3	10	1.5	15	3.0
2.1–4 m	5	1.8	12	2.5	16	2.7	19	2.5
4.1–8 m	13	3.7	25	4.0	28	3.5	25	3.1
>8 m	33	4.9	30	4.1	32	5.0	30	5.6
Shrub canopy cover (%)	1	1.1	1	0.6	4	3.1	6	3.2
Tree density (No./ha)								
2.5–7.6-cm dbh	387	150.2	308	162.0	1,040	361.5	935	212.4
7.7–15-cm dbh	284	54.5	255	72.2	443	171.6	530	115.3
15.1–23-cm dbh	204	95.4	147	40.6	178	30.1	337	79.4
23.1–38-cm dbh	176	79.9	77	24.2	111	25.6	193	34.2
38.1–53-cm dbh	4	4.0	17	5.5	28	13.6	32	10.1
>53-cm dbh			25	6.6	16	5.4	12	0.4
2.5–38-cm-dbh snags	111	74.7	73	30.1	211	136.8	194	31.5
>38.1-cm-dbh snags			7	6.4	2	1.4	12	8.3
Basal area (m^2/ha)								
Trees >30.5-cm dbh	2.0	1.0	15.8	2.8	12.7	3.5	13.1	4.01

ers to young or aggradation stage forest with little vertical diversity. This component describes the overall physiognomy of the stand whereas subsequent components relate to individual features of forest structure (i.e., understory, overstory).

2. Component 2 is a gradient with dense cover 0–2 m above the ground, numerous 2.5–15-cm-dbh trees and few large trees describing the positive end.
3. Component 3 defines a cline in occurrence of large trees (>53-cm dbh), high basal area, and upper canopy cover.
4. Component 4 defines a gradient in trees 15.1–38-cm dbh with a strong upper canopy.
5. Component 5 includes stands with large snags (>38-cm dbh) and little grass cover.
6. Component 6 is a gradient in forb ground cover.

A combination of principal Components 1 and 2 separates spruce–fir, lodgepole-pine, and mixed-conifer stands (Fig. 3). The simple structure of lodgepole-pine stands (compared to other types) is evident in Component 1, whereas spruce–fir and mixed conifer differ most in understory cover (Component 2). Douglas-fir forest is dispersed throughout the gradient occupied by spruce–fir and mixed conifer. The striking differences between lodgepole-pine stands and other forests is lost when viewed along gradients defined by Components 3 and 4. Unlike some regions of the Rockies, multiple-canopy-layered lodgepole-pine forest is common in the study area.

Relationship of Owl Use to Available Forest Structure.—In this analysis, we examined whether boreal owls in the RNRW showed evidence of nonrandom use of nesting habitat. We analyzed the results of vegetation measurements at 21 nest sites and 12 calling sites to determine whether the owls restricted nesting to particular habitats. Boreal owl breeding locations were found in mixed-conifer (39.4%), spruce–fir (18.2%), Douglas-fir (21.2%), and aspen (21.2%) stands. Lodgepole pine, which covers over 50% of the study area, was the only abundant vegetation type not used for nesting. The proportion of use in each type, however, could not be compared to the available proportion because owl surveys were not designed to sample

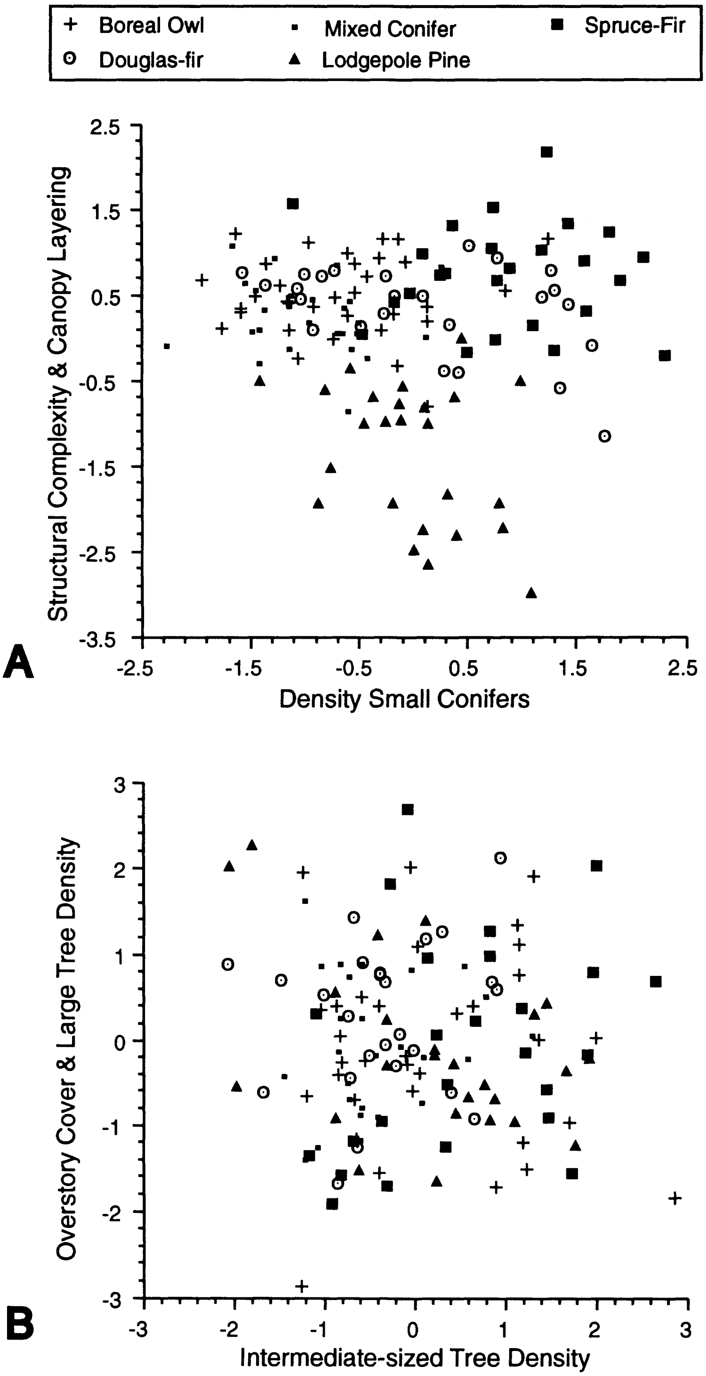

Fig. 3. Relationships among 4 vegetation cover types and boreal owl nest sites as defined by principal components analysis (PCA) of 21 forest structure variables. Each plot depicts the placement of 101 random plots and 33 boreal owl nest sites on 2 gradients defined by PCA. (A) Principal component 1 on *Y*-axis; principal component 2 on *X*-axis. (B) Principal component 3 on *Y*-axis; principal component 4 on *X*-axis.

Table 3. Six principal components derived from analysis of structural features measured at 101 random plots in 4 general vegetation types at Chamberlain Basin, Idaho, in 1985. Zero loading was 0.45 throughout.

Structural feature	Principal component 1	2	3	4	5	6
Basal area	0.668	−0.452	0.478			
Density trees 2.5–7.6-cm dbh		0.687				
Density trees 7.7–15-cm dbh	−0.733	0.481				
Density trees 15.1–23-cm dbh	−0.513			0.501		
Density trees 23.1–38-cm dbh				0.722		
Density trees 38.1–53-cm dbh	0.553					
Density trees >53-cm dbh	0.514	−0.487	0.477			
Density snags 2.5–38-cm dbh	−0.767					
Density snags >38-cm dbh					0.588	
Conifer cover 0–1 m	0.544	0.634				
Conifer cover 1.1–2 m	0.666	0.607				
Conifer cover 2.1–4 m	0.763					
Conifer cover 4.1–8 m	0.596					
Conifer cover >8 m			0.720	0.480		
Horizontal cover diversity	0.812					
Low conifer vertical diversity	0.605	0.745				
High conifer vertical diversity	−0.479		0.604			
Subshrub cover		0.467				
Grass cover					−0.555	
Forb cover						0.685
Eigenvalue	5.839	3.771	2.037	1.743	1.223	1.104
% variance explained	27.8	18.0	9.7	8.3	6.1	5.3
Cumulative %	27.8	45.8	55.5	63.8	69.9	75.2

types in proportion to their availability. Despite this problem, the high use of aspen, which accounted for <1% of forest vegetation cover, suggests strong selection for some components of aspen stands. The lack of nests in lodgepole-pine forest, which was surveyed more than other types, demonstrates avoidance of the most abundant vegetation type in the basin.

To compare vegetation structure at potential breeding sites and our random sites, we plotted boreal owl singing locations and nest sites with the stratified random sites on the first 4 principal components described above (Fig. 3). Owl sites were given a score on each principal component based on the vegetation analysis. The owl sites were then plotted in the space defined by the first 4 principal components of the vegetation analysis. The first 4 components accounted for 64% of the variance and were used to compare used sites with random sites. The resulting pattern indicates that boreal owls used structurally complex stands (Component 1) with less understory development than many available sites (Component 2) (Fig. 3). The nonrandom pattern of owl use on axes 1 and 2 contrasts with the pattern observed on subsequent axes. Owl sites included stands with a range of overstory cover and large tree densities (Component 3) closely matching the full range of available types. Likewise, owl use sites were scattered across the gradient describing large tree density (Component 4) (Fig. 3).

Description of Owl Breeding Sites in RNRW.—The above analysis suggests that boreal owls choose nest sites in a nonrandom manner; the physiognomy of owl use sites did not encompass the full range of available types. Nesting and calling sites never occurred in dense, even-age forest, and boreal owls never nested in lodgepole-pine forest—the most abundant cover type in the study area. Relative to stratified random sites, the most characteristic structural features of 33 nesting and calling sites were a high density of large trees (>38-cm dbh), open understory, and a multilayered canopy (Table 4).

All but 2 of 23 nests were in trees within

Table 4. Characteristics of forest structure measured at 33 boreal owl calling and nest sites in the Frank Church River of No Return Wilderness during 1984–88. Canopy cover is recorded in layers above the ground.

Structural feature	$\bar{x}$	±95% CL	Range	CV (%)
Tree density (No./ha)				
2.5–7.6-cm dbh	450	140	29–1,795	91
7.7–15-cm dbh	298	86	82–1,226	85
15.1–23-cm dbh	162	40	15–434	72
23.1–38-cm dbh	126	31	15–320	72
38.1–53-cm dbh	34	10	0–141	86
>53-cm dbh	23	6	0–64	76
2.5–38-cm-dbh snags	115	49	0–763	126
>38-cm-dbh snags	9	6	0–91	202
Basal area (m^2/ha)				
Trees >30.5-cm dbh	17.8	3.1	3.7–40.9	51
Tree canopy cover by height category (%)				
0–1 m	6	1.7	1–2	84
1.1–2 m	8	1.3	1–16	47
2.1–4 m	13	2.1	5–27	47
4.1–8 m	22	3.2	2–41	43
>8 m	30	4.3	4–5	42
Ground cover (%)				
Forb	12	4.0	1–55	73
Grass	8	3.1	4–50	50
Subshrub	7	27.0	0–26	111

a forest stand. One unusual nest occupied a lone spruce in a small boulder field. An extensive stand of large spruce and fir trees lay 16 m to the south of the 71-cm-dbh spruce. This nest failed 15 days after hatching 3 young. The second unusual nest was in a Douglas-fir snag in a 0.5-ha island of mature trees 20 m from extensive spruce–fir forest.

To describe forest stand sizes chosen for nesting, we estimated the area of the nest stand from the size of the area surrounding the nest tree that was characterized by homogeneous tree species and tree size class. Nest stands ranged from 0.8 to 14.6 ha and averaged 7.6 ± 3.96 ha. All 5 aspen nest stands ranged between 0.8 and 1.3 ha, whereas the smallest coniferous-forest nest stand covered 1.6 ha. All of these stands were surrounded by forest.

Nest Box Experiment.—During 4 years, owls were observed courting or nesting at only 3 nest boxes in the experiment. Although radio-tagged boreal owls frequently foraged and roosted near boxes in lodgepole pine, none nested there. In 1987, a pair of radio-marked owls fledged 2 young from a box in mixed conifer. A second box in mixed conifer fledged 2 young in 1988. In 1987, boreal owls nested in a box in Douglas-fir forest.

Nest Tree Characteristics.—During intensive studies in the RNRW, we found 23 boreal owl nests. Locating nests other than those used by radio-marked females was difficult. We found 6 nests occupied by unmarked owls, 2 of these in nest boxes. We found 3 nests when only the male member of the pair was radio marked and failed to find a nest for 6 radio-marked males. For the remaining 14 nests, either the female or both owls were radio marked.

We attributed the excavation of 18 of 19 natural nest cavities to pileated woodpeckers; a northern flicker probably excavated the other. Boreal owl nest-cavity entrances averaged 102 ± 12.41 mm high (range 64–150 mm) and 95 ± 11.89 mm wide (range 56–148 mm). Nest cavity condition ranged from recently constructed cavities without a feces layer (indicating no prior nesting by woodpeckers or other birds) to old cavities with a deep layer of dried feces, cone scales, and other debris. Inside, the cavities ranged from 7 to 50 cm deep ($\bar{x}$ = 31 ± 7.61 cm) and from 15 to 26 cm diameter ($\bar{x}$ = 19 ± 2.11 cm). The tree diameter at the cavity averaged 41 ± 5.21 cm (range 26–61 cm). Tree dbh averaged 64 ± 11.02 cm (range 33–112 cm).

Excluding nests in nest boxes, owls nested in ponderosa pine 10 times (53%), aspen 7 times (37%), and once each in Douglas-fir (5%) and Engelmann spruce (5%). Ten nests occupied snags, including 8 ponderosa pine, 1 aspen, and 1 Douglas-fir. Snag condition included 3 old branchless snags >11 m tall, 2 hard snags with sloughing bark and only large branches remaining, and 5 young snags with bark and complete limbs. Among nests in live trees, all but 2 cavities occurred in an open area on the tree bole; distance to foliage above the cavity averaged 3.8 ± 1.67 m (minimum 0.3 m). Over 75% of the cavities in live trees had no foliage below the cavity. For those that did, the minimum distance to foliage below was 0.6 m.

The owls chose relatively high cavities, averaging 12.7 ± 2.98 m and ranging from 6 to 25 m. Cavity height averaged 51% of tree height. Boreal owl nests in snags or trees with multiple cavities always occupied 1 of the uppermost cavities suggesting a preference for high nest sites.

The forest immediately around nest trees had an open structure. Density of 2.5–23-cm-dbh trees within a 0.01-ha plot around the nest tree averaged 398 ± 162/ha (range 0–1,482) (Table 5). The density of trees at nests was 3 times lower than the average at winter roost sites. Density of trees larger than 23.1-cm dbh averaged 212 ± 86/ha, similar to average densities measured at winter roost sites.

Nest sites occupied forest stands in 3 habitat series (based on Steele et al. 1981). We found 17% of nest sites in Engelmann spruce (*Picea engelmannii*) series, specifically the Engelmann spruce–common horsetail (*Equisetum arvense*) habitat type; 39% in the subalpine-fir series, specifically subalpine fir–twisted stalk (*Streptopus amplexifolius*), subalpine fir–bluejoint (*Calamagrostis canadensis*), subalpine fir–beargrass (*Xerophyllum tenax*), and subalpine fir–grouse whortleberry (*Vaccinium scoparium*) habitat types; and 44% in the Douglas-fir habitat series, specifically Douglas-fir–common snowberry (*Symphoricarpos albus*), Douglas-fir–pinegrass (*Calamagrostis rubescens*), and Douglas-fir–elk sedge (*Carex geyeri*) habitat types. The slope at the nest ranged from flat to 49%, averaging 28 ± 6%. Nest trees were distributed relatively evenly from bottoms

Table 5. Forest structure at 19 different nest trees used by boreal owls in the Frank Church River of No Return Wilderness during 1984–88. Tree densities are reported for 2-concentric circular plots—an inner circle 5.2-m radius and an outer "donut" extending from 5.2 m to 11.4 m.

Site characteristic	$\bar{x}$	±95% CL
Tree density (No./ha)		
Inner plot		
2.5–7.6-cm dbh	174	111.9
7.7–15-cm dbh	98	48.1
15.1–23-cm dbh	114	60.1
23.1–38-cm dbh	136	73.7
38.1–68-cm dbh	60	42.5
>68-cm dbh	11	15.6
Outer plot		
2.5–7.6-cm dbh	242	107.3
7.7–15-cm dbh	178	70.1
15.1–23-cm dbh	124	49.5
23.1–38-cm dbh	130	56.6
38.1–68-cm dbh	51	25.0
>68-cm dbh	10	7.8
Snag density (No./ha)		
2.5–38-cm dbh	79	42.2
>38-cm dbh	10	11.5
Basal area (m^2/ha)	33.7	3.62
Canopy cover (%)	55	7.7
Topographic features		
Distance to water (m)	201	98.9
Slope (%)	28	5.8

Table 6. Habitat types (Steele et al. 1981) of 194 winter and 342 summer roost sites used by boreal owls at Chamberlain Basin, Idaho, during 1984–87. We combined rare habitat types with those of similar moisture and structural characteristics.

Habitat types	Roost sites			
	Winter		Summer	
	No.	%	No.	%
Douglas-fir–blue huckleberry (*Pseudotsuga menziesii–Vaccinium globulare*)	4	2.1	3	0.9
Douglas-fir–warm sites with ponderosa pine (*Pinus ponderosa*) codominant in the overstory[a]	45	23.2	6	1.7
Douglas-fir–pinegrass (*Pseudotsuga menziesii–Calamagrostis rubescens*)				
Douglas-fir–white spirea (*Pseudotsuga menziesii–Spiraea betulifolia*)				
Douglas-fir–Oregon grape (*Pseudotsuga menziesii–Berberis repens*)				
Subalpine fir–hydric sites with lush understory of herbs and grasses[a]	12	6.2	16	4.7
Subalpine fir–twisted stalk (*Abies lasiocarpa–Streptopus amplexifolius*)				
Subalpine fir–bluejoint (*Abies lasiocarpa–Calamagrostis canadensis*)				
Subalpine fir–Canby's ligusticum (*Abies lasiocarpa–Ligusticum canbyi*)				
Subalpine fir or Douglas-fir–cold air drainage sites with a sparse understory of short shrubs[a]	25	12.9	2	0.9
Subalpine fir–dwarf huckleberry (*Abies lasiocarpa–Vaccinium caespitosum*)				
Douglas-fir–dwarf huckleberry (*Pseudotsuga menziesii–Vaccinium caespitosum*)				
Subalpine fir–twinflower (*Abies lasiocarpa–Linnaea borealis*)	9	4.6	5	1.5
Subalpine fir–mesic sites with tall shrubs[a]	9	4.6	8	2.3
Subalpine fir–Sitka alder (*Abies lasiocarpa–Alnus sinuata*)				
Subalpine fir–menziesia (*Abies lasiocarpa–Menziesia ferruginea*)				
Subalpine fir–high elevation mesic sites[a]	54	27.9	278	81.1
Subalpine fir–beargrass (*Abies lasiocarpa–Xerophyllum tenax*)				
Subalpine fir–blue huckleberry (*Abies lasiocarpa–Vaccinium globulare*)				
Subalpine fir–white spirea (*Abies lasiocarpa–Spiraea betulifolia*)				
Subalpine fir–harsh, high elevation, mesic sites with understory of short shrubs and grasses[a]	11	5.7	12	3.5
Subalpine fir–grouse whortleberry (*Abies lasiocarpa–Vaccinium scoparium*)				
Subalpine fir–elk sedge (*Abies lasiocarpa–Carex geyeri*)				
Subalpine fir–heartleaf arnica (*Abies lasiocarpa–arnica cordifolia*)				
Subalpine fir–pinegrass (*Abies lasiocarpa–Calamagrostis rubescens*)	25	12.9	12	3.5

[a] Represents sum of habitat types listed immediately below.

Table 7. Seasonal boreal owl roost characteristics measured at Chamberlain Basin, Idaho, during 1984–87. Tree densities were sampled from 2-concentric circular plots around the roost site. The inner circle was 0.0084 ha and outer "donut" was 0.0321 ha.

	Season			
	Winter (n = 189)		Summer (n = 241)	
Variable	$\bar{x}$	±95% CL	$\bar{x}$	±95% Cl
Canopy cover (%)	58.5	1.91	63.5	1.54
Basal area (m^2/ha)	26.0	2.03	29.8	1.87
Roost tree dbh (cm)	27.7	2.21	25.7	1.65
Tree density (No./ha)				
2.5–7.6-cm dbh—inner	797	120.5	1,380	142.4
2.5–7.6-cm dbh—outer	864	119.6	1,233	98.6
7.7–15-cm dbh—inner	561	81.3	897	78.8
7.7–15-cm dbh—outer	641	74.5	869	66.5
15.1–23-cm dbh—inner	261	32.0	341	33.5
15.1–23-cm dbh—outer	287	26.3	359	23.2
23.1–38-cm dbh—inner	130	20.2	181	21.3
23.1–38-cm dbh—outer	156	16.4	199	13.9
>38-cm dbh—inner	35	11.5	27	7.1
>38-cm dbh—outer	38	7.9	34	67.2
Snag density (No./ha)				
2.5–15-cm-dbh snags[a]	305	53.0	269	44.5
15.1–38-cm-dbh snags	37	7.9	49	8.8
>38-cm-dbh snags	2	1.2	8	2.4

[a] Snags measured in 0.04-ha-circular plot.

to upper slope positions (27% bottoms, 18% lower third, 14% midthird, 41% upper third). We failed to locate any nests on ridges.

Roost Habitat and Roosting Behavior

Unlike many species of owls, boreal owls roost at sites throughout their home range; rarely do they roost in the same stand on consecutive nights. We located consecutive daytime roosts of 14 owls on 159 occasions. In only 8 cases did owls use the same tree on consecutive days.

Pattern of Roost Habitat Use During Winter and Summer.—Habitat type of roost stands differed between seasons (χ^2 = 167.6, df = 7, $P < 0.001$) (Table 6) reflecting the use of more moist, higher elevation sites in summer. The topographic position of roost sites also differed between seasons (χ^2 = 63.7, df = 6, $P < 0.001$). Boreal owls roosted in bottoms significantly more often in winter (27 vs. 9%) than in summer and on mid- and upper slopes significantly more often in summer (12 vs. 26% midslopes, 29 vs. 41% upper slopes) than in winter.

We measured habitat structure at 430 roost sites of 24 radio-marked owls. Typical forest stands used for winter roosts had nearly 60% canopy cover, 26 m^2/ha basal area, 1,620 trees/ha having 2.5–23-cm dbh, and 165 trees/ha over 23.1-cm dbh (Table 7). Although similar to winter roosts, summer roosts (on average) occurred in more dense forest with greater cover. Microhabitat at roosts in winter and summer was different (MANOVA Wilks' F = 5.2; df = 7, 378; P = 0.0001) mainly due to differences in density of trees 2.5–23-cm dbh (Table 8). Winter and summer roosts differed in all aspects of forest structure measured. Individual boreal owls also differed significantly in roost habitat (MANOVA Wilks' F = 2.34; df = 154, 2,544; P = 0.0001) although the smaller F value indicates less evidence for an effect than the difference in seasons (Table 8). Individual owl roost sites differed most in the density of large trees, canopy height, and basal area. The effects of individual owl and

Table 8. Multivariate analysis of variance examining seasonal differences in boreal owl roost site characteristics measured at Chamberlain Basin, Idaho, during 1984–87. Sites from individual owls are treated as blocks and the interaction (season × owl) is included in the model. Univariate *F* and *P* values (based on type 1 sums of squares) suggest the roost characteristics most responsible for the significant seasonal effect (the "season" *F* and *P* values are included for the univariate summary). Tree density is estimated from an inner circle of 0.0084 ha and an outer "donut" of 0.0321 ha.

Source of variation	*F*	df	*P*
MANOVA (Wilks')			
Season	5.201	7; 378	<0.001
Owl	2.338	154; 2,544	0.001
Season × owl	1.145	119; 2,474	0.139
ANOVA for individual variables			
Canopy height	30.74	1	<0.001
Canopy cover (%)	16.96	1	<0.001
Basal area (m^2/ha)	8.30	1	0.004
Tree density (No./ha)			
2.5–23-cm dbh—inner	47.70	1	<0.001
2.5–23-cm dbh—outer	33.28	1	<0.001
23.1–68-cm dbh—inner	6.76	1	0.009
23.1–68-cm dbh—outer	13.91	1	<0.001

season did not interact ($P = 0.139$) indicating that the owl population as a whole consistently shifted roost habitat between seasons.

Unfortunately, finding an unequivocal answer to the question, "Does roost habitat differ between seasons?" is not straightforward. As we will show later, owls shifted home range slightly between winter and summer. Therefore, differences between seasons in roost habitat structure could result from changes in the range of available sites. To test whether shifts in home range would confound any conclusions about microhabitat use, we evaluated seasonal differences in the paired random sites measured in conjunction with the owl roost sites. This analysis included only the random, not the used sites. Any difference in microhabitat resulting from seasonal shifts in home range also should be apparent in the paired random plots. The random plots, then, act as a control for the confounding variable, home range shift.

The results support the assertion that owl roost habitat differs between summer

Table 9. Multivariate analysis of variance examining seasonal differences in paired-random roost-site characteristics. Random sites from individual owls are treated as blocks and the interaction (season × owl) is included in the model. Univariate *F* and *P* values (based on type 1 sums of squares) suggest the roost characteristics most responsible for the significant seasonal effect (the "season" *F* and *P* values are included for the univariate summary). Tree density is estimated from an inner circle of 0.0084 ha and an outer "donut" of 0.0321 ha.

Source of variation	*F*	df	*P*
MANOVA (Wilks')			
Season	1.630	7; 198	0.129
Owl	1.652	133; 1,318	0.001
Season × owl	1.162	56; 1,071	0.198
ANOVA for individual variables			
Canopy height	1.36	1	0.245
Canopy cover (%)	2.59	1	0.109
Basal area (m^2/ha)	0.01	1	0.912
Tree density (No./ha)			
2.5–23-cm dbh—inner	6.56	1	0.011
2.5–23-cm dbh—outer	19.03	1	<0.001
23.1–68-cm dbh—inner	0.65	1	0.420
23.1–68-cm dbh—outer	3.35	1	0.068

Table 10. Comparison of vegetation characteristics at roost sites at Chamberlain Basin, Idaho, with paired-random sites in winter ($n = 120$) and summer ($n = 123$) during 1984–87. Mean difference expressed as roost minus paired-random site. Tree density is estimated from an inner circle of 0.0084 ha and an outer "donut" of 0.0321 ha.

Variable	Winter			Summer		
	$\bar{x}$ difference	SE	P	$\bar{x}$ difference	SE	P
Canopy cover (%)	6.05	1.704	<0.001	7.59	1.503	<0.001
Basal area (m^2/ha)	12.20	6.774	0.07	11.80	5.801	0.04
Tree density (No./ha)						
2.5–7.6-cm dbh—inner	0.62	0.824	0.46	2.88	0.778	0.01
2.5–7.6-cm dbh—outer	2.52	2.902	0.39	−0.54	2.591	0.83
7.7–15-cm dbh—inner	0.68	0.529	0.21	1.20	0.545	0.03
7.7–15-cm dbh—outer	1.63	1.764	0.36	0.28	1.759	0.88
15.1–23-cm dbh—inner	−0.15	0.242	0.54	0.21	0.261	0.42
15.1–23-cm dbh—outer	−0.97	0.648	0.14	0.51	0.751	0.50
23.1–38-cm dbh—inner	0.21	0.159	0.19	0.44	0.142	<0.001
23.1–38-cm dbh—outer	0.55	0.404	0.18	0.82	0.321	0.01
38.1–53-cm dbh—inner	0.06	0.054	0.29	0.01	0.049	0.87
38.1–53-cm dbh—outer	0.18	0.166	0.30	0.07	0.132	0.62
53.1–68-cm dbh—inner	0.08	0.037	0.05	0.00	0.028	1.00
53.1–68-cm dbh—outer	0.08	0.051	0.15	0.00	0.057	1.00
>68-cm dbh—inner	0.03	0.021	0.10	−0.02	0.011	0.16
>68-cm dbh—outer	−0.06	0.054	0.29	−0.05	0.025	0.06
Snag density (No./ha)						
2.5–7.6-cm-dbh snags[a]	0.09	1.432	0.95	1.76	1.293	0.18
7.7–15-cm-dbh snags	−0.55	0.327	0.10	−0.35	0.313	0.27
15.1–23-cm-dbh snags	−0.39	0.177	0.03	−0.24	0.246	0.34
23.1–38-cm-dbh snags	0.28	0.114	0.01	−0.04	0.171	0.81
38.1–53-cm-dbh snags	0.00	0.031	1.00	0.07	0.068	0.34
53.1–68-cm-dbh snags	0.05	0.026	0.06	0.02	0.025	0.53
>68-cm-dbh snags	−0.02	0.021	0.42	0.01	0.008	0.32

[a] Snags measured in 0.04-ha-circular plot.

and winter (Table 9). No seasonal effect (MANOVA Wilks' $F = 1.63$; df = 7, 198; $P = 0.129$) or significant interaction between season and owl (MANOVA Wilks' $F = 1.16$; df = 56, 107; $P = 0.198$) appeared among random plots. The influence of individual owls was still apparent in the paired random plots (MANOVA Wilks' $F = 1.65$; df = 133, 1,318; $P = 0.0001$) but was rather weak judging by the low F value. An unbalanced sample across individuals may have contributed to the low F; however, the large number of degrees of freedom in this analysis suggests good power to detect even small differences in season—thus, we are confident in concluding that random plots showed no seasonal effect.

Roost Habitat Selection.—We located 882 roost sites from 1984 to 1987 and measured forest structure at 430 sites (Table 7). In both winter and summer, roost sites differed significantly from paired random sites (Hotelling's T^2; winter $F = 2.04$; df = 120, 106; $P = 0.021$; summer $F = 3.75$; df = 120, 106; $P < 0.0001$). Winter roosts had significantly greater canopy cover, more 23.1–38-cm-dbh snags in the inner plot, and fewer 15.1–23-cm-dbh snags than paired random sites (Table 10). Summer roosts had greater canopy cover, higher basal area, denser 2.5–15-cm-dbh trees in the inner plot, and denser 23.1–38 cm trees in both plots than paired random sites (Table 10).

Thus, we have stronger evidence for selection in summer than winter. Characteristics of summer roosts indicate the owls choose dense, shaded sites compared to paired random locations. When we compared temperature at the roost and in the nearest opening (both temperatures taken in the shade while the owl was roosting), roost sites were significantly cooler when

Table 11. Temperature differences between roost site and a nearby opening by temperature class for sites located in Chamberlain Basin, Idaho, during 1984–87. The negative mean difference indicates the roost was cooler than the paired reading.

Temperature in open (C)	n	$\bar{x}$ difference	SE of mean	t[a]	P
1–4	80	−0.07	0.04	1.69	0.10
4.1–10	101	−0.18	0.07	2.62	0.010
10.1–15	99	−0.39	0.09	4.13	<0.001
15.1–21	125	−0.56	0.09	5.82	<0.001
>21	80	−0.98	0.12	7.93	<0.001

[a] Student's t.

ambient temperatures exceeded 4 C. The difference in temperature increased with increasing ambient temperature (Table 11).

Roost Perch Characteristics.—During winter, boreal owls typically perched 7.2 ± 0.21 m high in a 27 ± 0.9-cm-dbh tree (n = 261). The ratio of perch height to the tree height averaged 0.4 (75% of winter roosts occurred in the lower half of the tree, 25% in the lowest quarter). Only 25% of the time did owls roost at or below the height of the lowest foliage. Seventy percent of roosts occupied the dominant tree species in the stand. When the owl had cover within 5 m of the perch, the distance to foliage above the owl averaged 0.8 ± 0.15 m, distance to foliage below averaged 0.7 ± 0.15 m, and distance to foliage at the side averaged 0.3 ± 0.06 m.

In summer, boreal owls continued to roost in the dominant tree species of the stand (66% of 378 roosts were in dominant tree species). Birds perched at a mean height of 5.9 ± 0.16 m in 25 ± 0.7-cm-dbh trees. Fifty percent of roosts occurred in the lower third of the roost tree. As in winter, 25% of the summer roosts occurred at or below the height of the lowest foliage. When the owl had cover within 5 m of the perch, the distance to foliage averaged 1.1 ± 0.14 m above the owl, 0.7 ± 0.10 m below, and 0.4 ± 0.06 m to the sides.

Roosting Behavior.—Daytime was a period of reduced activity for boreal owls. During daylight the owls spent the majority of time perched, eyes closed. On 16 occasions when we watched owls during daylight for 2 hours or more (total 46 hrs) they perched quietly with eyes closed 77% of the time. Periods of sleep rarely lasted more than 40 minutes before being interrupted by brief (2–5 min) periods of preening or actively looking about. The owls spent 6% of the observation period preening. The birds preened plumage and feet, stretched wings, and on some occasions shook the entire plumage. Except for the extended preening bout of 20–30 minutes, which always preceded leaving the roost to begin foraging at night, the owls preened only a small portion of the plumage during a preening bout.

In addition to preening and sleeping, owls spent 10% of the day actively looking around and 4% of the time eating. Hunting during daylight (1% of observation period) and simply moving from 1 roost perch to another (1%) encompassed only a short portion of the daytime activity but may be important. Owls exhibited gullar fluttering during 1% of the 46 hours of observation (occurred only on warmer days).

Boreal owls frequently consumed prey during the daytime. Our observations suggest that few (*see* below) of the prey eaten at the daytime roost were captured from the roost. We saw owls retrieve cached prey and eat some portion of the prey on 63 occasions, and we noted prey cached near roosting owls on 20 other occasions. Cached prey was usually stored in the fork of a tree branch.

Boreal owls tended to eat or possess cached prey more frequently in summer than winter. We observed owls with cached prey at 17% of summer roosts and 4% of winter roosts (χ^2 = 56, df = 1, n = 822, P = 0.00001). During summer the owls ate cached prey most frequently between 1200 and 1400 hours; 65% of observations of owls consuming cached prey occurred between 1120 and 1330 hours. In winter, we observed boreal owls eating prey throughout the day, although 66% occurred after noon. The distribution of observed feeding times differed from the distribution of our observed roost (radio tracking) location times for both winter and summer (winter χ^2 = 6.3, df = 3, P = 0.097; summer χ^2 = 7.7, df = 5, P = 0.176). For both tests we chose $\alpha = 0.20$, which is conventional for goodness-of-fit tests (White et al. 1982).

When feeding on prey at the roost, boreal owls rarely consumed prey whole. In fact, over half of cached prey observed at roosts were headless. The owls often consumed the forward half of the prey in pieces. On 4 occasions we saw the owl extract the intestines from the abdominal cavity and discard them before proceeding to consume the rear half of the prey. The rear half of mice and voles were then consumed whole.

Boreal owls seldom foraged for an extended period during daylight. On only 1 of 16 days (46 hrs), when we watched roosting owls for over 2 hours at their roost, did an owl begin hunting. The owl foraged for 15 minutes. In conjunction with our ordinary roost observations, we observed the owls on 882 occasions. On 13 days (2.9%) in winter and 33 days (7.4%) during summer, the owl began foraging in daylight during the normal ~1-hour observation period at the roost. We observed owls attack prey from their roost tree (excluding instances of extended daytime foraging) on 7 occasions.

Boreal owls moved to a different roost tree more frequently than they hunted during daylight. Owls moved during our roost observations on 16% of the days during summer and 8% in winter. These roost changes were distinguished from foraging by the lack of rapid head turning after the move and frequently by the owl closing its eyes soon after the flight. Most often the owl moved to an adjacent tree and rarely flew over 40 m.

Boreal owls normally roost alone and, as discussed in the section on movements, usually roost far from the nest and their mate. Although we radio marked both members of 5 mated pairs prior to nesting, we observed the owls roosting within 150 m of one another during the prenesting period on only 5.8% of the occasions ($n = 121$). One pair was found roosting together 4 times. On 14 March they roosted in adjacent trees 6.5 km from a nest site at which they had been courting since early February. The next day they again roosted within 30 m of each other, but this time about 200 m from the nest. On 18 March they roosted together 2.6 km from the nest although they had not been together the previous day or early the previous evening. On 27 March the pair roosted within 30 m of each other 3 km from the nest and were together near the nest by 2145 hours. This behavior suggests that these 2 owls traveled together during some nighttime movements, remaining together during journeys as far as 6.5 km.

We found unmated radio-marked owls roosting together twice. On 1 May 1986, 2 males roosted 120 m from each other on a hillside used often by both owls. On 29 June 1987, a female who was caring for nestlings roosted within 150 m of a male who had not bred that year. Within 5 days the 2 owls' roosts were no closer than 10 km.

Movements and Home Range Use

Daily Movements and Distance Covered.—Boreal owls lead a very mobile existence during both winter and summer. Although the limitations of travel within the wilderness prevented us from effectively following radio-marked owls during their foraging (Hayward 1987), locations of daytime roosts suggest the magnitude of the owls' daily movements. We used locations of consecutive daytime roosts and roost-to-nest distances as indications of the minimum travel distances. Although both measures certainly underestimate daily movement (Laundre et al. 1987), these indices do provide insight into the pattern of owl movements by describing minimum distances traversed.

We recorded distances between consecutive daytime roosts of 14 owls (7 females and 7 males) on 150 occasions over 4 years. Distance between roosts on consecutive days ranged from 0 to 6,935 m. Mean distances did not differ significantly between winter and summer (winter $\bar{x} = 1{,}540 \pm 446$ m, summer $\bar{x} = 934 \pm 348$ m). For this comparison we treated each radio-marked owl as a sample and consecutive

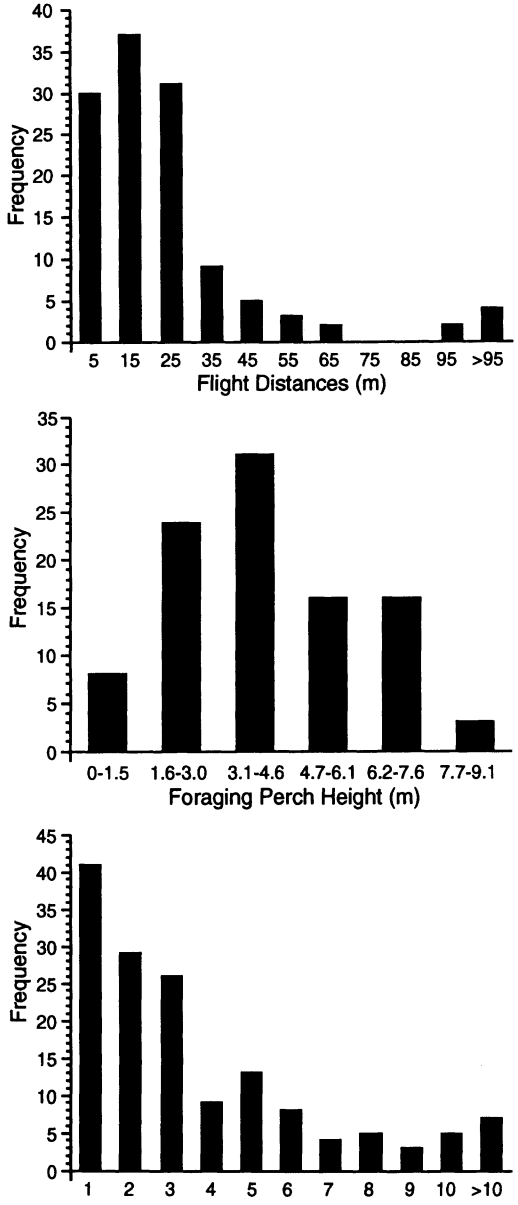

Fig. 4. Foraging behavior of boreal owls described by the distribution of flight distances, perch heights, and duration of search times for owls recorded during 18 foraging bouts in Chamberlain Basin, Idaho.

roost locations for an individual as subsamples.

The distance between male owl daytime roosts and nests also provides some insight on the magnitude of daily movements. Nesting males rarely roosted within 500 m of the nest, and over 85% of roosts were over 1,000 m from the nest (some up to 5,600 m away). The average roost-to-nest distance of 5 owls over the 4-year study was 1,729 ± 831 m.

Night radiotelemetry (triangulation) indicated that the roost-to-nest distances were accurate estimates of foraging distances. Following prey deliveries at the nest, 1 owl on 3 nights returned to the area of its day roost several kilometers from the nest. Four other males also left the nest stand immediately after prey deliveries to forage in areas distant from the nest.

Daily Movements and Foraging.—Boreal owls can be classified as sit-and-wait predators or searchers (as opposed to pursuers), but are very active while hunting. During a foraging bout, the birds move through the forest in an irregular or zigzag pattern, flying short distances between perches (Hayward 1987). They spend a majority of time perched; little time is spent actively pursuing prey. While perched, the owl constantly looks about with rapid head movements, apparently responding to forest sounds. We distinguished hunting activity from roosting by the rapid, jerking head movements of the bird and the bird's intense, wide-eyed appearance.

To quantify the strategy employed by boreal owls during foraging, we recorded flight distance between perches, time spent on each perch, and perch height during diurnal and nocturnal foraging bouts (Fig. 4). We observed hunting owls on 16 nights and recorded quantitative data on 10 occasions for 5 owls. These observations spanned 31.25 hours. After dark, the owl often was seen only intermittently despite our use of night vision goggles and beta-lights (Hayward 1987). We watched owls hunt during daytime 13 times and recorded quantitative results on 8 days for 4 owls. Diurnal observations spanned 7.2 hours. All quantitative results refer to foraging during summer (snow-free conditions) although we did pursue foraging birds in winter.

Owls flew an average of 25 ± 8 m between hunting perches in 123 recorded flight distances for 8 owls. The median distance between perches was 17 m, and over 90% of recorded flights were estimated to be <40 m (Fig. 4). In the forest environment we were unable to record long flights; therefore, our sample has a negative bias. We feel this bias influenced the observations only slightly. Over 75% of all flights were 25 m or less. Most often when we lost contact with an owl, the radio signal suggested that the bird made several fights with only brief perching periods.

Although the pattern of foraging flights varied, the owls often concentrated their activity in a relatively small area compared to the total length of all flights in a foraging bout. The owls we followed doubled back frequently and thus covered a relatively small area within several forest stands rather than a long narrow path.

While searching for prey, boreal owls perch on low branches. Perches during foraging observations averaged 4 ± 0.6 m high ($n = 114$). Seventy-five percent of 114 perches were <5.5 m and ranged from 0.5 to 9 m (Fig. 4).

Boreal owls search briefly from each hunting perch, but, as shown above, hunt a forest stand intensively by moving short distances between perches. We measured the duration of 150 hunting perches during 18 foraging bouts of 8 owls. Over 78% of perches were occupied <5 minutes, 64% for <3 minutes, and 27% for 1 minute or less (Fig. 4). Four percent of hunting perches were used >10 minutes. If an owl used a perch for >10 minutes, it either ceased foraging or was intently listening to a potential prey. One owl, observed hunting at night, watched a tight clump of branches in a lodgepole for 12 minutes prior to flying 2 m to attack a roosting passerine. Another owl, hunting during daylight, flew approximately 4 m to catch a southern red-backed vole (*Clethrionomys gapperi*) deep in a clump of common beargrass (*Xerophyllum tenax*) after watching the spot for 10 minutes.

Seasonal Movements.—Boreal owl activity shifted in elevation between summer and winter, but the change in activity did not involve a complete shift in home range. Each of 12 owls radio marked during both winter and summer used summer roost sites with a higher average elevation. The difference in average seasonal roost elevation was 186 ± 105 m ($n = 12$). Three owls (2 males and a female) used roosts during summer that averaged over 335 m higher than winter roosts.

Although the owls demonstrated a consistent tendency to concentrate use at higher elevations in summer, the range of elevations used in winter and summer overlapped completely. During winter, owls spent some time in the highest portions of the study area despite snow depths over 2 m. Likewise, in summer, owls roosted on some occasions in the lowest portion of their home range. The major difference in seasonal use was the proportion of time that birds spent in each elevation zone.

Female movements after the brooding period (nestling age of 20–26 days) were variable. During incubation and brooding, 8 radio-marked females were never found over 200 m from the nest tree. When the young reached 20–26 days old, females ceased occupying the cavity. We monitored 6 radio-marked females immediately following the brooding period. In 2 cases, females left the study area within 3 days of leaving the nest cavity. In both cases, the young fledged successfully. In 2 cases, females occupied the home ranges used prior to nesting and assisted in feeding the young at the cavity and after fledging. One female assisted feeding young at the nest and remained within 3 km of the nest for 2 weeks. She then moved 17 km within 3 days and settled in an area of <400 ha that she occupied for at least the next 2 months. In a final case, the nesting male abandoned the nest 1–3 days prior to the normal date for the female to leave the nest. She left the nest 1 day later on 10 July. Marks on an event recorder at the nest and fresh prey in the cavity on 15 and 19 July suggested that the female fed the young on 2 nights. By 22 July, however, she began using an area 4.5 km from the nest where she remained for 2 months.

Year-to-year Movements.—Movements of radio-marked owls provide some indi-

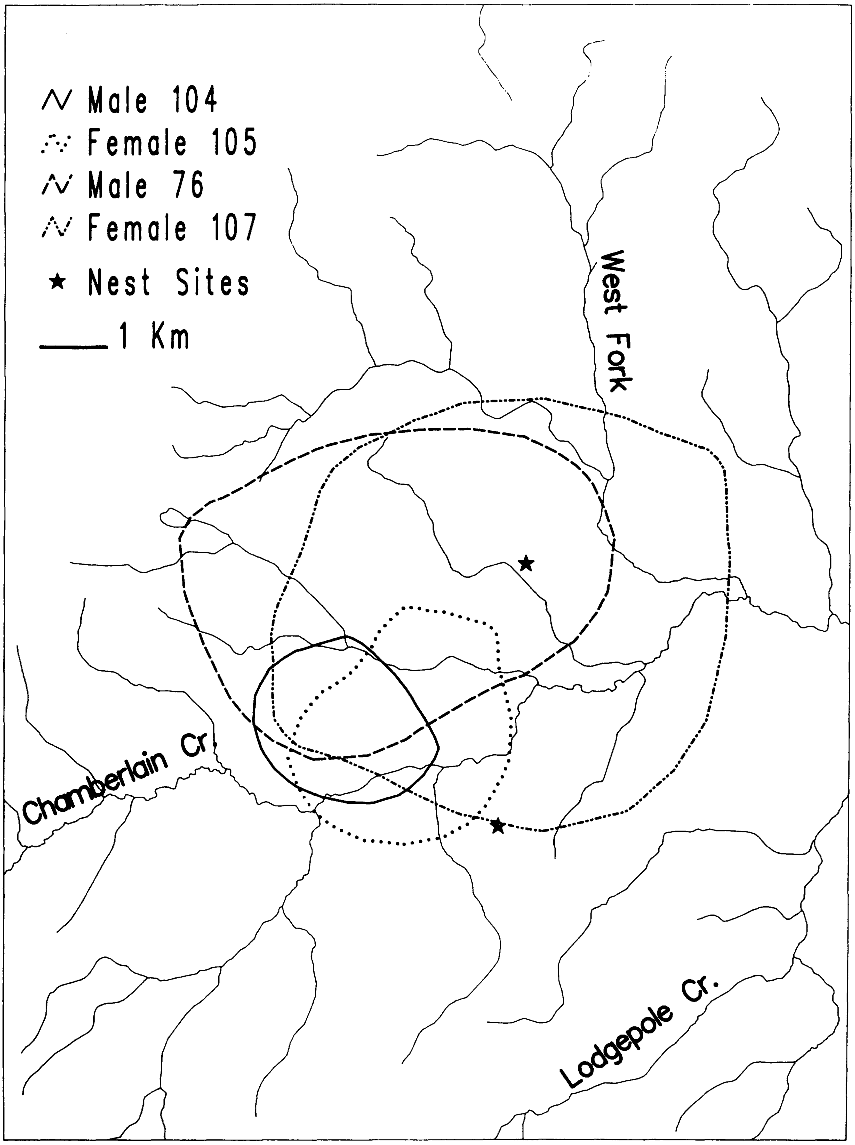

Fig. 5. Seasonal 95% utilization distributions of 4 boreal owls based on daytime roost locations in Chamberlain Basin, Idaho, during winter 1987. Owls 104 and 105 nested at the southern (lower on figure) site, and owls 76 and 107 initiated a nest at the northern site.

cation of the degree of site tenacity in the RNRW. We gathered evidence suggesting boreal owls in RNRW exhibit both year-to-year site tenacity and nomadic emigration. In late summer or autumn 1984–87, we placed new radio transmitters on 4, 5, 2, and 5 owls, respectively, in an effort to relocate the birds the following winter (4 months later). We observed male and female boreal owls who stayed in the basin for >1 year and used the same home range. We also documented emigration of adults from the population.

Using evidence from both radio-marked birds and banded individuals, we found 6 males and 4 females that remained in the basin for >1 year. The 6 radio-marked males each used portions of their original home range in the second year. For 2 males monitored through both winter and summer in 2 years (≥20 locations/season), seasonal home ranges from the 2 years

Table 12. Seasonal home range size (ha) for boreal owls in Chamberlain Basin, Idaho, during 1984–87. Areas are 95% utilization distributions calculated using the harmonic mean method (Samuel et al. 1985).

			Home range area					
			Winter		Summer		Combined	
Year	Owl	Sex	Size	No. of locations	Size	No. of locations	Size	No. of locations
1984	33	M					814	13
	34	F	320	10			607	20
	37	F	1,504	16			4,127	24
	42	M			610	16	1,166	24
1985	43	M	1,411	19	530	14	2,341	33
	76	M	1,282	24	229	19	2,581	43
	77	M	2,359	31	1,265	14	2,360	45
	84	M	946	12	1,421	17	2,141	29
1986	95	F	1,832	12				
	96	M			1,100	53	1,438	58
	97	M	2,001	13	2,386	27	6,876	40
	55	F			1,448	40		
1987	96	M	261	21	797	37	911	58
	76	M	2,259	28	1,520	45	1,761	73
	104	M	473	22	884	38	1,019	60
	105	F	826	17	747	21	1,122	38
	107	F	3,390	30	2,037	17	3,517	47
	117	M			1,884	20	1,161	26
	133	F			874	18		

matched closely. One of the males nested in cavities 1.3 km apart in consecutive years. Home range information was not as complete for the 4 females that were radio marked for more than 1 year; however, all 4 remained within the Chamberlain study area.

Each year we checked most nest cavities used by owls in previous years ($n = 23$). Although 2 nest trees were reoccupied, we never located an individual female in the same cavity she used previously. Nest sites of 2 individuals were known for 2 years. One female nested with different mates in 1984 and 1986 in cavities 1.4 km apart. The second female used cavities 7.6 km apart in 1987 and 1988.

In view of the evidence for year-round residency, several observations of emigration are important. As presented earlier, 2 females emigrated immediately after nesting and a third moved 17 km 2 weeks after nesting. We also witnessed the emigration of 2 males during a 2-week period in early February 1986. During the same 2-week period, another male died (1986 marked the low point in breeding effort and breeding success during the study). Both emigrating males wore new radio transmitters. Neither owl was relocated in the study area during repeated ground and air searches that winter. One of the males, first captured in February 1985 and monitored until February 1986, was relocated on 7 May 1986 near Upper Payette Lake, Valley County, Idaho, 80 km to the west of its former home range. The other male, first captured in February 1984, was monitored moving south from its home range and left the basin after living there at least 2 years.

Home Range Area.—We estimated the area used by boreal owls at Chamberlain during winter and summer from radio-marked owls in 1984–87 (Fig. 5). Mean winter home ranges averaged 1,451 ± 552 ha (adaptive kernel estimate 2,487 ± 1,218 ha) for 13 owls (5 females, 8 males) over the 4-year study. Summer ranges generally covered smaller areas; they averaged 1,182 ± 335 ha (adaptive kernel estimate 2,269 ± 1,644 ha) for 15 owls (4 females, 11 males). Year-round ranges averaged 2,048 ± 818 ha (adaptive kernel estimate 3,750 ± 1,645 ha) for 17 owls (4 females, 12 males) (Table 12). These values must be considered minimum home ranges. Our

Table 13. Diet of boreal owls at Chamberlain Basin, Idaho, estimated from pellets, cached prey, and nest contents of 31 owls in 1981 and 1984–88.

	Percent of prey items						Year-round	
	Winter			Summer			% of prey items	Biomass[a] (%)
	Male	Female	Total	Male	Female	Total		
Mammals								
Southern red-backed vole (*Clethrionomys gapperi*)	49	49	49	34	13	31	36	37
Northern pocket gopher (*Thomomys talpoides*)	8	6	7	10	16	11	10	26
Unidentified shrews (*Sorex* spp.)	15	6	12	11	10	11	11	3
Unidentified voles (*Microtus* spp.)	12	7	11	8	12	8	9	11
Deer mouse (*Peromyscus maniculatus*)	6	10	7	5	5	5	6	5
Heather vole (*Phenacomys intermedius*)	2	0	2	5	2	4	4	3
Northern flying squirrel (*Glaucomys sabrinus*)	1	14	5	0	tr[b]	tr	1	7
Yellow-pine chipmunk (*Tamias amoenus*)	2	0	1	2	2	2	2	3
Western jumping mouse (*Zapus princeps*)	0	0	0	3	0	2	2	1
Pika (*Ochotona princeps*)	0	0	0	tr	1	tr	tr	tr
Bushy-tailed woodrat (*Neotoma cinerea*)	0	0	0	0	1	tr	tr	tr
Unidentified weasel (*Mustela* spp.)	0	1	tr	0	0	0	tr	tr
Water vole (*Microtus richardsoni*)	0	0	tr	0	tr	tr	tr	tr
Birds	4	6	5	6	1	5	5	3
Insects	1	0	1	16	37	18	13	1
Total count	144	69	242	572	93	672	914	26,162

[a] Biomass calculated using values in Table 14.

[b] tr indicates <1%.

Table 14. Mean mass of small mammals used to calculate biomass of prey consumed by boreal owls. Measurements from individuals trapped in Chamberlain Basin, Idaho, 1984–87, except northern pocket gopher and northern flying squirrel values from Hall (1946).

Species	Age	Mass (g)	±95% CL	Sample size
Southern red-backed vole	Adult	26.5	0.74	100
Southern red-backed vole	Juvenile	12.2	0.67	48
Deer mouse	Adult	24.2	1.15	32
Deer mouse	Juvenile	11.5	2.16	6
Heather vole	Adult	22.7	3.47	9
Heather vole	Juvenile	13.8	5.01	3
Unidentified voles	Adult	30.0	3.01	12
Unidentified voles	Juvenile	12.2	2.11	9
Unidentified shrews		6.10	0.58	18
Western jumping mouse	Adult	20.81	2.57	14
Yellow-pine chipmunk		50.8	2.30	24
Northern pocket gopher		101.2		
Northern flying squirrel		140		

sample of each owl's activity was small, and (similar to other kernel estimators) the harmonic mean is biased low with small samples. Home range estimates, however, did not increase with sample size (*see* Hayward et al. 1987*b* for further discussion of seasonal home range use).

Home Range Overlap.—Our observations indicated that boreal owls did not use exclusive home ranges. Home ranges overlapped considerably, regardless of owl sex (e.g., Fig. 5). All of the owls we monitored in Ranch Creek or Flossie Creek drainages (where our sample was largest, $n = 13$) had a home range that overlapped some other monitored owl by >50%. Because they are most likely to defend a nesting territory, overlap of male home ranges is of greater interest (Lundberg 1979). In any year, male owls inhabiting the same drainage used the same forest stands intensively. During winter, when males establish small nesting *territories* (Mikkola 1983), distances between harmonic mean centers of activity (Samuel et al. 1985) were as low as 840 m. During summer, harmonic mean centers of activity were as close as 1,450 m, and neighboring nest sites were within 700 m. We found males roosting within 200 m of one another in both winter and summer. On 2 nights we captured 2 and 3 males at a single trapping site indicating common use of the site.

Food Habits

Boreal owls at Chamberlain killed a wide range of prey including small mammals, small birds, and insects (Table 13). The list of small mammals in the diet includes all species of mammals smaller than 50 g known to inhabit the basin except the water shrew (*Sorex palustris*). Because of incomplete skulls, bones of several large shrews found in pellets could not be identified, but were probably water shrews. Avian prey included 7 passerines and 1 Piciformes (Hayward 1989:46). Crickets numerically dominated insect remains.

The breadth of species represented in owls' diets contrasted with the narrow range of prey taken frequently. Small mammals accounted for 79.4% of 914 prey identified and over 95% of the biomass estimated from pellets and nest sites of 32 owls (Tables 13, 14). Southern red-backed voles and pocket gophers together accounted for over 63% of estimated prey biomass, underscoring the limited number of prey species important to the owls. Southern red-backed voles alone accounted for 36% of individual prey in our sample from winter and summer.

Diets of individual male owls differed significantly in both winter and summer (summer $\chi^2 = 55$, df = 25, $P < 0.01$; winter $\chi^2 = 20$, df = 10, $P = 0.025$). In our test of heterogeneity among individuals, we considered only those owls for whom we recorded 30 or more prey items in 1 season (6 males in summer, 3 males and 1 female in winter). We eliminated the single female from the winter sample because we wished to determine whether individuals of the same sex fed similarly within a season. In these tests we included only the 5 most common small mammal prey.

Major differences in diet among individuals involved unusual numbers of uncommon prey taken by particular owls. Although samples from some boreal owls included no avian prey, the diet of 1 male included 10% birds during summer (41% of all avian prey taken by 8 males during summer). This same male captured 54% of the chipmunks (*Tamias amoenus*) taken by males during summer. Two other individuals captured many western jumping mice (*Zapus princeps*) and crickets. Despite these differences, southern red-backed voles were the most frequent prey for all individuals in both winter and summer.

Seasonal Prey Use.—During winter, owls relied on southern red-backed voles for nearly 50% of prey (Table 13). The number of prey taxa available in winter is less than in summer and is reflected in fewer species in the winter diet. Northern pocket gophers (1 of the most frequent summer prey), western jumping mice, and yellow-pine chipmunks are all unavailable during the period of winter snow pack. (During early and late winter when snow

Table 15. Frequency of capture for 7 species of small mammal in 9 × 10 snap-trapping grids at 7 sites in Chamberlain Basin, Idaho, during 1984–87. Trap sites are labeled by dominant overstory vegetation. Each grid was run for 7 days during late May or early June for the number of years listed.

Small mammal species	Sage (4 yrs)		Wet meadow (3 yrs)		Lodgepole 1[a] (4 yrs)		Lodgepole 2[b] (3 yrs)		Douglas-fir (4 yrs)		Mixed conifer (5 yrs)		Spruce-fir (2 yrs)	
	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile
Southern red-backed vole	0	0	0	0	0	0	1	0	12	0	9	2	66	38
Deer mouse	3	0	0	0	3	0	1	0	7	1	7	3	0	0
Unidentified shrews[c]	0		16		4		1		0		1		71	
Unidentified voles	7	2	5	5	0	0	0	0	0	0	0	0	0	0
Western jumping mouse	2	0	1	0	8	0	0	0	0	0	3	0	0	0
Heather vole	0	0	0	0	1	0	0	0	0	1	0	0	1	0
Yellow-pine chipmunk	18	0	1	0	3	0	4	0	1	0	4	0	0	0

[a] Lodgepole-pine forest with lush understory vegetation and some downed logs.
[b] Lodgepole-pine forest with dry understory and no downed logs.
[c] Shrews were not aged.

cover is patchy, these species are captured and, therefore, are reported in the winter diet.) In spring, the earliest recorded dates for chipmunks or jumping mice in pellets or nest material were 14 March and 22 May, respectively. Pocket gophers were recorded in pellets by 15 March. Once available, pocket gophers were taken often and accounted for 15% of overall winter prey biomass (7% by frequency) despite the limited period they were available. Based on skeletal measurements (Janes and Barss 1985) of a sample of pocket gopher remains in pellets ($n = 17$), 41% of pocket gophers captured were 30 days old or younger.

Flying squirrels were the only prey taken much more often in winter than summer. Of 12 recorded flying squirrels, 11 were captured during winter, 10 of these by females. The squirrels represented 45% of prey biomass recorded for female owls during winter, indicating the importance of these prey during a period when other prey are less available.

During summer, southern red-backed voles continued to be the most frequent prey and accounted for 29% of biomass consumed. The owl summer diet was diverse compared to winter with the addition of chipmunks, jumping mice, and crickets. Crickets may be more important to owls than our sample suggests. We rarely found crickets during searches at nests for prey, but photographs of prey deliveries showed several crickets. Also, pellets composed largely of insects break apart more quickly and, therefore, are less likely to be found at roosts.

Yearly Variation.—The frequency of southern red-backed voles in prey samples was lower in 1986 (the year of poor owl breeding) than in other years. In 1986, 26% of prey were southern red-backed voles compared to 38, 44, and 45% in 1984, 1985, and 1987, respectively. As a result, deer mice (*Peromyscus maniculatus*), pocket gophers, and heather voles (*Phenacomys intermedius*) increased in pellets in 1986 compared with other years. The frequency of *Microtus* spp. remained relatively constant from 1984 to 1987 and averaged 11%. The frequency of the other common prey,

Table 16. Frequency of capture for small mammals in pit traps set year-round at Chamberlain Basin, Idaho, during 1984–87 in 7 vegetation types defined by dominant overstory. Three sites were trapped for each vegetation type except spruce bottom, where only 2 sites were trapped. Trap days in thousands are recorded in parentheses.

Small mammal species	Sage (2.9)	Wet meadow (2.8)	Lodgepole (3.3)	Mixed conifer (3.3)	Douglas-fir (3.0)	Spruce bottom (2.3)	Spruce–fir (2.4)
Southern red-backed vole	0	0	9	5	25	32	52
Unidentified shrews	25	45	19	28	16	77	45
Deer mouse	11	0	5	10	2	5	3
Unidentified voles	23	5	0	0	0	0	0
Western jumping mouse	4	1	1	0	0	1	0
Heather vole	0	0	2	1	0	0	3
Yellow-pine chipmunk	6	0	9	18	6	0	0
Northern pocket gopher	4	3	5	0	1	0	1

shrews, and birds also remained relatively constant.

Small Mammal Habitat Use—Comparison of Sampling Methods.—Because small mammals accounted for 80% of the boreal owl diet at Chamberlain, we used snap and pit trapping to determine habitat associations of small mammals (Tables 15, 16). Neither method can give a completely accurate ranking of species abundance within various habitats. By using both methods we sought to sample a broad range of mammal species and rank the importance of various vegetation types for each prey species.

Southern red-backed voles, shrews, and deer mice were the most frequently captured species by both methods (Tables 15, 16). To compare the 2 trapping methods, we examined capture rates for these 3 small mammals across 6 vegetation types using the 2 trapping methods. Capture rates of southern red-backed voles in pit and snap traps were strongly related (Spearman rank correlation, $r_s = 0.94$). The 2 trapping methods also resulted in similar ranking for shrew abundance ($r_s = 0.75$). Capture rate for deer mice differed ($r_s < 0.10$) mainly because pit traps in Douglas-fir forests captured few mice, whereas the number of snap-trapped mice was highest for this habitat. Otherwise the rank order of the habitats was similar for snap- and pit-trap samples of deer mice.

An important difference in pit-trap and snap-trap samples was the frequency of pocket gophers in the pit-trap sample (Tables 15, 16). Pit traps captured pocket gophers on 5 sites, whereas snap traps did not capture gophers.

Vegetation Association of Small Mammals.—Southern red-backed voles were captured in both pit and snap traps more frequently in spruce–fir forest than other types (Tables 15, 16). Average pit-trap capture rates across all years and sites were 2.4 times greater in upland spruce–fir than other types. Snap trapping indicated even larger differences in vole abundance between spruce–fir and other types. Average capture rates in spruce–fir exceeded all other forests by an order of magnitude. Southern red-backed voles occurred in all forest vegetation types but capture rates were low in lodgepole-pine and dry mixed-conifer forests. In both spruce–fir and Douglas-fir forests the biomass of southern red-backed vole captures exceeded that of other small mammals. We did not capture southern red-backed voles on nonforested sites.

We combined *Sorex vagrans* and *S. cinereus* for discussion of distribution and abundance. We captured shrews at all pit-trap sites, and variation in capture rates among habitats was lower for shrews than other small mammals (Table 16). Shrews were captured by both trapping methods most frequently in spruce–fir forests, including both spruce bottoms and upland spruce–fir forests. Using pit traps, more shrews than other mammals were captured in lodgepole-pine, mixed-conifer, and unforested habitats.

We captured deer mice in all vegetation types except wet meadow. Pit- and snap-

Table 17. Trend in boreal owl populations at Chamberlain Basin, Idaho, during 1984–87.

	Survey period			
Survey effort and results	1984 18 Jan–23 Apr	1985[a] 25 Jan–2 May	1986 14 Jan–7 May	1987 16 Feb–27 Apr
No. surveys	13	29	32	36
No. listening nights	5	16	5	19
Kilometers surveyed	64	195	217	218
% surveying or trapping nights males heard	56	51	8	49
% surveying nights calling males heard	62	48	6	53
No. calling males heard per survey km	0.24	0.07	0.02	0.14
No. owls captured				
winter	9	5	3	7
year	9	6	4	8

[a] Four people worked in 1985 and 1987; 2 people in 1984 and 1986.

trap results, however, differed in the rank order of deer mice captures across habitats. In pit-trap samples, deer mice were most frequently captured in sagebrush–bunchgrass habitats where they were the most frequently captured mammal. Deer mice also were captured often in mixed conifer by pit traps, but rarely in Douglas-fir. Snap-trapping results showed Douglas-fir as the most important habitat for deer mice followed by mixed conifer, lodgepole, and sagebrush–bunchgrass. During 2 years of snap trapping in the spruce–fir grid, no deer mice were captured.

We found chipmunks in all habitats except spruce–fir bottoms and spruce–fir uplands. Capture rates were the highest in the drier habitats—sagebrush–bunchgrass (snap trapping) and mixed conifer (pit trapping).

Montane voles (*Microtus montanus*) and long-tailed voles (*M. longicaudus*) used narrower ranges of habitat than other common small mammal species. Using both the pit and the snap traps, we caught *Microtus* spp. only in nonforested habitats. *Microtus* capture rates were higher at sagebrush–bunchgrass sites than in wet meadow.

Pocket gophers, jumping mice, and heather voles were caught in several habitats, but less frequently than the species discussed above. Dirt mounds and "soil ropes" seen throughout the study area suggested that pocket gophers used all but the most rocky habitats. Pit-trap capture rates of pocket gophers were highest in sagebrush–bunchgrass and lodgepole habitats. Jumping mice were found in a variety of forested and nonforested habitats, but were never captured at our Douglas-fir or spruce–fir sites (Table 16). Heather voles were rarely captured (8 individuals in pit and snap traps combined), yet they were found in all forest vegetation types. These voles did not occur in any of the nonforested types.

The water shrew was captured with a pit trap on a single site in a wet meadow. This site was inundated for several weeks each spring.

Owl Population Characteristics

Yearly Variation in Owl Abundance.—During the study, we recorded data that may be used to form several crude indices to breeding abundance and breeding productivity each year. Together these measures provide evidence of the minimum number of breeding owls and year-to-year changes in breeding effort or productivity at Chamberlain and Cold Meadows. Each of the indices discussed below is untested and, therefore, has unknown bias.

Nighttime surveys using playback of taped boreal owl calls during each winter show the yearly variation in breeding population abundance from 1984 to 1987 at Chamberlain and 1984 to 1986 at Cold Meadows (Tables 17, 18). At Chamberlain, 2 related indices show a slight decline in 1985 followed by a substantial decrease in breeding activity in 1986. The proportion

Table 18. Trend in boreal owl populations at Cold Meadows, Idaho, during 1984–86.

Survey effort and results	Survey periods 1984 19 Feb–4 Mar 3 Apr–11 Apr	1985 25 Feb–7 Mar 6 Apr–18 Apr	1986 27 Mar–11 Apr
No. surveys	17	20	11
Kilometers surveyed	74	100	70
% surveying nights calling males heard	40	20	18
No. calling males heard per survey km	0.20	0.07	0.09

of survey nights on which we heard boreal owls differed significantly among years ($\chi^2 = 20.2$, df = 3, $P < 0.0001$). The low number of successful nights in 1986 contributed most to the difference (cell $\chi^2 = 8.54$, $P = 0.0035$). If instead, we look at the number of male owls heard singing per mile surveyed, the same pattern is apparent—owl calling rate dropped from a high in 1984 to a low in 1986 with a recovery in 1987 (Table 17). This result suggests that differences between years in the number of miles surveyed in a given night or the total number of survey nights was not responsible for the observed trend. Each year of the study some new routes were surveyed, which added to the total survey mileage. Routes surveyed in earlier years were always resurveyed.

Survey results at Cold Meadows showed a similar pattern. The proportion of survey nights that we heard boreal owls was twice as large in 1984 as in 1985 or 1986, but the difference was not significant ($\chi^2 = 2.65$, df = 2, $P = 0.265$). The number of calling males heard per survey mile was higher in 1984 (Table 18) than in either of the following years.

The pattern of declining breeding populations of boreal owls from 1984 to 1986 before rebounding in 1987 also was reflected in our winter trapping success at Chamberlain. We captured 9 boreal owls in 1984, 5 in 1985, 3 in 1986, and 7 in 1987. Trapping effort for each year is difficult to quantify because our trapping strategy differed among years. The number of nights we trapped at sites where we had little confidence of capturing an owl differed among years. Using the number of trapping nights as an imperfect measure of effort, catch per unit effort during winter equaled 1.8 for 1984, 0.29 for 1985, 0.60 for 1986, and 1.67 for 1987. Again, 1985 and 1986 showed the lowest values.

As a final index to breeding population trend at Chamberlain, we used a method similar to spot mapping. Our earlier indices using number of males heard per mile surveyed or per survey night included owls heard calling from the same site during different surveys. We sought to remove this bias by defining the number of male territories located each year by assigning singing males heard in 1 forest stand on different surveys to 1 territory. We found 12 territories in 1984, 14 in 1985, 3 in 1986, and 19 in 1987. During the study, the area surveyed increased each year. An imperfect way of standardizing the number of territories is to divide by the number of kilometers surveyed. Territories located per kilometer surveyed equaled 0.65 in 1984, 0.24 in 1985, 0.04 in 1986, and 0.26 in 1987.

Yearly Variation in Productivity.—Productivity at nest sites was not constant from year to year. At Chamberlain we located 3 nests in 1984, 2 in 1985, 3 in 1986, and 8 in 1987. The number found was related to the owl breeding effort, the number of radio-marked owls, and our experience locating nests. From this small sample, we observed the rate of nest abandonment, clutch size, and number of young fledged per nest that fledged young.

In all but 2 cases, we observed the clutch size during incubation or at hatching, so we can only estimate minimum clutch size. During 1984–87, boreal owl clutches ranged from 2 to 4 eggs and averaged 3.3, 2.5, 2.5, and 3.5 eggs for each year 1984–

87, respectively. Only 2 nests during this period fledged 3 young (1 nest in 1984 and 1 nest in 1987); for the remaining nests, those which fledged young each produced 2 owlets. Clutch size and fledging rate, then, both indicated that 1984 and 1987 were more productive than the intervening 2 years.

Although clutch size and the number of fledglings per successful nest were higher in 1984 and 1987, complete nest failure appeared to contribute most to variation in productivity. Nests failed due to parental abandonment, nest predation, and death of an adult. The pattern of nest failure across years appeared to follow the other measures of breeding activity (calling) and productivity. During 1984–87, nest histories included complete success of 3 nests in 1984, abandonment of 1 of 2 nests in 1985, loss of all 3 nests in 1986, and loss of 6 of 8 nests in 1987. The timing and reasons for nest failure each year is important in evaluating trends in productivity. In years when other measures of productivity were low (1985, 1986), 3 of 4 nests that failed were abandoned early in the nesting cycle. The other nest failed due to predation. In good years (1984, 1987) predation was responsible for all nests that were lost.

Annual Survival.—During 1984–88, we monitored 24 adult boreal owls from late January through August using radiotelemetry. Each autumn we placed a fresh radio on any marked birds in an effort to recover the birds the following January. Ten owls were followed during more than 1 year. Records from the radio-marked owls give some insight into mortality rates and emigration from the population.

Radio packages certainly increase energy expenditure of the owls and may increase risk of predation. The extent to which radio packages reduced survival of marked owls cannot be assessed with our data. The longevity we witnessed in several marked owls suggests that the impact was not severe. One male and 1 female wore radio tags for 916 and 824 days, respectively, and were still alive at the end of the monitoring period. More importantly, we weighed each marked owl upon initial capture and at each recapture. Average recapture weights exceeded initial capture weights for 4 of 7 individuals who were weighed on 4 or more occasions. For owls recaptured on fewer occasions, body mass showed no trend. These results suggest that the radio transmitters did not prevent the owls from maintaining normal body mass. Whether the owls were forced to actively forage for longer periods, exposing themselves to predation, cannot be addressed.

During the study, 6 owls (3 males and 3 females) died. Two owls that died during winter (a male and a female) appeared to have starved; they showed no sign of injury. The male who had been monitored for 201 days was found within hours of his death and had lost 23 g (20% of body weight) in 12 days. Three birds (2 females and a male) were consumed in part or entirely by predators. Another male died of unknown causes.

A minimum of 5 marked owls (2 males and 3 females) was lost from the population through emigration. In 3 cases, we monitored the owls as they left the basin; 1 was relocated 80 km west, near Upper Payette Lake, Valley County, Idaho. Behavior prior to loss of contact led us to conclude that 2 other owls also emigrated.

We used information from the radio-marked owls to estimate survival of adult boreal owls (Heisey and Fuller 1985). Our analysis assumes that adult male and female boreal owls experience equal survival and that survival rate during any given year can be treated as constant throughout the year but that survival may differ between years. Because of relatively small samples within any year, estimates of yearly survival include extreme confidence intervals; therefore, we consider estimates for the entire study period only.

Because we monitored owls from midwinter through summer in most years, estimates of survival restricted to this period are most legitimate. Our analysis, then, treats only the fate of owls during this monitoring period. For instance, an owl that was remarked in autumn but could not be located the next year was not considered a loss because it survived during the monitoring period. If an owl was observed emigrating from the area, it was included in the analysis as described below.

It could be argued that owls who emigrate are rarely recruited to a new population. If we assume that all birds that emigrated during the monitoring period died and that no adult owls successfully immigrated into the population, then all emigration can be treated as a loss from the adult breeding population. These assumptions provide a conservative estimate of survival for adult, radio-marked owls during our investigation. Under these assumptions, finite survival from midwinter through summer, expressed as a yearly rate, averaged 20% (95% CI = 7–55%).

A liberal estimate of survival for adults in the owl population could be formed by assuming that all emigrating owls survived and that immigration into the population equals the emigration we observed. Under this assumption, only direct mortality is considered a loss. Finite survival from midwinter through summer expressed as a yearly rate averaged 46% (95% CI = 23–91%).

Breeding Chronology: Courtship.—At Chamberlain, male boreal owls began courtship singing by late January in most years. In 1984 and 1985, we began surveys during the last week of January and heard owls on the first or second night in both cases. The calling rate, measured as the number of owls heard per survey night, increased from January through March.

Female owls were observed at male calling sites early in the courtship period. On 5 February 1984, we captured a female at a calling site while a male called 10 m away. The same year, we heard a male and female together on 7 February. In 1987, we heard both members of a mated pair on 16 February (our first night surveying the route) and captured both owls on 20 February. The owls repeatedly visited the site to court throughout the winter and began nesting 20 April in a snag about 40 m from the capture site. In 1986 both members of a pair were heard calling at a nest cavity 2 months prior to egg laying.

Throughout courtship, the birds associated with one another mainly during nighttime rendezvous at the potential nest site; members of radio-marked, mated pairs rarely roosted together during daytime. Our observations suggest that pairs do not rendezvous at the nest every night during courtship (Hayward 1989:56–57). On nights when the owls do rendezvous, they meet at any time through the night rather than consistently meeting shortly after sundown.

Our observations suggest that at night, during the courtship period, male boreal owls sing most often within 150 m of a selected nest cavity, an exception being unmated owls early in the courtship period. During observations at nests of 4 radio-marked males on 14 nights prior to egg laying, the males never called farther than 150 m from the nest site. Usually the male sang from a tree adjacent to the nest tree or from inside the nest cavity. Because we were not following these males during foraging, however, these observations were not sufficient to determine if the males were singing at sites out of our hearing range. Other observations do suggest that early in the courtship period unmated owls call from several widely dispersed locations and that males visit stands where other males are singing. Even these dispersed singing locations were later determined to be sites used by boreal owls for nesting sometime during the study. On 14 February 1984, we captured 3 males while we played the boreal owl song at 1 site. One of the owls nested at the capture site that year, and 1 of the others moved about 6 km before the nesting season. On 18 February 1985, we captured 2 males when we played the courtship call at 1 site. One of these owls (BO77) nested in a stand 1.6 km from the capture site, and the other did not breed. The capture site had been used for nesting the previous year by a different male. Finally, on 31 March 1985, a male (BO84) was captured and radio tagged in the stand used by BO77 for nesting that year. BO84 was heard calling in the stand

Table 19. Nesting chronology of boreal owls at Chamberlain Basin, Idaho. In most cases laying date is estimated by aging nestlings found 1–4 days after hatching. We assumed in all cases that incubation lasted 29 days (Korpimaki 1981). First visit refers to the first time we observed eggs or young at the nest.

Year	Owl	Occupancy	Laying date	Hatching date	Fledging date	Female off	First visit
1984	34		17 May	15 Jun	7–12 Jul		Youngest 1 day old
	55		14 May	14 Jun[a]	15 Jul	4–6 Jul	Third egg laid
	58	18 Apr	7 May	5 Jun	2–8 Jul	22–26 Jul	Youngest 1 day old
1985	86	23 Apr	15 May	13 Jun	15 Jul	5 Jul	Youngest 1 day old
	87		20 May	18 Jun			Youngest 2 day old
1986	95	18 Apr	by 30 Apr[b]				
	55		24 May	22 Jun			During laying
1987	105	13–14 Apr	21 Apr	20 May	16–18 Jun	9 Jun	Youngest 2 days old
	107	16 Apr[c]					
	107	26–30 Apr	1 May	29 May			Youngest 4 days old
	115		12 Apr	11 May			Youngest 12 days old
	128	16 Apr[d]					
	133		17 Apr	16 May	14–15 Jun	6 Jun	Youngest 6 days old
1988	134		14 Apr	13 May			Unhatched egg
	135		17 Apr	18 May			Two unhatched eggs

[a] The date the third egg was laid and the date the first egg hatched were both known for this nest. The female had abandoned the nest for 2 days (between the second and third egg), which likely accounts for the long incubation period.

[b] Female abandoned the nest on 30 April without being disturbed, and 2 eggs had been laid.

[c] Female abandoned her first nest due to disturbance and initiated a second nest by 30 April.

[d] Female was killed by a predator away from the nest before laying.

on 7 April, but he failed to breed that year. On 24 March, when we played a tape-recorded boreal owl song at a calling site, 4 radio-marked owls, including the pair that nested at that site, moved to the stand.

Breeding Chronology: Nesting.—The end of courtship and beginning of nesting is not well defined because female boreal owls begin occupying the nest cavity prior to egg laying. We observed both the onset of nest occupancy and egg laying dates for 3 female owls. These birds began roosting in the nest cavity 10, 19, and 22 days prior to egg laying (Table 19). Early in this period, the female may spend some time away from the cavity foraging at night. Also the female may occupy the cavity intermittently. In 1987, a female occupied the cavity every other day for the first week of nest occupancy before spending consecutive days on the nest. On the other hand, in 1984, another female began roosting in the nest cavity on 18 April and appeared to occupy the site continuously for 16 days before egg laying. The use of the nest cavity for roosting demonstrates an important change in behavior associated with nesting. Other than female owls occupying cavities during nesting, we never found a boreal owl roosting in a cavity in 882 roost locations.

Females consistently occupied the nest cavity in mid-to-late April (Table 19). Six recorded occupancy dates over 4 years ranged from 13 to 23 April. In 1987, 1 bird laid eggs by 12 April indicating an earlier occupancy date. Recorded laying dates ranged from 12 April to 24 May. Although our sample is extremely small, there is some suggestion that laying dates are rather consistent within years and vary more between years (Table 19). A majority of our data on laying dates came from radio-marked birds, so we know these observations do not represent renesting dates.

Females occupy the nest cavity day and night through incubation and most of the nestling period. We recorded the date when females ceased occupying the nest cavity with their young for 5 owls over 4 years. Four of these owls raised 2 nestlings each and left the nest when the oldest nestling was 20–22, 17–21, 22, and 20 days old. The other female raised 3 nestlings and left the cavity when the oldest was 21 days old. At 4 nests where we recorded fledging dates, the oldest owlet fledged at 31, 32, 27–29, and 29–30 days old (Table 19).

DISCUSSION

Habitat Use

Nesting Habitat.—Our observation of nesting habitat demonstrated that boreal owls in the RNRW used sites that were not randomly chosen from among available habitats. Compared to available vegetation, breeding habitat use was concentrated in 2 types, mixed conifer and aspen. Forest structure at breeding sites also differed from the range of structures available. Furthermore, although not conclusive, a nest-box experiment implied avoidance of potential nest sites in lodgepole-pine stands when cavities were available elsewhere.

Boreal owl breeding habitat has not been studied quantitatively in Europe. Qualitative descriptions, however, include a wide range of habitats. In Germany, Jorlitschka (1988) noted breeding activity in old forest but not young forest. In Finland, Korpimaki (1981, 1988*a*) recorded nests in Norway spruce (*Picea abies*) and Scotch pine (*Pinus sylvestris*) forests, edges of bogs and lake shores, and in boxes on hay barns in mowed pastures. In Norway, boreal owls also nested in "areas dominated by cultivated land," bogland, and mixed Scotch pine and Norway spruce forest (Solheim 1983*a*:81). When suitable cavities are available (usually in artificial structures), boreal owls nest in forest clearcuts and forest with simple canopy structure (Sonerud 1986; H. Stein, Univ. of Oslo, Oslo, Norway, pers. commun.).

The relatively narrow range of breeding habitat use we observed contrasts with the variety of sites described in Europe. Several hypotheses could explain the differences in range of breeding sites used. Abundance and productivity differ between owl populations in Fennoscandia and the RNRW (Solheim 1983*b*, Korpimaki 1987*c*, this study). Theory (Fretwell and Lucas 1969) and empirical results (e.g., Alatalo et al. 1985) suggest that the range of habitats used is related to population abundance. If owl populations are more dense in Fennoscandia, some individuals may be forced to use suboptimal breeding habitat (thereby expanding the range of sites used) especially in peak breeding years. Alternatively, breeding-site use may differ between Fennoscandia and the RNRW largely due to the distribution of available cavities among habitats. In Fennoscandian study sites, a majority of boreal owls nest in artificial structures and biologists note the paucity of natural cavities (Korpimaki 1981, Lofgren et al. 1986). The distribution of available nest sites is largely dictated by human placement of nest structures, and, in peak years, over 40% of available nest boxes may be used (Lofgren et al. 1986). Therefore, in Fennoscandia, nest-site use likely reflects nest-site availability rather than preference. In the RNRW, our nest-box experiment suggested that nest cavities were not limited. Use of a restricted range of vegetation conditions for breeding, then, may reflect preference; however, the availability of cavities was not the same across sites, and vegetation types with numerous cavities were used most frequently. Finally, compared to Europe, few breeding sites have been located in the northern Rockies. Therefore, sampling considerations alone could explain the differences.

We speculate that the consistent use of mature forest for breeding in the RNRW does not result from preference for mature forest structure *per se*. Rather, the owls may key on forests with this structure when searching for cavities because the probability of encountering suitable nest cavities is highest in forest with this structure. Mature and older conifer forest and aspen stands both have high densities of large diameter cavities. We believe the density of suitable cavities was much lower in other forest vegetation types. Although we did not measure cavity density, based on our observations during 4 years of field work, we believe density of large cavities may be nearly 2 orders of magnitude greater in mixed-conifer and aspen forests (the vegetation types used for nesting) than in lodgepole or spruce–fir forests.

The mixed-conifer forest and aspen stands used by boreal owls at Chamberlain were unique forest sites. Both occurred in relatively small stands within the extensive lodgepole-pine, Douglas-fir, and spruce–fir forests. The occurrence of ponderosa pine within the subalpine-fir life zone is not common, and therefore the abundance of nest sites at Chamberlain may have been higher than is found in other areas of the northern Rockies. Aspen stands with large diameter individuals also are not common in the northern Rockies. At Chamberlain these stands generally occur on unique soil–topographic sites and exist as almost pure aspen.

Roosting Habitat.—Roost sites differed from breeding habitat and from paired random locations in the RNRW. Our observations showed that winter roosts differed only slightly from random sites, whereas the evidence for differences was greater for summer roosts. Site choice appears most critical in summer. Temperature measurements at roost sites and observations of owls gullar fluttering while roosting suggest that moderation of high summer temperatures may be an important function of summer roosts. We observed owls exhibiting gullar fluttering only during summer. On 2 occasions when we observed resting owls exhibiting gullar fluttering, the air temperature was only 18 and 23 C indicating that the birds can be heat stressed at moderate ambient temperatures.

Normal winter temperatures in the RNRW may not stress boreal owls, which are well adapted for cold (Mikkola 1983). The thermal neutral zone of the owl may encompass a majority of the temperatures experienced during winter in this region. Warm summer conditions, however, appeared to stress boreal owls. Behaviors to ameliorate warm conditions may involve roost stand selection.

Roost habitat characteristics have not been reported for European populations. Palmer (1986) measured habitat at 174 roosts of 3 boreal owls in Colorado during 1 year. He noted a difference in roost characteristics and random sites but did not examine this pattern by season.

Foraging Habitat.—Our direct observations of foraging owls were insufficient to describe patterns of foraging habitat use. Observations of roost sites and food habits, however, provide empirical information that can be used to make inferences concerning foraging habitat use.

If roost sites represent the end of a foraging bout as we suspect from our observations of foraging owls, we may use our sample of roost locations as a preliminary sample of foraging sites. Data from 257 winter and 376 summer roost sites indicate frequent use of spruce–fir forests. In summer, 67% of all roosts were in spruce–fir forest and 26% in lodgepole. Winter roosts occurred in spruce–fir forest 35% of the time and in lodgepole forests 38% of the time. In both seasons, spruce–fir stands used by owls were usually mature or older forest sites. The frequent use of spruce–fir forest contrasts with the availability of spruce–fir in the area. Spruce–fir covers <10% of the study area, whereas lodgepole covers >50%.

Owl food habits and data on habitat use by small mammals support the contention that spruce–fir forest is important foraging habitat. During both winter and summer, southern red-backed voles were the most frequently captured prey (Table 13), especially in years when the owls bred most successfully. Spruce–fir forest supported the greatest number of southern red-backed voles and highest small mammal biomass. Southern red-backed voles were more abundant in spruce–fir forests than any other habitat we trapped. We captured 9 times as many southern red-backed voles in an old spruce–fir forest than we captured in any other vegetation type. Spruce–fir forests used by boreal owls in the RNRW were also generally old forest stands.

Palmer's (1986) observations of boreal owls in Colorado suggest that older spruce–fir forest is used for hunting. Studies in

Norway also have noted the importance of mature spruce forest for foraging (Sonerud 1986, Sonerud et al. 1986). Based on direct observations and composition of diet, the authors suggested that during winter and summer boreal owls foraged primarily in older forest sites. In early spring, immediately following snow melt, however, the owls hunted clearcuts. Sonerud (1986) speculated that the owls favored mature forest during winter because snow conditions (uncrusted snow) facilitated access to prey. In summer, mature forest sites had less herbaceous cover than open sites, which allowed greater access to prey. During the short period following spring thaw, before herbaceous vegetation became dense, the owls shifted to openings where densities of voles exceeded densities in forested stands.

Home Range Area

Integrating our results concerning nesting, roosting, and foraging habitat use indicates that resources used by owls are not all provided by any single vegetation type. Because of the natural segregation of vegetation types in the landscape, habitat components used on a daily basis, especially in the courtship and breeding seasons, were dispersed geographically. Mixed conifer and stands used for nesting lie on the eastern side of the study area at lower elevations as do aspen stands that contain many potential nest cavities. Spruce–fir forests used for roosting and foraging, especially during summer, are concentrated at high elevations to the west. The geographic dispersion of habitats used for nesting, roosting, and foraging may be responsible, in part, for the large home ranges used by boreal owls in the RNRW.

The average winter and summer ranges for boreal owls compare with those cited by Lindstedt et al. (1986) for coyote (*Canis latrans*), red fox (*Vulpes vulpes*), fisher (*Martes pennanti*), and American marten (*Martes americana*), all of which feed on similar prey. Among owls, our estimates of boreal owl ranges compare to, and in some cases exceed, those reported for larger species. Thomas et al. (1990) reported median home ranges of spotted owls (*Strix occidentalis*) from studies in California, Oregon, and Washington that ranged from 571 to 4,021 ha; Forsman et al. (1984) reported home ranges <2,000 ha for spotted owls in Oregon; Hirons (1985*a*) reported Tawny owl (*Strix aluco*) territories <100 ha in England; Bull et al. (1988*a*) measured great gray owl (*Strix nebulosa*) home ranges that averaged 6,730 ha in Oregon; and Smith and Gilbert (1984) calculated home ranges for Eastern screech-owls (*Otus asio*) of under 150 ha. Boreal owl home ranges fell within the range reported for the largest owl, the eagle owl (*Bubo bubo*) of Europe (Mikkola 1983).

Several factors likely contribute to large boreal owl ranges. As noted above, no single vegetation type provides optimum nesting, roosting, and foraging habitat, and these vegetation types are geographically disjunct. Therefore, geographic features lead to a broad dispersion of resources forcing the owls to move long distances. In addition to the geographic distribution of habitats, low productivity of small mammals also may contribute to large owl ranges. Lindstedt et al. (1986) showed that home range size among carnivores is related to prey production. Results of our small mammal studies suggest that abundance of mice and voles is quite low compared to regions inhabited by boreal owls in Scandinavia. On our most productive snap-trap grid (an old spruce–fir forest), we caught an average of 8.3 voles/100 trap nights during a week of trapping each spring for 2 years. We also caught 5.6 shrews/100 trap nights, or 13.9 small mammals/100 trap nights. On the next most productive forest site, an old Douglas-fir forest, we caught 0.9 mammals/100 trap nights; this sample did not include any shrews. Our trapping rate at the spruce–fir site was less than snap-trap capture rates reported by Lofgren et al. (1986) and Korpimaki (1987*a,c*) during vole peaks during their studies of boreal owls in Europe. Lofgren et al. (1986) captured 16.6 voles/100 trap nights during vole peaks and 1.4 during low years when boreal owls failed to breed. Similarly, Korpimaki (1987*a,c*) captured up to 18 voles/100 trap nights in Finland during peak vole years. During all years, our trapping rates in habitats other than spruce–fir were lower than those observed by Lofgren et al. (1986) during low prey years. The habitats with low abundance of small mammals dominate the landscape at Chamberlain. Spruce–fir forest covered about 6% of our study area, and old stands that may support the greatest abundance of small mammals constitute only a portion of the spruce–fir forest. These results imply that small mammals are not abundant across much of the study area.

Further evidence of low prey availability comes from our observations of boreal owl courtship feeding. Courtship for some boreal owls lasted for over 3 months. During much of this period the pair met at the nest site several times a week. We suspect that males provided courtship food during these encounters. Beginning up to 2 weeks prior to laying, the female occupied the nest cavity continuously, receiving prey from her mate. Apparently the female requires an extended period of inactivity to accumulate reserves for egg laying. Hirons (1985*b*) has shown that female weight prior to egg laying in the tawny owl is strongly related to nesting success. The small clutches laid by boreal owls at Chamberlain after such lengthy courtship reinforces our contention that boreal owls do not easily obtain sufficient prey from habitats at Chamberlain.

In addition to low abundance of prey in most habitats, the few productive, old spruce–fir stands are relatively small and dispersed. The broad dispersion of good foraging sites could force the owls to use large home ranges. Because of long travel distances between old spruce–fir patches, the owls may hunt extensive areas of poorer forest rather than moving directly from 1 spruce–fir stand to the next.

Population Status

Reproduction and Survival.—Observations of low reproduction and survival for boreal owls over 4 years in the RNRW contrast with estimates of demographic characteristics for populations in Europe. In the RNRW, completed clutches averaged 3.1 eggs (range 2–4, $n = 11$), and fledging rate for nests that fledged young averaged 2.33 young (range 2–3, $n = 6$). Nest failure was an important component of low production; 10 of 16 nests failed. In Europe, boreal owls produce far more young. In Finland, during a 13-year study, Korpimaki (1987*a*) observed an average clutch of 5.6 ($n = 412$) and a fledging rate for successful nests of 3.9 ($n = 445$). He observed clutches as large as 10 and fledging rates up to 8 young/nest. Nest failure averaged 23%. Solheim (1983*b*) reported an average fledging rate of 4.8 young from first nests of polygamous boreal owls in Europe and biandrous females produced up to 12 young in a year. In Germany, Konig (1969) reported fledging rates of 4 young/nest in good vole years and 2.3 in poor years. His results suggest that German boreal owls produce as many young in poor years as the average production we observed on our study area.

Adult survival in Europe also exceeds estimated adult survival in the RNRW, although estimates for both regions are based on limited data. Our most liberal assumptions lead to an annual survival estimate of 46%. Franz et al. (1984) as cited by Korpimaki (1988*b*) reported female adult survival as 78%, and Sonerud (1988:180) referred to his own unpublished data to estimate survival of 62%.

Population Growth.—Interpreting the consequences of the demographic characteristics in terms of population growth is difficult without combining reproduction, mortality, and age of first reproduction in a model to examine population trend. Therefore, we used simple Leslie matrix projection models (Leslie 1945) to examine population status during our 4-year study. These projections suggest potential patterns of population growth for the population studied but must be viewed as hypotheses rather than statements of actual trend.

To examine a range of potential scenarios, we examined 4 models based on our estimates of demographic parameters for owls at Chamberlain. Each model included 9 age classes (Korpimaki 1988*b*) with complete mortality at age 9. Using this model structure, age class 7 includes <1% of the population in all scenarios.

We first assumed all nests fledge 2.33 young/year, no nests failed, and all females breed each year raising 1 brood/year and fledge 50% female offspring. We also assumed all females breed in the spring following birth. Adult survival was set at 46% (*see* Reproduction and Survival) assuming all emigration observed during the study was complemented by equal immigration. Finally, juvenile survival was assumed equal to our estimated adult survival. This model led to a stable population (λ = 0.99). The model assumed all nest failures observed during the study were human caused and, therefore, not representative of the population (we actually believe only 1 of 10 failures was human caused).

A more realistic model assuming some nest failure and higher juvenile mortality suggested negative population growth (λ = 0.65). Juvenile survival was set at half the adult rate and nest failure at 23% as measured by Korpimaki (1988*b*) rather than the higher failure rate we actually observed. For this model all other population parameters were set as in the first example.

These 2 models suggested that lambda (finite rate of increase) approaches 1 only when demographic characteristics of the population are assumed to be more favorable to population growth than estimates from Chamberlain during our 4-year-study period. Fecundity was higher than we observed in this study. Fecundity rates estimated for the RNRW do not include radiomarked females that did not attempt to mate. Therefore, these are optimistic estimates.

Our estimates of survival have broad confidence intervals, and, therefore, we cannot place too much certainty on the mean value. We, therefore, ran 2 more models using the 95% confidence intervals on the survival estimates used in the other simulations. Fecundity for all age classes was set at the average for all nests monitored at Chamberlain.

Using the upper limit on survival (91%) and assuming juvenile survival is half the adult rate results in slight positive growth (λ = 1.02). When survival is set at 23% (the lower limit) and juvenile survival at half the adult rate, population growth is strongly negative (λ = 0.28).

Although these analyses are based on limited demographic information from only 4 years, the pattern of population growth is supported by other information. Throughout their range, boreal owls prey almost exclusively on small mammals (Klaus et al. 1975, Korpimaki 1981, Schelper 1989), and breeding populations are limited by small mammal abundance (Lofgren et al. 1986, Korpimaki 1987*a*). During our investigations, small mammal populations were low relative to productive owl sites (Lofgren et al. 1986, Korpimaki 1987*c*). Furthermore, home range areas used by boreal owls were extremely large, implying limited prey availability (Lindstedt et al. 1986) and high energy expenditure.

Local and Regional Viability.—From these simulations, we suggest that if conditions during our study were representative of long-term patterns, and if our estimates of demographic parameters are accurate, the RNRW population may not be self-sustaining. We hypothesize that during good breeding years the population experiences modest positive growth, but, during poor years, population decline is significant. Under this hypothesis, the boreal owl population we studied, which occurs in a biological reserve (wilderness area), is a sink population, relying on immigration for continued existence (rescue effect, Brown and Kodric-Brown 1977).

On a broader scale, boreal owls may experience positive growth during most years in subpopulations that are more productive. Our current studies of boreal owls 80 km west of Chamberlain demonstrate higher mean clutch size over a 3-year period (3.6 ± 0.34) (G. D. and P. H. Hay-

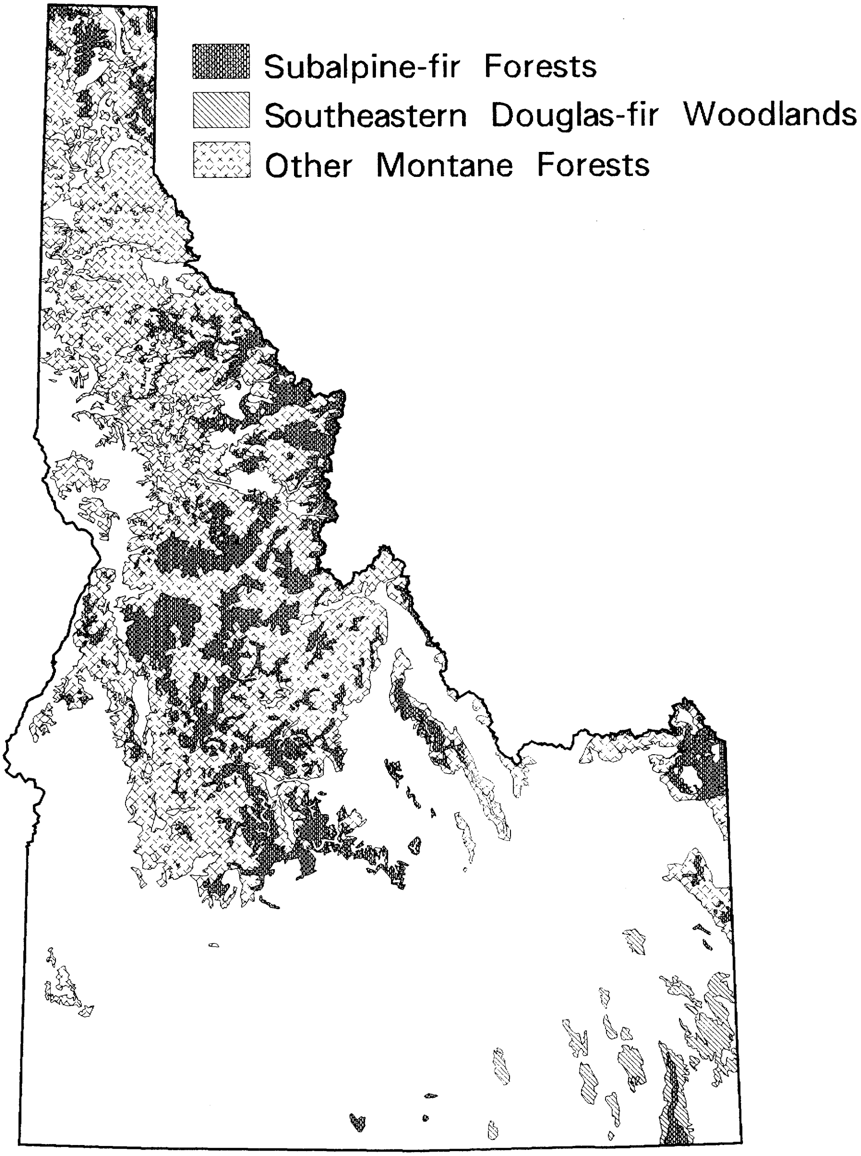

Fig. 6. Distribution of potential boreal owl habitat in Idaho based on results of regional owl surveys. Potential habitat is defined as forested sites in the subalpine-fir zone throughout the state and Douglas-fir woodland in southeastern Idaho. The map is based on data compiled by the Idaho Cooperative Fish and Wildlife Research Unit, University of Idaho, gap analysis program.

ward, unpubl. data). These more productive sites generally produce larger tree biomass and, therefore, are commercial forest lands where active forest management can be expected. Populations of boreal owls in wilderness, such as the RNRW, then, may in some situations depend on managed forest for long-term viability. Therefore, the impact of silvicultural practices in productive spruce–fir forest may influence not only the local boreal owl population but also distant populations, including those in "biological reserves." Subpopulations of boreal owls throughout the northern Rockies may be linked in a loose metapopulation (Opdam 1991) through juvenile dispersal and nomadic movements of adults.

Year-to-year movement patterns of boreal owls observed in the RNRW and in Europe would facilitate the dispersal required for the metapopulation dynamics we hypothesize. Wallin and Andersson (1981), Solheim (1983*a*), Lofgren et al. (1986), and Korpimaki (1986) have demonstrated that boreal owls in Europe exhibit both site tenacity and nomadic tendencies. In the most intensive investigation, Lofgren et al. (1986) showed that males were site tenacious throughout a prey cycle, whereas females were tenacious only during prey peaks. At Chamberlain, boreal owls showed a mixed strategy of site tenacity and nomadism (Hayward et al. 1987*b*). Clearly the owl's life history would facilitate recolonization of locally extinct habitats or the addition of individuals to sink populations through immigration. Production in some populations, however, must be sufficient to produce high net emigration rates.

Regional Population Perspectives

Our sample of owl breeding sites throughout the northern Rockies suggests that boreal owls occur primarily in high-elevation (1,400–3,100 m) coniferous forest, especially spruce–fir. These forests cover only a small portion of the landscape and occur as isolates in the expanse of forested land (Fig. 6). Therefore, even in the absence of forest fragmentation from deforestation, subpopulations of boreal owls exist as geographic isolates potentially connected through movements of adult owls or dispersal of juveniles.

Information from U.S. Forest Service (USFS) timber database suggests the limited extent of potential boreal owl habitat in the northern Rockies. Although the accuracy of this database has not been validated, it provides a first approximation from which to draw general patterns. Within USFS Region 1, ~9% of the forest land supports spruce–fir forest 15-cm dbh or larger (J. W. Laux, Timber Manage. Planner, Reg. 1, USFS, pers. commun.). On 7 forests in Idaho south of the Salmon River, spruce–fir forest covers ~7% of the forested landscape (H. A. Cheatham, Timber Manage., Reg. 4, USFS, pers. commun.). The spruce–fir forest within the region occurs in patches of varying extent separated from neighboring patches by landscapes supporting a range of vegetations (Fig. 6). The value of individual islands of spruce–fir forest to boreal owls will likely depend on the patch size, distance to nearest neighbor, vegetation (matrix) surrounding the patch, and the quality of habitat within the patch (MacArthur and Wilson 1963, Diamond 1975, Pimm et al. 1988).

How these landscape patterns interact to influence long-term viability of boreal owl populations is unknown. The limited extent of potential spruce–fir habitat suggests, however, that populations in this region may be vulnerable to reduction in suitable habitat (Stacey and Taper 1992). Loss of spruce–fir forest will reduce patch size, eliminate some habitat patches entirely, and increase the distance between suitable forest habitat. These processes have been linked to increased rates of local extinction (Connor and McCoy 1979, Burley 1989, Pimm and Gilpin 1989). Reductions in the size of forest habitat patches will likely reduce the numbers of owls in breeding populations. Because boreal owl populations inherently exist at low density (relative to passerines) and in this region may have low productivity and low survival, reduction in population size could lead to demographic instability and extinction due to stochastic events (Pimm and Gilpin 1989). Reductions in patch size also could lower the rate of immigration from neighboring populations, reduce the demographic linkage between populations, and potentially increase the rate of subpopulation turnover (Brown and Kodric-Brown 1977, Smith 1980). As pointed out by Stacey and Taper (1992), even suboptimal habitats may be important in maintaining persistence of a metapopulation. These habitats play a role in dispersal among subpopulations and as sites where floaters may exist prior to dispersal.

CONCLUSIONS

Prior to this investigation, little was known of the biology of boreal owls in North America. Several results appear most important. Those include the following:

1. Boreal owls in the northern Rocky Mountains occur in a relatively narrow life zone, breeding predominantly in forests of the spruce–fir zone at elevations from 1,400 to 3,100 m. Regional geography and the species' limited elevation distribution together result in a natural distribution pattern with numerous breeding populations isolated on patches of high-elevation forest. Metapopulation dynamics, therefore, may largely influence local and regional dynamics and have important implications for management. Forest fragmentation and removal of mature forest habitats on a regional scale may harm boreal owl populations.
2. Although we could not directly study foraging habitat, evidence suggests that mature and older spruce–fir forest is important for foraging. The importance of these forests is especially significant considering the fluctuations in prey populations experienced in the life zone occupied by boreal owls. Based on our data, however, it is difficult to separate selection for roosting and foraging habitat.
3. In the RNRW, no single forest habitat provides optimum conditions for nesting, roosting, and foraging. The owls selected habitat that differed in structure from stratified random locations for nesting and paired random sites for roosting. Forests chosen for nesting and roosting provide different resources, each meeting special needs of the owls.
4. Local geography and habitat heterogeneity strongly influenced the daily and seasonal movements of boreal owls. Habitats with abundant cavities that provided optimum nesting habitat were confined to low elevations on the eastern edge of the study area, whereas spruce–fir forest, which supported the highest prey populations and cool summer roosts, occurred at high elevations in the west. Therefore, resources used daily were not interspersed but segregated geographically.
5. Owls in the RNRW maintained large seasonal home ranges and moved long distances from day to day. The extensive movements result from several factors: the general low productivity of small mammals; dispersed distribution of habitats with abundant small mammals; and the distribution of habitats suitable for nesting, roosting, and foraging.
6. Boreal owl breeding populations and breeding success fluctuated during the study. These fluctuations may be tied to availability of small mammals, especially southern red-backed voles. The nature of fluctuations in demographic parameters has implications for assessing population viability and setting management guidelines.
7. Productivity of boreal owls in the RNRW is low compared to populations in Europe, and adult mortality may be high. Similar to populations in Europe, breeding populations and breeding success fluctuated during our study. Based on these demographic characteristics and results of Leslie projections, we suggest that this population is not self-sustaining. We hypothesize that the RNRW population may, in the long term, persist only because of linkage with a larger metapopulation.

MANAGEMENT RECOMMENDATIONS

Results of our study should not be used to make inferences to boreal owls outside the populations sampled. We measured nesting habitat at sites in Idaho and Montana, but the majority of our results come from owls in the RNRW. Boreal owls in the RNRW appear to occupy unproductive habitat, and the patterns observed may or may not apply to populations throughout the northern Rocky Mountains. In an academic sense, then, until we investigate other populations in the region and examine boreal owl ecology in managed forest, we are uncertain how the owls will respond to changing environments in the region. The need for further research is obvious and we discuss priorities below.

The manager, however, must make decisions regarding forest management despite our limited understanding of boreal owl ecology. In the comments that follow, we have not attempted to develop a conservation plan but rather present basic management ideas based on the boreal owl populations we studied.

Distribution and Abundance

Biologists should consider all forested sites in the spruce–fir zone (*Abies lasiocarpa*) as potential boreal owl habitat. Forests within 100–200 m elevation below this zone also will support breeding boreal owls and may be the most important nesting habitats. Playback surveys conducted from January to April offer the most efficient means of determining general geographic distribution patterns. A single year of negative results, however, does not constitute evidence that a site is not occupied. Absence can never be established, and the yearly variation in calling rates we observed suggest that several years of surveys will be necessary to establish presence or absence.

Any attempt to monitor trends in abundance or productivity will require development of specific monitoring methods. Playback survey methods developed for spotted owls (Forsman 1983, Thomas et al. 1990) should not be used to monitor boreal owl population trend. Spotted owls and boreal owls have different territorial systems (Mikkola 1983, Forsman et al. 1984) and, therefore, have different singing behavior. Because we do not understand what factors (aside from breeding density) influence singing rates in boreal owls, playback surveys are not a suitable intensive monitoring tool. The probability of an individual responding to playback depends on the time of night, current weather conditions, past weather conditions (that influence snowpack, plant phenology, and small mammal availability), the individual owl's physiological condition, degree of competition for nest sites from other male owls, and whether the owl has attracted a mate. Lundberg (1979) suggested that the number of boreal owls singing may be inversely related to breeding success. Despite problems in using playback surveys to monitor *abundance* of individual populations, playback surveys conducted each year, over a large region, may be useful in detecting regional trends by assessing population presence in numerous subpopulations. We are currently exploring the use of nest boxes to monitor owls on a local scale (Hayward et al. 1992). An acceptable monitoring scheme will require a valid sampling design conducted at a scale that matches the goals of the monitoring plan.

Habitat Management

Nesting Habitat.—Management of nesting habitat will largely involve snag management, management of large woodpeckers, and aspen management. Maintenance of mature aspen stands dispersed across the landscape can be important in managing boreal owl nesting habitat. Even aspen stands that cover small areas are important because of the high use of aspen by primary cavity nesters including pileated woodpeckers (G. D. and P. H. Hayward, pers. obs.). Aspen should exceed 33-cm dbh, however, to support a cavity large enough for a boreal owl. In conifer forest, potential nest snags should be >38-cm dbh and part of a forest stand. Our evidence suggests that small stands, less than a hectare, are suitable. Snags in mature or older conifer forest or aspen stands should be retained. We recommend leaving clumps of trees around large snags within clearcut units in landscapes where timber harvest is extensive. Leave-strips along stream corridors should not be viewed as the major element of a network for potential boreal owl nesting habitat; few nest sites located in our studies were in low topographic positions. Because the owls do not defend large nesting territories, potential nest stands may be under 1 km apart.

In Europe nearly 90% of some populations nest in nest boxes (Lofgren et al. 1986). This proven tool may be necessary to maintain boreal owls on some forests. Before adopting boxes as standard management tools for degraded forest landscapes, however, we must evaluate how box dispersion, density, and other factors influence boreal owls and other members of the forest community. In Europe, predation at nest boxes is not uncommon (Sonerud 1985), and our experience with nest boxes indicates American marten predation at nest boxes can be high (G. D. and P. H. Hayward, pers. obs.). In any case, because nest-box programs ignore requirements of species other than the target species, nest boxes should not be considered a desirable mitigation tool for future timber-harvest operations but may be useful in mitigating past mistakes.

Roosting and Foraging Habitat.—Roosting habitat can be maintained through management of foraging habitat. Suitable winter roosting habitat appears to be met by any sawtimber or older coniferous forest. For summer roosts the owls need cool sites found most commonly in mature and older spruce–fir forests. Summer roosting habitat must be well dispersed because the owls use large home ranges and roost throughout their home range.

Managing foraging habitat will be the most important challenge to the forest manager. Boreal owls are food limited (Lofgren et al. 1986, Korpimaki 1987*a*), and reductions in prey availability would have negative consequences for populations in marginal habitat. Silvicultural prescriptions should be designed to maintain stands with abundant small mammals and stand structure that permits owl foraging flights. Clearcuts, large meadows, and young forest stands are not quality foraging habitat.

A variety of silvicultural treatments could maintain suitable foraging habitat. In each case, several objectives should be pursued: maintenance of some forest structure to facilitate hunting; prevention of dense thicket-like timber that inhibits owl mobility; and maintenance of a microenvironment suitable for small mammal populations, especially southern red-backed voles. Slash treatment should assure that large woody debris is maintained in the system for small mammals (Fogel and Trappe 1978, Maser et al. 1978, Martell 1981).

In cases where current stand structure and silvicultural objectives dictate an even-aged system, we suggest either an irregular shelterwood or group shelterwood system. In both systems, clumps of overstory trees are left after the preparatory and seed cuts. To promote owl foraging habitat, some of the clumps should be retained after the overstory removal. During the rotation, intermediate treatments that open the regenerating stand will facilitate development of suitable hunting conditions. In lodgepole-pine forest, where a shelterwood is not feasible, small-patch clearcutting is recommended using patch sizes that approach group selection techniques.

In older spruce–fir forest, uneven-aged forest management (such as group selection) would provide for owl foraging habitat and permit timber harvest. On spruce–fir land units allocated to timber production, some form of partial cutting is desirable from the perspective of many resources, including boreal owl habitat management. Although wood-fiber production in spruce–fir forest is low relative to other coniferous forests, watershed and aesthetic values are high. Partial cutting maintains these values and protects soil productivity (Alexander 1987). A wide variety of wildlife, including American marten (Buskirk et al. 1989), fisher (Jones 1991), and great gray owls (Bull et al. 1988*b*) also would benefit directly or indirectly from maintenance of forest cover.

"Although uneven-aged cutting methods—individual tree and group selection—have seldom been used in spruce–fir forests, they appear to simulate the natural dynamics of these forests" (Alexander 1987:59). Group selection systems are easier to design and therefore may be preferable, especially in stands that are naturally patchy. All tree sizes, including some very large trees, should be represented in the postharvest stand. This will assure production of large snags for nesting, large woody debris for small mammals, and clumps with high canopy cover for roosting.

A review of 14 national forest plans from Regions 1 and 4, however, demonstrates a reluctance to initiate uneven-aged management in many spruce–fir stands. Our interpretation of 14 plans in 1989 indicated even-aged management would dominate on all but 1 forest.

An acceptable alternative to group selection, which may be easier to administer and design, is small-patch or strip clearcutting. To maintain forest structure suitable for owl foraging habitat, a portion of any harvested watershed (including areas outside the riparian area) should be managed on a long rotation. Mature stands should be well distributed through the watershed.

Landscape Scale Perspective.—Because boreal owls use large home ranges, population densities are low, and patches of suitable habitat are relatively small (Fig. 6), immigration is likely important to maintain individual populations (Stacey and Taper 1992). To facilitate movement between subpopulations, quality habitat must be well distributed throughout the species' geographic range. The area necessary to support a population is unknown but likely exceeds 1,000 km^2. Throughout the landscape, only a modest (unknown) percent of the area must remain high quality foraging habitat, but quality stands must be well dispersed. In areas managed to support boreal owls, we recommend that a portion of a watershed be maintained in mature or older forest and over half be forested with stands older than saplings.

Many individual boreal owl populations must be maintained because of the small area occupied by each and, therefore, the potential probability for local extinction. Because spruce–fir forests are naturally limited due to geography and fire history, the manager will be challenged to maintain suitable habitat to support the species in regions with high timber harvest.

Research

An outline of research priorities must recognize that prior to our investigation only 2 ecological studies (Bondrup-Nielsen 1978, Palmer 1986) addressed boreal owl habitat use in North America. As a major avian predator of small mammals in spruce–fir forests of the northern Rocky Mountains, boreal owls play an important role in these forests. Study of small mammal habitat use and population dynamics should be a high priority. A wide range of predators (American marten, fisher, coyote, and forest owls) rely on relatively few small mammal species in spruce–fir forest. But, how *Clethrionomys* and *Phenacomys* populations can be managed through silvicultural treatments is unknown. The literature on habitat use by *Phenacomys,* in particular, is limited (McAllister and Hoffmann 1988).

Foraging habitat use by boreal owls must be investigated in managed forest to understand how the species uses stands of differing structure. We suggest an experimental approach that creates landscape level treatments on managed forest lands. Treatments should include even- and uneven-aged management systems comparing foraging behavior and productivity of owls using alternate treatments.

To assess the regional status of boreal owls, basic population demography must be studied. The goal should be a database sufficient to build demographic models of owl populations from several sites with varying productivity and survival. To build a reliable metapopulation model, information must be obtained on the extent of adult and juvenile movements. The ultimate goal must be a spatially-explicit regional metapopulation model. We are currently working on such an approach.

Finally, alternative methods to monitor populations and habitat must be explored. It may be impractical to monitor population trend for a species that naturally exists at low densities in inaccessible habitat. Therefore, methods to efficiently monitor presence or absence and some measures of habitat quality may be most desirable.

LITERATURE CITED

ALATALO, R. V., A. LUNDBERG, AND S. ULFSTRAND. 1985. Habitat selection in the pied flycatcher *Ficedula hypoleuca*. Pages 59–85 *in* M. L. Cody, ed. Habitat selection in birds. Academic Press, New York, N.Y.

ALEXANDER, R. R. 1987. Ecology, silviculture, and management of the Engelmann spruce–subalpine fir type in the central and southern Rocky Mountains. U.S. For. Serv., Agric. Handb. 659, Washington, D.C. 144pp.

ALLDREDGE, J. R., AND J. T. RATTI. 1986. Comparison of some statistical techniques for analysis of resource selection. J. Wildl. Manage. 50:157–165.

ARNO, S. F. 1979. Forest regions of Montana. U.S. For. Serv. Res. Pap. INT-218. 39pp.

AVERY, T. E. 1975. Natural resource measurements. McGraw Hill Book Co., Inc., New York, N.Y. 339pp.

BAUER, H. L. 1943. The statistical analysis of chaparral and other plant communities by means of transect samples. Ecology 24:45–60.

BONDRUP-NIELSEN, S. 1978. Vocalizations, nesting, and habitat preferences of the boreal owl (*Aegolius funereus*) in North America. M.S. Thesis, Univ. Toronto, Ont., Can. 151pp.

BONHAM, C. D. 1989. Measurements for terrestrial vegetation. John Wiley & Sons, Inc., New York, N.Y. 338pp.

BORMAN, F. H., AND M. F. BUELL. 1964. Old-age stand of hemlock–northern hardwood forest in central Vermont. Bull. Torrey Bot. Club 91:451–465.

BROWN, J. H., AND A. KODRIC-BROWN. 1977. Turnover rates in insular biogeography: effects of immigration on extinction. Ecology 58:445–449.

BULL, E. L. 1987. Capture techniques for owls. Pages 291–293 *in* R. W. Nero, C. R. Knapton, and R. J. Hamre, eds. Biology and conservation of northern forest owls: symposium proceedings. U.S. For. Serv. Gen. Tech. Rep. RM-142. 309pp.

———, M. G. HENJUM, AND R. S. ROHWEDER. 1988*a*. Home range and dispersal of great gray owls in northeastern Oregon. J. Raptor Res. 22:101–106.

———, ———, AND ———. 1988*b*. Nesting and foraging habitat of great gray owls. J. Raptor Res. 22:107–115.

BURLEY, T. V. 1989. Extinction in nature reserves: the effect of fragmentation and the importance of migration between reserve fragments. Oikos 55:75–81.

BUSKIRK, S. W., S. C. FORREST, M. G. RAPHAEL, AND H. J. HARLOW. 1989. Winter resting site ecology of marten in the central Rocky Mountains. J. Wildl. Manage. 53:191–196.

CARLSSON, B. G., B. HORNFELDT, AND O. LOFGREN. 1987. Bigyny in Tengmalm's owl *Aegolius funereus*: effect of mating strategy on breeding success. Ornis Scand. 18:237–243.

CASWELL, H. 1989. Matrix population models: construction, analysis, and interpretation. Sinauer Assoc., Inc., Sunderland, Mass. 328pp.

CATLING, P. M. 1972. A study of the boreal owl in southern Ontario with particular reference to the irruption of 1968–69. Can. Field-Nat. 86:223–231.

CONNOR, E. F., AND E. D. MCCOY. 1979. The statistics and biology of the species–area relationship. Am. Nat. 113:791–833.

CRAMP, S. 1977. Handbook of the birds of Europe, the Middle East, and North Africa: the birds of the western Palearctic. Vol. 4. Oxford Univ. Press, Oxford, U.K. 960pp.

DANIEL, T. W., J. A. HELMS, AND F. S. BAKER. 1979. Principles of silviculture. McGraw-Hill Book Co., Inc., New York, N.Y. 500pp.

DAUBENMIRE, R. F. 1959. A canopy-coverage method of vegetation analysis. Northwest Sci. 33:43–46.

DEJAIFVE, P. A., C. NOVOA, AND R. PRODON. 1990. Habitat et densité de la Chouette de Tengmalm *Aegolius funereus* a l'extrémité orientale des Pyrénées. Alauda 58:267–273.

DIAMOND, J. M. 1975. Assembly of species communities. Pages 342–444 *in* M. L. Cody and J. M. Diamond, eds. Ecology and evolution of communities. Harvard Univ. Press, Cambridge, Mass.

DIGBY, P. G., AND R. A. KEMPTON. 1987. Multivariate analysis of ecological communities. Chapman and Hall, New York, N.Y. 206pp.

EVANS, D. L., AND R. N. ROSENFIELD. 1977. Fall migration of boreal owls. Loon 49:165–167.

FINKLIN, A. I. 1988. Climate of the Frank Church, River of No Return Wilderness, Central Idaho. U.S. For. Serv. Gen. Tech. Rep. INT-240. 221pp.

FOGEL, R., AND J. M. TRAPPE. 1978. Fungus consumption (*Mycophagy*) by small animals. Northwest Sci. 52:1–31.

FORSMAN, E. D. 1983. Methods and materials for locating and studying spotted owls. U.S. For. Serv. Gen. Tech. Rep. PNW-162. 8pp.

———, E. C. MESLOW, AND H. M. WIGHT. 1984. Distribution and biology of the spotted owl in Oregon. Wildl. Monogr. 87. 64pp.

FRANZ, A., T. MEBS, AND E. SEIBT. 1984. Zur populations biologic des Rauhfusskauzes (*Aegolius funereus*) in sudliehen Westfalen und in angrenzenden Gebieten anhand von Beringungsergebnissen. Vogelwarte 32:260–269.

FRETWELL, F. D., AND H. L. LUCAS. 1969. On territorial behavior and other factors influencing habitat distribution in birds. I. Theoretical development. Acta Biotheor. 19:16–36.

HALL, E. R. 1946. Mammals of Nevada. Univ. California Press, Los Angeles. 448pp.

HAYWARD, G. D. 1987. Betalights: an aid in the nocturnal study of owl foraging habitat and behavior. J. Raptor Res. 21:98–102.

———. 1989. Habitat use and population biology of boreal owls in the northern Rocky Mountains, USA. Ph.D. Thesis, Univ. Idaho, Moscow. 113pp.

———. 1991. Using population biology to define oldgrowth. Wildl. Soc. Bull. 19:111–116.

———, AND E. O. GARTON. 1983. First nesting record for boreal owl in Idaho. Condor 85:501.

———, AND ———. 1988. Resource partitioning among forest owls in the River of No Return Wilderness, Idaho. Oecologia (Berlin) 75:253–265.

———, AND P. H. HAYWARD. 1991. Body measurements of boreal owls in Idaho and a discriminant model to determine sex of live specimens. Wilson Bull. 103:497–500.

———, ———, AND E. O. GARTON. 1987*a*. Revised breeding distribution of the boreal owl in the northern Rocky Mountains. Condor 89:431–432.

———, ———, AND ———. 1987*b*. Movements and home range use by boreal owls in central Idaho. Pages 175–184 *in* R. W. Nero, C. R. Knapton, and R. J. Hamre, eds. Biology and conservation of northern forest owls: symposium proceedings. U.S. For. Serv. Gen. Tech. Rep. RM-142.

———, R. K. STEINHORST, AND P. H. HAYWARD. 1992. Monitoring boreal owl populations with nest boxes: sample size and cost. J. Wildl. Manage. 56:777–785.

HEISEY, D. M., AND T. K. FULLER. 1985. Evaluation of survival and cause-specific mortality rates using telemetry data. J. Wildl. Manage. 49:668–674.

HIRONS, G. J. M. 1985*a*. The effects of territorial behavior on the stability and dispersion of tawny owl (*Strix aluco*) populations. J. Zool. Lond. (B) 1:21–48.

———. 1985*b*. The importance of body reserves for successful reproduction in the tawny owl (*Strix aluco*). J. Zool. Lond. (B) 1:1–20.

HURLBERT, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Monogr. 54:187–211.

JAMES, F. C. 1971. Ordinations of habitat relationships among breeding birds. Wilson Bull. 83:215–236.

JANES, S. W., AND J. M. BARSS. 1985. Predation by three owl species on northern pocket gophers of different body mass. Oecologia (Berlin) 67:76–81.

JOHNSON, R. A., AND D. W. WICHERN. 1982. Applied multivariate statistical analysis. Prentice-Hall, Inc., Engelwood Cliffs, N.J. 594pp.

JONENIAUX, A., AND G. DURAND. 1987. Forest management and population ecology of Tengmalm's owl. Rev. Ecol. Terre et la Vie, Suppl. 4:84–87.

JONES, J. 1991. Habitat use by fisher (*Martes pennati*) in north-central Idaho. M.S. Thesis, Univ. Idaho, Moscow. 132pp.

JORLITSCHKA, W. 1988. Untersuchung zur Habitatstruktur von Revieren des Rauhfußkauzes *Aegolius funereus* im Nordschwarzwald. Die Vogelwelt 109:152–155.

KLAUS, V. S., H. MIKKOLA, AND J. WIESNER. 1975. Aktivitat und Ernahrung des Rauhfußkauzes *Aegolius funereus* (L.) wahrend der Fortpflanzungsperiode. Zool. Jb. Syst. 102:485–507.

KONIG, C. 1969. Six year observations of a population of Tengmalm's owl *Aegolius funereus*. J. Ornithol. 110:133–147.

KORPIMAKI, E. 1981. On the ecology and biology of Tengmalm's owl (*Aegolius funereus*) in southern Ostrobothnia and Suomenselka, western Finland. Acta Univ. Ouluensis Ser. A. Sci. Rerum Nat. 118 Biol. 13. 84pp.

———. 1984. Population dynamics of birds of prey in relation to fluctuations in small mammal populations in western Finland. Ann. Zool. Fenn. 21: 287–293.

———. 1986. Gradients in population fluctuations of Tengmalm's owl *Aegolius funereus* in Europe. Oecologia (Berlin) 69:195–201.

———. 1987*a*. Clutch size, breeding success and brood size experiments in Tengmalm's owl *Aegolius funereus*: a test of hypothesis. Ornis Scand. 18:277–284.

———. 1987*b*. Breeding performance of Tengmalm's owl *Aegolius funereus*: effects of supplementary feeding in peak vole year. Ibis 131: 51–56.

———. 1987*c*. Selection for nest hole shift and tactics of breeding dispersal in Tengmalm's owl *Aegolius funereus*. J. Anim. Ecol. 56:185–196.

———. 1988*a*. Effects of territory quality on occupancy, breeding performance and breeding dispersal in Tengmalm's owl. J. Anim. Ecol. 57: 97–108.

———. 1988*b*. Effects of age on breeding performance of Tengmalm's owl *Aegolius funereus* in western Finland. Ornis Scand. 19:21–26.

———, AND K. NORRDAHL. 1991. Numerical and functional responses of kestrels, short-eared owls, and long-eared owls to vole densities. Ecology 72:814–826.

LAUNDRE, J. W., T. D. REYNOLDS, S. T. KNICK, AND I. V. BALL. 1987. Accuracy of daily point-relocations in assessing real movement of radio-marked animals. J. Wildl. Manage. 51:937–940.

LESLIE, P. H. 1945. On the use of matrices in certain population mathematics. Biometrika 33:183–212.

LINDSEY, A. A. 1956. Sampling methods and community attributes in forest ecology. For. Sci. 2:287–296.

LINDSTEDT, S. L., B. J. MILLER, AND S. W. BUSKIRK. 1986. Home range, time and body size in mammals. Ecology 67:413–418.

LOFGREN, O., B. HORNFELDT, AND B. G. CARLSSON. 1986. Site tenacity and nomadism in Tengmalm's owl (*Aegolius funereus* (L.)) in relation to cyclic food production. Oecologia (Berlin) 69: 321–326.

LUNDBERG, A. 1979. Residency, migration and a compromise: adaptations to nest site scarcity and food specialization in three Fennoscandian owl species. Oecologia (Berlin) 41:273–281.

MACARTHUR, R. H., AND E. O. WILSON. 1963. An equilibrium theory of insular zoogeography. Evolution 17:373–387.

MARTELL, A. M. 1981. Food habits of southern red-backed voles (*Clethrionomys gapperi*) in northern Ontario. Can. Field-Nat. 95:325–328.

MASER, C., J. M. TRAPPE, AND R. A. NUSSBAUM. 1978. Fungal–small mammal interrelationships with emphasis on Oregon coniferous forests. Ecology 59:799–809.

MCALLISTER, J. A., AND R. S. HOFFMANN. 1988. *Phenacomys intermedius*. Mammalian Species No. 305, Am. Soc. of Mammal. 8pp.

MCCLELLAND, B. R. 1977. Relationships between hole-nesting birds, forest snags, and decay in western larch/Douglas-fir forests of the northern Rocky Mountains. Ph.D. Thesis, Univ. Montana, Missoula. 395pp.

MENDENHALL, W., L. OTT, AND R. L. SCHEAFFER. 1971. Elementary survey sampling. Duxbury Press, Belmont, Calif. 245pp.

MIKKOLA, H. 1983. Owls of Europe. Buteo Books, Vermillion, S.D. 397pp.

MUELLER-DOMBOIS, D., AND H. ELLENBERG. 1974. Aims and methods of vegetation ecology. John Wiley & Sons, Inc., New York, N.Y. 547pp.

MYSTERUD, I. 1970. Hypothesis concerning characteristics and causes of population movements in Tengmalm's owl (*Aegolius funereus* (L.)). Nytt Mag. Zool. (Oslo) 18:49–74.

NORBERG, R. A. 1964. Studies on the ecology and etology of Tengmalm's owl (*Aegolius funereus*) Var Fagelvarld 23:228–244.

OPDAM, P. 1991. Metapopulation theory and habitat fragmentation: a review of holarctic breeding bird studies. Landscape Ecol. 5:93–106.

PALMER, D. A. 1986. Habitat selection, movements, and activity of boreal and saw-whet owls. M.S. Thesis, Colorado State Univ., Fort Collins. 101pp.

———, AND R. A. RYDER. 1984. The first documented breeding of boreal owl in Colorado. Condor 86:215–217.

PIELOU, E. C. 1984. The interpretation of ecological data: a primer on classification and ordination. John Wiley & Sons, Inc., New York, N.Y. 263pp.

PIMM, S. L., AND M. E. GILPIN. 1989. Theoretical issues in conservation biology. Pages 287–305 *in* J. Roughgarden, R. M. May, and S. A. Levin, eds. Perspectives in ecological theory. Princeton Univ. Press, Princeton, N.J. 394pp.

———, H. L. JONES, AND J. DIAMOND. 1988. On the risk of extinction. Am. Nat. 132:757–785.

SAMUEL, M. D., D. J. PIERCE, E. O. GARTON, L. J. NELSON, AND K. R. DIXON. 1985. User's manual for program HOME RANGE. Univ. Idaho, For., Wildl., and Range Exp. Stn. Tech. Rep. 15. 70pp.

SCHELPER, V. W. 1989. Zur Brutbiologie, Ernahrung und Populationsdynamik des Rauhfußkauzes *Aegolius funereus* im Kaufunger Wald (Sudniedersachsen) Vogelkd. Ber. Niedersachsen. 21:33–53.

SCHOENER, T. W. 1981. An empirically based estimate of home range. Theor. Popul. Biol. 20: 281–325.

SCOTT, J. M., B. C. CSUTI, J. D. JACOBI, AND J. E. EVENS. 1987. Species richness: a geographical approach to protecting future biological diversity. BioScience 37:782–788.

SILVERMAN, B. W. 1986. Density estimation for statistics and data analysis. Chapman and Hall, London, U.K. 323pp.

SMITH, A. T. 1980. Temporal changes in insular populations of the pika (*Ochotona princeps*). Ecology 61:8–13.

SMITH, D. G., AND R. GILBERT. 1984. Eastern screech owl home ranges and use of suburban habits in southern Connecticut. J. Field Ornithol. 55:322–329.

SOLHEIM, R. 1983*a*. Breeding frequency of Tengmalm's owl *Aegolius funereus* in three localities in 1974–78. Proc. Nord. Congr. Ornithol., Kobenhaven, Norway 3:79–84.

———. 1983*b*. Bigyny and biandry in the Tengmalm's owl *Aegolius funereus*. Ornis Scand. 14: 51–57.

SONERUD, G. A. 1985. Nest hole shift in Tengmalm's owl *Aegolius funereus* as defense against nest predation involving long term memory in the predator. J. Anim. Ecol. 54:179–192.

———. 1986. Effect of snow cover on seasonal changes in diet, habitat, and regional distribution of raptors that prey on small mammals in boreal zones of Fennoscandia. Holarctic Ecol. 9:33–47.

———. 1988. Dispersal of Tengmalm's owl *Aegolius funereus* in relation to prey availability and nesting success. Ornis Scand. 19:175–181.

———, R. SOLHEIM, AND B. V. JACOBSEN. 1986. Home-range use and habitat selection during hunting in a male Tengmalm's owl *Aegolius funereus*. Fauna Norvegica, Ser. C. Cinclus 9:100–106.

STACEY, P. B., AND M. TAPER. 1992. Environmental variation and the persistence of small populations. Ecol. Appl. 2:18–29.

STAUFFER, D. F., AND S. R. PETERSON. 1985. Seasonal habitat use by ruffed and blue grouse in southeastern Idaho. J. Wildl. Manage. 49:459–466.

STEELE, R., R. D. PFISTER, R. A. RYKER, AND J. A. KITTAMS. 1981. Forest habitat types of central Idaho. U.S. For. Serv. Gen. Tech. Rep. INT-114. 138pp.

STRICKLER, G. S. 1959. Use of the densiometer to estimate density of forest canopy on permanent sample plots. U.S. For. Serv. Res. Note PNW-180. 5pp.

THOMAS, J. W., E. D. FORSMAN, J. B. LINT, E. C. MESLOW, B. R. NOON, AND J. VERNER. 1990. A conservation strategy for the northern spotted owl. Rep. of the Interagency Sci. Comm. to address the conservation of the northern spotted owl, Portland, Oreg. U.S. Gov. Print. Off., Washington, D.C. 427pp.

VOOUS, K. H. 1988. Owls of the northern hemisphere. Massachusetts Institute of Technology Press, Cambridge. 320pp.

WALLIN, K., AND M. ANDERSSON. 1981. Adult nomadism in Tengmalm's owl *Aegolius funereus*. Ornis Scand. 12:125–126.

WHELTON, B. D. 1989. Distribution of the boreal owl in eastern Washington and Oregon. Condor 91:712–716.

WHITE, G. C., D. R. ANDERSON, K. P. BURNHAM, AND D. L. OTIS. 1982. Capture–recapture and removal methods for sampling closed populations. U.S. Dep. Energy, Los Alamos Natl. Lab., N.M. LA-8787-NERP. UC-11. 235pp.

WILLIAMS, D. F., AND S. E. BRAUN. 1983. Comparison of pitfall and conventional traps for sampling small mammal populations. J. Wildl. Manage. 47:841–845.

WORTON, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. Ecology 70:164–168.

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APPENDIX

Vegetation characteristics recorded at boreal owl nest sites, potential breeding sites, and random sites in the Frank Church River of No Return Wilderness to evaluate habitat at breeding locations.

Variable	Measurement
% shrub cover 0–1 m high	Mean of 8 line-intercept transects
% shrub cover 1.1–2 m high	Mean of 8 line-intercept transects
% deciduous cover 2.1–4 m high	Mean of 8 line-intercept transects
% deciduous cover 4.1–8 m high	Mean of 8 line-intercept transects
% deciduous cover >8 m high	Mean of 8 line-intercept transects
% conifer cover 0–1 m high	Mean of 8 line-intercept transects
% conifer cover 1.1–2 m high	Mean of 8 line-intercept transects
% conifer cover 2.1–4 m high	Mean of 8 line-intercept transects
% conifer cover 4.1–8 m high	Mean of 8 line-intercept transects
% conifer cover >8 m high	Mean of 8 line-intercept transects
% cover grass	Mean of 16 0.1-m^2-rectangular plots
% cover forb	Mean of 16 0.1-m^2-rectangular plots
% cover subshrub	Mean of 16 0.1-m^2-rectangular plots
Density trees 2.5–7.6-cm dbh	Mean of 16 83-m^2-circular plots
Density trees 7.7–15-cm dbh	Mean of 16 83-m^2-circular plots
Density trees 15.1–23-cm dbh	Mean of 16 83-m^2-circular plots
Density trees 23.1–38-cm dbh	Mean of 16 83-m^2-circular plots
Density trees 38.1–53-cm dbh	Mean of 16 83-m^2-circular plots
Density trees >53-cm dbh	Mean of 16 83-m^2-circular plots
Density trees >30.5-cm dbh	Mean of 4 variable-radius plots
Basal area of trees >15-cm dbh	Mean of 4 variable-radius plots

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Faunal indicator taxa selection for monitoring ecosystem health

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Abstract

Maintaining healthy ecosystems is a prerequisite for conserving biodiversity. The complex nature of ecosystems often necessitates the use of indicator taxa to monitor ecosystem health. However, ambiguous selection criteria and the use of inappropriate taxa have brought the utility of indicator taxa under question. This review compiles existing selection criteria from the literature, evaluates inconsistencies among these criteria, and proposes a step-wise selection process. In addition, 100 vertebrate and 32 invertebrate taxa documented in the conservation science literature as indicators of ecosystem health are examined to assess how well they adhere to the referenced criteria. Few vertebrate taxa fulfill multiple criteria, as most are highly mobile generalists that lack established tolerance levels and correlations with ecosystem changes. Most suggested invertebrate taxa also lack correlations to ecosystem changes, but satisfy other selection criteria. However, invertebrate taxa are often suggested at high taxonomic levels, encompassing many species, making it difficult to measure specific attributes, and potentially including many unnecessary and even inappropriate species. © 2000 Elsevier Science Ltd. All rights reserved.

Keywords: Conservation; Guidelines; Monitoring; Vertebrates; Invertebrates

1. Introduction

The complexity of ecosystems has forced conservation biologists to develop alternative methods to monitor change that would be too costly or difficult to measure directly (Landres et al., 1988; Meffe and Carroll, 1997). One such method is the use of indicator taxa, which are species or higher taxonomic groups whose parameters, such as density, presence or absence, or infant survivorship, are used as proxy measures of ecosystem conditions. For example, indicator taxa have been used to evaluate toxicity levels, abundance of specific resources, levels of biodiversity, target taxa status, endemism levels, and ecosystem health (Temple and Wines, 1989; Wilcove, 1989; Croonquist and Brooks, 1991; van Franeker, 1992; Kremen et al., 1993; Kushlan, 1993; Maho et al., 1993; Bortone and Davis, 1994; Anderson-Carnahan et al. 1995; Louette et al., 1995; Cherel and Weimerskirch, 1995; Harris, 1995; Nyholm, 1995; Faith and Walker, 1996).

This paper focuses on utilization of taxa as appropriate indicators for assessing general ecosystem health. This use of indicator taxa is important to biological conservation yet lacks well-established methodologies. The goal of monitoring ecosystem health is to identify chemical, physical and/or biological changes due to human impacts (Hughes et al., 1992). The term ecosystem health has been hotly debated in the literature (Jamieson, 1995; Lackey, 1995; Rapport, 1995a; Wicklum and Davies, 1995; Callicott and Mumford, 1997; Simberloff, 1998). While some condone complete abdication of the term, ecosystem health remains a widely used concept and many papers reviewed here used the term. We prefer Rapport's (1995a,b) definition which states ecosystem health as the absence of signs of ecosystem distress, an ecosystem's ability to recover with speed and completeness (resilience), and/or a lack of risks or threats pressuring the ecosystem composition, structure, and/or function. Kolasa and Pickett (1992) also suggest that measures of ecosystem health should be based on a pre-defined scale using a baseline condition.

Multiple monitoring methods are usually required to address complex ecosystems (Soule, 1985). The index of biotic integrity (Karr, 1981), developed to assess degradation in streams, uses an array of ecological measures,

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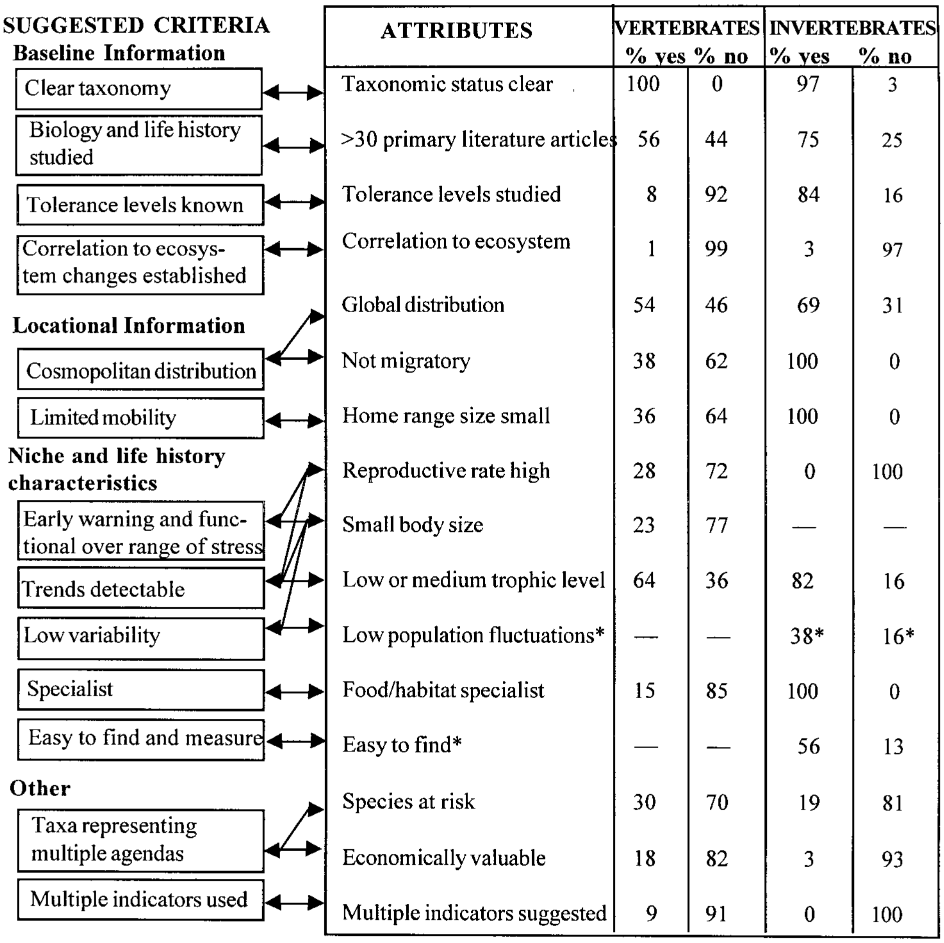
one of which is indicator taxa (Fausch et al., 1990). If selected correctly a set of complementary indicator taxa may provide early warning of problems and help monitor change over large temporal and spatial scales (Rapport, 1992; Harris 1995).

While use of indicator taxa has become integrated into many ecosystem health monitoring programs, the appropriateness of using indicator taxa has been questioned. Simberloff (1998), for example, proposes focusing on keystone species rather than indicator taxa because of concern about the true utility of indicator taxa. Unclear guidelines, dubious assumptions about the ability of indicator taxa to represent other ecosystem trends, and difficulties in differentiating human impact and non-human related changes have led indicator taxa to be the subject of much debate (Rosenberg et al., 1986; Landres et al., 1988; Fausch et al., 1990; Pearman et al, 1995; Taper et al., 1995; Simberloff, 1998).

The objectives of this review are: (1) to evaluate various criteria that have been proposed by others in the conservation science literature for selecting indicator taxa, (2) to offer a step-wise process for indicator taxa selection, and (3) to test the criteria against the indicator taxa that biologists and natural resource managers are currently using to monitor ecosystem health.

1.1. Review of the suggested criteria for selecting indicator taxa

We found nine articles published in the last 13 years that outline criteria for selecting indicator taxa (Soule, 1985; Hellawell, 1986; Landres et al., 1988; Kelly and Harwell, 1990; Noss, 1990; Regier, 1990; Pearson and Cassola, 1992; Johnson et al., 1993; Kremen, 1994). The indicator selection criteria reviewed applied to conservation-oriented efforts but not necessarily to indicators of ecosystem health. However, significant overlap in criteria of reviewed papers suggests that indicator taxa for ecological monitoring share the same basic requirements. We focus on 13 selection criteria mentioned in more than one reference. These criteria fall into four general categories (Fig. 1): (1) baseline information, (2)



ATTRIBUTES	VERTEBRATES % yes	VERTEBRATES % no	INVERTEBRATES % yes	INVERTEBRATES % no
Taxonomic status clear	100	0	97	3
>30 primary literature articles	56	44	75	25
Tolerance levels studied	8	92	84	16
Correlation to ecosystem	1	99	3	97
Global distribution	54	46	69	31
Not migratory	38	62	100	0
Home range size small	36	64	100	0
Reproductive rate high	28	72	0	100
Small body size	23	77	—	—
Low or medium trophic level	64	36	82	16
Low population fluctuations*	—	—	38*	16*
Food/habitat specialist	15	85	100	0
Easy to find*	—	—	56	13
Species at risk	30	70	19	81
Economically valuable	18	82	3	93
Multiple indicators suggested	9	91	0	100

* Only some authors categorized suggested taxa by Population Fluctuation or Easy to Find.

Fig. 1. Suggested criteria for selecting indicator taxa of ecosystem health and attributes used to assess if taxa fulfill the criteria are in columns 1 and 2, respectively. Columns 3 and 4 include summary results of measured attributes for 100 suggested vertebrate and 32 suggested invertebrate taxa. Where percents do not add up to 100, not all taxa were categorized.

location information, (3) niche and life history attributes, and (4) other. Except where discussed, the reviewed criteria apply to both single species and higher level taxon. Not all useful indicator taxa will necessarily fit all criteria, but each taxon, selected as part of a complementary set, should satisfy multiple criteria.

Adequate baseline information means the biology, taxonomy, and tolerance of a taxon's measurable characteristics should be understood (Hellawell, 1986; Landres et al., 1988; Kelly and Harwell, 1990; Regier, 1990; Pearson and Cassola, 1992; Johnson et al., 1993; Kremen, 1994). Studies asserting cause and effect assure that observed changes in the characteristics measured for the indicator taxa are induced by human actions (Landres et al., 1988). In addition, clear correlation between the taxon's response to impacts and ecosystem changes should be established (Landres et al., 1988; Kelly and Harwell, 1990; Pearson and Cassola, 1992; Kremen 1994).

Location information is also important in selecting indicator taxa. Reviewed literature indicate that selected taxa should have a cosmopolitan distribution to assist in cross-comparisons of sites (Hellawell, 1986; Noss, 1990; Regier, 1990; Pearson and Cassola, 1992; Johnson et al., 1993). An indicator taxon should also have limited mobility, so that the taxon is less likely to be able to avoid disturbances (Landres et al., 1988; Johnson et al., 1993). For example, changes in a population of migratory birds could be due to impacts in any part of their migratory route, not just at the study site.

Specified niche and life history characteristics should also be considered for each indicator taxon. A taxon should have low variability both genetically and ecologically, so neither random fluctuations in populations nor species adaptations hinder detection of impacts (Hellawell, 1986; Landres et al., 1988; Noss, 1990; Johnson et al., 1993). In addition, the life history of the selected indicator taxa should be such that it will both be able to provide early warning and be effective over a wide range of stress (Soule, 1985; Kelly and Harwell, 1990; Noss, 1990). Finally, the life history characteristic specialized/endemic is a debated criterion in the literature review (Landres et al., 1988; Pearson and Cassola, 1992; Johnson et al., 1993; Kremen, 1994).

Other considerations proposed for selecting indicator taxa include cost effectiveness, ease of detection and mensuration, and ability to detect and quantify changes (Hellawell, 1986; Kelly and Harwell, 1990; Noss, 1990; Regier, 1990; Pearson and Cassola, 1992; Johnson et al., 1993; Kremen, 1994). Inadequate sample size, lack of statistical power, or inability to detect the difference between environmental variation and changes induced by human impacts can prevent effective hypothesis testing. Another proposed criterion is the selection of taxa that serve other agendas, such as social, political or economic priorities. Using species at risk, flagship species, or otherwise prioritized species also as an indicator taxon is controversial and debated in the literature (Soule, 1985; Hellawell, 1986; Landres et al., 1988; Kelly and Harwell, 1990; Pearson and Cassola, 1992). Finally, use of a set of complementary indicator taxa where each selected taxon can satisfy multiply criteria is critical (Soule, 1985; Landres et al., 1988; Kelly and Harwell, 1990; Noss, 1990).

1.2. Critique of suggested criteria

The criteria outlined above suffer from several problems. Some of the criteria are unclear, conflict with one another, or are disputed among authors. Also, these criteria have never been prioritized in order of importance, and most of the criteria are difficult to determine for most taxa because there are few measurable attributes associated with the suggested criteria.

The criterion, cosmopolitan, conflicts with the recommendation for endemic and specialized taxa because no one taxon can be both endemic and cosmopolitan at the same taxonomic level. While one possible solution is to select higher ranking taxa; thereby assessing their distribution at the taxonomic level suggested and evaluating their level of endemism at the species level, use of higher taxonomic levels is not necessarily appropriate. Instead, use of low ranking taxa is important to minimize the possibility of including inappropriate species. Selection of high taxonomic level taxa is also problematic because species within a taxon can vary dramatically across sites making comparisons difficult, and a taxon may only be an appropriate indicator in part of its range (Landres et al., 1988). Given that cosmopolitan can conflict with the criterion of specialization and endemic, and that cross-site comparisons are potentially limited even with cosmopolitan distribution of a taxon, we recommend that cosmopolitan be considered a secondary criterion to be considered after the other criteria have been satisfied (Table 1).

Two other criteria, early warning and low variability, are both important but can also be conflicting. For example, large bodied, high trophic level, generalist vertebrates with low reproductive rates are indicative of taxa that have low population variability, but these same characteristics imply limited ability to provide early warning of impacts. We suggest that indicators should be selected in such a way that early warning detection is maximized while minimizing unpredictable fluctuations in populations (Table 1). Some invertebrates, such as Collembola and Odonata, satisfy both the early warning and low variability criteria (Brown, 1991).

The criterion, specialization, is disputed in the literature. While specialists are argued to be more information rich (Pearson and Cassola, 1992), there is concern that they may not adequately represent ecosystem complexity (Landres et al., 1988). We believe that specialization

Table 1
Step-wise decision-making framework for selecting indicator taxa[a]

Step 0	Decide what ecosystem attribute(s) indicator taxa should reflect.
Step 1	Make a list of all species in the area that best satisfy the baseline information criteria.
Step 2	From this initial list, retain species that best meet the suggested niche and life history criteria.
Step 3	Remove species that may respond to changes occurring outside the system of interest.
Step 4	Use only those species that can be easily detected and monitored with available funds.
Optional step	Reduce the list further by selecting taxa in the list with cosmopolitan distributions and/or that represent other agendas of interest.
Step 5	Select a set of complementary indicator taxa from different taxonomic groups so that all selection criteria are met by more than one taxon.

[a] See Fig. 1, column 1 for a detailed list of criteria.

should be considered an important criterion because generalists, like more mobile species, can potentially avoid impacts by switching food sources or altering their habitat use, thereby failing to respond to the impact. For example, coyotes (*Canis latrans*) rely on different prey items depending on land use patterns, allowing them to persist in areas of human development where eradication programs no longer exist (Quinn, 1997). A set of complementary specialist indicator taxa representing a spectrum of ecosystem niches would potentially allow for detection of fine scale impacts and earlier detection, while use of multiple generalists is unlikely to lead to enhanced fine scale or early detection.

The importance of selecting indicator taxa that also fulfill political, economic or social agendas is also a criterion of contention. Simultaneously servicing other agendas may increase support for the project (Pearson and Cassola, 1992), but attempts to address more than one agenda with the same taxon too often pre-empts the selection of more appropriate indicators that would satisfy the recommended criteria (Landres et al., 1988). In addition, use of taxa servicing other agendas opens the door to a host of other problems. Economically important taxa, such as harvested or hunted species, are affected by off-take that can confound data collected for detecting changes in ecosystem health (Merenlender et al., 1998).

Similarly, use of taxa identified at risk of extinction as indicators of ecosystem health can be problematic. First, species at risk are often difficult to study because the precarious status of the taxon prohibits further disruption, impeding necessary studies to establish baseline information. Second, endangered populations are often at low densities or have restricted distributions, which result in reduced sampling size and statistical power for trend analyses. Third, taxa at risk may not function as an indicator of ecosystem change over a range of human impacts. Given the potential pitfalls, selection of indicator taxa that represent other agendas should be secondary to the other criteria. To improve the indicator taxon selection process and prioritization of criteria, we provide a step-wise process to select the best available indicator taxa in Table 1.

Another potential problem with the compiled criteria is that the concepts are not easily translatable to land managers. In other words, many of the criteria are conceptual and often lack easily measurable attributes, so it is difficult for practitioners to make unbiased evaluations of potential indicator taxa. We identified attributes for which information is relatively easy to gather to determine if a particular taxon meets each suggested selection criteria. A list of these identified attributes are described in the methods section and are cross referenced to the published suggested selection criteria in Fig. 1. The cost of using a particular taxon is not included because cost is dependent on the location, circumstances, and resources available. Many of the attributes collected are not independent variables, but instead serve together to provide evidence as to whether the taxa satisfy suggested selection criteria. We used these attributes to examine how indicator taxa found in the literature satisfied the suggested selection criteria.

2. Methods

2.1. Literature search and restrictions

A literature search was conducted focusing on the primary biological literature published in English in the past 10 years that explicitly identified taxa as indicators of ecosystem health. The phrases bio-indicators and management indicator species were accepted only when authors explicitly stated that a taxon was used for assessment, management, or monitoring of ecosystem health. Terms considered equivalent to ecosystem health when discussed in the context of monitoring human impacts include: water quality, disturbance, impacts, landuse change impacts, health of ecosystem, health condition, ecosystem deterioration, forest health, ecosystem, state of ecosystem, ecosystem degradation, intact ecosystem, and the effects of land management. Only indicator taxa suggested for or used in the field were analyzed, and studies referring solely to a specific impact of ecosystem health such as of the effect of building a road, were not included because we focused on indicator taxa that potentially address multiple impacts.

Papers that referred to high level taxonomic groups such as birds, fish, or macroinvertebrates as indicators (e.g. Hocutt, 1981; Morrison, 1986; Adamus, 1995) had to be excluded because analysis of all of the species falling into such a high taxonomic level would prove unwieldy. We also did not analyze biotic indices or similar procedures because our analysis required a concrete list of species, and we wanted to focus on indicator taxon selection and not the host of other techniques included in these methods. Finally, we did not include flora in the review because life history characteristics of plants can not be directly compared with those of animals.

Information for attributes associated to the list of criteria (Fig. 1) was collected from a number of sources. When available, data were collected from the reference in which the indicator taxon was suggested. Other sources were consulted to obtain information on conservation status and life history attributes (Appendixes A and B). When a suggested taxon included multiple species, the characteristics of the majority of the group were considered to hold true for the entire taxon. Details on the attributes collected for each identified indicator taxon are listed below.

2.2. Baseline information and research

2.2.1. Taxonomic status

When neither the author nor other sources reviewed claimed the suggested indicator to be taxonomically unclear at the taxonomic level used, the taxonomy was recorded as established.

2.2.2. Estimation of information from primary literature

We conducted a keyword search in Biosis (1997) for the common and scientific names of each taxon. We tabulated the relative number of citations as an estimate of the amount of primary literature available on each indicator taxon used. Any search result of greater than 30 references was considered to reflect adequate baseline information, regardless of the applicability of the references to ecosystem health.

2.2.3. Tolerance

This category reflects an indicator taxon's ability to withstand a broad range of human impacts. A more tolerant taxon would not show any measurable change as a result of small or medium impacts. If the author(s) suggesting the indicator taxa tested or cited studies testing the tolerance, the tolerance was considered addressed. Adequacy of the studies was not examined. If tolerance had been established in previous studies but was not cited by the author, it would not have been recorded.

2.2.4. Correlation to other biota

Similar to tolerance, this category addresses whether changes in each indicator taxon have been correlated with ecosystem changes. The suggested taxon satisfied this criterion if the author performed or cited studies that established this correlation. Adequacy of studies was not examined, and studies not cited by the author went undetected.

2.3. Locational information

2.3.1. Distribution

Three categories were used to define the taxon's distribution: local, regional, and global. Local was defined as a biogeographic unit (e.g. mountain range) or specific local habitat type (e.g. California redwood forest) and regions as entire continents or sub-continent (e.g. Mongolia, North India, or North America). Global classification required presence of the taxon on multiple continents.

2.3.2. Migratory

Any taxon that had a defined seasonal shift in non-contiguous habitats in any part of the taxon's range was classified as migratory.

2.3.3. Home range

Range was considered only for terrestrial mammals because the migratory criteria covered invertebrates, herps, birds, and aquatic mammal mobility. All invertebrates reviewed in this paper are classified as having small home range because their home ranges are small compared to the extent of most ecosystem health study sites. The median of nineteen randomly selected terrestrial mammals, 700 ha, served as a rough division point between large and small home ranges (van Gelder, 1982).

2.4. Niche and life history characteristics

2.4.1. Trophic level

Low trophic level taxa included only herbivorous species of which the adults were potentially prey for other species. Medium trophic level included omnivorous and carnivorous taxa of which the adults were also potential prey to other species. High trophic level was defined as carnivorous taxa of which adults were not actively hunted by any non-human species. For example, lake trout are terminal predators in the Great Lakes system and therefore classified as high trophic level (Marshall et al., 1987).

2.4.2. Specialist vs generalist

Vertebrate habitat specialists include all species defined as such by authors or other references, or any taxon found in only one habitat type as defined by Miller (1951). Invertebrate taxa were classified as specialists if authors or others defined them as such, or if they utilized only one substrate type (Merrit and Cummins, 1996). Food specialists were defined as monophagous or oligophagous.

2.4.3. Reproductive rates

As all invertebrates, fishes, and amphibians reviewed in this paper fall into the life history strategy of producing many young, these animals were all classified as having high reproductive rates. To separate different reproductive strategies among birds and mammals, the median of 19 randomly selected mammals and birds, 5.5 per year, served as a division point between high and low reproductive rates (Burt and Grossenheider, 1980; van Gelder, 1992).

2.4.4. *Damped fluctuations*

This category was only applied to invertebrate taxa where authors categorized taxa as having low or high population fluctuations. Such fluctuations in populations might be the consequence of environmental factors such as resource availability or weather or a result of population densities.

2.4.5. *Body size*

The median length of 19 randomly selected mammals and birds, 29 cm, served as a rough division point between large and small body sizes for mammals and birds. Whereas bird and mammal body size is indicative of life history strategies, this is less well-established for amphibians, fishes, and invertebrates, so these groups were not classified by body size.

2.4.6. *Easy to find*

Some authors suggesting invertebrate indicator taxa classified the taxa as easy or difficult to find, so we included their classifications in the compilation of data.

2.5. *Other agendas served by suggested indicators and implementation*

2.5.1. *Taxon at risk*

The vertebrate taxon was considered at risk if it was listed in IUCN red data list, in state, federal, or other countries' listings.

2.5.2. *Economic value*

If stated by author or other references, or there is a known market for the taxon, it was included in this category. Hunting was considered an economic value.

2.5.3. *Other indicators used or suggested*

The author(s) suggested or used multiple indicator species.

3. Results and discussion

Appendixes A and B list the 100 vertebrate and 32 invertebrate indicator taxa reviewed. The results for whether or not each taxon met the attributes and corresponding selection criteria are summarized in Fig. 1. Baseline information for suggested indicator taxa is generally inadequate. The taxonomy is the only baseline information criterion that appears to be well-established for almost all of the taxa at the taxonomic level suggested, although taxonomy may not be clear at the species level. Forty-four percent of reviewed vertebrates and 25% of reviewed invertebrates failed the biology and life history criteria, based on apparent lack of primary references. Studies of physical tolerance levels of the vertebrate taxa were only executed or referred to 8% of the vertebrate articles. In contrast, authors cited or completed physical tolerance studies on invertebrate indicators 84% of the time. In both vertebrates and invertebrates only 1 and 3% of the taxa, respectively, referred to studies correlating changes in the indicator status with changes to the ecosystem. While such research may be difficult, establishing tolerance levels and correlating changes of the indicator with the ecosystem is critical for indicators to be informative about ecosystem health. Otherwise, determining the magnitude of an impact and how the ecosystem health is affected when a taxon indicates an impact remains unclear.

Not all of the proposed indicator taxa satisfied the locational criteria either. Only 54% of the vertebrates and 69% of the invertebrates reviewed have a global distribution, although this criterion, as discussed earlier, should be of secondary importance. More importantly, many suggested vertebrate taxa are highly mobile. Fifty-nine of the 67 avian taxa and three of the 16 mammalian taxa migrate, and seven of the 11 suggested terrestrial mammals fall into the large home range category. If mobile taxa are used, characteristics monitored should reflect conditions of the study site. For example, nestling success in the study area of some migratory birds is the type of data that may be attributed more easily to local conditions, despite their migratory status. Invertebrates reviewed in this paper are, for the most part, unable to move long distances, and their inability to escape adverse local conditions contributes to their potential value as indicators.

Vertebrate taxa also fared poorly in fulfilling suggested niche and life history criteria. Eighty-five percent of the vertebrates are generalists, failing the specialist criterion. In contrast, all of the invertebrate taxa are specialists and have high reproductive rates, and only 16% are categorized as high trophic level, attributes associated with biota that are likely to satisfy the early warning criterion. These same attributes, however, are also often indicative of populations that tend to fluctuate, failing the low variability criterion. This can make it difficult to differentiate between natural population fluctuations (noise) and population decline due to human impacts (signal). Only five of 17 invertebrates classified have populations that fluctuate in number.

In contrast to invertebrates, 72% of the vertebrates have low reproductive rates, 24% are large-bodied, and

36% are high trophic level taxa. These attributes suggest that these taxa, while their populations do not fluctuate much, may not be useful early warning detectors because longevity and low reproductive rates of these taxa make for slow changes in population structure and number. Large bodies and high trophic level status also indicate that such taxa are probably found at low densities (Blueweiss et al. 1978) and more susceptible to local extinction (Shaffer, 1981). Susceptibility to local extinction means that these taxa may also fail to fulfill the criterion of monitoring a wide range of stresses. In addition, low-density taxa can prove to be more difficult to find and quantitatively assess. Although no authors recommending vertebrate taxa classified taxa by ease of finding, 18 of 22 invertebrates are categorized as easy to find.

A number of suggested indicator taxa are also priorities on political, social, or economic agendas. Thirty percent of vertebrate taxa and 19% of invertebrate taxa include at least one species considered at risk. Additionally, 18% of the vertebrates have some economic value compared to only one invertebrate taxon. Reviewed taxa may also serve other agendas such as flagship (Dietz et al., 1994; Hunter and Sulzer, 1996) or umbrella species (Clark et al., 1996), but we only quantify at risk and economically valuable taxa making this estimate very conservative. As many of these taxa fail to satisfy the other criteria, selection of taxa servicing other agendas may be inhibiting selection of taxa that would be more appropriate as indicators of ecosystem health.

Finally, use of multiple indicator taxa is suggested for 91% of the vertebrate indicator taxa; all of the proposed invertebrate taxa are suggested as part of a set of complementary indicator taxa. That no single taxa can accurately reflect ecosystem health is well understood. Poor selection of multiple indicator taxa, however, will still lead to poor monitoring of ecosystem health. Each indicator should embody as many of the criteria as possible to create the most effective set of complementary indicator taxa.

While it may appear that invertebrates satisfy more criteria, this result is confounded by the higher level taxonomy of most suggested invertebrates as compared to vertebrates. All but four vertebrates reviewed were suggested at the species level, while the suggested invertebrates represented seven orders, 12 families, four subfamilies, and nine species. This difference in the level of taxonomy makes direct comparisons between invertebrates and vertebrates difficult. High taxonomic level suggestions such as beetles or birds may lead to inclusion of inappropriate taxa as indicators and unclear outcomes. In addition, the analysis of high level taxon may not reveal inclusion of poor indicator taxa because of generalizations made for each taxon at the level suggested.

Despite the limitations of most indicators reviewed, several proposed indicator taxa fulfill important multiple criteria that might make them more appropriate for future use. For example, Coleoptera: Cicindelidae (Pearson and Cassola, 1992), Lepidoptera: Morphinae and Satyrinae (Brown, 1991), Hymenoptera: Apoidae, Vespidae, and Sphecidae (Brown, 1991) are easy to find specialists. They are also relatively immobile, have detectable trends, and baseline studies examining tolerance levels and correlation of the taxa changes in the ecosystem are available. Few suggested vertebrate taxa satisfy multiple criteria. However, the spring peeper (*Pseudacris crucifer*) (Hecnar and M'Closkey, 1996), a vertebrate specialist, has limited mobility and adequate baseline information, and has shown detectable changes in local abundance. Few negative indicators, taxa that may increase in abundance with increased levels of human disturbance, were suggested. Yet negative indicators, such as Diptera, brown-headed cowbirds, and Norway rats, are potentially easier to find, quantitatively measure, and manipulate. Negative indicators might also provide data over a larger range of stress although, as with all indicator taxa, changes in the taxa need to be correlated to changes in the ecosystem (Landres et al., 1988).

4. Conclusion

This review demonstrates that there is room for improvement in selecting both vertebrate and invertebrate taxa that better satisfy the criteria put forth in the conservation science literature for identifying useful indicator taxa. The major shortcomings of the invertebrates reviewed include failure to establish correlation between changes in the indicator taxa and the ecosystem and selecting taxa at high taxonomic level, which potentially increases the number of inappropriate species and noise in the data. In general, the vertebrates reviewed lack established tolerance levels and correlation with changes in the ecosystem. Most suggested vertebrates are also low density, highly mobile generalists, and service other agendas while lacking other characteristics desirable for indicator taxa.

Our efforts revealed that published criteria for selection of indicator taxa are often unclear and conflicting in many cases. Failure of proposed indicator taxa to adhere to suggested criteria may be, in part, due to unclear criteria. Our step-wise framework clarifies and prioritizes selection criteria and assures that each taxon in a complementary set fulfill a majority of the criteria. This framework could eliminate the shortcomings associated with past selection of indicator taxa, so that a set of complementary indicator taxa may better serve as a tool for conservation of biological resources. Moving toward clear and objective selection of indicator taxa is one way to enhance the utility of indicator taxa. However, successful use of indicator taxa also depends on clear and established understanding of what indicator taxa will indicate, and on establishment of objective metrics of ecosystem health that can be associated to indicator taxa.

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Appendix A

Table A1
List of vertebrate indicator taxa reviewed

Suggested vertebrate taxa[a]		Reference
Chaetodontidae	Butterflyfishes	Hourigan et al., 1988
Micropterus dolomieu	Small-mouth bass	Ecosystem Objectives Committee, 1990
Salvelinus namaycush	Lake trout	Marshal et al., 1987; Ecosystem Objectives Committee, 1990
Stizostedian vitreum	Walleye	Ecosystem Objectives Comittee, 1990
Ambystoma maculatum	Spotted salamander	Hecnar and M'Closkey, 1996
Notophthalmus viridescens	Fire salamander	Hecnar and M'Closkey, 1996
Bufo americanus	American toad	Hecnar and M'Closkey, 1996
Bufo boreas	Western toad	Kiester and Eckhardt, 1994
Hyla versicolor	Grey tree frog	Hecnar and M'Closkey, 1996
Rana sylvatica	Wood frog	Hecnar and M'Closkey, 1996
Rana pipiens	Northern leopard frog	Hecnar and M'Closkey, 1996
Rana pretiosa	Spotted frog	Kiester and Eckhardt, 1994
Rana catesbeiana	Bull frog	Hecnar and M'Closkey, 1996
Rana clamitans	Green frog	Hecnar and M'Closkey, 1996
Rana palustris	Pickerel frog	Hecnar and M'Closkey, 1996
Pseudacris crucifer	Spring peeper	Hecnar and M'Closkey, 1996
Pseudacris triseriata	Western chorus frog	Hecnar and M'Closkey, 1996
Histrionicus histrionicus	Harlequin duck	Kiester and Eckhardt, 1994
Pelecanus occidentalis californicus	California brown pelican	Davis, 1989
Phalacrocorax auritus	Double-crested cormorant	Davis, 1989
Ardeidae	Herons	Spalding and Frederick, 1995
Ardea herodias	Great blue heron	Kiester and Eckhardt, 1994
Ardea cinerea	Grey heron	Bharucha and Gofte, 1990
Ardea purpurea	Purple heron	Bharucha and Gofte, 1990
Egretta	Egrets	Spalding and Frederick, 1995
Egretta garzetta	Smaller egret	Bharucha and Gofte, 1990
Ciconia ciconia	White necked stork	Bharucha and Gofte, 1990
Mycteria leucocephela	Painted stork	Bharucha and Gofte, 1990
Anastomus oscitans	Asian open-billed stork	Bharucha and Gofte, 1990
Phoenicopterus ruber	Lesser flamingo	Bharucha and Gofte, 1990
Threskiornithinae	Ibises	Spalding and Frederick, 1995
Plegadis falcinellus	Glossy ibis	Bharucha and Gofte, 1990
Plataleinae	Spoonbills	Spalding and Frederick, 1995
Platalea leucorodia	Eurasian spoonbill	Bharucha and Gofte, 1990
Anthropoides virgo	Demoiselle crane	Bharucha and Gofte, 1990
Grus grus	Eurasian common crane	Bharucha and Gofte, 1990
Anas penelope	Eurasian wigeon	Bharucha and Gofte, 1990
Anas crecca	Common teal	Bharucha and Gofte, 1990
Anas clypeara	Northern shoveller	Bharucha and Gofte, 1990
Anas acuta	Northern pintail	Bharucha and Gofte, 1990
Gallinago stenura	Painted pintail	Bharucha and Gofte, 1990
Aythya ferina	Common pochard	Bharucha and Gofte, 1990
Aythya australis	White-eyed pochard	Bharucha and Gofte, 1990

continued on next page

Table A1 (*continued*)

Suggested vertebrate taxa[a]		Reference
Aythya fuligula	Tufted pochard	Bharucha and Gofte, 1990
Tadorna ferruginea	Ruddy shelduck	Bharucha and Gofte, 1990
Porphyrio porphyrio	Purple moorhen	Bharucha and Gofte, 1990
Fulica atra	Common coot	Bharucha and Gofte, 1990
Himantopus himantopus	Black-winged stilt	Bharucha and Gofte, 1990
Charadrius alexandrinus	Snowy plover	Davis, 1989
Charadrius dubius	Little-ringed plover	Bharucha and Gofte, 1990
Charadrius alexandrinus	Kentish plover	Bharucha and Gofte, 1990
Limosa limosa	Black-tailed godwit	Bharucha and Gofte, 1990
Numenius arquata	Eurasian curlew	Bharucha and Gofte, 1990
Tringa nebularia	Common greenshank	Bharucha and Gofte, 1990
Tringa achropus	Green sandpiper	Bharucha and Gofte, 1990
Actitis macularia	Spotted sandpiper	Bharucha and Gofte, 1990
Actitis hypoleucos	Common sandpiper	Bharucha and Gofte, 1990
Calidris temminckii	Temminck stint	Bharucha and Gofte, 1990
Larus argentatus	Herring gull	Bharucha and Gofte, 1990
Larus brunnicephalus	Brown headed gull	Bharucha and Gofte, 1990
Larus occidentalis	Western gull	Davis, 1989
Synthliboramphus hypoleuca	Xantu murrelet	Davis, 1989
Ptychoramphus aleuticus	Cassin auklet	Davis, 1989
Haliaeetus leucocephalus	Bald eagle	Kiester and Eckhardt, 1994; USDA, 1996; Suring and Sidle, 1987
Harpia harpyja	Harpy eagle	Albuquerque, 1994
Aquila pomarna	Spotted eagle	Bharucha and Gofte, 1990
Circus cyaneus	Northern marsh harrier	Bharucha and Gofte, 1990
Accipiter gentilis	Northern goshawk	Suring and Sidle, 1987; Rissler, 1995; AFSEE, 1996
Pandio haliaetus	Osprey	Bharucha and Gofte, 1990; Suring and Sidle, 1987
Falco sparverius	American kestrel	Davis, 1989; Johnson-Duncan et al., 1986
Dendragapus obscurus	Blue grouse	Suring and Sidle, 1987;Kiester and Eckhardt, 1994
Strix occidentalis caurina	Northern spotted owl	Doak, 1989; Rissler, 1995 Foster, Thomas, Korth, Bowmer, 1995
Otus kennicotti	Western screech owl	Johnson-Duncan et al., 1986
Micrathene whitneyi	Elf owl	Johnson-Duncan et al., 1986
Glaucidium brasilianum	Ferruginous pygmy-owl	Johnson-Duncan et al., 1986
Aegolius funeleus	Boreal owl	Kiester and Eckhardt, 1994
Athene cunicularia	Burrowing owl	Johnson-Duncan et al., 1986
Chaetura vauxi	Vaux swift	Kiester and Eckhardt, 1994
Sphyrapicus ruber	Red-breasted sapsucker	Kiester and Eckhardt, 1994; USDA, 1996
Picoide villosus	Hairy woodpecker	Kiester and Eckhardt, 1994; USDA, 1996
Certhia americana	Brown creeper	Suring and Sidle, 1987; Kiester and Eckhardt, 1994; USDA, 1996
Ixoreus naevius	Varied thrush	Kiester and Eckhardt, 1994
Melospiza melodia	Song sparrow	Davis, 1989
Loxia sp.	Crossbill	Kiester and Eckhardt, 1994; USDA, 1996
Ursus americanus	Black bear	Suring and Sidle, 1987; Kiester and Eckhardt, 1994; USDA, 1996
Ursus arctos	Brown bear	Suring and Sidle, 1987; Kiester and Eckhardt, 1994; USDA, 1996
Canis lupus ligoni	Alexander archipelago wolf	Suring and Sidle, 1987; Kiester and Eckhardt, 1994; USDA, 1996
Martes americana	Marten	Suring and Sidle, 1987; Kiester and Eckhardt, 1994; USDA, 1996
Lutra canadensis	River otter	Suring and Sidle, 1987; Kiester and Eckhardt, 1994; USDA, 1996
Urocyon littoralis	Island fox	Davis, 1989
Zalophus californianus	California sea lion	Davis, 1989
Callorhinus ursinus	Northern fur seal	Davis, 1989
Phoca vitulina richardsi	Harbor seal	Davis, 1989
Pagophilus groenlandicus	Harp seal	Timoshenko, 1995
Mirounga angustirostris	Northern elephant seal	Davis, 1989
Tamiasciurus hudsonicus	Red squirrel	Suring and Sidle, 1987; Kiester and Eckhardt, 1994; USDA, 1996
Glaucomys sabrinus	Northern flying squirrel	Kiester and Eckhardt, 1994
Peromyscus maniculatus	Deer mouse	Davis, 1989
Odocoileus hemionus sitkansis	Sitka black-tailed deer	Suring and Sidle, 1987; Kiester and Eckhardt, 1994; USDA, 1996
Oreamnos americanus	Mountain goat	Suring and Sidle, 1987; Kiester and Eckhardt, 1994; USDA, 1996

[a] Additional sources were used to gather information on life history and conservation status, including Stebbins, 1962; Stebbins, 1985; Herald, 1972; Breen, 1974; Carlander, 1977; Burt and Grossenheider, 1980; Lee et al., 1980; Terres, 1980; Frost, 1985; Perrins and Middletin, 1985; Wheeler, 1985; National Geographic Society, 1987; Ehrlich et al., 1988; Capulo, 1989; Dubois, 1990; Parker, 1990; Sibley and Monroe, 1990; Nowak, 1991; del Hoyo et al., 1992; van Gelder, 1982; Witt, 1992; Frtjord, 1993; Weigant and Steinhaus, 1993; Brauning et al., 1994; Kostyushin, 1994; Maedlow and Mayer, 1996; Ogilvie, 1996; Ranner et al., 1995; Garrison et al., 1996; Rasmussen, 1996; Rodger, 1996; USFWS, 1996; WCMC, 1997.

Appendix B

Table B1
List of invertebrate indicator taxa reviewed

Suggested invertebrate taxa[a]		Reference
Oligochaeta: Lumbricus terrestris	Earth worm	Xiaoming and Grizelle, 1995
Bivalvia: Macoma balthica	Clam	Wilson, 1994
Amphipoda: Pontoporeia hoyi	Benthic anthropod	Ecosystem Objectives Committee, 1990
Araneae: Erigone dentipalpis	Spider	Pristavko and Zhukovets, 1988
Araneae: Oedothorax apicatus	Spider	Pristavko and Zhukovets, 1988
Araneae: Pachygnata degeeri	Spider	Pristavko and Zhukovets, 1988
Araneae: Xerolycosa miniata	Spider	Pristavko and Zhukovets, 1988
Araneae: Pardosa pullata	Spider	Pristavko and Zhukovets, 1988
Homoptera: Membracidae, Cercopidae	Tree hoppers, froghoppers, spittlebugs	Brown, 1991
Coleoptera: Carabidae, Cicindelidae, Elateridae, Cerambycidae	Of beetles	Brown, 1991
Coleoptera: Carabidae	Carabid beetles	Pizzolotto, 1994
Coleoptera: Cicindelidae	Tiger beetles	Pearson and Cassola, 1992
Diptera	Flies	Brown, 1991; Resh, 1995
Diptera: Chironomidae	Midges	Saether, 1979; Hannaford and Resh, 1995; Brown, 1991
Lepidoptera: Arctiidae	Tiger moths, footman moths	Brown, 1991
Lepidoptera: Bait-attracted Nymphalinae	Brush-footed butterflies	Brown, 1991
Lepidoptera: Heliconiini, Ithomiinae	Heliconine and ithomiine butterflies	Brown, 1991
Lepidoptera: Morphinae, Satyrinae	Morpho butterflies, wood nymphs, satyrs	Brown, 1991
Lepidoptera: Papilionidae, Pieridae	Swallow tails, whites, sulphurs orange-tips	Brown, 1991
Lepidoptera: Satyrinae: Henotesia	Satyrs, wood nymphs	Kremen, 1994
Lepidoptera: Sphingidae, Saturnoidae	Hawk and silk moths	Brown, 1991
Hymenoptera: Formicidae	Ants	Brown, 1991
Hymenoptera: Apoidea, Vespidae, Sphecidae	Bees, vespid and sphecid wasps	Brown, 1991
Hemip.: Coreidae, Pentatomidae, Cygaeidae, Tingidae, Myridae	True bugs	Brown, 1991
Collembola	Spring tails	Brown, 1991
Ephemeroptera	Mayflies	Eaton and Lenat, 1991; Resh, 1995
Ephemeroptera: Cinygmula	Mayfly	Jackson and Resh, 1988
Ephemeroptera: Hexagenia limbata	Burrowing mayfly	Brown, 1991
Isoptera	Termites	Brown, 1991
Odonata	Dragonflies, damsel fllies	Brown, 1991
Plecoptera	Stoneflies	Eaton and Lenat, 1991; Resh, 1995
Trichoptera	Caddisflies	Eaton and Lenat, 1991; Resh, 1995

[a] Additonal source were used to provide information life history and conservation status, including Borror et al., 1976; Merrit and Cummins, 1996; WCMC, 1997.

References

Adamus, P.R., 1995. Validating a habitat evaluation method for predicting avian richness. Wildlife Society Bulletin 23, 743–749.

AFSEEE (Association of Forest Service Employees for Environmental Ethics), 1996. Tongass in Transition: Blueprint for a Sustainable Future. The AFSEEE-Sponsored Alternative Plan for the Tongass National Forest in Southeast Alaska, AFSEEE, Eugene, OR.

Albuquerque, J.L.B., 1995. Observations of rare raptors in southern atlantic rainforest of Brazil. Journal Field Ecology 66, 363–369.

Anderson-Carnahan, L.S., Foster, M., Thomas-Korth, W., Bowmer, K.H., 1995. Selection of a suitable cladoceran species for toxicity testing in turbid waters. Australian Journal of Ecology 20, 28–33.

Bharucha, E.K., Gogte, P.P., 1990. Avian profile of a man-modified aquatic ecosystem in the backwaters of the Ujjani Dam. Bombay Natural Hist. Society 87, 73–87.

Biosis, 1997. Biosis Inc., Philadelphia, PA.

Blueweiss, L., Fox, H., Kudzma, V., Nakashima, D., Peters, R., Sams, S., 1978. Relationships between body size and some life history parameters. Oecologia 37, 257–272.

Borror, D.J., DeLong, D.M., Triplehorn, C.A., 1976. An Introduction to the Study of Insects, Holt, Rinehart and Winston, New York.

Bortone, S.A., Davis, W.P., 1994. Fish intersexuality as indicator of environmental stress. Bioscience 44, 165–172.

Brauning, D.W., Brittingham, M.C., Gorss, D.A., Leberman, R.C., Master, T.C., Muvihill, R.S., 1994. Pennsylvania's breeding birds of special concern: a listing rational and status update. Journal of the Pennsylvania Academy of Science 68, 3–28.

Breen, J.F., 1974. Encyclopedia of Reptiles and Amphibians. T.F.H. Publications, Neptune City, NJ.

Brown, K.S., 1991. Conservation of neotropical environments: insects as indicators. In: Collins, N.M., Thomas, J.A. (Eds.), The Conservation of Insects and their Habitats. Academic Press, London, pp. 349–404.

Burt, W.H., Grossenheider, R.P., 1980. A Field Guide to Mammals. (Peterson Field Guides). Houghton Mifflin Company, Boston, MA.

Callicott, J.B., Mumford, K., 1997. Ecological sustainability as a conservation concept. Conservation Biology 11, 32–40.

Capulo, M., 1989. Guide to Reptiles and Amphibians of the World, Simon and Schuster, New York.

Carlander, K.D., 1977. Handbook of Freshwater Fishery Biology, vols. 1–2. Iowa State University Press. Ames, IA

Cherel, Y., Weimerskirch, H., 1995. Seabirds as indicators of marine resources: black-browed albatrosses feeding on ommastrephid squids in Kerguelen waters. Marine Ecology Progress Series 129, 295–300.

Clark, T.W., Paquet, P.C., Curlee, A.P., 1996. Special section: large carnivore conservation in the Rocky Mountains of the United States and Canada. Conservation Biology 10, 936–939.

Croonquist, M.J., Brooks, R.P., 1991. Use of avian and mammalian guilds as indicators of cumulative impacts in riparian-wetland areas. Environmental Management 15, 701–714.

Davis, G.E., 1989. Design of a long-term ecological monitoring program for Channel Islands National Park, California. Natural Areas Journal 9, 80–89.

del Hoyo, J., Elliot, A., Sargatal, J. (Eds.), 1991. Handbook of the Birds of the World, vols. 1–3. Lynx Ediciones, Barcelona.

Dietz, J.M., Dietz, L.A., Nagagata, E.Y., 1994. The effective use of flagship species for conservation of biodiversity: the example of lion tamarins in Brazil. In: Olney, P.J.S., Mace, G.M., Feistner, A.T.C (Eds.), Creative Conservation: Interactive Managment of Wild and Captive Animals. Chapman and Hall, London, pp. 32–49.

Doak, D., 1989. Spotted owls and old growth logging in the Pacific Northwest. Conservation Biology 3, 389–396.

Dubois, P.J., 1990. Com homologation national list of rare birds in France 1989. Alauda 58, 245–266.

Eaton, L.E., Lenat, D.R., 1991. Comparison of a rapid bioassessment method with North Carolina's qualitative macroinvertebrate collection method. J. N. Am. Bentol. Soc 10, 335–338.

Ecosystem Objectives Committee, 1990. Final report of the Ecosystem Objectives Committee, March 1990. Report to the Great Lakes Science Advisory Board, Windsor, ON.

Ehrlich, P.R, Dobkin, D.S., Wheye, D., 1988. The Birder's Handbook: A Field Guide to the Natural History of North American Birds, Simon and Schuster, New York.

Faith, D.P., Walker, P.A., 1996. Environmental diversity: on the best-possible use of surrogate data for assessing the relative biodiversity of sets of areas. Biodiversity and Conservation 5, 399–415.

Fausch, K.D., Lyons, J., Karr, J.R., Angermeier, P.L., 1990. Fish communities as indicators of environmental degradation. In: Adama, S.M. (Ed.), Biological Indicators of Stress in Fish. American Fisheries Symposium 8, Bethesda, MD, pp 123-145.

Frost, D.R., 1985. Amphibian Species of the World: A Taxonomic and Geographical Reference. Allen Press/Association of Systematics Collections, Lawrence, KS.

Frtjord, K., 1993. Food Habits of Arctic foxes *Alopex lagopus* on the western coast of Suabard. Arctic 46, 49–54.

Garrison, B.A., Hunting, K.W., Sernka, K.J., 1996. Training Manual for the California Wildlife Habitat Relationships System, CWHR Database, ver. 5.2. California Interagency Wildlife Task Group/ California Department of Fish and Game, Sacramento, CA

Hannaford, M.J., Resh, V.H., 1995. Variability in macroinvertebrate rapid-bioasessment surveys and habitat assessment in a northern California stream. J. N. Am Benthol. Soc. 14, 430–439.

Harris, J.H., 1995. The use of fish in ecological assessments. Australian Journal of Ecology 20, 65–80.

Hecnar, S.J., M'Closkey, R.T., 1996. Regional dynamics and the status of amphibians. Ecology 77, 2091–2097.

Hellawell, J.M., 1986. Biological Indicator of Freshwater Pollution and Environmental Management, Elsevier Applied Science Publishers, London.

Herald, E.S., 1972. Fishes of North America, Doubley and Company, New York.

Hocutt, C.H., 1981. Fish as indicators of biological integrity. Fisheries 6, 28–31.

Hourigan, T.F., Tricas, T.C., Reese, E.S., 1988. Coral reef fishes as indicators of environmental stress in coral reefs. In: Soule, D.F., Kleppel, G.S. (Eds.), Marine Organisms as Indicators. Springer-Verlag, New York, pp. 107–135.

Hughes, R.M., Whittier, T.R., Thiele, S.A., Pollard, J.E., Peck, D.V., Paulsen, S.G., Mcmullen, D., Lazorchak, J., Larsen, D.P., Kinney, W.L., Kaufmann, P.R., Hedtke, S, Dixit, S.S., Collins, G.B., Baker, J.R., 1992. Lake and stream indicators for the United States Environmental Protection Agency's environmental monitoring and assessment program. In: McKenzie, D.H., Hyatt, D.E., McDonald, V.J. (Eds.), Ecological Indicator, I. Elsevier Applied Science, London, pp. 305–335.

Hunter, M.L., Sulzer, A., 1996. Fundamentals of Conservation Biology, Blackwell Science, Cambridge.

Jackson, J.K., Resh, V.H., 1988. Sequential decision plans in monitoring benthic macroinvertebrates: cost savings, classification accuracy, and development of plans. Can. J. Fish. Aquat. Sci 45, 280–286.

Jamieson, D., 1995. Ecosystem health: some preventative medicine. Environmental Values 4, 333–344.

Johnson-Duncan, E.E., Duncan, D.K., Johnson, R.R., 1986. Small nesting raptors as indicators of change in the southwest desert. Proceeding of the Southwest Raptor Management Symposium and Workshop, 21–24 May 1986. National Wildlife Federation Scientific and Technical Series No. 11, University of Arizona, Tucson, AZ.

Johnson, R.K., Widerholm, T., Rosenberg, D.M., 1993. Freshwater biomonitoring using individual organisms, populations, and species assemblages of benthic macroinvertebrates. In: Rosenberg, D.M., Resh, V.H. (Eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York, pp. 40–158.

Karr, J.R., 1981. Assessment of biotic integrity using fish communities. Fisheries 6, 21–27.

Kelly, J.R., Harwell, M.A., 1990. Indicators of ecosystem recovery. Environmental Management 14, 527–545.

Kiester, A.R., Eckhardt, C., 1994. Review of Wildlife Management and Conservation Biology on the Tongass National Forest: A Synthesis with Recommendations. USDA, Forest Service, Pacific Northwest Research Station, Corvallis, OR.

Kolasa, J., Pickett, S.T.A., 1992. Ecosystem stress and health: an expansion of the conceptual basis. Journal of Aquatic Ecosystem Health 1, 7–13.

Kostyushin, V.A., 1994. New record of rare and protected bird species in Nikolaev Oblast. Vestink Zoologii 5, 47.

Kremen, C., 1994. Biological inventory using target taxa: a case study of the butterflies of Madagascar. Ecological Applications 4, 407–422.

Kremen, C., Colwell, R.K., Erwin, T.L., Murphy, D.D., Noss, R.F., Sanjayan, M.A., 1993. Terrestrial arthropod assemblages: their use in conservation planning. Conservation Biology 7, 796–808.

Kushlan, J.A., 1993. Colonial waterbirds as bioindicators of environmental change. Colonial Waterbirds 16, 223–251.

Lackey, R.T., 1995. Ecosystem health, biological diversity, and sustainable development: research that makes a difference. Renewable Resources Journal 13, 8–13.

Landres, P.B., Verner, J., Thomas, J.W., 1988. Critique of vertebrate indicator species. Conservation Biology 2, 316–328.

Lee, D.S., Gilbert, C.R., Hocult, C.H., Jenkins, P.E., McAllister, D.E., Stauffer, J.R., 1980. Atlas of North American Freshwater Fishes, Publication #1980-12 of the North Carolina Biological Survey, North Carolina State Museum of Natural History, NC.

Louette, M., Bijnenes, L., Agenenong'a, D.U., Fotso, R.C., 1995. The utility of birds as bioindicators: case studies in equatorial Africa. Beig. J. Zool 125, 157–165.

Maedlow, W., Mayr, C., 1996. Population trends of some scarce breeding birds in Germany 1990-1994. Vogelwelt 117, 249–260.

Maho, Y.L., Gendner, J.P., Challet, E., Bost, C.A., Gilles, J., Verdon, C., Plumere, C., Robin, J.P., Handrich, Y., 1993. Undisturbed breeding penguins as indicators of changes in marine resources. Marine Ecology Progress Series 95, 1–6.

Marshall, T.R., Ryder, R.A., Edwards, C.J., Spangler, G.R., 1987. Using the lake trout as an indicator of ecosystem health- application of the dichotomous key. Great Lakes Fishery Commission, Technical Report No. 49, Ann Arbor, MI.

Meffe, G.K., Carroll, C.R., 1997. Principles of Conservation Biology, 2nd Ed. Sinauer Associates Sunderland, MA.

Merenlender, A., Kremen, C., Rakotondratsima, M., Weiss, A., 1998. Monitoring impacts of natural resource extraction on lemurs of the Masoala Peninsula, Madagascar. Conservation Ecology [online] 12,www.consecol.org/vol2/iss2/art5

Merrit, R.W., Cummins, K.W., 1996. An Introduction to the Aquatic Insects of North America, 3rd ed. Kendall/Hunt, Dubuque, IA.

Miller, A.H., 1951. An analysis of the distribution of birds of California. University of California Publ. Zool 50, 531–644.

Morrison, M.L., 1986. Bird populations as indicators of environmental change. Current Ornithology 3, 429–451.

National Geographic Society, 1987. Field Guide to the Birds of North America, 2nd ed. The National Geographic Society, Washington, DC.

Noss, R.F., 1990. Indicators for monitoring biodiversity: a hierarchical approach. Conservation Biology 4, 355–364.

Nowak, R.M., 1991. Walker's Mammals of the World, 5th ed., vols. I–II. Johns Hopkins University Press, Baltimore, MD.

Nyholm, N.E., 1995. Monitoring of terrestrial environmental metal pollution by means of free- living insectivorous birds. Annali di Chimica 85, 343–351.

Ogilvie, M., 1996. Rare breeding birds in the United Kingdom in 1994. British Birds 89, 387–417.

Parker, S.P. (Ed.), 1990. Grizmek's Encyclopedia of Mammals. vols. 1 and 4. McGraw-Hill, New York.

Pearman, P.B., Guerrero, M., Sisk, T.D., Murphy, D.D., 1995. Correlation patterns among groups proposed as biological indicators: what do they indicate. Supplement to the Bulletin of the Ecological Society of America 76, 375.

Pearson, D.L., Cassola, F., 1992. World-wide species richness patterns of tiger beetles (Coleoptera Cicindelidae) indicator taxon for biodiversity and conservation studies. Conservation Biology 6, 376–391.

Perrins, C.M., Middletin, A.L.A., 1985. The Encyclopedia of Birds, Equinox Ltd, New York.

Pizzolotto, R., 1994. Soil arthropods for faunal indices in assessing changes in natural value resulting from human disturbance.. In: Boyle, T.J.B., Boyle, C.E.B. (Eds.), Biodiversity, Temperate Ecosystems, and Global Change, NATO ASI Series, vol.20. Springer-Verlag, Berlin, pp. 291–313.

Pristavko, V.P., Zhukovets, E.M., 1988. Spiders (Araneida) as objects of ecological monitoring in the Berezinskiy Nature Reserve. Entomological Review 67, 26–32.

Quinn, T., 1997. Coyote (*Canis latrans*) food habits in three urban habitat types of western Washington. Northwest Science 71, 1–5.

Ranner, A., Laber, J., Berg, H.M., 1995. Record of rare and remarkable species in Australia 1980–1990. First report of the Avifaunistic Commission of Birdlife Australia. Egretta 38, 59–98.

Rapport, D.J., 1992. Evaluating ecosystem health. Journal of Aquatic Ecosystem Health 1, 15–24.

Rapport, D.J., 1995a. Ecosystem health: more than a metaphor. Environmental Values 4, 287–387.

Rapport, D.J., 1995b. Ecosystem health: an emerging integrative science. In: Rapport, D.J., Gaudet, C.L., Calow, P. (Eds.), Evaluating and Monitoring the Health of Large-Scale Ecosystems. Springer, New York, pp. 5–33.

Rasmussen, P.A.F., 1996. Rare birds in Denmark and Greenland in 1994. Dannsk Ornitologisk Forenigs Tids-Krift 90, 141–152.

Regier, H., 1990. Workgroup issue paper: indicators and assessment of the state of fisheries. Environmental Monitoring and Assessment 15, 289–294.

Resh, V.H., 1995. Freshwater benthic macroinvertebrates and rapid assessment procedures for water quality monitoring in developing and newly industrialized countries. In: Davis, W.S., Simon, T.P. (Eds.), Assessment and Criteria Tools for Water Resource Planning and Decision Making. Lewis, London, pp. 167–177.

Rissler, L.J., 1995. Resource partitioning of two indicator species (northern spotted owl and northern goshawk): analysis of forest structure. Bulletin of the Ecological Society of America 72, 227.

Rodger, J.A., 1996. Birds. In: Pritchard, P.C.H. (Ed.), Rare and Endangered Biota of Florida, vol. V. University Press of Florida, Gainesville, FL.

Rosenberg, D.M., Hanks, H.V., Lehmkuhl, D.M., 1986. Importance of insects in environmental impact assessment. Environmental Management 10, 773–783.

Saether, O.A., 1979. Chironomid communities as water quality indicators. Holarctic Ecology 2, 65–74.

Shaffer, M.L., 1981. Minimum population sizes for species conservation. Bioscience 31, 131–134.

Sibley, C.G., Monroe, B.L. Jr., 1990. Distribution and Taxonomy of Birds of the World, Yale University Press, New Haven, CT.

Simberloff, D., 1998. Flagships, umbrellas, and keystones: is single-species management passe in the landscape era. Biological Conservation 83, 247–257.

Soule, M.E., 1985. Biodiversity indicators in California: taking nature's temperature. California Agriculture 49, 40–44.

Spalding, M.G., Frederick, P.C., (1995). The everglades: what can indicator species tell us about ecosystem health? Proceedings: American Association of Zoo Veterinarians, 12–17 August, 1995. Wildlife Disease Association, American Association of Wildlife Veterinarians, Joint Conference.

Stebbins, R.C., 1962. Amphibians of Western North America. University of California Press, Berkeley, CA.

Stebbins, R.C., 1985. Western Reptiles and Amphibians, 2nd ed. Peterson Field Guides. Houghton Mifflin Company, Boston.

Suring, L.H., Sidle, W.B., 1987. Selection and use of management indicator species on national forests in Alaska. Proceedings of the Western Raptor Management Symposium and Workshop 26–28 October 1987. National Wildlife Federation Scientific and Technical Series No. 12, Boise, IN.

Taper, M.L., Bohning-Gaese, K., Brown, J.H., 1995. Individualistic responses of bird species to environmental change. Oecologia 101, 478–486.

Temple, S.A., Wiens, J.A., 1989. Bird populations and environmental changes: can birds be bio-indicators. American Birds 43, 260–270.

Terres, J.K., 1980. Encyclopedia of North American Birds, Alfred A Knopf, New York.

Timoshenko, Y.K., 1995. Harp seals as indicators of the Barents Sea ecosystem. In: Blix, A.S., Walloe, L., Ultang, O. (Eds.), Whales, Seals, Fish and Man; International Symposium on the Biology of Marine Mammals in North East Atlantic, Tromso, Norway, November 29–December 1, 1994. Elsevier Science, Amsterdam, pp. 509–523.

USDA 1996. Tongass Land Management Plan Revision. Final Environmental Impact Statement. Part 1: Summary, Chapters 1 and 2, and Chapter 3 (Physical and Biological Environment). United States Department of Agriculture, Forest Service R10-MB-338b, Juneau, AK.

USFWS 1996. Endangered and Threatened Wildlife and Plants 50 CFR 17.11 and 17.12. USFWS, Department of Interior, United States Government.

van Franeker, J.A., 1992. Top predators as indicator for ecosystem events in the confluence zone and marginal ice zone of the Weddell and Scotia seas, Antarctica, November 1988 to January 1989 (EPOS Leg 2). Polar Biology 12, 93–102.

van Gelder, R.G., 1982. Mammals of the National Parks, John Hopkins University Press, Baltimore, MD.

WCMC (World Conservation Monitoring Centre), 1997. Electronic Animals Database 1997. IUCN Red List of Threatened Animals. IUCN Species Survival Commission.

Weigant, W.M., Steinhaus, G.H., 1993. Commissie Dwaalgasten Nederlandse Avifauna. Rare and scarce birds in Netherlands in 1993. Limosa 69, 13–22.

Wheeler, A., 1985. The World Encyclopedia of Fishes, Macdonald Books, Great Britain.

Wicklum, D., Davies, R.W., 1995. Ecosystem health and integrity. Can. J. Bot. 73, 997–1000.

Wilcove, D.S., 1989. Protecting biodiversity in multiple-use lands: lessons from the US Forest Service. TREE 4, 385–388.

Wilson, J.G., 1994. The role of bioindicators in estuarine management. Estuaries 17, 94–101.

Witt, K., 1992. Population changes of some selected endangered breeding bird species of Western Germany 1977 to 1989. Vogelwelt 113, 289–310.

Xiaoming, Z., Grizelle, G., 1995. Earthworm as an indicator of changes in ecosystem processes in reforested tropical lands. Bulletin of the Ecological Society of America 76, 290.

Toward Meaningful Snag-Management Guidelines for Postfire Salvage Logging in North American Conifer Forests

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Abstract: *The bird species in western North America that are most restricted to, and therefore most dependent on, severely burned conifer forests during the first years following a fire event depend heavily on the abundant standing snags for perch sites, nest sites, and food resources. Thus, it is critical to develop and apply appropriate snag-management guidelines to implement postfire timber harvest operations in the same locations. Unfortunately, existing guidelines designed for green-tree forests cannot be applied to postfire salvage sales because the snag needs of snag-dependent species in burned forests are not at all similar to the snag needs of snag-dependent species in green-tree forests. Birds in burned forests have very different snag-retention needs from those cavity-nesting bird species that have served as the focus for the development of existing snag-management guidelines. Specifically, many postfire specialists use standing dead trees not only for nesting purposes but for feeding purposes as well. Woodpeckers, in particular, specialize on wood-boring beetle larvae that are superabundant in fire-killed trees for several years following severe fire. Species such as the Black-backed Woodpecker (*Picoides arcticus*) are nearly restricted in their habitat distribution to severely burned forests. Moreover, existing postfire salvage-logging studies reveal that most postfire specialist species are completely absent from burned forests that have been (even partially) salvage logged. I call for the long-overdue development and use of more meaningful snag-retention guidelines for postfire specialists, and I note that the biology of the most fire-dependent bird species suggests that even a cursory attempt to meet their snag needs would preclude postfire salvage logging in those severely burned conifer forests wherein the maintenance of biological diversity is deemed important.*

Keywords: Black-backed Woodpecker, cavity-nesting birds, crown fire, mixed-severity fire, *Picoides arcticus*, salvage logging, stand-replacement fire

Hacia Directrices Significativas para la Gestión de Raigones en la Cosecha de Salvamento en Bosques de Coníferas de Norte América

Resumen: *Las especies de aves en el oeste de Norte América que están restringidas a, y por lo tanto más dependientes de, bosques de coníferas severamente quemados durante los primeros años después de un incendio dependen en alto grado de la abundancia de raigones en pie para sitios de percha, sitios de anidación y recursos alimenticios. Por lo tanto, el desarrollo y aplicación de directrices apropiadas para la gestión de raigones es crítico para la implementación de operaciones de cosecha posteriores al fuego en las mismas localidades. Desafortunadamente, las directrices existentes diseñadas para bosques verdes no se pueden aplicar a la venta de salvamento post fuego porque las necesidades de las especies dependientes de raigones en bosques quemados no son similares a las necesidades de las especies dependientes de raigones en bosques verdes. Las aves en bosques quemados tienen necesidades de retención de raigones muy diferentes a las de especies de aves que anidan en oquedades que han fungido como el centro para el desarrollo de las directrices de gestión de raigones existentes. Específicamente, muchos especialistas post fuego utilizan árboles muertos en pie no solo para propósitos de anidación sino también para propósitos alimenticios. En particular, los pájaros carpinteros se especializan en larvas de escarabajos perforadores de madera que son superabundantes en árboles*

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muertos por fuego durante varios años después de un incendio severo. Especies como Picoides arcticus *están casi restringidas a bosques severamente quemados. Más aun, los estudios de cosecha de salvamento post fuego revelan que la mayoría de los especialistas post fuego están completamente ausentes de bosques quemados con cosecha de salvamento (aun parcial). Hago un llamado para el desarrollo y utilización de directrices de retención de raigones más significativas para especialistas post fuego, y noto que la biología de la mayoría de las especies de aves dependientes de fuego sugiere que aun un intento apresurado de satisfacer sus necesidades de raigones excluiría la cosecha de salvamento post fuego en estos bosques de coníferas severamente quemados en los que se considera importante el mantenimiento de la diversidad biológica.*

Palabras Clave: aves andantes en oquedades, cosecha de salvamento, fuego de dosel, fuego de reemplazo de árboles, fuego de severidad mixta, *Picoides arcticus*

Density and Temporal Dynamics of Snags in Conifer Forests

Snags are standing dead trees from which most leaves and limbs have fallen (Thomas 1979; Thomas et al. 1979) and are usually the result of death due to lightning, fire, wind, disease, or insects. The U.S. Forest Service (USFS), Forest Inventory and Analysis program, defines a snag as a dead tree that is at least 22.5 cm dbh and 2 m tall. However defined, snags occur naturally in forests and play a crucial role in the ecology of forested ecosystems. A given snag will persist in the forest for years and will provide nesting, foraging, and roosting habitat for numerous species (Thomas et al. 1979; Harmon et al. 1986; Bull 2002).

Snag densities and characteristics vary significantly with forest type (Scott et al. 1980; Harris 1999) and forest age (Cline et al. 1980; Spies et al. 1988; Ohmann et al. 1994; Flanagan et al. 2004). Nevertheless, the focus of discussion about snags and other kinds of coarse woody debris has been tied strongly to the issue of old-growth rather than young, postdisturbance forests. If one focuses on North American conifer forests born of fire, one realizes that snag densities are uniquely high in forests recently disturbed by stand-replacement fire (Harmon et al. 1986; Everett et al. 1999). Indeed, in burned forest 1 year after a fire, the mean basal area of snags has been predicted (Spies et al. 1988) and observed (Drapeau et al. 2002) to be four times that in burned forests more than 1 year after a fire. Nearly 70% of all snags in landscapes dominated by stand-replacement fires occur in forests that are <20 years old (Lehmkuhl et al. 2003).

The main source of recruitment of snags also varies with forest type and stage of succession. Specifically, disease, beetles, and lightning probably account for the creation of snags in most forest types of advanced ages (Spies et al. 1988). In contrast, most of the snags that occur in early postdisturbance forest conditions are created by the disturbance event itself (Drapeau et al. 2002). As a consequence of these differences in the way snags are created, relatively few snags are at advanced decay stages in early postfire forests, whereas the reverse is true in older forests, where most snags show more advanced stages of decay (Drapeau et al. 2002). Even though advanced-decay-stage snags may be relatively rare in early postdisturbance forests, such forests still harbor some snags produced by disease, insects, or lightning before the disturbance event, and they constitute some of the most important wildlife snags therein. Regardless of origin, all the snags in early postdisturbance forests represent important legacies that are passed from one forest generation to the next by virtue of having survived the disturbance event as standing organic structures (Franklin et al. 2000).

Snags as Important Wildlife Resources

Remarkably, at least one fourth of all bird species in western forests (McClelland et al. 1979) and perhaps even as much as 45% of native North American bird populations (Balda 1975; Scott et al. 1980) are snag-dependent; that is, they require the use of snags at some point in their life cycle. Of the 102 terrestrial vertebrate species that occur in Washington State, over half (56) nest or den only in (require) the boles of dead or dying trees (Wilhere 2003). Moreover, an astounding two thirds of all wildlife species use deadwood structures or woody debris for some portion of their life cycles (Brown 2002). Such facts are clearly the driving force behind the development of snag-retention guidelines for managed lands. For birds in severely burned forests, the importance of snags goes well beyond the nesting needs of cavity-nesting species. By my own calculations (Hutto 1995), at least 60% of the species that nest in severely burned conifer forests use snags as nest sites, and virtually all those species nest only in or on snags.

The most valuable wildlife snags in green-tree forests are relatively large, as evidenced by the disproportionate number of cavities in larger snags (Lehmkuhl et al. 2003), and are relatively deteriorated (Drapeau et al. 2002). In burned conifer forests, the most valuable wildlife snags are also significantly larger than expected owing to chance, and are more likely to be thick-barked (ponderosa pine [*Pinus ponderosa* P.& C. Lawson], western larch [*Larix occidentalis* Nutt.], and Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco]) than thin-barked (Englemann spruce [*Picea engelmannii* Parry ex Engelm.], true firs [*Abies* sp.], and lodgepole pine [*Pinus contorta* Dougl. ex Loud.]) tree species (Hutto 1995; Saab & Dudley 1998; Kreisel & Stein 1999; Powell 2000; Haggard & Gaines 2001). In addition, broken-top snags (trees that were already snags before the fire event) are used as nest sites in recently burned conifer forests disproportionately often by both primary and secondary cavity-nesting bird species (Hutto 1995; Saab & Dudley 1998; Haggard & Gaines 2001).

The high value of large, thick-barked snags in severely burned forests has as much to do with the feeding opportunities as it does the nesting opportunities they provide birds. The phenomenal numerical response of woodpeckers of numerous species (Fig. 1a) that occupy recently burned conifer forests during both the breeding (Harris 1982; Murphy & Lehnhausen 1998; Powell 2000) and nonbreeding (Kreisel & Stein 1999) seasons is most certainly associated with the dramatic increase in availability of wood-boring beetle larvae (Muona & Rutanen 1994; Rasmussen et al. 1996; Saint-Germain et al. 2004) that serve as a superabundant food resource for woodpeckers. A marked increase in numbers of seed-eating bird species after crown fires is also undoubtedly related to the increased availability of seed resources after cones of blackened pine, fir, and spruce species open in response to fire (Fig. 1b). This helps explain why, in contrast with snags in green-tree forests, valuable wildlife snags in burned conifer forests include not only relatively soft snags (used for nesting by both cavity-nesting and open-cup-nesting species) but also snags that are at the sounder end of the snag decay continuum because the latter are what both beetles and birds require for feeding purposes (Nappi et al. 2003) and what many bird species use for nesting purposes. Consequently, burn specialists such as the Black-backed Woodpecker (*Picoides arcticus*), which depends on snags for both feeding and nesting, settle in areas with higher snag densities than expected owing to chance (Harris 1982; Saab & Dudley 1998; Kotliar et al. 2002).

*Figure 1. (a) Marks on burned trees left by woodpeckers that fed extensively on wood-boring beetle larvae in the snags. (b) Clark's Nutcracker (*Nucifraga columbiana*), a postfire specialist that has evolved a sublingual seed pouch that can hold more than 100 seeds, is extracting seeds from an underappreciated seed source—severely burned ponderosa pine.*

Broader Ecological Context for Snags in Severely Burned Conifer Forests

Uniquely high snag densities characterize severely burned conifer forests, and that makes for unique ecological conditions as well. But to what extent do severely burned forests occur naturally in western North American landscapes? Some (e.g., Skinner 2002) argue that the snag densities in postfire conifer forests are unnaturally high or that crown fires themselves are an unnatural product of our well-intentioned but misguided fire suppression policies of the past. Most conifer forest types, however, include crown-fire events as part of their natural fire regimes, and most are well within the natural range of variation (Romme 1982; Johnson et al. 2001; Johnson et al. 2003). Even low-elevation ponderosa pine forests are typified by an unspecified amount of mixed-severity fire (Arno 2000; Arno & Allison-Bunnell 2002), and at least some crown-fire events are perfectly natural, if rare, occurrences in those forest types as well (Agee 1993; Brown & Sieg 1996; Shinneman & Baker 1997; Brown et al. 1999; Veblen 2000; Ehle & Baker 2003; Schoennagel et al. 2004).

Even if the spatial scale over which stand-replacement fires occur in the lower-elevation conifer forest types is greater now than in the historic past, that is not to say that the presence of crown fires represents a process that is unnatural. Severe fires are clearly natural, and they constitute an important part of the fire regimes associated with most western conifer forest types (Arno 1980;

Heinselman 1981; DeByle et al. 1987; Arno 2000). Nevertheless, because severe fires are infrequent and numerically rare relative to the number of low-severity fire events that occur in the lower-elevation conifer forests of the West, the presence and importance of severe fires is currently underappreciated (Shinneman & Baker 1997; Baker & Ehle 2001; Ehle & Baker 2003; Baker & Shinneman 2004). This failure to appreciate the natural role of severe events may lead to well-intentioned but misguided management. Specifically, although detailed descriptions of a given fire regime might acknowledge the presence of frequent to infrequent occurrences of severe events, a given regime (i.e., low-severity regime) tends to be labeled by the most frequent kind of fire instead of by what might be a less frequent but biologically important component of the regime. Consequently, fire management tends to be focused primarily on restoration of the more common and not the least common type of fire in a given system. Heinselman (1981, 1985) anticipated this problem more than 20 years ago, when he argued strongly that restoration efforts should include the maintenance of stand-replacement regimes and stand-replacement events within low-severity regimes in at least the more remote portions of our public lands.

Besides the growing body of evidence that large, infrequent events are ecologically significant and not out of the range of natural variation (Foster et al. 1998; Turner & Dale 1998), an evolutionary perspective also yields some insight into the "naturalness" of severely burned forests. Specifically, there are unique biological and physical attributes that are relatively restricted to severely burned forests, indicating that infrequent events are not only natural, but biologically important as well. In a review of all published information on the effects of fire on forest birds of western North America, Kotliar et al. (2002) found that nine bird species are typically more abundant in burned than in unburned forests, as evidenced by a meta-analysis of results involving species that occurred in at least three studies prior to 2002. That number of species grows to 14 if one considers data from species that occurred in fewer than three studies, and that number more than doubles if fire severity is taken into account in the analysis (Smucker et al. 2005). Earlier studies of severely burned conifer forests throughout western Montana (Hutto 1995) suggest that as many as 15 species are nowhere more abundant than in recently and severely burned conifer forest. Some of these species (Black-backed Woodpecker, American Three-toed Woodpecker [*Picoides dorsalis*], Olive-sided Flycatcher [*Contopus cooperi*], Clark's Nutcracker [*Nucifraga columbiana*], and Mountain Bluebird [*Sialia currucoides*]) are even relatively narrowly restricted in their habitat distributions to, and presumably relatively dependent on, burned forest conditions (Hutto 1995; Hobson & Schieck 1999; Nappi 2000; Kotliar et al. 2002).

The life cycles of most wood-boring beetle species are 2–3 years, so the window of opportunity for many bird species that use recently burned forest is extremely narrow. Indeed, populations of timber-drilling woodpeckers peak at perhaps 4 years after a burn and then decline to near zero another 6 years after that (Taylor & Barmore 1980; Hoyt & Hannon 2002; Kotliar et al. 2002). These bird responses are unique to severe fires. Therefore, although less severe understory fires may be more frequent, they may also be less critical to the maintenance of some bird populations. The naturalness and importance of crown fires is reinforced by the fact that the bird species that are always more common in burned than in unburned forests are also more common in the more severely than in the less severely burned portions of those forests (Kotliar et al. 2002; Smucker et al. 2005).

Information on bird response to severe fire represents only a fraction of the biological uniqueness associated with recently burned conifer forests in western North America, as indicated by the large number of additional examples (Agee 1993; Whelan 1995; Brown & Smith 2000; Smith 2000; Arno & Allison-Bunnell 2002; Fisher & Wilkinson 2005) of positive responses of both plants (e.g., various *Pinus*, *Ceanothus, Arctostaphylos, Ribes, Dracocephalum, Corydalis, Geranium,* morel mushroom) and animals (e.g., numerous beetle species, boreal toad [*Bufo boreas*], spotted frog [*Rana pretiosa*], deer mouse [*Peromyscus maniculatus*], moose [*Alces alces*]) to what continues to be labeled "catastrophic" fire. The dramatic positive response of so many plant and animal species to severe fire and the absence of such responses to low-severity fire in conifer forests throughout the U.S. West argue strongly against the idea that severe fires are unnatural. The biological uniqueness associated with severe fires could emerge only from a long evolutionary history between a severe-fire environment and the organisms that have become relatively restricted in distribution to such fires. The retention of those unique qualities associated with severely burned forest should, therefore, be of highest importance in management circles. Yet, everything from the system of fire-regime classification, to a preoccupation with the destructive aspects of fire, to the misapplication of snag-management guidelines have led us to ignore the obvious: we need to retain the very elements that give rise to much of the biological uniqueness of a burned forest—the standing dead trees.

Postfire Salvage Logging and Its Effect on Snag-Dependent Species

So what happens if all snags are removed from a recently burned forest (Fig. 2a)? Research results on the ecological effects of a complete salvage harvest are consistent and overwhelmingly negative (McIver & Starr 2000). With respect to the avifauna, data suggest that there is no way to conduct a complete salvage harvest and retain suitable

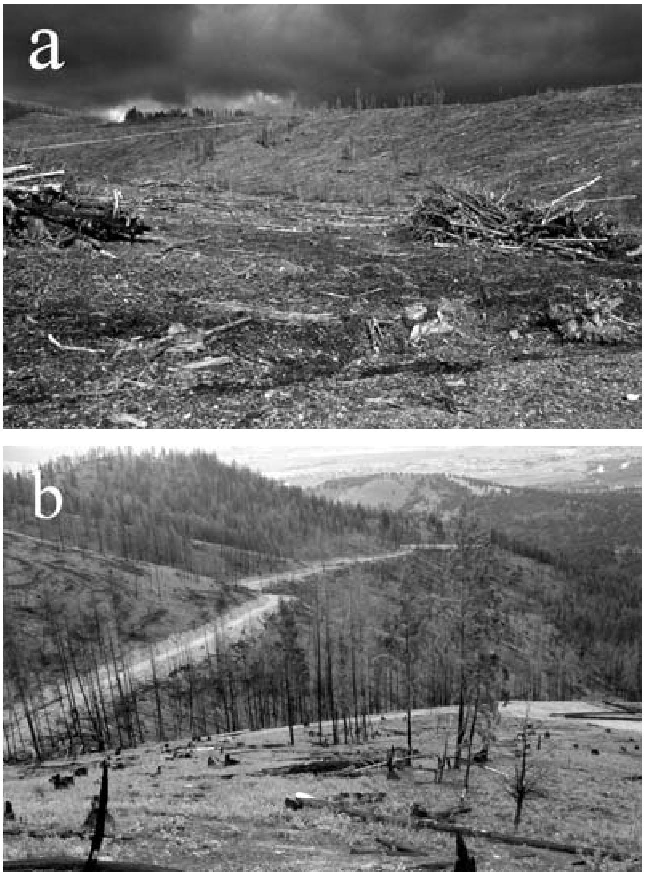

Figure 2. Burned forests that were (a) completely and (b) partially salvage logged.

conditions for species that would otherwise have occupied that patch of burned forest (Kotliar et al. 2002). This is especially true for bird species that depend on snags.

What about a less intensive, incomplete harvest (Fig. 2b)? The answer undoubtedly depends on the extent of harvest under consideration, and we currently lack data from a broad range of salvage intensities to be able to plot the precise relationship between snag density and bird density. Nonetheless, it is clear from existing data that incompletely logged burned forests still decrease the suitability of those forests for most cavity-nesting bird species (Kotliar et al. 2002). Most important, all existing studies of the effects of partial salvage logging on bird communities (Caton 1996; Hitchcox 1996; Saab & Dudley 1998; Haggard & Gaines 2001; Morissette et al. 2002) show negative effects on species that are most restricted to burned forests. For example, Black-backed and American Three-toed Woodpeckers are not only more abundant in uncut burned forests than in salvage-logged forests, they are frequently entirely absent from burned forests that have been incompletely salvage logged. The only bird species that may benefit from partial salvage logging (American Kestrel [*Falco sparverius*], Lewis's Woodpecker [*Melanerpes lewis*] and Western Bluebird [*Sialia mexicana*]) are not nearly as restricted in their distributions to burned forest conditions; they commonly occur in naturally open, unburned, low-elevation conifer forests as well (Saab & Dudley 1998; Haggard & Gaines 2001; Saab & Vierling 2001). Thus, it may be possible to develop methods of harvest that will mitigate negative effects on a handful of cavity-nesting bird species, but apparently not the most fire-dependent ones. In general, the very bird species that are most restricted to postfire conditions appear to be affected most negatively by postfire fuel-reduction logging or salvage logging (Kotliar et al. 2002; Morissette et al. 2002).

Perhaps there is a way to retain some of the ecological value associated with a burned forest in the face of partial salvage, and the finding that at least some species may benefit from partial salvage some of the time (Saab & Dudley 1998; Haggard & Gaines 2001; Kotliar et al. 2002) is encouraging. Nonetheless, the implementation of an adaptive-management cycle that is tightly coupled with a solid monitoring program will be needed to determine whether any level of salvage logging is compatible with the retention of the unique ecological values associated with severely burned forests (Robichaud et al. 2000; Hutto & Young 2002; Hutto 2004). So far, there are practically no data bearing on the effects of alternative styles of partial salvage logging because there has been neither the will nor the financial support needed to gain such knowledge.

Inadequacy of Current Snag Guidelines

Current snag-retention guidelines for most North American plant community types fall between 1 and 8 snags/ha. These guidelines emerged primarily from a consideration of the nesting requirements of cavity-nesting vertebrate species in the now classic *Blue Mountains* book (Thomas 1979). The retention of 8 snags/ha was judged to support 100% of the maximum population density of any of the woodpecker species that occur in the Blue Mountains area (Thomas 1979: Appendix 22). Bull et al. (1997) concluded that about 10 snags/ha in ponderosa pine and mixed-conifer forests should support viable populations of cavity-nesting birds. Thus, most current U.S. National Forest guidelines generally converge on the recommendation to retain 6–10 trees/ha, as do guidelines for Washington State, the Ontario Ministry of Natural Resources, the U.S. Army Corps of Engineers, and many other land-management agencies.

It has been acknowledged that snag guidelines should be sensitive to forest type and forest age because "the wildlife species that use snags are influenced by the stage of forest succession in which the snag occurs" and by the breakdown stage of the snag (Thomas et al. 1979). Moreover, snag types, sizes, and densities vary significantly with vegetation type (Harris 1999; Harmon 2002; White et al. 2002). Therefore, it follows necessarily that the desired snag types and densities will differ with both plant community type and successional stage and that we need as great a variety of guidelines as there are community types and successional stages (Bull et al. 1997; Everett et al. 1999; Rose et al. 2001; Kotliar et al. 2002; Lehmkuhl et al. 2003). Unfortunately, we have generally failed to adjust snag-retention recommendations to specific forest age, and nowhere is that failure more serious than for those special plant community types that were ignored in the development of the generic guidelines—recently burned conifer forests. Such forests are characterized by uniquely high densities of snags (Angelstam & Mikusinski 1994; Hutto 1995; Agee 2002; Drapeau et al. 2002), and snag use by most woodpeckers in burned forests requires high snag densities because they nest in and feed from burned snags.

These facts have been overlooked in the development and implementation of meaningful snag-management guidelines. Indeed, these guidelines have generally converged toward an average of 6–7 trees/ha because that number was deemed more than adequate to meet the nesting requirements of cavity-nesting wildlife species (Thomas et al. 1979:69). Snag guidelines were not originally developed with an eye toward non-nesting uses of snags or from an attempt to mirror snag densities that typically occur on unmanaged reference stands. Snag guidelines are still much narrower than numerous authors have suggested they ought to be, and we currently run the risk of managing coarse woody debris with uniform standards across historically variable landscapes, which is entirely inappropriate. Instead, we should be managing for levels of coarse woody debris that more accurately mirror levels characteristic of the natural disturbance regime (Agee 2002). Clearly, we need more data on what might constitute meaningful snag targets for all forest types and successional stages, and those targets should be set on the basis of reference conditions from natural postdisturbance forests, not from managed forest stands and certainly not from consideration of only a single aspect of an organism's life history.

Newer guidelines that are appropriate for snag-dependent species that occupy standing dead forests at the earliest stage of succession are beginning to trickle in (Saab & Dudley 1998; Haggard & Gaines 2001; Saab et al. 2002; Kotliar et al. 2002), and authors suggest that 200–300 snags/ha may better address the needs of wildlife in burned forests. The issue has yet to receive the serious management attention it deserves, but the comprehensive review of habitat needs of vertebrates in the Columbia River Basin (Wisdom et al. 2000) and the recently developed DecAID modeling effort in Washington and Oregon represent important efforts toward providing that kind of management guidance (Marcot et al. 2002).

Current Postfire Management Decisions Related to Snag Retention

The following points regarding management decisions apply to western forest types that experience crown fire as at least a minor component of their fire regimes (and that is virtually all western forest types).

(1) The USFS uses fire as a motivation to harvest trees. This is evident because in most cases where postfire logging is proposed they had not already sold green-tree harvests in those particular areas prior to the time of fire disturbance. Even though land managers are becoming more aware of the overwhelmingly negative ecological impacts of postfire salvage logging, the management has not shifted correspondingly toward less salvage harvesting. Instead, the most common justification for such harvests seems to have shifted recently from "salvaging" what economic value there might be to preventing another catastrophic fire (McIver & Starr 2000). Recent modifications of legislation and regulations by provincial governments in Canada (cited in Nappi et al. 2003) and by the U.S. government as well (Healthy Forests Restoration Act) expedite or even provide incentives for salvage logging. Such legislation provides no commitment to meaningful snag retention on burned forest lands. This failure to appreciate the value of burned forests to ecosystem sustainability is exacerbated by the fact that industrial lands (and most state lands) are, and probably always will be, completely salvage logged after fire because the value of those lands to those landowners lies entirely with the potential for short-term economic gain. The onus lies squarely on public land managers to provide the necessary protection of snag resources on burned forest land, and that has yet to happen.

(2) The usual agency response to questions about the amount or kind of burned trees to leave is that it does not really matter because they propose taking only a small proportion of what burned, so there must be plenty left for wildlife. Although that could be true, there is no scientific basis for such a conclusion. The volume of burned timber needed to enable populations to expand enough so that they can weather the next hiatus without fire in a particular area is unknown.

(3) If a partial salvage is proposed, the level of snag retention is generally based on a gross misapplication of current snag guidelines. In short, meaningful snag-management guidelines for burned forests are lacking

because the general public and the land management agencies that act on behalf of the public do not recognize the biological value of snags in burned conifer forests.

Toward Solutions

Any postfire salvage logging operation that requires a consideration of the maintenance of biological diversity will have to deal with the facts associated with salvage logging, which are unprecedented in terms of how consistently negative the ecological effects of salvage logging are. Because postfire bird communities soon after fire are biologically unique and the most unique elements are lost after any kind of salvage harvest, postfire salvage logging (especially a complete harvest; illustrated in Fig. 2a) is clearly ill advised. Even though existing science-based data suggest that very little, if any, salvage logging should be conducted in burned forests, salvage logging will not cease any time soon. So what guidelines should be followed in the meantime until education about the benefits of burned forests takes a greater hold on the public psyche? As a general guideline for any kind of timber harvesting (green-tree or salvage harvesting), public land managers should always strive to emulate natural processes with harvesting that occurs on public lands (Hejl et al. 1995; Hobson & Schieck 1999; Hejl et al. 2002; Kotliar et al. 2002; Arno & Feidler 2005). Thus, snag-retention guidelines for salvage sales should be no different; they should be based on an explicitly identified postharvest emulation target that reflects the kind of natural disturbance process and stand structure that one hopes to emulate through the harvest process.

On patches that are harvested, cutting should either be intense enough that the result will emulate a later stage of succession that corresponds with the proposed level of snag retention or cutting should be low enough in intensity that there will be no significant ecological effect. Clearly, the only way to extract higher wood volumes from salvage timber sales would be to claim to be mimicking not the earliest stage of succession but something more like a forest 10 years after a fire, where natural snag attrition due to blowdown might be used to justify a much smaller snag-retention target. In no case would a complete salvage harvest mimic any stage of natural forest succession.

Even if managers take relatively few trees and do a good job of mimicking the numbers and kinds of snags in, say, a forest 10 years after a fire, it should be clear that if the naturally occurring earliest stage of succession (a forest 0–5 years after a fire, which normally contains hundreds of burned snags per hectare) is not managed for, then managers will have failed to maintain an important component of biological diversity: all the unique plants and animals that depend on those first few years of natural (postfire) succession. The retention on the landscape of some burned forest 0–5 years after a fire at any one point in time should be a management priority because that is the narrow window of time during which the biologically unique early postfire conditions become established and persist. And because there is less of that forest age than what was historically available due to successful fire suppression during the past half century (Gruell 1983; Hessburg et al. 2000), these forests should be valued at least as much as the small amounts of old-growth that are left. These facts alone seem justification enough to remove all burned forests from consideration for harvest, but the opposite tendency currently prevails.

An alternative strategy might be to salvage harvest only that number of snags that would still allow the special ecological conditions (such as those that I have described herein for birds) to be retained. Unfortunately, the only way to mimic natural snag densities for harvests that seek to mimic the very earliest stage of succession (immediately after a fire) would be to leave close to the same number of burned trees per unit area that would occur through a stand-replacement disturbance event. The numbers of standing dead trees per hectare immediately following stand-replacement fire number in the hundreds, of course (Everett et al. 1999), so snag guidelines should recommend perhaps 50 times the number currently recommended in the most commonly used guidelines. On top of that, the densities of snags in patches used by birds for cavity nesting (Harris 1982; Saab & Dudley 1998) and feeding (Kreisel & Stein 1999) are significantly higher than what is randomly available in early postfire forests, so even if guidelines were built on "average" snag densities associated with recently burned forests, they might still fall short of the densities actually needed by these birds. I hasten to add that I am only scratching the surface of this issue by concentrating my attention on the needs of birds. Even more stringent guidelines might follow from a consideration of the needs of snag-dependent, pyrophilous insects and spiders, for example (Nappi et al. 2004).

A partial salvage harvest that produces little or no ecological damage will be difficult to achieve because of the sensitivity of early postfire specialists to any disturbance, as described earlier. Unfortunately, we currently have no data on the relationship between levels of harvest and ecological consequences, as measured by birds, plants, or whatever ecological response variable one wishes to use. This lack of information led me to suggest earlier (Hutto 1995) that the safest strategy (if salvage logging must occur) may be to take some and leave some large patches of untouched burned forest across the landscape. As others (Hannon & Drapeau 2005) have noted, the unknown with this approach, however, lies with the mystery of how much to leave. In response to this question, Nappi et al. (2004) make it clear in their recent paper dealing with the effects of salvage logging in the boreal forest that it is dangerous to rest the maintenance of biodiversity on the assumption that the unharvested portion may compensate adequately for the intensively salvaged portion of burned forests.

Conclusions

The ecological cost of salvage logging speaks for itself, and the message is powerful. I am hard pressed to find any other example in wildlife biology where the effect of a particular land-use activity is as close to 100% negative as the typical postfire salvage-logging operation tends to be. If input from biologists is ever to have an influence on policy, this should certainly be one of those instances. Yet largely economic interests have apparently compelled legislators to ignore such facts and pass recent legislation in the United States and Canada that will serve to expedite salvage logging. Existing science-based data suggest that there is little or no biological or ecological justification for salvage logging. McIver and Starr (2000) note that because of this, the justification for salvage logging has begun to shift toward arguments related to rehabilitation or restoration, but those sorts of justifications also reflect a lack of appreciation that severe fires are themselves restorative events and that rehabilitation occurs naturally as part of plant succession (Lindenmayer et al. 2004). Interference with the natural process of plant succession by planting or spraying to speed the process of succession toward narrow timber-producing or old-growth goals, as some suggest (e.g., Sessions et al. 2004), is also incompatible with a holistic public-land-management goal of working within the constraints of a natural system. All things that characterize a severe disturbance event, including soil erosion and sometimes insufferably slow plant recovery, are precisely the things that constitute "rehabilitation" for those organisms that need those aspects of disturbance events at infrequent intervals to sustain their populations.

The profound failure of many decision makers to appreciate the ecological value of burned forests stems from their taking too narrow a view of what forests provide. The general belief that "dead and dying timber ought to be harvested and put to use" (Schwennesen 1992) prevailed prior to the infamous salvage rider of 1995 (U.S. Congress 1995), and it apparently still prevails today in many management circles. Land managers, politicians, and the public-at-large need to gain a better appreciation of the unique nature of burned forests as ecological communities, how sensitive the process of succession is to conditions immediately following the disturbance event (Platt & Connell 2003), and how important the legacy of standing deadwood is to the natural development of forests (Franklin et al. 2000). Nowhere are soils, special plants, or wildlife more sensitive to the proposition of tree harvesting than in a burned forest. And nowhere is the consideration of ecology more blatantly absent than in decisions to salvage log. Education to these facts is needed at all levels.

It is time for conservation biologists and enlightened land managers to educate others to the fact that there is an ecological benefit in staying out of forests that have been recently restored by natural stand-replacement fire. There are plenty of green-tree forest stands to harvest in a sustainable fashion while offering boons to local economies (especially in the urban interface), so economic arguments should not interfere with the responsible decision to celebrate the benefits of a natural restoration event when it happens and to harvest timber outside the biologically unique and rare severely burned forests.

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Literature Cited

Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Covelo, California.

Agee, J. K. 2002. Fire as a coarse filter for snags and logs. General technical report PSW-GTR-181:235-368. U.S. Department of Agriculture Forest Service, Albany, California.

Angelstam, P., and G. Mikusinski. 1994. Woodpecker assemblages in natural and managed boreal and hemiboreal forest—a review. Annales Zoologici Fennici **31:**157–172.

Arno, S. F. 1980. Forest fire history in the Northern Rockies. Journal of Forestry **78:**460–465.

Arno, S. F. 2000. Fire regimes in western forest ecosystems. Pages 97–120 in J. K. Brown, and J. K. Smith, editors. General technical report RMRS-GTR-42-volume 2. Effects of fire on flora. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Arno, S. F., and S. Allison-Bunnell. 2002. Flames in our forest. Island Press, Washington, D.C.

Arno, S. F., and C. E. Feidler. 2005. Mimicking nature's fire: restoring fire-prone forests in the West. Island Press, Washington, D.C.

Baker, W. L., and D. Ehle. 2001. Uncertainty in surface-fire history: the case of ponderosa pine forests in the western United States. Canadian Journal of Forest Research **31:**1205–1226.

Baker, W. L., and D. J. Shinneman. 2004. Fire and restoration of pinon-juniper woodlands in the western United States: a review. Forest Ecology and Management **189:**1–21.

Balda, R. P. 1975. The relationship of secondary cavity nesters to snag densities in western coniferous forests. Wildlife habitat technical bulletin 1. U.S. Department of Agriculture Forest Service, Albuquerque, New Mexico.

Brown, J. K., and J. K. Smith. 2000. Wildland fire in ecosystems: effects of fire on flora. General technical report RMRS-GTR-42-volume 2. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Brown, P. M., and C. H. Sieg. 1996. Fire history in interior ponderosa pine communities of the Black Hills, South Dakota, USA. International Journal of Wildland Fire **6:**97–105.

Brown, P. M., M. R. Kaufmann, and W. D. Shepperd. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology **14:**513–532.

Brown, T. K. 2002. Creating and maintaining wildlife, insect, and fish habitat structures in dead wood. General technical report PSW-GTR-181:883-892. U.S. Department of Agriculture Forest Service, Albany, California.

Bull, E. L. 2002. The value of coarse woody debris to vertebrates in the Pacific Northwest. General technical report PSW-GTR-181:171-178. U.S. Department of Agriculture Forest Service, Albany, California.

Bull, E. L., C. G. Parks, and T. R. Torgersen. 1997. Trees and logs important to wildlife in the interior Columbia River basin. General technical report PNW-GTR-391:1-55. U.S. Department of Agriculture Forest Service, Portland, Oregon.

Caton, E. L. 1996. Effects of fire and salvage logging on the cavity-nesting bird community in northwestern Montana. University of Montana, Missoula, Montana.

Cline, S. P., A. B. Berg, and H. W. Wight. 1980. Snag characteristics and dynamics in Douglas-fir forests, western Oregon. Journal of Wildlife Management **44:**773–786.

DeByle, N. V., C. D. Bevins, and W. C. Fischer. 1987. Wildfire occurrence in aspen in the interior western United States. Western Journal of Applied Forestry **2:**73–76.

Drapeau, P., A. Nappi, J.-F. Giroux, A. Leduc, and J.-P. Savard 2002. Distribution patterns of birds associated with snags in natural and managed eastern boreal forests. Pages 193–205 in W. F. Laudenslayer Jr., P. J. Shea, B. E. Valentine, C. P. Weatherspoon, and T. E. Lisle, editors. Proceedings of the symposium on the ecology and management of dead wood in western forests. General technical report PSW-GTR-181. U.S. Department of Agriculture Forest Service, Albany, California.

Ehle, D. S., and W. L. Baker. 2003. Disturbance and stand dynamics in ponderosa pine forests in Rocky Mountain National Park, USA. Ecological Monographs **73:**543–566.

Everett, R., J. Lehmkuhl, R. Schellhaas, P. Ohlson, D. Keenum, H. Riesterer, and D. Spurbeck. 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:**223–234.

Fisher, J. T., and L. Wilkinson. 2005. The response of mammals to forest fire and timber harvest in the North American boreal forest. Mammal Review **35:**51–81.

Flanagan, P. T., P. Morgan, and R. L. Everett. 2004. Snag recruitment in subalpine forests of the North Cascades, Washington State. General technical report PSW-GTR-181:517-525. U.S. Department of Agriculture Forest Service, Albany, California.

Foster, D. R., D. H. Knight, and J. F. Franklin. 1998. Landscape patterns and legacies resulting from large, infrequent forest disturbances. Ecosystems **1:**497–510.

Franklin, J. F., D. Lindenmayer, J. A. MacMahon, A. McKee, J. Magnuson, D. A. Perry, R. Waide, and D. Foster. 2000. Threads of continuity. Conservation Biology in Practice **1:**8–16.

Gruell, G. E. 1983. Fire and vegetative trends in the Northern Rockies: interpretations from 1871-1982 photographs. General technical report INT-158:1-117. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Haggard, M., and W. L. Gaines. 2001. Effects of stand-replacement fire and salvage logging on a cavity-nesting bird community in Eastern Cascades, Washington. Northwest Science **75:**387–396.

Hannon, S. J., and P. Drapeau. 2005. Bird responses to burning and logging in the boreal forest of Canada. Studies in Avian Biology **30:**97–115.

Harmon, M. E. 2002. Moving towards a new paradigm for woody detritus management. General technical report PSW-GTR-181:929-944. U.S. Department of Agriculture Forest Service, Albany, California.

Harmon, M. E., et al. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research **15:**133–302.

Harris, M. A. 1982. Habitat use among woodpeckers in forest burns. M.S. thesis. University of Montana, Missoula, Montana.

Harris, R. B. 1999. Abundance and chacteristics of snags in western Montana forests. General technical report RMRS-GTR-31:1-19. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Heinselman, M. L. 1981. Fire and succession in the conifer forests of northern North America. Pages 374–405 in D. C. West, H. H. Shugart, and D. B. Botkin, editors. Forest succession: concepts and applications. Springer-Verlag, New York.

Heinselman, M. L. 1985. Fire regimes and management options in ecosystems with large high-intensity fires. General technical report INT-182:101-109. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Hejl, S. J., R. L. Hutto, C. R. Preston, and D. M. Finch. 1995. Effects of silvicultural treatments in the Rocky Mountains. Pages 220–244 in T. E. Martin, and D. M. Finch, editors. Ecology and management of neotropical migratory birds. Oxford University Press, New York.

Hejl, S. J., D. E. Mack, J. S. Young, J. C. Bednarz, and R. L. Hutto. 2002. Birds and changing landscape patterns in conifer forests of the north-central Rocky Mountains. Studies in Avian Biology **25:**114–130.

Hessburg, P. F., B. G. Smith, R. B. Salter, R. D. Ottmar, and E. Alvarado. 2000. Recent changes (1930s-1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management **136:**53–83.

Hitchcox, S. M. 1996. Abundance and nesting success of cavity-nesting birds in unlogged and salvage-logged burned forest in northwestern Montana. [M.S. Thesis], University of Montana, Missoula, Montana.

Hobson, K. A., and J. Schieck. 1999. Changes in bird communities in boreal mixedwood forest: harvest and wildfire effects over 30 years. Ecological Applications **9:**849–863.

Hoyt, J. S., and S. J. Hannon. 2002. Habitat associations of black-backed and three-toed woodpeckers in the boreal forest of Alberta. Canadian Journal of Forest Research **32:**1881–1888.

Hutto, R. L. 1995. The 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. 2004. Northern Region Landbird Monitoring Program: a program designed to monitor more than long-term population trends. General technical report PSW-GTR-191:956-962. U.S. Department of Agriculture Forest Service, Albany, California.

Hutto, R. L., and J. S. Young. 2002. Regional landbird monitoring: perspectives from the northern Rocky Mountains. Wildlife Society Bulletin **30:**738–750.

Johnson, E. A., K. Miyanishi, and S. R. J. Bridge. 2001. Wildfire regime in the boreal forest and the idea of suppression and fuel buildup. Conservation Biology **15:**1554–1557.

Johnson, E. A., A. M. Gill, R. A. Bradstock, A. Granstrom, L. Trabaud, and K. Miyanishi. 2003. Towards a sounder fire ecology. Frontiers in Ecology and the Environment **1:**271–276.

Kotliar, N. B., S. J. Hejl, R. L. Hutto, V. A. Saab, C. P. Melcher, and M. E. McFadzen. 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.

Kreisel, I. J., and S. J. Stein. 1999. Bird use of burned and unburned coniferous forests during winter. Wilson Bulletin **111:**243–250.

Lehmkuhl, J. F., R. L. Everett, R. Schellhaas, P. Ohlson, D. Keenum, H. Riesterer, and D. Spurbeck. 2003. Cavities in snags along a wildfire chronosequence in eastern Washington. Journal of Wildlife Management **67:**219–228.

Lindenmayer, D. B., D. R. Foster, J. F. Franklin, M. L. Hunter, R. F. Noss, F. A. Schmiegelow, and D. Perry. 2004. Salvage harvesting policies after natural disturbance. Science **303:**1303.

Marcot, B. G., K. Mellen, J. L. Ohmann, K. L. Waddell, E. A. Willhite, B. B. Hostetler, S. A. Livingston, C. Ogden, and T. Dreisbach. 2002. The DecAID repository: background information for DecAID, the decayed wood advisor for managing snags, partially dead trees, and down wood for biodiversity in forests of Washington and Oregon. U.S. Department of Agriculture Forest Service, Pacific Northwest

Research Station, Portland, Oregon. Available from http://www.fs.fed.us/wildecology/decaid/decaid_background/decaid_home.htm (accessed April 2006).

McClelland, B. R., S. S. Frissell, W. C. Fischer, and C. H. Halvorson. 1979. Habitat management for hole-nesting birds in forests of western larch and Douglas-fir. Journal of Forestry **77:**480–483.

McIver, J. D., and L. Starr. 2000. Environmental effects of postfire logging: literature review and annotated bibliography. General technical report PNW-GTR-486:1-72. U.S. Department of Agriculture Forest Service, Portland, Oregon.

Morissette, J. L., T. P. Cobb, R. M. Brigham, and P. C. James. 2002. The response of boreal forest songbird communities to fire and post-fire harvesting. Canadian Journal of Forest Research **32:**2169–2183.

Muona, J., and I. Rutanen. 1994. The short-term impact of fire on the beetle fauna in boreal coniferous forest. Annales Zoologici Fennici **31:**109–121.

Murphy, E. G., and W. H. Lehnhausen. 1998. Density and foraging ecology of woodpeckers following a stand-replacement fire. Journal of Wildlife Management **62:**1359–1372.

Nappi, A. 2000. Distribution des pics et utilisation des arbres morts dans les forêts soumises aux perturbations naturelles en pessière à mousses. M.S. thesis. Université du Québec, Montreal.

Nappi, A., P. Drapeau, J.-F. Giroux, and J.-P. Savard. 2003. Snag use by foraging black-backed woodpeckers (*Picoides arcticus*) in a recently burned eastern boreal forest. Auk **120:**505–511.

Nappi, A., P. Drapeau, and J. P. L. Savard. 2004. Salvage logging after wildfire in the boreal forest: is it becoming a hot issue for wildlife? Forestry Chronicle **80:**67–74.

Ohmann, J. L., W. C. McComb, and A. A. Zumrawi. 1994. Snag abundance for primary cavity nesting birds on nonfederal forest lands in Oregon and Washington. Wildlife Society Bulletin **22:**607–620.

Platt, W. J., and J. H. Connell. 2003. Natural disturbances and directional replacement of species. Ecological Monographs **73:**507–522.

Powell, H. 2000. The influence of prey density on post-fire habitat use of the black-backed woodpecker. M.S. thesis. University of Montana, Missoula, Montana.

Rasmussen, L. A., G. D. Amman, J. C. Vandygriff, R. D. Oakes, A. S. Munson, and K. E. Gibson. 1996. Bark beetle and wood borer infestation in the Greater Yellowstone area during four postfire years. Research paper INT-RP-487:1-10. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Robichaud, P. R., J. L. Beyers, and D. G. Neary. 2000. Evaluating the effectiveness of postfire rehabilitation treatments. General technical report RMRS-GTR-63:1-85. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Romme, W. H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecological Monographs **52:**199–222.

Rose, C., B. G. Marcot, T. K. Mellen, J. L. Ohmann, K. Waddell, D. Lindley, and B. Schreiber. 2001. Decaying wood in Pacific Northwest forests: concepts and tools for habitat management. Pages 580–623 in D. Johnson and T. O'Neil, editors. Wildlife-habitat relationships in Oregon and Washington. Oregon State University Press, Corvallis, Oregon.

Saab, V., R. Brannon, J. Dudley, L. Donohoo, D. Vanderzanden, V. Johnson, and H. Lachowski. 2002. Selection of fire-created snags at two spatial scales by cavity-nesting birds. General technical report PSW-GTR-181:835-848. U.S. Department of Agriculture Forest Service, Albany, California.

Saab, V. A., and J. G. Dudley. 1998. Responses of cavity-nesting birds to stand-replacement fire and salvage logging in ponderosa pine/Douglas-fir forests of southwestern Idaho. Research paper RMRS-RP-11:1-17. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Saab, V. A., and K. T. Vierling. 2001. Reproductive success of Lewis's woodpecker in burned pine and cottonwood riparian forests. Condor **103:**491–501.

Saint-Germain, M., P. Drapeau, and C. Hebert. 2004. Comparison of Coleoptera assemblages from a recently burned and unburned black spruce forests of northeastern North America. Biological Conservation **118:**583–592.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience **54:**661–676.

Schwennesen, D. 1992. Timber leaders lobby for bill to speed up salvage efforts. Missoulian **13 June:** A1, A10.

Scott, V. E., J. A. Whelan, and P. L. Svoboda. 1980. Cavity nesting birds and forest management. General technical report INT-86:311-324. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Sessions, J., P. Bettinger, R. Buckman, M. Newton, and J. Hamann. 2004. Hastening the return of complex forests following fire: the consequences of delay. Journal of Forestry **102:**38–45.

Shinneman, D. J., and W. L. Baker. 1997. Nonequilibrium dynamics between catastrophic disturbances and old-growth forests in ponderosa pine landscapes of the Black Hills. Conservation Biology **11:**1276–1288.

Skinner, C. N. 2002. Influence of fire on the dynamics of dead woody material in forests of California and southwestern Oregon. General technical report PSW-GTR-181:445-454. U.S. Department of Agriculture Forest Service, Albany, California.

Smith, J. K. 2000. Wildland fire in ecosystems: effects of fire on fauna. General technical report RMRS-GTR-42-volume 1. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Smucker, K. M., R. L. Hutto, and B. M. Steele. 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecological Applications **15:**1535–1549.

Spies, T. A., J. F. Franklin, and T. B. Thomas. 1988. Coarse woody debris in Douglas-fir forests of western Oregon and Washington. Ecology **69:**1689–1702.

Taylor, D. L., and W. J. Barmore Jr. 1980. Post-fire succession of avifauna in coniferous forests of Yellowstone and Grand Teton National Parks, Wyoming. General technical report INT-86:130-145. U.S. Department of Agriculture Forest Service, Ogden, Utah.

Thomas, J. W. 1979. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Agriculture handbook 553. U.S. Department of Agriculture Forest Service, Washington, D.C.

Thomas, J. W., R. G. Anderson, C. Maser, and E. L. Bull. 1979. Snags. Pages 60–77 in J. W. Thomas, editor. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Agriculture handbook 553. U.S. Department of Agriculture, Forest Service, Washington, D.C.

Turner, M. G., and V. H. Dale. 1998. Comparing large, infrequent disturbances: what have we learned? Ecosystems **1:**493–496.

U.S. Congress. 1995. Emergency supplemental appropriations for additional disaster assistance, for anti-terrorism initiatives, for assistance in the recovery from the tragedy that occurred at Oklahoma City, and recissions act of 1995. Public Law 104-19, 104th Congress (27 July).

Veblen, T. T. 2000. Disturbance patterns in in southern Rocky Mountain forests. Pages 31–54 in R. L. Knight, F. W. Smith, S. W. Buskirk, W. H. Romme, and W. L. Baker, editors. Forest fragmentation in the southern Rocky Mountains. University Press of Colorado, Boulder, Colorado.

Whelan, R. J. 1995. The ecology of fire. Cambridge University Press, New York.

White, D. E., T. Atzet, P. A. Martinez, and L. A. McCrimmon. 2002. Dead wood and fire relationships in southwestern Oregon western hemlock forests. General technical report PSW-GTR-181:479-487. U.S. Department of Agriculture Forest Service, Albany, California.

Wilhere, G. F. 2003. Simulations of snag dynamics in an industrial Douglas-fir forest. Forest Ecology and Management **174:**521–539.

Wisdom, M. J., et al. 2000. Source habitats for terrestrial vertebrates of focus in the Interior Columbia Basin: broad-scale trends and management implications. General technical report PNW-GTR-485-volume 1. U.S. Department of Agriculture Forest Service, Portland, Oregon.

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THE EFFECTS OF POSTFIRE SALVAGE LOGGING ON CAVITY-NESTING BIRDS

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Abstract. We investigated the effects of postfire salvage logging on cavity-nesting birds by comparing nest densities and patterns of nest reuse over a three-year period in seven logged and eight unlogged patches of mixed-conifer forest in the Blackfoot-Clearwater Wildlife Management Area, Montana. We found 563 active nests of 18 cavity-nesting birds; all species were found nesting in the uncut burned forest plots, but only eight nested in the salvage-logged plots. All except one species nested at a higher density in the unlogged areas, and half of the species were significantly more abundant in the unlogged plots. Every timber-drilling and timber-gleaning species was less abundant in the salvage-logged plots, including two of the most fire-dependent species in the northern Rocky Mountains—American Three-toed (*Picoides dorsalis*) and Black-backed (*P. arcticus*) Woodpeckers. Lower abundances in salvage-logged plots occurred despite the fact that there were still more potential nest snags per hectare than the minimum recommended number needed to support maximum densities of primary cavity-nesters, which suggests that reduced woodpecker densities are more related to a reduction in food (wood-boring beetle larvae) than to nest-site availability. Because cavities were present in only four of 244 randomly selected trees, and because frequency of cavity reuse by secondary cavity-nesters was higher in salvage-logged than in unlogged plots, nest-site limitation may be a more important constraint for secondary cavity-nesters in salvage-logged areas. These results suggest that typical salvage logging operations are incompatible with the maintenance of endemic levels of most cavity-nesting bird populations, especially populations of primary cavity-nesting species.

Key words: *Black-backed Woodpecker, cavity-nesting birds, fire, healthy forests, salvage logging.*

Efectos de la Extracción Forestal Post-Incendio sobre las Aves que Anidan en Cavidades

Resumen. Investigamos los efectos de la extracción forestal post-incendio sobre las aves que anidan en cavidades por medio de una comparación de la densidad de nidos y los patrones de su reutilización. El estudio se realizó durante un periodo de tres años, en fragmentos de bosque mixto de coníferas quemados con y sin extracción forestal (siete y ocho fragmentos, respectivamente) en el Área de Manejo de Fauna Silvestre Blackfoot-Clearwater, Montana. Encontramos 563 nidos activos pertenecientes a 18 especies; todas las especies se registraron anidando en los fragmentos de bosque quemado sin extracción, pero sólo ocho de ellas anidaron en los fragmentos quemados con extracción. Con sólo una excepción, la densidad de nidos de todas las especies fue mayor en los fragmentos quemados sin extracción forestal y la mitad de las especies fueron significativamente más abundantes en los fragmentos sin extracción. Todas las especies que obtienen su alimento perforando la corteza o buscando en ella fueron menos abundantes en los fragmentos con extracción, incluyendo dos de las especies más dependientes de los incendios en las Montañas Rocallosas—*Picoides dorsalis* y *P. arcticus*. Las abundancias fueron menores en los fragmentos con extracción post-incendio, a pesar de contar con un número mayor de troncos por hectárea con potencial para construir nidos, en relación con el número mínimo recomendado para mantener densidades máximas de aves que anidan en cavidades. Esto sugiere que la disminución de las densidades de los pájaros carpinteros está más relacionada con una disminución en su alimento (larvas perforadoras de escarabajos), que con la disponibilidad de sitios de anidación. Ya que se detectaron cavidades en sólo cuatro de los 244 árboles seleccionados al azar, y dado que la tasa de reutilización de cavidades por parte de las especies de anidación secundaria fue mayor en los sitios con extracción que en sitios sin extracción, la restricción de sitios de anidación puede ser un factor limitante más importante para las especies de anidación secundaria en los sitios con extracción forestal post-incendio. Estos resultados sugieren que las operaciones típicas de extracción son incompatibles con el mantenimiento de los niveles

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endémicos de la mayoría de las poblaciones de aves que anidan en cavidades, especialmente para poblaciones de especies de anidación primaria.

INTRODUCTION

Severe, stand-replacing fire is arguably the most ecologically important disturbance agent in conifer forests of the northern Rocky Mountains (Gruell 1983, Pierce et al. 2004). These fires generally leave a mosaic of green, unburned areas interspersed with large areas of standing dead trees, or snags (Turner and Romme 1994, Turner et al. 2003). The ecological importance of this type of forest habitat is reflected in the fact that the avian community found in forests following stand-replacing fires is comprised, in part, of species (e.g., Black-backed Woodpecker [*Picoides arcticus*], American Three-toed Woodpecker [*Picoides dorsalis*], Olive-sided Flycatcher [*Contopus cooperi*], and Mountain Bluebird [*Sialia currucoides*]) that are not only relatively abundant in, but are also relatively restricted to, severely burned forest patches (Hutto 1995, Hutto and Young 1999, Kotliar et al. 2002, Smucker et al. 2005). These species depend heavily on the abundant standing dead trees associated with severely burned forests because they provide: (1) perches from which foraging sallies are initiated, (2) food for specialized beetle larvae, which themselves serve as prey for numerous woodpecker species (Powell 2000), and (3) nesting sites (Hutto 1995).

In unburned forest, the importance of snags to cavity-nesting birds has been clearly demonstrated (Dickson et al. 1983, Marcot 1983, Scott and Oldemeyer 1983, Zarnowitz and Manuwal 1985, Schreiber and deCalesta 1992) and recommendations for snag retention during timber harvests have been incorporated into current management plans (Thomas 1979, Raphael and White 1984, Zarnowitz and Manuwal 1985, Morrison and Raphael 1993). In burned forests, however, there are still no generally accepted snag retention guidelines for postfire conditions, even though the needs of snag-dependent species in burned forests are not the same as the needs of snag-dependent species in unburned forests (Hutto 1995, 2006). The lack of snag retention guidelines for burned forests is undoubtedly a reflection of the prevailing view that forest fires are "environmental disasters" (Taylor 1995) and that such fires leave behind fire-damaged timber that, other than as salvaged wood, lacks value. Consequently, salvage logging is a common management practice in the western United States after high-severity, stand-replacing fires, as it is elsewhere in the world after severe disturbance events (Lindenmayer et al. 2004).

Because burned forests are valuable to snag-dependent wildlife (and other plants and animals), and because some of these species appear to be nearly restricted in their distribution to such conditions (Hutto 1995, Hobson and Schieck 1999, Nappi 2000, Kotliar et al. 2002), managers should be especially concerned about mitigating possible negative effects of salvage logging on fire-dependent plants and animals. We need information on the effects of different levels of salvage logging to be able to assess whether any level of salvage logging is compatible with wildlife needs and, if so, which kinds and minimum numbers of snags are needed to support fire-dependent species after stand-replacing fires (Hutto 1995, 2006, Nappi et al. 2004).

Unfortunately, given the inherent danger associated with severe forest fires, the prospects of obtaining information on the effects of salvage logging from multiple, independently created experimental treatment and control plots will always be limited (Hargrove and Pickering 1992, Whelan 1995, Andersen et al. 1998, van Mantgem et al. 2001). Therefore, generalizations are most likely to emerge from meta-analyses of individual studies conducted when opportunities for study after unplanned fire events presented themselves (Kotliar et al. 2002). We capitalized on one of these rare opportunities to obtain empirical data on the effects of salvage logging when a large fire burned most of the Blackfoot-Clearwater Wildlife Management Area in western Montana. By working with the State Department of Fish, Wildlife and Parks and a local timber company, we were able to design a study that included a series of otherwise similar treatment (cut) and control (uncut) plots that allowed us to: (1) assess and compare breeding, cavity-nesting bird abundance in salvage-logged vs. unlogged burned forest patches, (2) assess and compare characteristics of nest trees and randomly

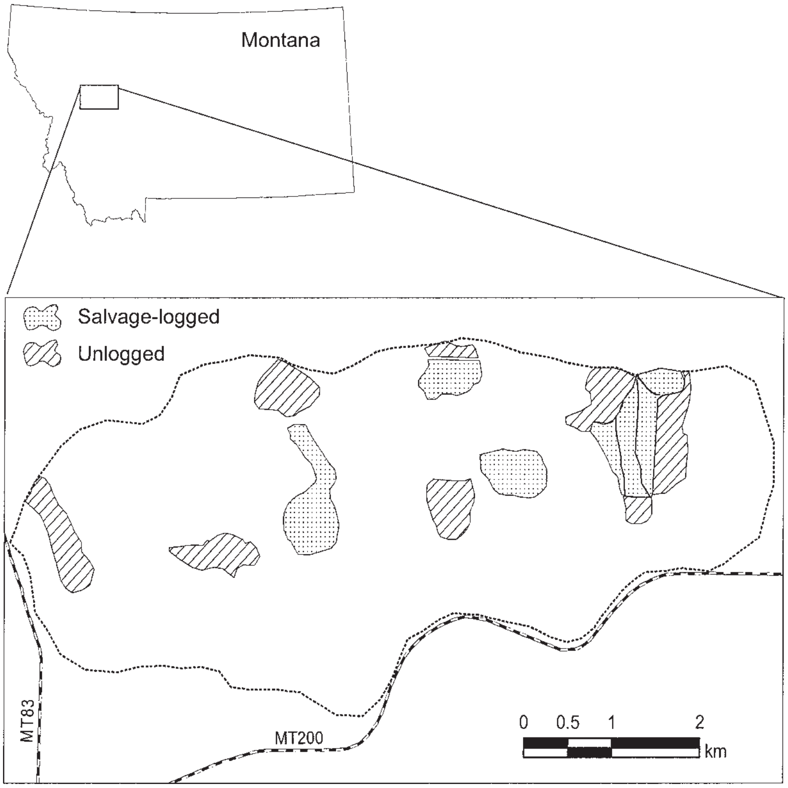

FIGURE 1. Map of the field plots used to study the effects of postfire salvage logging on cavity-nesting birds in the Blackfoot-Clearwater Wildlife Management Area, Montana. The fire boundary is indicated by the dotted line. Most of the area between plots is grassland, and MT83 and MT200 are state highways.

selected trees from salvage-logged and unlogged plots, and (3) document the reuse of cavities over a three-year period.

METHODS

STUDY AREA

This study took place on the Blackfoot-Clearwater Wildlife Management Area, which is located 80 km east of Missoula, Montana, at an elevation of approximately 1200 m. A severe fire in October of 1991 burned approximately 1600 ha of grassland and mixed-conifer forest. The burned forest areas consisted mainly of 50- to 150-year-old second-growth conifers (Douglas-fir [*Pseudotsuga menziesii*], western larch [*Larix occidentalis*], and ponderosa pine [*Pinus ponderosa*]) with occasional pockets of broadleafed deciduous trees (*Populus* spp.). Approximately 275 ha were salvage-logged in the winter following the fire (1991–1992) in a pattern in which seven salvage-logged plots (totaling approximately 134 ha) were interspersed with eight unlogged control plots (totaling approximately 148 ha; Fig. 1). The individual plots averaged 23.9 ± 11.7 SD ha in size (range = 7–36 ha), and were delineated either on the basis of isolation of the unit or on the basis of road locations and timber harvest prescriptions provided by Champion Timber Company. The extent of salvage logging varied among plots, but in most cases all merchantable (>15 cm dbh, >4.5 m tall) fire-killed timber was removed. Most unlogged and salvage-logged forest plots were surrounded by grasslands.

NEST ABUNDANCE AND CAVITY REUSE

We located active cavity nests by searching the 15 forest plots during the breeding season from mid-May through mid-July of 1993, 1994, and 1995. We walked parallel transects (spaced at 20 m intervals) through the plots to systematically search the plots for active nests. Plots were of a size that required between one and two days' search time to cover the entire plot. We searched plots at least twice during each breeding season, and walked transects in opposite directions from visit to visit to maximize our chances of finding all active nests. Nests were located by finding cavities directly and observing subsequent activity, following adult birds to nests, or hearing young birds in the nest. We recorded activity of adult birds at the nest (building, incubating, or feeding) at the time the nest was found and the type of cavity used (excavated or natural). The first time an active nest was found, it was considered a new nest. In subsequent years, the same nest cavities were checked for activity. Cavities active in subsequent years were considered reused nests, and cavities harboring a second pair of breeding birds of the same or a different species in a single year were classified as repeat nests.

VEGETATION AND TREE MEASUREMENTS

To characterize vegetation structure associated with the two treatments, we measured vegetation characteristics surrounding 132 randomly located trees in unlogged plots and 112 randomly located trees in salvage-logged plots. Sample trees were selected by locating random grid points on high-resolution aerial photos. If a random point fell on a tree, that tree was selected as a sample tree; otherwise the point was discarded. Random points were generated until all sample trees had been identified. Sample trees were located on the ground with the aid of aerial photos.

We recorded several characteristics associated with, and vegetation conditions surrounding, nest trees and randomly selected trees. Tree characteristics included size (diameter at breast height, measured with a dbh tape), height (measured with a clinometer), species, status (live or dead; intact or with a broken or dead top), percent bark remaining (estimated visually), and presence of a nest cavity. To assess vegetation characteristics surrounding focal trees, stakes were placed 15 m from the tree in the four cardinal directions. We made ocular estimates of the percent of bare ground (bare soil), ground cover (herbaceous plants <25 cm tall), low shrub cover (herbaceous plants >25 cm tall plus woody plants <0.5 m), and tall shrub cover (woody plants >0.5 m tall) in each quarter and then averaged the four estimates. These estimates were made from above, so the four estimates summed to 100%. Burn severity of the vegetation within 50 m of a sample tree was estimated using a subjective scale of one to five (after Hutto 1995), with 1 = 100% of the trees with green foliage or needles, 2 = most (>60%) trees with green foliage or needles, and most of the rest with brown needles, 3 = 40%–60% of the trees brown-needled, and the rest either green- or black-needled, 4 = most (>60%) trees black-needled, and the rest with brown needles, and 5 = all trees blackened. All vegetation measurements (including fire severity) on logged plots reflected conditions after cutting, so it was not possible to compare preharvest conditions on treatment and control plots.

STATISTICAL ANALYSES

We compared vegetation characteristics associated with the two treatments using Mann-Whitney *U*-tests for numerical data and chi-square likelihood ratios for categorical data. A Bonferonni correction for experiment-wise error was applied to adjust the *P*-value for multiple simultaneous tests. We tested for a difference in the distribution of nest abundance across species by treatment type using a chi-square likelihood ratio after combining numbers of nests across plots. Nest abundance was converted to nest density by averaging the number of nests per plot for each year of the study. We tested for differences in nest density over the three years of our study using repeated measures ANOVA. Nest densities were transformed by the inverse of the square root to achieve normality.

For each species with a minimum of nine nests, we compared characteristics of nest trees and randomly selected trees in both the unlogged and salvage-logged plots. Differences in numerical variables were tested with Mann-Whitney *U*-tests, and differences in distributions of categorical variables were tested with chi-square likelihood ratios. A Bonferonni correction for experiment-wise error was applied to all univariate tests for each species. When the same species nested in the same cavity in two or more years, the characteristics associated with that nest tree were included only once in the analysis.

For each primary cavity-nesting species, the suitability of randomly selected trees for nesting

TABLE 1. The percentage of nests found by each of three different methods and by status at the time they were found in unlogged and salvage-logged burned forest, Montana, 1993–1995. Most nests were discovered by following adult birds to the cavity, and most had either eggs or young when they were found.

	Unlogged	Salvage-logged
Method of location ($n = 300$)		
Following adult bird to nest	58	48
Finding cavity directly	21	23
Seeing or hearing adult or young at nest	21	29
Nest status ($n = 392$)		
Building, excavating, or advertising	17	18
Incubating	32	43
Feeding young	51	39

was assessed by first calculating the range of each of five characteristics associated with nest trees. If the value of a characteristic associated with a random tree fell within the range of values obtained for nest trees, that random tree was considered "suitable" for that characteristic. Overall suitability was then determined by combining suitability information from all the individual tree characteristics. If values of all characteristics of a random tree fell within the range of values from nest trees, that random tree was considered "suitable" for that species. The proportion of random trees classified as suitable for each species (all plots combined within each treatment) was then converted into an absolute number of suitable trees per hectare by multiplication with the estimated number of trees per hectare in each treatment (360 trees per ha in unlogged plots and 134 trees per ha in salvage-logged plots).

RESULTS

NEST ABUNDANCE

We found most nests between 8 and 23 June in both salvage-logged and unlogged areas. Most nests were discovered by following adult birds to the cavity, and most had either eggs or young when they were found (Table 1). Eighty-three percent of nests in unlogged plots and 77% of nests in salvage-logged plots (80% overall) were in cavities originally excavated by primary cavity-nesting birds.

We found 563 active nests distributed among 18 cavity-nesting species (Table 2). All 18 species nested in unlogged areas, whereas only eight species nested in salvage-logged areas (Table 2). The total number of active nests found in unlogged plots was almost three times higher than the total number of nests in salvage-logged plots, even though the area searched and search effort was similar between plot types (Table 2). The combined nest density was significantly higher in unlogged than in salvage-logged areas in each year of our study (treatment effect: $F_{1,13} = 8.3$, $P = 0.02$), and the mean density of all species combined increased significantly over the three years of study in both the unlogged and salvage-logged plots (year effect: $F_{2,26} = 4.2$, $P = 0.02$; Fig. 2). All cavity-nesting species except the Black-capped Chickadee (*Poecile atricapillus*) nested at a higher density in the unlogged than the salvage-logged areas, and half of all cavity-nesting species were significantly more abundant in the unlogged plots (Table 2). The most abundant species in unlogged areas (Northern Flicker [*Colaptes auratus*], House Wren [*Troglodytes aedon*], and Mountain Bluebird) were also the most abundant in salvage-logged areas, and the vast majority of nests in both salvage-logged and unlogged plots belonged to ground- and shrub-foraging species (73% and 93%, respectively) and to secondary cavity-nesting species (63% and 65%, respectively). Nevertheless, the relative abundances of all cavity-nesting species differed significantly between the two treatments ($\chi^2_{17} = 59.4$, $P < 0.001$), primarily because five timber-drilling woodpecker species and the two timber-gleaning nuthatch species nested only in the unlogged plots (Table 2).

CAVITY REUSE

Only 34% of the cavities excavated in 1993 were reused by cavity-nesting birds in 1994 (Table 3). In 1995, 36% of cavities used for nesting during either or both of the two previous years were reused. The frequency of cavity reuse in salvage-logged areas was 50% in 1994 and 44% in 1995. In contrast, the frequencies of reuse in unlogged areas were substantially lower—30% in 1994 and 33% in 1995 (Table 3). In 1995, a small percentage of cavities (2.6%) were used twice in one breeding season, and Mountain Bluebird, House Wren, and North-

TABLE 2. Total number of nests and mean nest densities (number per 40 ha ± SE) in unlogged and salvage-logged plots in burned forest in Montana. Numbers were tallied across three years from 1993 to 1995. We discovered a greater number of nests in unlogged than in salvage-logged plots for all but one species, and the mean nest densities of most cavity-nesting species were also significantly higher in the unlogged plots. Bonferonni-adjusted *P*-values are from Mann-Whitney *U*-tests for differences in mean density between the two treatments.

				Unlogged		Salvage-logged		
Species	Species code	Nesting guild[a]	Foraging guild[b]	Total no. nests	Mean density	Total no. nests	Mean density	*P*
American Kestrel (*Falco sparverius*)	AMKE	2°	GF	6	0.7 ± 0.1	5	0.4 ± 0.1	0.53
Lewis' Woodpecker (*Melanerpes lewis*)	LEWO	1°	AI	2	0.2 ± 0.1	0	0	0.35
Williamson's Sapsucker (*Sphyrapicus thyroideus*)	WISA	1°	TD	9	0.5 ± 0.1	0	0	0.07
Red-naped Sapsucker (*Sphyrapicus nuchalis*)	RNSA	1°	TD	11	0.5 ± 0.1	0	0	0.03
Downy Woodpecker (*Picoides pubescens*)	DOWO	1°	TD	4	0.4 ± 0.1	0	0	0.03
Hairy Woodpecker (*Picoides villosus*)	HAWO	1°	TD	15	1.5 ± 0.2	5	0.6 ± 0.1	0.07
American Three-toed Woodpecker (*Picoides dorsalis*)	TTWO	1°	TD	9	0.8 ± 0.1	0	0	< 0.01
Black-backed Woodpecker (*Picoides arcticus*)	BBWO	1°	TD	10	0.9 ± 0.2	0	0	< 0.01
Northern Flicker (*Colaptes auratus*)	NOFL	1°	GF	81	7.8 ± 0.9	51	4.9 ± 0.4	0.43
Tree Swallow (*Tachycineta bicolor*)	TRSW	2°	AI	27	2.8 ± 0.5	5	0.4 ± 0.1	0.05
Black-capped Chickadee (*Poecile atricapillus*)	BCCH	1°	FS	1	0.1 ± 0.0	1	0.1 ± 0.0	0.90
Mountain Chickadee (*Poecile gambeli*)	MOCH	1°	FS	10	0.8 ± 0.2	0	0	0.01
Red-breasted Nuthatch (*Sitta canadensis*)	RBNU	1°	TG	8	0.3 ± 0.1	0	0	0.03
White-breasted Nuthatch (*Sitta carolinensis*)	WBNU	1°	TG	6	0.4 ± 0.1	0	0	0.05
House Wren (*Troglodytes aedon*)	HOWR	2°	GF	126	11.5 ± 1.4	43	4.6 ± 0.4	0.24
Western Bluebird (*Sialia mexicana*)	WEBL	2°	GF	2	0.2 ± 0.1	0	0	0.18
Mountain Bluebird (*Sialia currucoides*)	MOBL	2°	GF	70	7.1 ± 0.5	29	2.6 ± 0.3	< 0.01
European Starling (*Sturnus vulgaris*)	EUST	2°	GF	17	2.1 ± 0.4	10	0.8 ± 0.2	0.52
Combined species totals				414	38.6 ± 3.1	149	14.6 ± 1.0	< 0.01

[a] 1° = primary cavity-nesting species, 2° = secondary cavity-nesting species (Thomas 1979).

[b] GF = ground-shrub forager, TD = timber driller, AI = aerial insectivore, FS = foliage searcher, TG = timber gleaner (Bock and Lynch 1970).

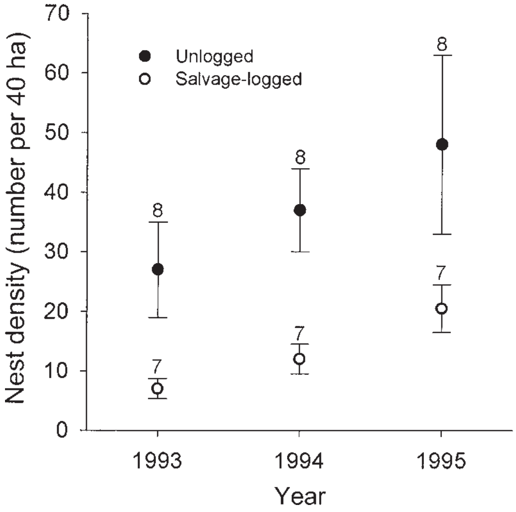

FIGURE 2. In each of the three years of study, mean nest density of all cavity-nesting species averaged over all plots was significantly higher in unlogged than in salvage-logged treatments in the Blackfoot-Clearwater Wildlife Management Area, Montana. Sample sizes appear above the error bars.

ern Flicker were the three species that most often used reused cavities for nesting. More than a quarter (27%) of Northern Flicker nests were located in previously used cavities. Although Northern Flickers may have excavated these cavities originally, it is clear they were not creating new nest cavities each breeding season.

There were 128 cavities known by observation to have been excavated by primary cavity-nesting birds. Of these, 58 were reused at least once during subsequent years, and half of those were Northern Flicker cavities being reused by flickers (Table 4). Despite the high frequency of reuse, Northern Flickers provided more cavities for other secondary cavity-nesting species than any other primary cavity-nesting species. Additionally, Northern Flicker cavities appeared to be the only cavities suitable for American Kestrels (*Falco sparverius*), possibly because flickers excavated relatively large entrance holes in relatively decayed snags. A small number of Red-naped (*Sphyrapicus nuchalis*) and Williamson's (*S. thyroideus*) Sapsuckers reused cavities excavated by Red-naped Sapsuckers (Table 4). Most of the Hairy (*Picoides villosus*), American Three-toed, and Black-backed Woodpecker cavities were reused by House Wrens and Mountain Bluebirds, which together used over a quarter of the 58 cavities originally made by primary cavity-nesting birds.

VEGETATION CHARACTERISTICS

Unlogged plots had significantly larger and taller trees, a higher density of trees, trees with more bark, higher live-tree density, and a lower proportion of intact snags than salvage-logged plots (Table 5). The burn severity surrounding randomly selected trees in unlogged plots was also slightly less than the burn severity surrounding randomly selected trees in logged plots (Table 5), although the index of severity that we used was undoubtedly affected by the removal of some green and brown-needled trees from the salvage-logged plots.

CHARACTERISTICS AND AVAILABILITY OF NEST TREES

Of the 12 species for which we found nine or more nests, all but one (Mountain Chickadee [*Poecile gambeli*]) nested in trees that were larger and taller than what was available in the salvage-logged plots; only the American Kestrel and Northern Flicker used larger trees than

TABLE 3. Numbers of new, reused (used in previous years by the same or different species), or repeat (used earlier in the same breeding season by the same or different species) nests across three years in unlogged and salvage-logged plots following fire in Montana. The proportion of cavities that were reused was higher in salvage-logged than in unlogged plots.

	Unlogged			Salvage-logged			
Year	New nests	Reused nests	Repeat nests	New nests	Reused nests	Repeat nests	Total
1993	96	N/A	N/A	26	N/A	N/A	122
1994	108	29	0	33	13	0	183
1995	108	68	5	46	26	5	258
Total	312	97	5	105	39	5	563

TABLE 4. Number of cavities first excavated by a given primary cavity-nesting bird species that were subsequently used by each of several primary and secondary cavity-nesting species in years following initial cavity excavation in Montana, 1993–1995. The cavities of seven species were reused by the same or by another species, and the Northern Flicker provided the greatest number of cavities that were reused.

Excavating species[a]	Reuse species[a]									Total reused	Total excavated	% reused	% reused by another species	No. cavities used all 3 yr
	MAKE	WISA	RNSA	NOFL	TRSW	WBNU	HOWR	MOBL	EUST					
LEWO				1						1	2	50	100	0
WISA				1	1					2	8	25	100	0
RNSA		2	2							4	11	36	50	0
HAWO				3			3	2		8	12	67	100	2
TTWO							1	4	1	6	10	60	100	2
BBWO				2		1	2	1		6	12	50	100	1
NOFL	3			22	2			2	2	31	73	42	29	7
Totals (percent)	3 (5)	2 (3)	2 (3)	29 (50)	3 (5)	1 (2)	6 (10)	9 (16)	3 (5)	58	128	48	35	12

[a] Species codes are given in Table 2.

what would be expected by chance in the unlogged plots (Fig. 3). Deciduous and broken-topped trees were also used for nesting by most species more often than would be expected due to chance (Fig. 4).

The percentage of randomly selected trees that were deemed suitable for nesting varied widely among the primary cavity-nesting species (Table 6). For some species, a single tree characteristic excluded a large number of randomly selected trees from being considered suitable. For example, Red-naped Sapsuckers nested in deciduous trees only, and because deciduous trees comprised a small percentage of random trees, the proportion of suitable nest trees for sapsuckers was close to zero in both unlogged and salvage-logged areas (Table 6). The estimated density of available nest trees ranged from zero for Red-naped Sapsuckers in salvage-logged plots, to 232 trees per ha for Hairy Woodpeckers in unlogged plots (Table 6).

Nest tree availability was not as easily assessed for secondary cavity-nesters because the presence of cavities was difficult to discern and because new cavities were undoubtedly created after we measured characteristics of randomly selected trees during the first year of our study. Nevertheless, cavities were present initially in only four of 244 randomly selected trees and only in the unlogged plots.

DISCUSSION

The densities of cavity nests in this study (14.6 and 38.6 nests per 40 hectares in salvage-logged and unlogged plots, respectively) are at the upper end of cavity nest densities that have been reported for other burned areas (Bock and Lynch 1970, Taylor and Barmore 1980, Caton 1996). Nevertheless, the higher abundance and species richness of cavity-nesting birds in unlogged compared to salvage-logged burned areas in this study is a result that is consistent with virtually all studies to date on this topic (Blake 1982, Harris 1982, Raphael and White 1984, Lyon and Marzluff 1985, Caton 1996, Saab and Dudley 1998, Haggard and Gaines 2001, Morissette et al. 2002). The single exception comes from a longer-term study in ponderosa pine-dominated communities in Idaho (V. Saab, R. Russell, and J. Dudley, USDA Forest Service, unpubl. data), where the overall density of cavity-nesting birds was observed to

TABLE 5. Vegetation characteristics associated with randomly selected trees in plots that were either unlogged (*n* = 132 trees) or salvage-logged (*n* = 112 trees) after a 1991 fire in the Blackfoot-Clearwater Wildlife Management Area, Montana. Tree sizes, tree densities, and percentage of live trees were significantly greater in unlogged than in salvage-logged plots. Means ± SE are given for numerical variables, and frequency distributions are given for categorical variables. Asterisk indicates a significant ($P < 0.05$) difference between unlogged and salvage-logged plots.

Vegetation characteristic	Unlogged	Salvage-logged
DBH (cm)*	30.7 ± 1.1	20.9 ± 0.9
Height (m)*	16.2 ± 0.4	12.7 ± 0.4
Bark (%)*	97 ± 1	94 ± 1
Tree species		
Pinus contorta	2%	0%
Pinus ponderosa	30%	19%
Larix occidentalis	10%	15%
Pseudotsuga mensiezii	53%	63%
Populus spp.	5%	3%
Tree status*		
Intact snag	74%	88%
Broken snag	5%	5%
Broken live	0%	1%
Intact live	21%	6%
Tree density (15 m radius)		
10–40 cm dbh*	23.7 ± 1.4	9.1 ± 0.8
>40 cm dbh*	1.6 ± 0.2	0.0 ± 0.0
Bare ground (%)	9 ± 1	8 ± 1
Ground cover (%)	33 ± 2	35 ± 2
Low shrub (%)	56 ± 2	55 ± 2
Tall shrub (%)	3 ± 3	2 ± 0
Burn severity*		
100% green	18%	2%
>60% green	5%	3%
40%–60% brown	25%	9%
>60% black	24%	25%
100% black	28%	62%

be slightly greater in salvage-logged plots in the first five years following fire, but not during the next five years. The combined density of all cavity-nesting birds increased over the three-year period of this study, primarily due to the creation of cavities by primary cavity-nesting species and the use of those same cavities in subsequent years by both primary and secondary cavity-nesters.

Of particular note is the fact that six of eight woodpecker species nested only in unlogged burned forest; they were entirely absent from salvage-logged areas. This is consistent with reports (Hutto 1995, Hutto and Young 1999) that, in the northern Rocky Mountains, two of these species (Black-backed and American Three-toed Woodpeckers) are not only more abundant in burned conifer forests than in any other vegetation type, but are also relatively restricted to such conditions. This result is also consistent with results from all of the previously cited studies on the effects of salvage logging on cavity-nesting birds. The number of Black-backed and American Three-toed Woodpecker nests decreased from the third to the fifth year after fire, which is similar to what Harris (1982), Caton (1996), Saab et al. (2004) and Saab et al. (V. Saab, R. Russell, and J. Dudley, USDA Forest Service, unpubl. data) found in their multiyear studies of postfire forests.

Preharvest tree density in salvage-logged areas (estimated by adding stump density to existing tree density) was 271 trees per ha, which was still well below the tree density in unlogged areas (360 trees per ha). It is therefore likely that salvage-logged and unlogged areas were somewhat different prior to the onset of fire and subsequent salvage-logging. There is always the possibility that some preharvest difference between plot types (e.g., a difference in average fire severity, slope, or aspect) might have contributed to the differences in bird densities between plot types; however, the presence of treatment replication in this study and the complete agreement between our results and the results of others who have looked at the effects of salvage logging strongly suggest that the significant reduction in abundance of cavity-nesting timber-drillers and timber-gleaners in salvage-logged areas was due primarily to the reduction in number of recently killed snags. This raises the question of whether it was a reduction in number of potential nest sites or some other aspect of habitat quality such as food availability that was affected most by the removal of standing dead trees.

Snags have traditionally been viewed as valuable for cavity-nesting birds exclusively in terms of their potential to be used as nest sites. Indeed, snag-management guidelines were originally based on meeting the nesting requirements of cavity-nesting birds (Thomas 1979). If nest trees were limiting bird densities in salvage-logged plots, then the density of suitable nest trees should have decreased to a number well below 6–8 trees per ha—the minimum number believed necessary to support maximum population densities of most cavity-nesting species

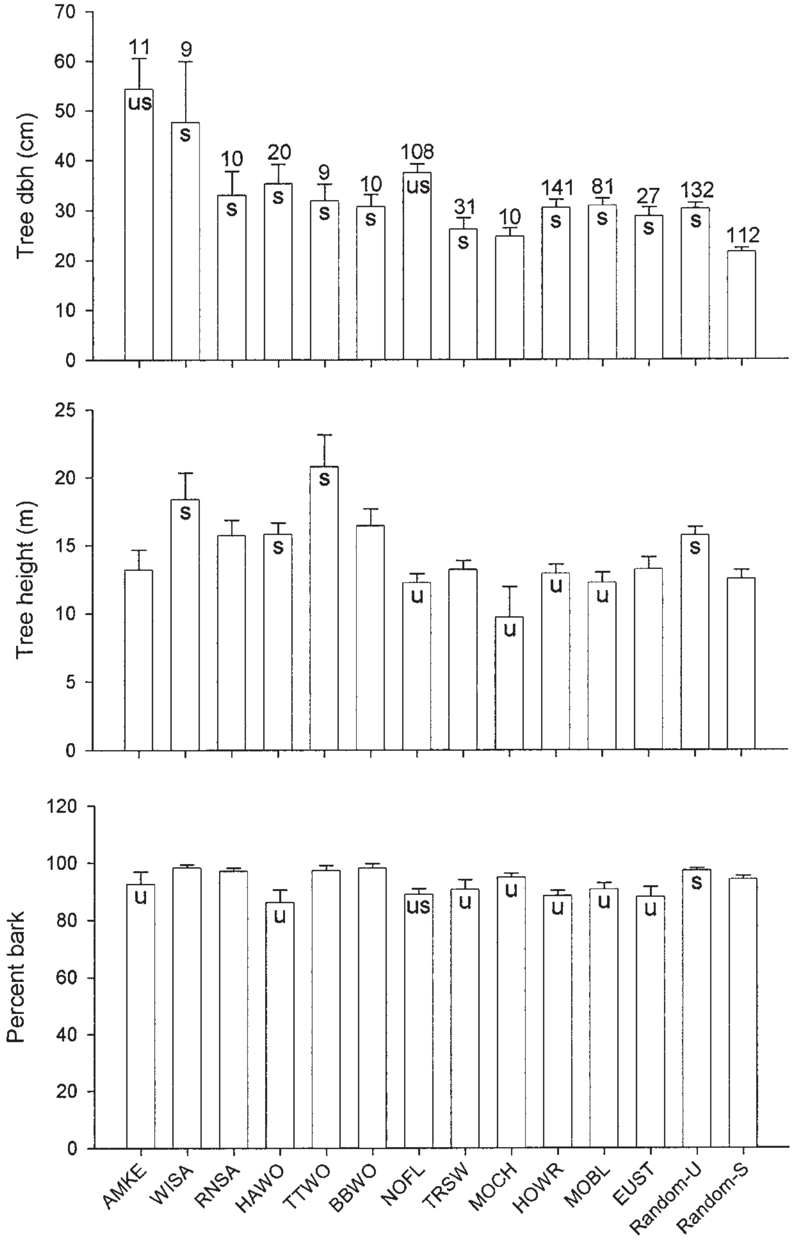

FIGURE 3. Mean nest tree characteristics (± SE) for cavity-nesting bird species (codes given in Table 2) and for randomly selected trees in unlogged (Random-U) and salvage-logged (Random-S) plots in the Blackfoot-Clearwater Wildlife Management Area, Montana. Nest trees generally differed significantly in one or more respects from randomly selected trees, especially in salvage-logged plots. Sample sizes are given above bars in the upper bar chart. Small letters indicate significant differences from randomly selected trees in salvage-logged (s) or unlogged (u) plots ($P < 0.05$, Mann-Whitney *U*-tests with Bonferonni correction for multiple comparisons).

(Thomas 1979:appendices 22, 23). We tested this hypothesis by calculating the percentage of randomly selected trees in both uncut and salvage-logged plots that could be considered suitable for nesting by each species. Most cavity-nesting species used snags rather than

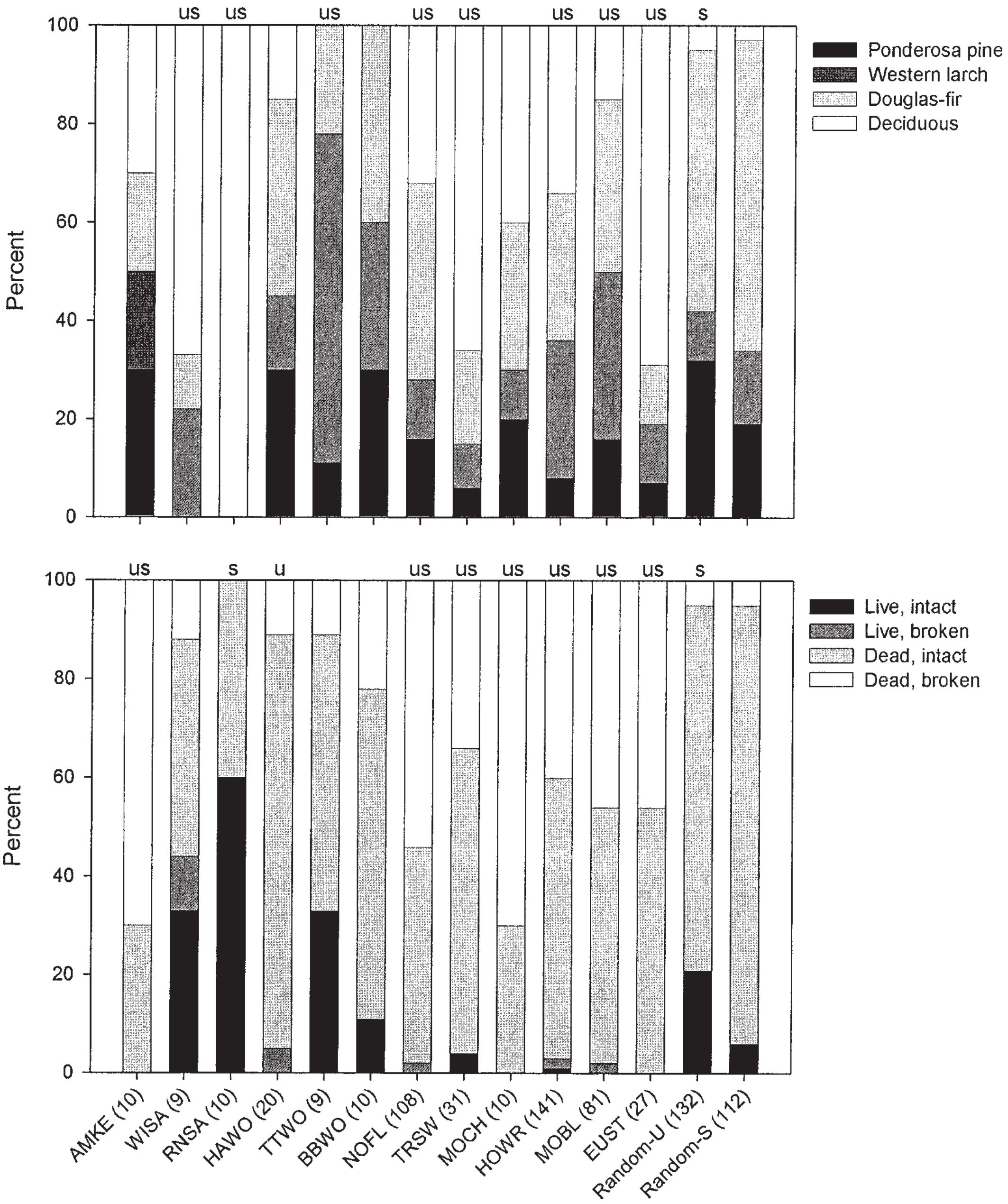

FIGURE 4. Frequency distributions of tree species and status of trees used by cavity-nesting bird species (codes given in Table 2) and of randomly selected trees in unlogged (Random-U) and salvage-logged (Random-S) plots. Most cavity-nesters used deciduous trees and broken-topped snags more than expected due to chance. Sample sizes are given at the bottom of the figure. Small letters indicate significant differences from randomly selected trees in salvage-logged (s) or unlogged (u) plots ($P < 0.05$, chi-square likelihood ratio with Bonferonni correction for multiple comparisons).

live trees for nesting, and several species used broken-topped snags more than expected based on their availability. All but one species (Mountain Chickadee) used nest trees that had significantly larger diameters than randomly selected trees in salvage-logged areas. The

TABLE 6. Percentages of randomly selected trees in unlogged (n = 132) and salvage-logged (n = 112) plots that were considered suitable for primary cavity-nesting species according to each of five different tree characteristics and all five characteristics combined in unlogged (U) and salvage-logged (S) plots in burned forest in Montana, 1993–1995. See Table 2 for Latin names of bird species. Only species with at least nine nests were included. The density of potentially suitable nest trees for each species was much greater in unlogged plots, and was greater than the minimum density needed to support maximum population densities of most cavity-nesting species in both unlogged and salvage-logged plots.

		Tree characteristic												Estimated number per ha	
		DBH		Height		Bark		Status		Species		Combined			
Species	*n*	U	S	U	S	U	S	U	S	U	S	U	S	U	S
Williamson's Sapsucker	9	67	21	93	94	95	92	99	99	68	81	39	10	142	14
Red-naped Sapsucker	10	60	26	67	70	95	92	95	94	5	3	3	0	11	0
American Three-toed Woodpecker	9	73	31	67	31	96	92	100	99	93	97	48	11	172	14
Black-backed Woodpecker	10	58	21	85	77	98	93	100	99	97	93	48	14	172	19
Hairy Woodpecker	20	99	87	85	77	98	96	78	93	100	100	64	63	232	84
Northern Flicker	108	91	61	99	100	79	93	99	99	100	100	62	53	224	71
Mountain Chickadee	10	52	38	80	96	95	92	99	100	78	93	36	41	128	55

selective removal of economically valuable larger-diameter trees clearly reduced the density and average size of snags in salvage-logged areas and, therefore, reduced the amount of suitable nesting habitat. Nevertheless, the decline in density of snags that might be used as nest sites by those bird species that were most negatively affected by logging still far exceeded the 6–8 snags per ha recommended as a minimum number needed to maintain maximum population densities.

These results highlight the fact that we need to appreciate snags as food resources as well as nest-site resources and that, for timber-drilling woodpecker species in particular, the number of snags needed to meet food resource needs appears to be much greater than the number needed to meet nesting requirements. Most woodpecker species in this study are relatively ''tree-dependent'' foragers because they forage on dead trees for beetle larvae. Burned snags are known to serve as sources of superabundant bark and wood-boring beetle larvae (Muona and Rutanen 1994, Rasmussen et al. 1996, Hart 1998), which are the primary food resource for tree-dependent foragers in burned forests (Powell et al. 2002, Nappi et al. 2003). Salvage logging undoubtedly reduces foraging opportunities for these birds, and may be the primary reason that overall habitat suitability declines for them (Caton 1996). The total disappearance of most timber-drilling and timber-gleaning birds in this study also underscores the need for a modification of current snag management guidelines (Hutto 2006) to better recognize the importance of retaining numerous snags as sources for food (beetle larvae) in burned forests.

In contrast with primary cavity-nesting birds, secondary cavity-nesters are often thought to be limited by the availability of nest sites (Brush 1983, Brawn and Balda 1988, Walankiewicz 1991). Because two-thirds of the cavities we discovered were not reused during the three years of this study, either something other than nest sites limited the density of secondary cavity-nesting birds, or many of the apparently suitable nest cavities were not really suitable at all (see also Welsh and Capen 1992). Other animals (e.g., northern flying squirrels [*Glaucomys sabrinus*] and red squirrels [*Tamiasciurus hudsonicus*]) were observed using previously active cavities, and they are known to displace secondary cavity-nesting birds or depredate their nests. The location of cavities near other nesting birds of the same or different species may also make cavities unsuitable for nesting. For example, birds may avoid nesting near American Kestrel nests because they (or their nestlings) might experience higher rates of predation. Indeed, American Kestrels were observed to prey on other secondary cavity-nesting species in this study. Other predators known to occur in the study area, and that could have affected the suitability of cavities for secondary cavity-nesters, include striped skunks (*Mephitis mephitis*), black bears (*Ursus americana*), and weasels (*Mustela* spp.). Other factors such as a lack of foraging opportunities could also limit secondary cavity-nesting bird densities in the face of what appear to be abundant nesting opportunities.

Assuming that secondary cavity-nesters are limited by nesting opportunities, it is noteworthy that the frequency of cavity reuse was higher in salvage-logged than in unlogged plots, possibly reflecting a greater level of nest-site limitation in the salvage-logged areas. More than 80% of the nests of secondary cavity-nesters were located in cavities excavated by woodpeckers, indicating a strong reliance on primary cavity-nesting birds for cavity excavation (see also Dobkin et al. 1995). In unlogged areas, the continuous creation of roosting and nesting cavities by primary cavity-nesting species may provide abundant new cavities for secondary cavity-nesting birds to use. In contrast, fewer breeding primary cavity-nesters in salvage-logged areas create fewer new cavities, and this may force secondary cavity-nesting birds to reuse a smaller number of older cavities, which could also affect their nest success in salvage-logged forests.

In conclusion, the cavity-nesting bird community as a whole was clearly negatively affected by salvage logging in our study area in the early years following stand-replacing fire, and the main problem for primary cavity-nesting bird species does not appear to have been availability of snags as nest sites. Rather, the reduction in numbers of trees that harbored important food resources seemed to be to the detriment of the most fire-dependent cavity-nesting bird species. Especially noteworthy was the absence of several postfire specialist woodpecker species from salvage-logged areas and the general decrease in nesting densities of timber-drilling and timber-gleaning species compared to their densities in unlogged burned areas. These results are especially important in light of recent modifications to legislation and regulations by provincial governments in Canada (cited in Nappi et al. 2003) and by the U. S. government (Healthy Forests Restoration Act of 2003) to expedite and provide incentives for salvage logging on top of already documented increases in dead tree harvest in the West (Duncan 2002). Because early postfire cavity-nesting bird communities are biologically unique, and because we lose the most fire-dependent elements of that community through salvage harvests such as the ones included in this study, the merit of postfire salvage logging should be reexamined, especially when there are other, less ecologically sensitive options available for timber harvest. Although research on the effects of different levels and styles of salvage logging are urgently needed, the findings that have already emerged from this and from other studies (McIver and Starr 2000, Kotliar et al. 2002, Beschta et al. 2004, Karr et al. 2004) are entirely consistent, and they suggest that it may be difficult to retain the ecological integrity of a burned forest in the face of most kinds of postfire salvage logging.

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LITERATURE CITED

ANDERSEN, A. N., R. W. BRAITHWAITE, G. D. COOK, L. K. CORBETT, R. J. WILLIAMS, M. M. DOUGLAS, A. M. GILL, S. A. SETTERFIELD, AND W. J. MULLER. 1998. Fire research for conservation management in tropical savannas: introducing the Kapalga fire experiment. Australian Journal of Ecology 23:95–110.

BESCHTA, R. L., J. J. RHODES, J. B. KAUFFMAN, R. E. GRESSWELL, G. W. MINSHALL, J. R. KARR, D. A. PERRY, F. R. HAUER, AND C. A. FRISSELL. 2004. Postfire management on forested public lands of the western United States. Conservation Biology 18:957–967.

BLAKE, J. G. 1982. Influence of fire and logging on nonbreeding bird communities of ponderosa pine forests. Journal of Wildlife Management 46:404–415.

BOCK, C. E., AND J. F. LYNCH. 1970. Breeding bird populations of burned and unburned conifer forest in the Sierra Nevada. Condor 72:182–189.

BRAWN, J. D., AND R. P. BALDA. 1988. Population biology of cavity nesters in northern Arizona: do nest sites limit breeding densities? Condor 90:61–71.

BRUSH, T. 1983. Cavity use by secondary cavity-nesting birds and response to manipulations. Condor 85:461–466.

CATON, E. L. 1996. Effects of fire and salvage logging on the cavity-nesting bird community in northwestern Montana. M.Sc. thesis, University of Montana, Missoula, MT.

DICKSON, J. G., R. N. CONNER, AND J. H. WILLIAMSON. 1983. Snag retention increases birds in a clearcut. Journal of Wildlife Management 47:799–809.

DOBKIN, D. S., A. C. RICH, J. A. PRETARE, AND W. H. PYLE. 1995. Nest-site relationships among cavity-nesting birds of riparian and snowpocket aspen woodlands in the northwestern Great Basin. Condor 97:694–707.

DUNCAN, S. 2002. Postfire logging: is it beneficial to a forest? Science Findings 47:1–5.

GRUELL, G. E. 1983. Fire and vegetative trends in the northern Rockies: interpretations from 1871–1982 photographs. USDA Forest Service General Technical Report INT-158.

HAGGARD, M., AND W. L. GAINES. 2001. Effects of stand-replacement fire and salvage logging on a cavity-nesting bird community in eastern Cascades, Washington. Northwest Science 75: 387–396.

HARGROVE, W. W., AND J. PICKERING. 1992. Pseudoreplication: a *sine qua non* for regional ecology. Landscape Ecology 6:251–258.

HARRIS, M. A. 1982. Habitat use among woodpeckers in forest burns. M.Sc. thesis, University of Montana, Missoula, MT.

HART, S. 1998. Beetle mania: an attraction to fire. BioScience 48:3–5.

HOBSON, K. A., AND J. SCHIECK. 1999. Changes in bird communities in boreal mixedwood forest: harvest and wildfire effects over 30 years. Ecological Applications 9:849–863.

HUTTO, R. L. 1995. The 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 20:984–993.

HUTTO, R. L., AND J. S. YOUNG. 1999. Habitat relationships of landbirds in the Northern Region, USDA Forest Service. USDA Forest Service General Technical Report RMRS-GTR-32.

KARR, J. R., J. J. RHODES, G. W. MINSHALL, F. R. HAUER, R. L. BESCHTA, C. A. FRISSELL, AND D. A. PERRY. 2004. The effects of postfire salvage logging on aquatic ecosystems in the American West. BioScience 54:1029–1033.

KOTLIAR, N. B., S. J. HEJL, R. L. HUTTO, V. A. SAAB, C. P. MELCHER, AND M. E. MCFADZEN. 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.

LINDENMAYER, D. B., D. R. FOSTER, J. F. FRANKLIN, M. L. HUNTER, R. F. NOSS, F. A. SCHMIEGELOW, AND D. PERRY. 2004. Salvage harvesting policies after natural disturbance. Science 303:1303.

LYON, L. J., AND J. M. MARZLUFF. 1985. Fire's effects on a small bird population, p. 16–22. *In* J. E. Lotan and J. K. Brown [EDS.], Fire's effects on wildlife habitat–symposium proceedings. USDA Forest Service General Technical Report INT-186.

MARCOT, B. G. 1983. Snag use by birds in Douglas-fir clearcuts, p. 134–139. *In* J. W. Davis, G. A. Goodwin, and R. A. Ockenfels [EDS.], Snag habitat management: proceeding of the symposium. USDA Forest Service General Technical Report RM-99.

MCIVER, J. D., AND L. STARR. 2000. Environmental effects of postfire logging: literature review and annotated bibliography. USDA Forest Service General Technical Report PNW-GTR-486.

MORISSETTE, J. L., T. P. COBB, R. M. BRIGHAM, AND P. C. JAMES. 2002. The response of boreal forest songbird communities to fire and post-fire harvesting. Canadian Journal of Forest Research 32:2169–2183.

MORRISON, M. L., AND M. G. RAPHAEL. 1993. Modeling the dynamics of snags. Ecological Applications 3:322–330.

MUONA, J., AND I. RUTANEN. 1994. The short-term impact of fire on the beetle fauna in boreal coniferous forest. Annales Zoologici Fennici 31:109–121.

NAPPI, A. 2000. Distribution des pics et utilisation des arbres morts dans les forêts soumises aux perturbations naturelles en pessière à mousses. M.Sc. thesis, Université du Québec, Montreal, Quebec.

NAPPI, A., P. DRAPEAU, J.-F. GIROUX, AND J.-P. SAVARD. 2003. Snag use by foraging Black-backed Woodpeckers (*Picoides arcticus*) in a recently burned eastern boreal forest. Auk 120: 505–511.

NAPPI, A., P. DRAPEAU, AND J.-P. L. SAVARD. 2004. Salvage logging after wildfire in the boreal forest: is it becoming a hot issue for wildlife? Forestry Chronicle 80:67–74.

PIERCE, J. L., G. A. MEYER, AND A. J. T. JULL. 2004. Fire-induced erosion and millennial-scale climate change in northern ponderosa pine forests. Nature 432:87–90.

POWELL, H. 2000. The influence of prey density on post-fire habitat use of the Black-backed Woodpecker. M.Sc. thesis, University of Montana, Missoula, MT.

POWELL, H. D. W., S. J. HEJL, AND D. L. SIX. 2002. Measuring woodpecker food: a simple method for comparing wood-boring beetle abundance among fire-killed trees. Journal of Field Ornithology 73:130–140.

RAPHAEL, M. G., AND M. WHITE. 1984. Use of snags by cavity-nesting birds in the Sierra Nevada. Wildlife Monographs 86.

RASMUSSEN, L. A., G. D. AMMAN, J. C. VANDYGRIFF, R. D. OAKES, A. S. MUNSON, AND K. E.

GIBSON. 1996. Bark beetle and wood borer infestation in the Greater Yellowstone area during four postfire years. USDA Forest Service Research Paper INT-RP-487.

SAAB, V. A., J. DUDLEY, AND W. L. THOMPSON. 2004. Factors influencing occupancy of nest cavities in recently burned forests. Condor 106:20–36.

SAAB, V. A., AND J. G. DUDLEY. 1998. Responses of cavity-nesting birds to stand-replacement fire and salvage logging in ponderosa pine/Douglas-fir forests of southwestern Idaho. USDA Forest Service Research Paper RMRS-RP-11.

SCHREIBER, B., AND D. S. DECALESTA. 1992. The relationship between cavity-nesting birds and snags on clearcuts in western Oregon. Forest Ecology and Management 50:299–316.

SCOTT, V. E., AND J. L. OLDEMEYER. 1983. Cavity-nesting bird requirements and responses to snag cutting, p. 19–23. *In* J. W. Davis, G. A. Goodwin, and R. A. Ockenfels [EDS.], Snag habitat management: proceeding of the symposium. USDA Forest Service General Technical Report RM-99.

SMUCKER, K. M., R. L. HUTTO, AND B. M. STEELE. 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecological Applications 15:1535–1549.

TAYLOR, C. 1995. The politics of salvage timber. Journal of Forestry 93:60.

TAYLOR, D. L., AND W. J. BARMORE. 1980. Post-fire succession of avifauna in coniferous forests of Yellowstone and Grand Teton National Parks, Wyoming, p. 130–145. *In* R. M. DeGraaf [ED.], Workshop proceedings: management of western forests and grasslands for nongame birds. USDA Forest Service General Technical Report INT-86.

THOMAS, J. W. 1979. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Agriculture Handbook No. 553, USDA Forest Service, Washington, DC.

TURNER, M. G., AND W. H. ROMME. 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology 9:59–77.

TURNER, M. G., W. H. ROMME, AND D. B. TINKER. 2003. Surprises and lessons from the 1988 Yellowstone fires. Frontiers in Ecology and the Environment 1:351–358.

VAN MANTGEM, P., M. SCHWARTZ, AND M. B. KEIFER. 2001. Monitoring fire effects for managed burns and wildfires: coming to terms with pseudoreplication. Natural Areas Journal 21: 266–273.

WALANKIEWICZ, W. 1991. Do secondary cavity-nesting birds suffer more from competition for cavities or from predation in a primeval deciduous forest? Natural Areas Journal 11:203–212.

WELSH, C. J. E., AND D. E. CAPEN. 1992. Availability of nesting sites as a limit to woodpecker populations. Forest Ecology and Management 48:31–41.

WHELAN, R. J. 1995. The ecology of fire. Cambridge University Press, New York.

ZARNOWITZ, J. E., AND D. A. MANUWAL. 1985. The effects of forest management on cavity-nesting birds in northwestern Washington. Journal of Wildlife Management 49:255–263.

esa ECOSPHERE

Toward a more ecologically informed view of severe forest fires

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Abstract. We use the historical presence of high-severity fire patches in mixed-conifer forests of the western United States to make several points that we hope will encourage development of a more ecologically informed view of severe wildland fire effects. First, many plant and animal species use, and have sometimes evolved to depend on, severely burned forest conditions for their persistence. Second, evidence from fire history studies also suggests that a complex mosaic of severely burned conifer patches was common historically in the West. Third, to maintain ecological integrity in forests born of mixed-severity fire, land managers will have to accept some severe fire and maintain the integrity of its aftermath. Lastly, public education messages surrounding fire could be modified so that people better understand and support management designed to maintain ecologically appropriate sizes and distributions of severe fire and the complex early-seral forest conditions it creates.

Key words: early succession; ecological integrity; ecological system; fire management; fire regime; forest resilience; forest restoration; severe fire; wildfire.

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INTRODUCTION

The spatiotemporal expression of fire events over time in any landscape produces a "fire regime" that influences ecosystem dynamics in that area (Heinselman 1981, Kilgore 1981). Even though the various characteristics of a fire regime (Table 1) are continuous in nature, the traditional approach in representing this variation has been to create a small number of discontinuous categories. Fire regimes in western North America, for example, are often classified into as few as three categories: (1) low-severity, (2) mixed-severity, and (3) high-severity or stand-replacement (Agee 1998, Brown 2000). Our attempt to categorize fire regimes is ". . . an oversimplification…for the convenience of humans" (Sugihara et al. 2006; p. 62), and has had the unfortunate consequence of minimizing rather than emphasizing variation in fire behavior and fire outcomes among vegetation types and across spatial scales (Morgan et al. 2014). In reality, relatively few forest types fit entirely within either of the two extremes—the low-severity (e.g., some interior ponderosa pine) or the stand-replacement (e.g., Rocky Mountain lodgepole pine) categories. Instead, as a simple analysis

Table 1. Characteristics or descriptors often used to describe disturbance regimes (from Keane 2013).

Disturbance Characteristic	Description	Example
Agent	Factor causing the disturbance	Fire is an agent that can kill trees
Source, Cause	Origin of the agent	Lightning is a source for wildland fire
Frequency	How often the disturbance occurs or its return time	Years since last fire (scale dependent)
Intensity	A description of the magnitude of the disturbance agent	Wildland fire heat output
Severity	The level of impact of the disturbance on the environment	Fuel consumption in wildland fires; change in biomass
Size	Spatial extent of the disturbance	Tree kill can occur in small patches or across entire landscapes
Pattern	Patch size distribution of disturbance effects; spatial heterogeneity of disturbance effects	Fire can burn large regions but weather and fuels can influence fire intensity and therefore the patchwork of tree mortality
Seasonality	Time of year of that disturbance occurs	Spring burn vs. fall burn
Duration	Length of time of that disturbances occur	Fires can burn for a day or for an entire summer
Interactions	Disturbance types may interact with each other, or with climate, vegetation and other landscape characteristics	Mountain pine beetles may create fuel complexes that facilitate or exclude wildland fire
Variability	The spatial and temporal variability of the above factors	Each of the above characteristics has variation associated with it

using LANDFIRE data (Rollins 2009, <http://www.landfire.gov>) reveals, roughly 85% of all forested lands within the western US fit within the mixed-severity category, which includes proportions of low-, moderate-, and high-severity (lethal to more than 70% of all trees) fire that vary widely across vegetation types and biophysical settings.

Agee (1993) captured the essence of this important idea in a graph depicting the proportion of low-, moderate-, and high-severity fire across the range of fire regimes (Fig. 1). Note that change from one fire regime to the next (movement along the x-axis) is accompanied not by the sudden appearance of a different fire severity, but by continuous changes in the proportions of each fire severity category. Thus, fire regimes blend imperceptibly into one another. More importantly, except for the two end points on the graph where the proportion of high-severity fire would be either 0% or 100%, most fire regimes consist of a mix of fire severities so, technically speaking, they fit best within a mixed-severity regime (Fig. 2). It is not the presence of a particular fire severity, but the proportion (and, presumably, the distribution and patch sizes) of each severity component that distinguishes regimes. Indeed, empirical data drawn from recent fires across the western United States between 1984 and 2008 (Fig. 3) reveal this continuous variation in proportions of different fire severities among fires. Thus, a more continuous view of fire regimes might be a better way to appreciate the infinite variability in fire behavior among forest types and geographic locations, and it might also promote a greater appreciation of severe fire as an integral

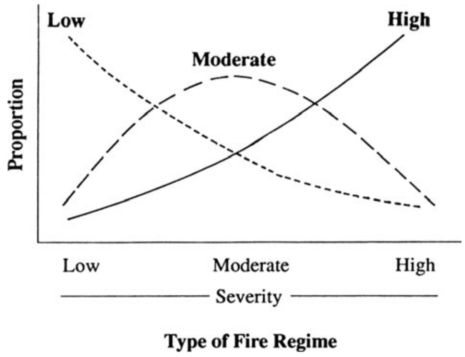

Fig. 1. This graph (from Agee 1993) illustrates that fire regimes are not characterized by the presence of only one kind of fire. Rather, it is the relative frequency of low-, moderate-, and high-severity fire in an average burn that varies among fire regimes.

Fig. 2. Mixed-severity fires (fires that leave recognizable patches of low-severity, medium-severity, and high-severity effects) typify the majority of mixed-conifer forest systems in the western United States. The brown-needled and blackened areas harbor unique sets of plant and animal species found in no other forest conditions. This photograph of the North Fork of the Blackfoot River was taken 10 months after the 1988 Canyon Creek fire in Montana. Many fire-dependent plant and animal species were present in the more severely burned areas until they were helicopter logged, suggesting that unburned forests might be a better alternative for timber harvest.

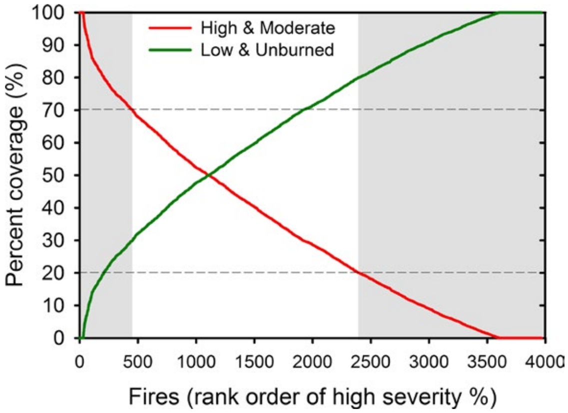

Fig. 3. The percent area within a fire perimeter that burned at low (green line) and at moderate to high (red line) severity is shown for a series of 3696 fires that burned in the western United States between 1984 and 2008 (after Belote 2015). The figure shows that the proportions of each severity category are continuously variable and that high-severity fire is a natural part of most forest fires in the West.

part of mixed- and high-severity conifer forest fire regimes.

Accordingly, we highlight the need for better information on the historical patterns and abundances of high-severity patches in different forest types. This is an important discussion because, even though our National Cohesive Wildland Fire Management Strategy (Wildland Fire Executive Council 2014) acknowledges that many fire regimes exist and that management needs to accommodate that variation and the variety of habitat such variation produces, contemporary fire management is focused heavily on the exclusion (prevention and suppression, collectively) or mitigation of severe fire. When either of those fails, management efforts seem to shift toward speeding the "recovery" of the forest after severe fire. With respect to the latter, there are repeated attempts to introduce legislation designed to expedite logging after fire (salvage logging). Although the removal of dead trees is justified near roads and structures for safety reasons, and although postfire logging can capture economic value of wood that would otherwise be lost, such logging has been shown to carry significant ecological costs (Hutto 2006, Lindenmayer and Noss 2006, Swanson et al. 2011, Lindenmayer and Cunningham 2013, DellaSala et al. 2015). The ecological benefits and necessity of severe fire (and its aftermath) has widespread implications for the flora and fauna that depend on the presence of burned forest conditions. Ecologically sound fire management includes land management designed to ensure the maintenance of ecologically appropriate mixes of fire severities within the forested landscapes of western North America while protecting homes and lives at the same time (Perry et al. 2011). An ecologically informed view of severe fire requires recognition that it is a natural component of many western conifer forests (Heinselman 1981, Arno 2000). Moreover, the severe-fire component must have been large enough and frequent enough to have favored the evolution of specialization by various plant and animal species to conditions that occur in the aftermath of severe fire. We offer the following points in an effort to better recognize and include severe fire as an integral part of fire management in mixed-conifer forest systems:

SEVERELY BURNED FORESTS CREATE BIOLOGICALLY UNIQUE CONDITIONS THAT CANNOT BE CREATED BY OTHER KINDS OF DISTURBANCES OR THROUGH ARTIFICIAL MEANS

Patterns in the habitat associations of plant and animal species can provide definitive evidence that severe fire plays an essential role in the ecology of mixed-conifer forests (Hutto et al. 2008). Specifically, if a plant or animal species occurs only in burned forest conditions created by severe fire events, then it cannot be using burned forest conditions merely opportunistically. Instead, the species must have evolved to depend on such conditions because it occurs rarely, if ever, in unburned habitat (Swanson et al. 2011, DellaSala et al. 2014). For example, some moss and lichen species are relatively restricted to severely burned forest conditions (Ahlgren and Ahlgren 1960), as are the fire morel mushroom (*Morchella elata*) and Bicknell's geranium (*Geranium bicknellii*) in forests throughout the West (Heinselman 1981, Pilz et al. 2004). The black-backed woodpecker (*Picoides arcticus*) is emblematic of a species that is relatively restricted to early successional conditions created by high-severity fire (Hutto 1995, Dixon and Saab 2000, Hoyt and Hannon 2002). Black-backed woodpeckers are attracted to postwildfire conditions because of the abundance of larvae of a number of wood-boring beetle species that are attracted to the fire-killed trees (Murphy and Lehnhausen 1998, Rota et al. 2015). Several of these beetle species are themselves relatively restricted to recently burned forests (Saint-Germain et al. 2004*a,b*, Boucher et al. 2012). Importantly, black-backed woodpeckers are significantly more likely to occur in the more severely burned portions of a mixed-severity fire (Hutto 2008, Latif et al. 2013). Although black-backed woodpeckers are known to occur outside severely burned forests on rare occasions, detailed study of survival and reproductive success shows that they exhibit growing populations only in forests recently burned by summer wildfires (Rota et al. 2014). The adaptations of thick bark, branch shedding, and serotiny in *Pinus* are thought to have evolved in response to a period of more intense crown fires in the mid-Cretaceous (He et al. 2012), and those adaptations also reflect the severe-fire backdrop against which pine, Douglas-fir, and larch are thought to thrive.

Many additional animal species, while not as narrowly restricted to burned forest conditions, clearly benefit from the burned forest conditions created by severe fires in mixed-conifer forests throughout the West (Hutto et al. 2015). For example, nest survival of white-headed woodpeckers is significantly higher in burned (wildfire) compared to unburned forest (Hollenbeck et al. 2011, Lorenz et al. 2015). In aquatic systems, severe fire events can rejuvenate stream habitats by causing large amounts of gravel, cobble, woody debris, and nutrients to be imported, resulting in increased production and aquatic insect emergence rates (Benda et al. 2003, Burton 2005, Malison and Baxter 2010, Ryan et al. 2011, Jackson et al. 2015). These changes can, in turn, affect food web dynamics in a way that results in higher growth rates in young trout, including young coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) (Heck 2007) and rainbow trout (*Oncorhynchus mykiss*) (Rosenberger et al. 2011). Indeed, nonnative fish populations declined and native trout densities increased 3 yr after a severe fire in the Bitterroot River watershed, Montana, indicating that severe fire may help ensure ecological integrity of some western streams (Sestrich et al. 2011). In addition, native amphibians such as boreal toads (*Bufo boreas*) thrive in areas that burn severely (Dunham et al. 2007, Hossack and Corn 2007) and use severely burned areas more than expected due to chance (Hossack and Corn 2007, Guscio et al. 2008), as do some bat species (Buchalski et al. 2013).

These strong associations between organisms and severely burned forest patches suggests that many plant and animal species have evolved to rely on recurring severe wildfire events, and further indicates that severe fire events are a natural and important part of the fire regimes associated with many western mixed-conifer forest types. In other words, if one or more species occupy severely burned forests to the exclusion of other forest types (and if they do not tend to occupy forests disturbed through artificial means), then a severely burned forest would have to be considered natural, and would necessarily lie within the historical range of variation (Hutto et al. 2008). Moreover, a more intimate understanding of the biology of those plants and animals (e.g., knowledge of dispersal processes and patterns, foraging ecology, home-range sizes) can provide insight into the historical spatial scales at which severe fire operated across the broader landscape.

FIRE HISTORY STUDIES SUGGEST THAT SEVERE FIRE IS AN INTEGRAL COMPONENT OF MOST FIRE REGIMES

In addition to the definitive evidence provided above, a growing body of fire history information points to the same conclusion—severe fire was historically, and is currently, an important component of many western conifer forest systems. At one end of the fire regime spectrum, conifer forests in the warmer, drier geographic areas in western North America are commonly characterized by frequent, low-severity fires that killed primarily juvenile trees historically, resulting in the maintenance of open pine forests with low densities of mature trees (Covington and Moore 1994*a,b*). Nevertheless, mixed and stand-replacement fires were possible even in these forest types after long inter-fire intervals, such as after an especially cold, wet period similar to what occurred during the Little Ice Age (Brown et al. 1999, Sherriff and Veblen 2007, Williams and Baker 2012, Odion et al. 2014, Hanson et al. 2015). At the other end of the fire regime spectrum, cooler, moister forest types, such as lodgepole pine forests, support fire regimes dominated by severe fire events (Brown and Smith 2000), although mixed- and low-severity fires are known to occur in these types as well (Barrett et al. 1991).

Between these two extremes lie the vast majority of mixed-conifer forest types in western North America. These include everything from the xeric, low-elevation, mixed ponderosa pine and Douglas-fir forest types to mesic, high-elevation, spruce-fir forest types. Unlike the forest types that are dominated by either the absence or presence of severe fire, mixed-conifer forests are best characterized by fire regimes of variable, or mixed severity (see Baker 2009: fig. 7.1), which means that the presence of sizable proportions of the three classes of fire severity characterize the fires that burn in those forest systems (Sherriff and Veblen 2006, 2007, Baker et al. 2007, Hessburg et al. 2007, Klenner et al. 2008, Perry et al. 2011, Schoennagel et al. 2011). Importantly, extreme weather (e.g., high temperature, low humidity, high wind speed) rather than quantity of woody fuels often exerts the greatest influence on fire severity and extent across that broad range of mixed-conifer forest types (Johnson et al. 2003, Schoennagel et al. 2004, Lydersen et al. 2014, Williams et al. 2015). This means that, in contrast with the situation in low-elevation or xeric-type ponderosa pine forests in some areas of the southwestern United States (Keane et al. 2008), the amount of high-severity fire in other mixed-conifer forest types is less likely to have departed significantly from historical ranges of variability, even though those forests may have experienced measurable twentieth century changes in fuels due to fire exclusion, timber harvest, and cattle grazing (e.g., Baker et al. 2007, Dillon et al. 2011, Marlon et al. 2012, Miller et al. 2012, Odion et al. 2014, Sherriff et al. 2014). We recognize the lack of relevant historical information on landscape-level distributions and spatial scales of different classes of fire severity for many forest types and regions, but severely burned forest patches have probably always occurred naturally, even in pure ponderosa pine forests of the Southwest, as Cooper (1961) and Weaver (1943) described long ago. We also know that, at least throughout the northern half of the western United States, the extent of severe-fire patches must have been both substantial enough in area and frequent enough to support those plant (e.g., lodgepole pine) and animal (e.g., wood-boring beetle and woodpecker) species that evolved to depend on severe fire itself or on the resulting severely burned forest conditions.

MAINTAINING ECOLOGICAL INTEGRITY MEANS ACCOMMODATING A BROAD SPECTRUM OF FIRE SEVERITIES, INCLUDING SEVERE FIRE AND ITS AFTERMATH, IN MOST MIXED-CONIFER FORESTS

We have now established two important facts: severe fire (moderate-to-high burn severity) is a natural agent of disturbance in many mixed-conifer forest types, and such fire is thought to be ecologically necessary for the presence or success of many plant and animal species. These two facts make it clear that management to maintain the ecological integrity of any ecosystem that harbors species that depend on severe fire as a disturbance agent will have to integrate severe fire and its effects into management goals. Moreover, if we better considered distribution patterns, home range sizes, movement patterns, and other animal adaptations that reflect the environment within which they evolved (e.g., Hutto et al. 2008), we could gain considerable insight into historical spatial scales under which severe fire operated as well. We are not questioning or attempting to discredit the evidence that some forest systems were historically dominated by low-severity fire; rather, we are encouraging land managers to also pay close attention to maintaining amounts and distributions of higher severity fire consistent with ecological integrity in our western mixed-conifer forests. The current science, management, and policy challenge for ecosystem managers is to estimate and incorporate amounts of low-, moderate-, and high-severity fire in a manner that maintains ecological integrity (Hessburg et al. 2007, Perry et al. 2011, Baker 2015).

While many fire ecologists understand the importance of more severe fire in forest ecosystems, politicians and the public at large have yet to reach the same understanding. Recent increases in the amount of forested area burned by wildfire over the past three decades in western North American forests (Westerling et al. 2006, Dennison et al. 2014) signaling what many believe to be the emergence of a new age of megafires (Attiwill and Binkley 2013), has created increased movement toward pre and postfire land management activities designed to reduce fire severity, mimic fire effects without the use of fire, or speed the recovery of a forest after fire. These activities may provide some societal benefits, but they can have real costs in terms of the way they negatively affect the ecological integrity of mixed-conifer forests born of mixed-severity fire. Removed from locations that pose a clear and immediate threat to human lives and property, the ecological costs associated with forest thinning may outweigh stated benefits by large margins. We highlight two types of land management (beyond fire suppression itself) that can have significant negative effects on fire-dependent species and, therefore, can interfere with our ability to maintain the ecological integrity of fire-dependent conifer forests: prefire fuel treatments and postfire salvage logging.

Prefire harvest treatments

We know a great deal about the effects of fuel treatments and restoration harvests on forest structure and vegetation recovery, but we know little about the ecological effects of such treatments on the prefire responses of most plant and animal species, and virtually nothing about postfire responses of the most fire-dependent plant and animal species after a treatment subsequently burns in a wildfire. This is because such treatments are rarely accompanied by "ecological effects monitoring," which, in contrast with implementation monitoring (evaluating whether a management activity was implemented) and effectiveness monitoring (evaluating whether the management activity achieved the stated goal), is specifically designed to address whether there are unforeseen negative ecological consequences of a management treatment (Hutto and Belote 2013).

Fuel treatments designed to restore fire-prone ecosystems should do so in the proper fire regime context; more specifically, they should produce appropriate postfire plant and animal responses when fire returns to the forest. Thus, treatments appropriate for dry forests that were historically maintained by a low-severity fire regime may be inappropriate for forests maintained by a mixed-severity fire regime. One serious negative consequence of canopy fuel reduction in forests that evolved with mixed-severity fire could be that fire-dependent species requiring high densities of large standing-dead trees created by the severe-fire component may not recruit after a subsequent fire. For example, the fire-dependent black-backed woodpecker was found to be even less abundant in mixed-conifer forests that were thinned before fire than in the same forest types logged after fire, even though the two pathways support similar standing dead tree densities. This is probably because birds rarely colonize thinned forests that burn, but they still make the best of a bad situation when trees are removed after they have already colonized a densely stocked, severely burned forest (Hutto 2008). Recent research on postfire soil conditions shows that soil C and N response following wildfire also depends on whether there have been fuel treatments, so the assessment of fuel treatment effects needs to include postfire response and not simply postharvest response (Homann et al. 2015). It has been suggested (e.g., Franklin and Johnson 2014) that variable-retention harvests could be designed to emulate early-seral conditions following natural disturbance events in forests born of mixed-severity fire, thereby avoiding the negative consequences associated with other tree harvesting methods. Unfortunately, that strategy is unlikely to satisfy the needs of those fire-dependent animal species that require high densities of fire-killed trees immediately following severe fire (Schieck and Song 2006, Hutto 2008, Reidy et al. 2014).

Postfire salvage logging

Salvage logging after fire is intended to recover economic value of timber that would otherwise be lost, to ensure human safety, and to reduce the risk of future fires. Unfortunately, salvage harvesting activities undermine the ecosystem benefits associated with fire (Lindenmayer et al. 2004, Lindenmayer and Noss 2006, Swanson et al. 2011). For example, postfire salvage logging removes dead, dying, or weakened trees, but those are precisely the resources that provide nest sites and an abundance of food in the form of beetle larvae and bark surface insects (Hutto and Gallo 2006, Koivula and Schmiegelow 2007, Saab et al. 2007, 2009, Cahall and Hayes 2009). No fire-dependent bird species has ever been shown to benefit from salvage logging (Hutto 2006, Hanson and North 2008). The ecological effects of salvage logging on aquatic ecosystems are also largely negative (Karr et al. 2004). In fact, the demonstrated negative ecological effects associated with postfire salvage logging are probably the most consistent and dramatic of any wildlife management effects ever documented for any kind of forest management activity (Hutto 2006). Therefore, because the National Forest Management Act and other legal mandates require public land managers to maintain the integrity of the larger ecological system, burned forests should perhaps be given special consideration compared with green-tree forests. Specifically, they could receive a low priority ranking when it comes to timber harvest decisions (with the obvious exception of small harvests associated with roads and other areas where safety or infrastructure are legitimate concerns). Timber can be harvested from many green-tree forests in a manner that imposes relatively little ecological cost in comparison with the costs associated with logging in burned forest (Lindenmayer and Cunningham 2013).

How do we move toward a more ecologically informed view of forest fires?

The ecological costs associated with some of the more commonly employed pre and postfire management activities in the western United States probably increase substantially as one moves from the low-elevation or xeric ponderosa pine or woodland forest types, where trees were widely spaced and severe fire historically played a spatially restricted role, to the broad array of more densely stocked mixed-conifer forest types, where severe fire historically played a major role. Therefore, a thorough understanding of the historical fire regime associated with any particular vegetation type or land area (as determined from multiple lines of evidence concerning regionally specific fire history) is critically important for land managers who concern themselves with the issues of wildfire risk, ecological restoration, or maintenance of the diversity of native species (Schoennagel and Nelson 2011). More specifically, quantification of appropriate fire rotations and proportions of low-, moderate-, and high-severity fire for any given forest landscape is critical for enlightened land management. For example, in some xeric ponderosa pine forest types, ecosystem restoration activities designed to decrease the severity of wildfire may be ecologically appropriate. The same management activities are not likely to be ecologically appropriate in many mixed-conifer forests, however, because key indicator species evolved to depend on significant amounts of severe fire in those forest types (Schoennagel et al. 2004, Hutto 2008, Klenner et al. 2008, Baker 2012, 2015, Williams and Baker 2012, Odion et al. 2014).

Land and fire managers are now facing future fires that many hypothesize will become larger and contain larger proportions of more severely burned patches under warming climate conditions (Rocca et al. 2014). Problems associated with climate change, however, must be solved through efforts directed toward the causes of climate change and not toward the symptoms of climate change. Any perceived problem with future changes in fire behavior cannot be solved by redoubling our effort to treat this particular climate change symptom by installing widespread fuel treatments that do nothing to stop the warming trend, and do little to reduce the extent or severity of weather-driven fires (Gedalof et al. 2005). Therefore, fuel management efforts to reduce undesirable effects of wildfires outside the xeric ponderosa pine forest types could be more strategically directed toward creating fire-safe communities (Calkin et al. 2014, Kennedy and Johnson 2014). A management emphasis directed toward altering conditions in and immediately adjacent to human communities is very different from an emphasis directed toward treating massive amounts of fuel on more remote public lands. Fuel treatment efforts more distant from human communities may carry the negative ecological consequences we outlined earlier and do little to stop or mitigate the effects of fires that are increasingly weather driven (Rhodes and Baker 2008, Franklin et al. 2014, Moritz et al. 2014, Odion et al. 2014).

Public land managers face significant challenges balancing the threats posed by severe fire with legal mandates to conserve wildlife habitat for plant and animal species that are positively associated with recently burned forests. Nevertheless, land managers who wish to maintain biodiversity must find a way to embrace a fire-use plan that allows for the presence of all fire severities in places where a historical mixed-severity fire regime creates conditions needed by native species while protecting homes and lives at the same time. This balancing act can be best performed by managing fire along a continuum that spans from aggressive prevention and suppression near designated human settlement areas to active "ecological fire management" (Ingalsbee 2015) in places farther removed from such areas. This could not only save considerable dollars in fire-fighting by restricting such activity to near settlements (Ingalsbee and Raja 2015), but it would serve to retain (in the absence of salvage logging, of course) the ecologically important disturbance process over most of our public land while at the same time reducing the potential for firefighter fatalities (Moritz et al. 2014). Severe fire is not ecologically appropriate everywhere, of course, but the potential ecological costs associated with prefire fuels reduction, fire suppression, and postfire harvest activity in forests born of mixed-severity fire need to considered much more seriously if we want to maintain those species and processes that occur only where dense, mature forests are periodically allowed to burn severely, as they have for millennia.

Another integral part of moving toward an ecologically informed perspective of forest fire involves getting the public, politicians, and policy-makers to better recognize and appreciate the critical role that severe fire plays in many forest systems. This has been difficult, and this difficulty has been exacerbated by public messages about severe fire that are uniformly negative. Progress toward allowing fires to burn is difficult unless the public begins to receive a message that differs markedly from the message that Smokey the Bear is sending them now. Fires in our wildlands are fundamentally natural and beneficial, so we must learn to live in a way that allows naturally occurring fires, including severe fires, to burn while minimizing risk to human property and lives (Calkin et al. 2014). That is a vastly different message from one that says severe fires are fundamentally bad and that we have to do everything in our power to prevent and suppress them, or from one that says severely burned forests are places where we should expedite efforts to capture residual economic value through "salvage" logging. We challenge ecologists and managers to pay greater attention to the degree of variation in fire regimes within mixed-conifer forests and to recognize that prefire thinning and postfire "restoration" activities may not always be compatible with maintenance of the ecological integrity of conifer forests that depend on complex mixed-severity fire disturbance.

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LITERATURE CITED

Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Covelo, California, USA.

Agee, J. K. 1998. The landscape ecology of western forest fire regimes. Northwest Science 72 :24–34.

Ahlgren, I. F., and C. E. Ahlgren. 1960. Ecological effects of forest fires. Botanical Review 26:483–533.

Arno, S. F. 2000. Fire regimes in western forest ecosystems. Pages 97–120 *in* J. K. Brown, and J. K. Smith, editors. Effects of fire on flora. USDA Forest Service General Technical Report RMRS-GTR-42-volume 2, Ogden, Utah, USA.

Attiwill, P., and D. Binkley. 2013. Exploring the megafire reality: a 'Forest Ecology and Management' conference. Forest Ecology and Management 294:1–3.

Baker, W. L. 2009. Fire ecology in Rocky Mountain landscapes. Island Press, Washington, D.C., USA.

Baker, W. L. 2012. Implications of spatially extensive historical data from surveys for restoring dry forests of Oregon's eastern Cascades. Ecosphere 33:23.

Baker, W. L. 2015. Are high-severity fires burning at much higher rates recently than historically in dry-forest landscapes of the western USA? PLoS One 10:e0136147.

Baker, W. L., T. T. Veblen, and R. L. Sherriff. 2007. Fire, fuels and restoration of ponderosa pine-Douglas fir forests in the Rocky Mountains, USA. Journal of Biogeography 34:251–269.

Barrett, S. W., S. F. Arno, and C. H. Key. 1991. Fire regimes of western larch-lodgepole pine forests in Glacier National Park, Montana. Canadian Journal of Forest Research 21:1711–1720.

Belote, R. T. 2015. Contemporary patterns of burn severity heterogeneity from fires in the northwestern U.S. Pages 252–256 *in* R. E. Keane, M. Jolly, R. Parsons, and K. Riley, editors. Proceedings of the large wildland fires conference; 19–23 May 2014, Missoula, Montana. USDA Forest Service Proceedings RMRS-P-73, Missoula, Montana, USA.

Benda, L., D. Miller, P. Bigelow, and K. Andras. 2003. Effects of post-wildfire erosion on channel environments, Boise River, Idaho. Forest Ecology and Management 178:105–119.

Boucher, J., E. T. Azeria, J. Ibarzabal, and C. Hébert. 2012. Saproxylic beetles in disturbed boreal forests: temporal dynamics, habitat associations, and community structure. Ecoscience 19:328–343.

Brown, J. K. 2000. Introduction and fire regimes. Pages 1–7 *in* J. K. Brown, and J. K. Smith, editors. Wildland fire in ecosystems: effects of fire on flora. General Technical Report RMRS-GTR-42-vol. 2. USDA Forest Service Rocky Mountain Research Station, Ogden, Utah, USA.

Brown, J. K., and J. K. Smith. 2000. Wildland fire in ecosystems: effects of fire on flora. USDA Forest Service General Technical Report RMRS-GTR-42-vol. 2, Ogden, Utah, USA.

Brown, P. M., M. R. Kaufmann, and W. D. Shepperd. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology 14:513–532.

Buchalski, M. R., J. B. Fontaine, P. A. III Heady, J. P. Hayes, and W. F. Frick. 2013. Bat response to differing fire severity in mixed-conifer forest California, USA. PLoS One 8:e57884.

Burton, T. A. 2005. Fish and stream habitat risks from uncharacteristic wildfire: observations from 17 years of fire-related disturbances on the Boise National Forest, Idaho. Forest Ecology and Management 211:140–149.

Cahall, R. E., and J. P. Hayes. 2009. Influences of postfire salvage logging on forest birds in the Eastern Cascades, Oregon, USA. Forest Ecology and Management 257:1119–1128.

Calkin, D. E., J. D. Cohen, M. A. Finney, and M. P. Thompson. 2014. How risk management can prevent future wildfire disasters in the wildland-urban interface. Proceedings of the National Academy of Sciences of the United States of America 111:746–751.

Cooper, C. F. 1961. Pattern in ponderosa pine forests. Ecology 42:493–499.

Covington, W. W., and M. M. Moore. 1994a. Postsettlement changes in natural fire regimes and forest structure: ecological restoration of old-growth ponderosa pine forests. Journal of Sustainable Forestry 2:153–182.

Covington, W. W., and M. M. Moore. 1994b. Southwestern ponderosa pine forest structure: changes since Euro-American settlement. Journal of Forestry 92:39–47.

DellaSala, D. A., M. L. Bond, C. T. Hanson, R. L. Hutto, and D. C. Odion. 2014. Complex early seral forests of the Sierra Nevada: What are they and how can they be managed for ecological integrity? Natural Areas Journal 34:310–324.

DellaSala, D. A., D. B. Lindenmayer, C. T. Hanson, and J. Furnish. 2015. In the aftermath of fire: logging and related actions degrade mixed- and high-severity burn areas. Pages 313–347 *in* D. A. DellaSala, and C. T. Hanson, editors. The ecological importance of mixed-severity fires: nature's phoenix. Elsevier Inc., Amsterdam, The Netherlands.

Dennison, P. E., S. C. Brewer, J. D. Arnold, and M. A. Moritz. 2014. Large wildfire trends in the western

United States, 1984–2011. Geophysical Research Letters 41:2928–2933.

Dillon, G. K., Z. A. Holden, P. Morgan, M. A. Crimmins, E. K. Heyerdahl, and C. H. Luce. 2011. Both topography and climate affected forest and woodland burn severity in two regions of the western US, 1984 to 2006. Ecosphere 2:130.

Dixon, R. D., and V. A. Saab. 2000. Black-backed Woodpecker (*Picoides arcticus*), The Birds of North America Online (A. Poole Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online:http://bna.birds.cornell.edu/bna/species/509

Dunham, J. B., A. E. Rosenberger, C. H. Luce, and B. E. Rieman. 2007. Influences of wildfire and channel reorganization on spatial and temporal variation in stream temperature and the distribution of fish and amphibians. Ecosystems 10:335–346.

Franklin, J. F., and N. K. Johnson. 2014. Lessons in policy implementation from experiences with the Northwest Forest Plan, USA. Biodiversity and Conservation 23:3607–3613.

Franklin, J. F., R. K. Hagmann, and L. S. Urgenson. 2014. Interactions between societal goals and restoration of dry forest landscapes in western North America. Landscape Ecology 29:1645–1655.

Gedalof, Z., D. L. Peterson, and N. J. Mantua. 2005. Atmospheric, climatic, and ecological controls on extreme wildfire years in the northwestern United States. Ecological Applications 15:154–174.

Guscio, C. G., B. R. Hossack, L. A. Eby, and P. S. Corn. 2008. Post-breeding habitat use by adult boreal toads (*Bufo boreas*) after wildfire in Glacier National Park, USA. Herpetological Conservation and Biology 3:55–62.

Hanson, C. T., and M. P. North. 2008. Postfire woodpecker foraging in salvage-logged and unlogged forests of the Sierra Nevada. Condor 110:777–782.

Hanson, C. T., R. L. Sherriff, R. L. Hutto, D. A. DellaSala, T. T. Veblen, and W. L. Baker. 2015. Setting the stage for mixed- and high-severity fire. Pages 3–22 *in* D. A. DellaSala, and C. T. Hanson, editors. The ecological importance of mixed-severity fires: nature's phoenix. Elsevier, Amsterdam, The Netherlands.

He, T., J. G. Pausas, C. M. Belcher, D. W. Schwilk, and B. B. Lamont. 2012. Fire-adapted traits of *Pinus* arose in the fiery Cretaceous. New Phytologist 194:751–759.

Heck, M. P. 2007. Effects of wildfire on growth and demographics of coastal cutthroat trout in headwater streams. M.S. Thesis. Oregon State University, Corvallis, OR, USA.

Heinselman, M. L. 1981. Fire and succession in the conifer forests of northern North America. Pages 374–405 *in* D. C. West, H. H. Shugart, and D. B. Botkin, editors. Forest succession: concepts and applications. Springer-Verlag, New York, New York, USA.

Hessburg, P. F., R. B. Salter, and K. M. James. 2007. Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology 22:5–24.

Hollenbeck, J. P., V. Saab, and R. W. Frenzel. 2011. Habitat suitability and nest survival of white-headed woodpeckers in unburned forests of Oregon. Journal of Wildlife Management 75:1061–1071.

Homann, P., B. Bormann, B. Morrissette, and R. Darbyshire. 2015. Postwildfire soil trajectory linked to prefire ecosystem structure in Douglas-fir forest. Ecosystems 18:260–273.

Hossack, B. R., and P. S. Corn. 2007. Responses of pond-breeding amphibians to wildfire: short-term patterns in occupancy and colonization. Ecological Applications 17:1403–1410.

Hoyt, J. S., and S. J. Hannon. 2002. Habitat associations of Black-backed and Three-toed woodpeckers in the boreal forest of Alberta. Canadian Journal of Forest Research 32:1881–1888.

Hutto, R. L. 1995. 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 20:984–993.

Hutto, R. L. 2008. The ecological importance of severe wildfires: some like it hot. Ecological Applications 18:1827–1834.

Hutto, R. L., and R. T. Belote. 2013. Distinguishing four types of monitoring based on the questions they address. Forest Ecology and Management 289:183–189.

Hutto, R. L., and S. M. Gallo. 2006. The effects of postfire salvage logging on cavity-nesting birds. Condor 108:817–831.

Hutto, R. L., C. J. Conway, V. A. Saab, and J. R. Walters. 2008. What constitutes a natural fire regime? Insight from the ecology and distribution of coniferous forest birds in North America. Fire Ecology 4:115–132.

Hutto, R. L., M. L. Bond, and D. A. DellaSala. 2015. Using bird ecology to learn about the benefits of severe fire. Pages 55–88 *in* D. A. DellaSala, and C. T. Hanson, editors. The ecological importance of mixed-severity fires: nature's phoenix. Elsevier, Amsterdam, The Netherlands.

Ingalsbee, T. 2015. Ecological fire use for ecological fire management: managing large wildfires by design. Pages 120–127 *in* R. E. Keane, M. Jolly, R. Parsons, and K. Riley, editors. Proceedings of the large wildland fires conference; 19–23 May 2014, Missoula, Montana. USDA Forest Service Proceedings RMRS-P-73, Missoula, Montana, USA.

Ingalsbee, T., and U. Raja. 2015. The rising costs of wildfire suppression and the case for ecological fire use. Pages 348–371 *in* D. A. DellaSala, and C. T. Hanson, editors. The ecological importance of mixed-severity fires: nature's phoenix. Elsevier Inc., Amsterdam, The Netherlands.

Jackson, B. K., S. M. P. Sullivan, C. V. Baxter, and R. L. Malison. 2015. Stream-riparian ecosystems and mixed- and high-severity fire. Pages 118–148 *in* D. A. DellaSala, and C. T. Hanson, editors. The ecological importance of mixed-severity fires: nature's phoenix. Elsevier Inc., Amsterdam, The Netherlands.

Johnson, E. A., A. M. Gill, R. A. Bradstock, A. Granstrom, L. Trabaud, and K. Miyanishi. 2003. Towards a sounder fire ecology. Frontiers in Ecology and the Environment 1:271–276.

Karr, J. R., J. J. Rhodes, G. W. Minshall, F. R. Hauer, R. L. Beschta, C. A. Frissell, and D. A. Perry. 2004. The effects of postfire salvage logging on aquatic ecosystems in the American West. BioScience 54:1029–1033.

Keane, R. E. 2013. Disturbance regimes and the historical range of variation in terrestrial ecosystems. Pages 568–581 *in* A. L. Simon, editor. Encyclopedia of biodiversity (Second Edition). Academic Press, Waltham, MA, USA.

Keane, R. E., J. K. Agee, P. Z. Fulé, J. E. Keeley, C. Key, S. G. Kitchen, R. Miller, and L. A. Schulte. 2008. Ecological effects of large fires on US landscapes: benefit or catastrophe? International Journal of Wildland Fire 17:696–712.

Kennedy, M. C., and M. C. Johnson. 2014. Fuel treatment prescriptions alter spatial patterns of fire severity around the wildland–urban interface during the Wallow Fire, Arizona, USA. Forest Ecology and Management 318:122–132.

Kilgore, B. M. 1981. Fire in ecosystem distribution and structure: western forests and scrublands. Pages 58–89 *in* H. A. Mooney, T. M. Bonnicksen, N. L. Christensen, J. E. Lotan, and W. A. Reiners, editors. Fire regimes and ecosystem properties. USDA Forest Service General Technical Report WO-26, Washington, District of Columbia, USA.

Klenner, W., R. Walton, A. Arsenault, and L. Kremsater. 2008. Dry forests in the Southern Interior of British Columbia: historic disturbances and implications for restoration and management. Forest Ecology and Management 256:1711–1722.

Koivula, M. J., and F. K. A. Schmiegelow. 2007. Boreal woodpecker assemblages in recently burned forested landscapes in Alberta, Canada: Effects of post-fire harvesting and burn severity. Forest Ecology and Management 242:606–618.

Latif, Q. S., V. A. Saab, J. G. Dudley, and J. P. Hollenbeck. 2013. Ensemble modeling to predict habitat suitability for a large-scale disturbance specialist. Ecology and Evolution 3:4348–4364.

Lindenmayer, D. B., and S. A. Cunningham. 2013. Six principles for managing forests as ecologically sustainable ecosystems. Landscape Ecology 28:1099–1110.

Lindenmayer, D. B., and R. F. Noss. 2006. Salvage logging, ecosystem processes, and biodiversity conservation. Conservation Biology 20:949–958.

Lindenmayer, D. B., D. R. Foster, J. F. Franklin, M. L. Hunter, R. F. Noss, F. A. Schmiegelow, and D. Perry. 2004. Salvage harvesting policies after natural disturbance. Science 303:1303.

Lorenz, T. J., K. T. Vierling, J. M. Kozma, J. E. Millard, and M. G. Raphael. 2015. Space use by white-headed woodpeckers and selection for recent forest disturbances. Journal of Wildlife Management, 79:1286–1297. (DOI: 10.1002/jwmg.957).

Lydersen, J. M., M. P. North, and B. M. Collins. 2014. Severity of an uncharacteristically large wildfire, the Rim Fire, in forests with relatively restored frequent fire regimes. Forest Ecology and Management 328:326–334.

Malison, R. L., and C. V. Baxter. 2010. The fire pulse: wildfire stimulates flux of aquatic prey to terrestrial habitats driving increases in riparian consumers. Canadian Journal of Fisheries and Aquatic Sciences 67:570–579.

Marlon, J. R., et al. 2012. Long-term perspective on wildfires in the western USA. Proceedings of the National Academy of Sciences of the United States of America 109:E535–E543.

Miller, J. D., C. N. Skinner, H. D. Safford, E. E. Knapp, and C. M. Ramirez. 2012. Trends and causes of severity, size, and number of fires in northwestern California, USA. Ecological Applications 22: 184–203.

Morgan, P., R. E. Keane, G. K. Dillon, T. B. Jain, A. T. Hudak, E. C. Karau, P. G. Sikkink, Z. A. Holden, and E. K. Strand. 2014. Challenges of assessing fire and burn severity using field measures, remote sensing and modelling. International Journal of Wildland Fire 23:1045–1060.

Moritz, M. A., et al. 2014. Learning to coexist with wildfire. Nature 515:58–66.

Murphy, E. G., and W. H. Lehnhausen. 1998. Density and foraging ecology of woodpeckers following a stand-replacement fire. Journal of Wildlife Management 62:1359–1372.

Odion, D. C., et al. 2014. Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS One 9:e87852.

Perry, D. A., P. F. Hessburg, C. N. Skinner, T. A. Spies, S. L. Stephens, A. H. Taylor, J. F. Franklin, B. McComb, and G. Riegel. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. Forest Ecology and Management 262:703–717.

Pilz, D., N. S. Weber, M. C. Carter, C. G. Parks, and R. Molina. 2004. Productivity and diversity of morel mushrooms in healthy, burned, and insect-damaged forests of northeastern Oregon. Forest Ecology and Management 198:367–386.

Reidy, J. L., F. R. Thompson Iii, and S. W. Kendrick. 2014. Breeding bird response to habitat and landscape factors across a gradient of savanna, woodland, and forest in the Missouri Ozarks. Forest Ecology and Management 313:34–46.

Rhodes, J. J., and W. L. Baker. 2008. Fire probability, fuel treatment effectiveness and ecological tradeoffs in western U.S. public forests. Open Forest Science Journal 1:1–7.

Rocca, M. E., P. M. Brown, L. H. MacDonald, and C. M. Carrico. 2014. Climate change impacts on fire regimes and key ecosystem services in Rocky Mountain forests. Forest Ecology and Management 327:290–305.

Rollins, M. G. 2009. LANDFIRE: a nationally consistent vegetation, wildland fire, and fuel assessment. International Journal of Wildland Fire 18:235–249.

Rosenberger, A. E., J. B. Dunham, J. M. Buffington, and M. S. Wipfli. 2011. Persistent effects of wildfire and debris flows on the invertebrate prey base of rainbow trout in Idaho streams. Northwest Science 85:55–63.

Rota, C. T., J. J. Millspaugh, M. A. Rumble, C. P. Lehman, and D. C. Kesler. 2014. The role of wildfire, prescribed fire, and mountain pine beetle infestations on the population dynamics of black-backed woodpeckers in the black hills, South Dakota. PLoS One 9:e94700.

Rota, C. T., M. A. Rumble, C. P. Lehman, D. C. Kesler, and J. J. Millspaugh. 2015. Apparent foraging success reflects habitat quality in an irruptive species, the Black-backed Woodpecker. Condor 117:178–191.

Ryan, S. E., K. A. Dwire, and M. K. Dixon. 2011. Impacts of wildfire on runoff and sediment loads at Little Granite Creek, western Wyoming. Geomorphology 129:113–130.

Saab, V. A., R. E. Russell, and J. Dudley. 2007. Nest densities of cavity-nesting birds in relation to post-fire salvage logging and time since wildfire. Condor 109:97–108.

Saab, V. A., R. E. Russell, and J. G. Dudley. 2009. Nest-site selection by cavity-nesting birds in relation to postfire salvage logging. Forest Ecology and Management 257:151–159.

Saint-Germain, M., P. Drapeau, and C. Hebert. 2004a. Comparison of Coleoptera assemblages from a recently burned and unburned black spruce forests of northeastern North America. Biological Conservation 118:583–592.

Saint-Germain, M., P. Drapeau, and C. Hebert. 2004b. Landscape-scale habitat selection patterns of *Monochamus scutellatus* (Coleoptera: Cerambycidae) in a recently burned black spruce forest. Environmental Entomology 33:1703–1710.

Schieck, J., and S. J. Song. 2006. Changes in bird communities throughout succession following fire and harvest in boreal forests of western North America: literature review and meta-analyses. Canadian Journal of Forest Research 36:1299–1318.

Schoennagel, T., and C. R. Nelson. 2011. Restoration relevance of recent National Fire Plan treatments in forests of the western United States. Frontiers in Ecology and the Environment 9:271–277.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience 54:661–676.

Schoennagel, T., R. L. Sherriff, and T. T. Veblen. 2011. Fire history and tree recruitment in the Colorado Front Range upper montane zone: implications for forest restoration. Ecological Applications 21:2210–2222.

Sestrich, C. M., T. E. McMahon, and M. K. Young. 2011. Influence of fire on native and nonnative salmonid populations and habitat in a western Montana basin. Transactions of the American Fisheries Society 140:136–146.

Sherriff, R. L., and T. T. Veblen. 2006. Ecological effects of changes in fire regimes in *Pinus ponderosa* ecosystems in the Colorado Front Range. Journal of Vegetation Science 17:705–718.

Sherriff, R. L., and T. T. Veblen. 2007. A spatially-explicit reconstruction of historical fire occurrence in the ponderosa pine zone of the Colorado Front Range. Ecosystems 10:311–323.

Sherriff, R. L., R. V. Platt, T. T. Veblen, T. L. Schoennagel, and M. H. Gartner. 2014. Historical, observed, and modeled wildfire severity in montane forests of the Colorado Front Range. PLoS One 9:e106971.

Sugihara, N. G., J. W. van Wangtendonk, and J. Fites-Kaufman. 2006. Fire as an ecological process. Pages 58–74 *in* N. G. Sugihara, J. W. van Wangtendonk, J. Fites-Kaufman, K. E. Shaffer, and A. E. Thode, editors. Fire in California's ecosystems. University of California Press, Berkeley, California, USA.

Swanson, M. E., J. F. Franklin, R. L. Beschta, C. M. Crisafulli, D. A. DellaSala, R. L. Hutto, D. B. Lindenmayer, and F. J. Swanson. 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment 9:117–125.

Weaver, H. 1943. Fire as an ecological and silvicultural factor in the ponderosa pine region of the Pacific Slope. Journal of Forestry 41:7–14.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increases western U.S. forest wildfire activity. Science 313:940–943.

Williams, M. A., and W. L. Baker. 2012. Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests. Global Ecology and Biogeography 21:1042–1052.

Williams, A. P., et al. 2015. Correlations between components of the water balance and burned area reveal new insights for predicting forest fire area in the southwest United States. International Journal of Wildland Fire 24:14–26.

Inventory of Flammulated Owls Breeding in the East Kootenay 2003

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Introduction

The Flammulated Owl has the distinction of being the only insectivorous and migratory owl in British Columbia. This small owl is distributed in western North America from Mexico north to British Columbia. In BC, Flammulated owls nest in the dry Douglas-fir forests of interior BC from May to September and migrate to the southern portions of their North American range for the winter.

Flammulated Owls (FLOW) are blue-listed in British Columbia and nationally listed as a species of Special Concern (COSEWIC 2003). Threats to their nesting habitat are a primary concern. The Flammulated Owls demography and life history make it vulnerable to changes in nesting habitat quality and availability. Flammulated Owls require a complex of four habitat components to meet their nesting requirements. These are thick shrubs or young trees for security, small grassy openings for foraging, large mature trees for roosting and snags with cavities for nesting (Reynolds and Linkhart 1992). All four habitat components should be available within the owl's home range, which may range from 2.2-15.9 ha in size (Cannings and van Woudenberg, 2003). This complex is generally found in old or mature Douglas-fir forests with a variable and open stand structure. Forest management activities, ecosystem restoration, prescribed burning or fire suppression may result in the loss of some or all of these critical nesting habitat components. Successful management of the FLOW nesting habitat complex requires integrating local information on nesting habitat requirements with these potential threats and identifying opportunities to maintain or improve nesting habitat.

Background

In British Columbia, Flammulated Owls have only been intensively studied in the Kamloops area. Inventory efforts expanded in the late 1990's and confirmed the species' presence in the Cariboo, Williams Lake, Lillooet, Merritt, Princeton and the Nicola Valley. The Rocky Mountain Trench was historically known as part of the species range, but its presence and distribution there were only confirmed through inventories conducted in 2000 and 2001 (van Woudenberg et al. 2000; Addison and Christie 2002). Prior to the current study, two FLOW nest sites had been found in the East Kootenay.

One nest was discovered during a cavity nesting study and another was located during nest searches conducted in 2001 (Addison and Christie 2002). Both nests were located on Grainger Mountain near the Whiteswan Rd. Surveys conducted in 2000 and 2001 detected owls on several transects in the region but resources were inadequate for more detailed work locating nests and describing nesting habitat.

Several habitat management activities underway in the East Kootenay may have negative affects on FLOW and their nesting habitat. Thinning and prescribed burning are being used to decrease the effects of forest ingrowth from fire suppression. Management treatments to restore and maintain Ungulate Winter Range may also degrade FLOW habitat. Snag availability and recruitment can be reduced by firewood cutting and forest management activities.

Objectives

The goal of this study was to obtain additional information on the distribution, numbers and nesting habitat requirements of Flammulated Owls in the East Kootenay. The objectives of this study were to:

1) Inventory FLOW using call playback surveys on previously identified transects in the East Kootenay
2) Search areas with calling owls to locate nest trees
3) Document nesting habitat characteristics in the East Kootenay

Methods

Study Area

The study area extended from the Wildhorse River near Fort Steele north to Mt Swansea near Invermere. The study area consisted of seven survey transects which had been previously selected and surveyed in 2000 and or 2001 (Addison and Christie 2002). All transects were located on the east side of the Kootenay trench with surrounding habitat consisting of mature or old forests in the Interior Douglas-fir Biogeoclimatic zone.

Transects with previous owl detections (in 2000 and 2001) and located between Cranbrook and Invermere were selected for the study.

Call Playback surveys

Call Playback surveys were used to determine the number and location of calling male Flammulated Owls. Survey methods followed the RIC raptor survey standards as used by van Woudenberg et al. (2000) and Addison and Christie (2002). Survey transects ranged from 4-12km in length with call playback stations located every 500m along the transect. At each station surveyors listened for spontaneously calling owls for five minutes. If no FLOW were heard during this period a 30sec recording of FLOW calls was played on a portable stereo followed by 1 minute of listening for response calls. The recording was played up to three times at each station. For each owl detected during the surveys we recorded: the compass bearing of calls, the estimated distance of calls, the species of owl, and whether calls were spontaneous or induced. GPS locations were taken at each survey station and calling owl locations were plotted on 1:20,000 forest cover maps following surveys.

Call Playback surveys for Flammulated owls were conducted between May 22 and June 6, 2003 on 8 transects. Surveys were conducted between 22:00 and 03:00 hours. Weather conditions were recorded during surveys and surveys were aborted if wind or rain conditions hampered our ability to hear or locate owls. Our objective was to survey each transect once under ideal survey conditions. The Mt Swansea transect was repeated on June 6 because of windy conditions encountered on May 30. The Whiteswan transect was surveyed over two days due to its length.

Locations of calling owls were plotted on 1:20,000 scale maps. Preliminary estimates of minimum numbers of owls and their density (estimated number of individuals/km of transect) on the transects were derived from the survey data. These estimates provide an index of the owl population at transects but do not indicate absolute population numbers. Numbers of calling males detected can change based on breeding status of males, and can be inflated by non-resident owls migrating through the area.

Mapping of the data revealed apparent "clusters" of owl detections which subsequently proved to be repeat detections of the same bird, heard from different stations. The number of spontaneous versus elicited responses was noted. Owl numbers and densities presented are estimates based on preliminary surveys.

Nest Searches

Follow up nest searches were conducted in areas that had high potential for FLOW nesting based on call playback surveys and habitat characteristics. Sites where owl calls were close to survey stations were given a higher priority for searching because plotted owl locations were likely to be more accurate than distant calls. Owl clusters where owls were heard from 2 or more survey stations were also given a higher priority for nest search efforts. Survey data from 2000/2001 was also considered and sites that had been occupied by owls over multiple years were given a high priority because territory occupancy may be an important indicator of habitat quality (Linkhart and Reynolds 1997). The Whiteswan transect was not the focus of major nest search efforts in 2003 because 2 nests had been previously found there

During nest searches we recorded waypoints with the GPS at regular intervals, checked all potential nesting snags encountered (by scraping the tree) and recorded snag species, the diameter at breast height (dbh), decay class and cavity presence. We also noted the general quality of foraging, roosting and security habitat encountered during nest searches.

Nesting habitat characteristics

We returned to nest trees to document nesting habitat 3-13 days after nests were located. Nest trees were described according to British Columbia Resource Inventory Committee Standards. Habitat parameters at each nest tree were recorded and included the type of structure, the aspect of the nest site, and the height of the nest above the ground. Nest trees themselves were classified using the tree attributes for wildlife codes for Appearance, Crown condition, Bark retention and wood condition following the standards of 'Field Manual for Describing Terrestrial Ecosystems' (Ministry of Forests and Ministry of Environment 1998). Nest tree species, dbh, height, and decay class were also recorded.

Forest structure around nest trees was determined by establishing nested 7.98-m radius (0.02 ha) and 25-m radius (0.19 ha) plots centred on the nest tree. All plots were measured using slope corrected distances. Site characteristics recorded for the 25-m radius plot include the elevation, slope, aspect and topography. Structural stage was described using the following categories: shrub; pole/sapling; young forest; mature forest; and old forest (Ministry of Forests and Ministry of Environment 1998).

Detailed vegetation characteristics were measured within the 7.98-m radius plot. The percentage cover of each plant species occurring in the tree, shrub, herb and moss layers was estimated (Ministry of Forests and Ministry of Environment 1998).

Tree and stand structural characteristics were measured within the 25-m radius plot. Tree species, diameter at breast height (dbh), tree height and top condition were determined for all trees with a dbh >10 cm within this plot. The height of one tree from each layer was measured using a clinometer and meter tape, and then estimated for the remaining trees. Snags and live trees with potential nesting structures within the plot were classified using the tree attributes for wildlife, categories for Appearance, Crown condition, Bark retention and Wood condition and Wildlife Use (Ministry of Forests and Ministry of Environment 1998). For trees < 10 cm in dbh, stem density was recorded within the 25-m radius plot for each tree species.

Coarse woody debris (CWD) was assessed in the nest patch by establishing one 25-m radial transect. CWD diameter and decay class was measured for each piece > 7.5 cm in diameter crossing the transect line (Ministry of Forests and Ministry of Environment 1998). Summary variables describing live tree and snag density height and diameter were calculated for each nest site. These data, in addition to describing the present stand structure, also provide some insight into stand dynamics creating this structure.

Results

Call Playback surveys

A total of 105 Flammulated owl detections were heard on 56 km of survey transects (Table 1). We estimate these detections to represent a total of 64 individual owls. Densities ranged from 0.8 to 2.0 owls per km and averaged 1.1 birds/km. The highest owl densities were recorded at the Brewery Creek and Mt Swansea transects (Table 1). This density is not representative of the East Kootenay as a whole because our surveys were focused on the best Flammulated Owl areas identified in previous surveys in 2000 and 2001 (van Woudenberg et al. 2000 and Addison and Christie 2002).
The majority (82%) of Flammulated Owls heard were calling spontaneously; only 18 % of all detections were elicited by playing a recording. We found that elicited responses were more numerous during poor survey conditions when spontaneous calls were difficult to hear. We also found that elicited owl calls tended to be more mobile.

Barred Owls, Great Horned Owls and Northern Pygmy Owls were also heard during call playback surveys. A total of seven Barred Owls were heard from 4 of the 8 transects. On the Brewery Creek and Lakit transects Barred owls were heard calling very close to Flammulated Owl calling locations (Figures1, 7). On the Whiteswan and Lazy Lake transects, however, Barred Owls were located on parts of the transect that had few or no Flammulated Owls (Figures 3, 5).

Common Poorwills were heard calling at 4 stations on the Lakit transect (LK5, SG1, 2, 3). As there are few records of this species in the East Kootenay details of Common Poorwill detections are described in Appendix 2. Poorwill calls appeared to be associated with extensive antelope brush /boulder habitat with well developed leaf litter.

Nest Search Efforts and Results

A total of 98 person hours of nest searching was conducted at 13 owl cluster locations (Table 2). Of the 13 sites that were searched, 9 (69%) had all of the nesting habitat features required for Flammulated Owls. Four nest trees were located. At the other four sites, owls were calling from locations that lacked some nesting habitat features. At the Whiteswan 8 km site an owl was heard calling from the searched location repeatedly on 2 survey dates. The site had regenerating forest on the lower slope with a couple of very obvious emergent snags. The understory at this site was thick and shrubby and not suitable for foraging. At the Lazy Lake 4 km site a FLOW was heard calling at the site from 5 different locations. The owl locations were up to 500m apart so it is possible that there was some location error or that the owl was moving around at this site (Figure 3). We found all components of FLOW nesting habitat at this site but not in close association with each other. Locations with foraging and security cover lacked snags and vice versa. At the Mt Swansea 2.5 km site we had multiple owl detections close to the survey station but the calls were responses to broadcast calls and the bird appeared to be moving and following the recording (Figure 4). There was a lack of suitable snags and foraging habitat in the area searched. At the Brewery Creek 4.5 km site we heard two owls calling from two stations. Barred owls were heard in the area, and the denser forest and thick shrub cover on the lower portions of this site appeared more suitable for Barred than for Flammulated Owls (Figure 1a).

As a result of these nest search efforts, 9 of the 13 owl cluster sites identified were recommended as candidate WHAs for Flammulated Owls. These sites include the four nest sites found, as well as five additional stands (Table 2). Due to time limitations we could not survey many areas that had high potential for nesting.

Owl Behaviour at nest sites

Nest occupancy was observed from June 19 to July 16 (Table 3). Owls responded quickly to scratching of the snags by appearing at the cavity entrance. They responded readily to scraping of nest trees in both the morning and afternoon (Table 3). At the Brewery Creek nest the owl reacted to the noise of falling rocks and appeared at the nest cavity before the snag was scraped. When we returned to nest trees, 3-13 days later, owls were still present at nests and were visible at the nest cavity briefly when we were close to the nest tree.

Nesting habitat characteristics

FLOW nests were located in trees ranging from 36.9 to 58 cm in diameter and 5.5 to 22 m tall. Three nest trees were dead (Appearance codes 3-5) and one tree was live but declining (Appearance code 2) (Table 4) Nesting snags all had a high proportion of remaining branches (See Photos Appendix 3). The live nest tree had a broken top and decay in the bole of the tree. Nest trees were classified as decay classes 2, 3 and 5 and Appearance codes 2, 4 and 5 (Table 4).

Nesting cavities appeared to be Pileated woodpecker cavities due to the large size of cavity openings. At Mt Swansea the cavity had a distinctive keyhole cavity shape. Nest cavities were located from 3.9 to 10 metres above the ground. The three Douglas-fir nest trees all had broken tops and nest cavities were located just below the top of the snag or tree. The Ponderosa Pine nest snag had an intact top and the nest cavity was located about half way up the tree. Nest cavities were oriented to the east, south and west (Table 4). Flow nest sites were located between 1068 to 1156 m elevation. All nest sites were on steep slopes (62-78%) with south to west aspects (Table 5). Habitat at nests were classified as drier site associations of the Kootenay dry mild Interior Douglas-fir variant (IDFdm2) (Braumandl and Curran 1992). Three nest sites were classified as subxeric/ submesic site associations (IDFdm2-03) and one site was classified as xeric (IDFdm2-02). All nest sites were in mature or old structural stages (structural stage 6 and 7 respectively). FLOW nest stands had low to moderate canopy cover (10-32%) of predominantly Douglas-fir. Shrub cover at nest stands ranged from 14-66%. Dominant shrub species were Douglas-fir, Saskatoon (*Amelanchier alnifolia*), Soopolalie (*Shepherdia canadensis*) common and rocky mountain juniper (*Juniperus communis, Juniperus scopulorum*). Herb cover at nest sites ranged from 14-41% with Bluebunch wheatgrass (*Agropyron spicatum*) dominant at all sites. Tree density (>10 cm dbh) at nest plots ranged from 361-621 stems per hectare (Table 6).. Lazy Lake and Mt Swansea nest plots were considered mature structural stage stands and both were dominated by smaller diameter stems < 40 cm in dbh (Figure 9). Mt Swansea had the lowest densities of trees < 10 cm (Table 6) and also low frequency of trees in the 10-15 cm dbh category as seen in the frequency histogram (Figure 9). Trees in the smallest diameter class (<10cm) ranged in density from 15/ha at Mt Swansea to 1065 at Maus Creek. Brewery Creek and Maus Creek had a wider range of stem diameters represented in the stands and were the only plots with stems in the 50-69cm and >70 cm diameter classes (Table 6, Figure 9). Despite the differences in tree size frequency among sites, the average stem diameter (for stems >10cm dbh) was consistent across all four sites ranging from 19.4 to 21.1 cm (Figure 1). Standard deviation in tree diameter was higher at Maus Creek and Brewery Creek (old structural stage) than Mt Swansea and Lazy Lake (mature structural stage). Snag density ranged from 15-76 snags >10cm dbh/ha. Mt Swansea which had the lowest tree density had the highest density of snags. Average snag diameter ranged from 21.9 to 47.9 cm dbh. Preferred habitat characteristics for Flammulated Owl nesting habitat include the presence of large snags >64 cm in dbh or at a minimum snags >35cm dbh (Cannings and van Woudenberg 2003). Larger snags (> 35cm in dbh) were present all of the four nest plots (Table 6). Snags at all sites averaged <10m in height except at Maus Creek where all snags were >35cm in dbh and averaged 18.7 m tall.

Discussion

Owl detections

Call playback surveys detected owls on all transects surveyed in 2003 which is consistent with the results from 2000/2001 surveys. Many of the calling owl locations from 2003 overlapped with locations from 2000 and 2001. To facilitate comparisons in future years a database of survey and owl location data should be maintained. The estimates of owl detections per km are preliminary. These numbers are intended for comparison among transects and should not be extrapolated to other areas. Relative population numbers could be determined with repeated call playback surveys combined with stratification of sampled habitat (RIC 2001).

Nest Searches

Nest search efforts were highly successful in 2003. The potential for locating more nests is high and can be done cost effectively. We were not able to search all of the potential owl clusters and calling male locations identified in the study. Many of the potential search areas were also occupied in 2000/2001. Nests were located in 4 of 9 areas searched that had suitable nesting habitat. It is possible that nests were missed at these sites due to early failure of nests, or failure of the owls to respond to scraping of Wildlife trees. Search efforts could be improved by conducting repeated call playback surveys to identify mated males that stop broadcasting calls once the nest is established (van Woudenberg 1999). This would better identify the timing of nest initiation and ensure that searches were conducted during the brood period when females will appear at the cavity entrance. If nest location is a major goal of future surveys it would be beneficial to locate calling stations closer to potential nesting locations whenever possible. This would decrease the error in triangulating bird calling locations.

We identified 4 locations where calling males were heard in habitat that was not suitable for nesting. Unmated male owls are known to continue calling late into the breeding season (van Woudenberg 1999). These locations had snags but the snags were within a forest context unsuitable for foraging and/or security. Unmated males may call from these sites due to a lack of suitable nesting habitat and fail to attract mates in these locations.

Nesting characteristics in the East Kootenay

Throughout North America Flammulated Owls are reported to nest in Pileated Woodpecker and Northern Flicker cavities in Ponderosa Pine and Douglas-fir snags and live trees (van Woudenberg 1999). The nest trees in this study are consistent with this trend. Other studies have indicated a preference for Ponderosa Pine snags in Oregon and in the Kamloops area (Bull et al. 1990, van Woudenberg 1999). In this sample 3 of 4 nests were in Douglas-fir wildlife trees and these nest stands lacked Ponderosa Pine wildlife trees. Our observations during nest searches indicate that Douglas-fir snags were more abundant at our study areas than Ponderosa Pine. However, Ponderosa pine Wildlife trees generally have more cavities especially Pileated Woodpecker Cavities.

Preference of snag species is difficult to determine with only 6 known nest sites in the East Kootenay (3 Douglas-fir and 3 Ponderosa Pine).

Nest trees in this study (range 36.9-58 cm dbh) were slightly smaller in diameter than sizes reported for the Kamloops area (mean 60.6 range 49-82cm Cannings and van Woudenberg 2003). Trees and snags in the 35-60 dbh range can contribute important nesting and roosting habitat in this area. Nest plots at two sites had no trees >50cm available. DBH in itself is not important as long as the tree provides the cavity structure and crown closure needed for nesting and roosting (Table 6). Nest cavity height was similar to that reported for Kamloops (mean 8.9m Cannings and van Woudenberg 2003).

The topography of FLOW nest sites in the East Kootenay is consistent with data reported for other locations (Bull et al 1990). Nests were in a limited elevational band from 1060 to 1160m that coincides sub-xeric and sub-mesic sites on steep slopes with west to south aspects. Slopes at nest stands in the East Kootenay were steeper than described for the Kamloops area (van Woudenberg 1999). The three steepest nest stands has small talus streams creating grassy openings, a feature that was also observed at nest sites in the Cariboo area (van Woudenberg 1999). The specific elevation and topography of FLOW nest sites may occur because these conditions are predictors of dryer site associations where ideal stand structure occurs. The use of steep terrain on warm aspects may also provide warmer less humid micro-climates that are optimal for nocturnal foraging (vanWoudenberg 1999). Steep slopes are generally inaccessible to firewood cutters and may have high densities of potential nesting cavities.

The density if stems>10cm dbh ranged from 361-621 stems/ha in East Kootenay nest stands. Comparison of densities across studies is difficult when the dbh cut-off is not given but this range appears similar to numbers reported for New Mexico (504-589) and Oregon (330) (McCallum 1994). In Kamloops nest stands had densities ranging from 2472-2837 stems/ha including stems <10cm dbh (van Woudenberg 1999). For comparison the East Kootenay nests range from 376-1686 stems/ha when all stems <10cm are included. Stem density was highly variable among the four nests. At the low range Mt Swansea had very low <10 cm stem density and a low frequency of stems 10-15cm in dbh. This structure was the result of burning and thicket removal done at the site 15-20 years ago to improve ungulate winter habitat (P. Davidson pers.com.). Although there were no thickets adjacent to the nest tree there was a very large thicket about 30m away (outside of the nest plot). The low stem density at the Mt Swansea nest would probably not be adequate for nesting without the security provided by the adjacent thicket.

Ungulate winter range management guidelines recommend restoring stand structure to either Open range (5-75 stems per hectare) or Open forest (76-400 stems per hectare), however the minimum stem dbh for these rules is not known Both of these stem densities are well below the stand densities reported for FLOW nests in the East Kootenay and elsewhere. UWR objectives are likely to degrade the present habitat suitability for FLOW in the East Kootenay. Interim guidelines for maintaining FLOW habitat given these conflicting management objectives should be developed to maintain current habitat suitability. Additional research on nesting productivity in relation to stem density treatments is needed to recommend a management strategy.

Snag density at nest sites ranged from 20-82 snags per hectare for all snags and 5-20/hectare for large snags (>35cm dbh). There is little information on snag densities at FLOW nest sites in the literature other than a recommendation of 5 snags/hectare for Washington (Hays and Roderick 2002). More information on snag density at nest sites is needed in order to maintain current habitat suitability and ensure that management activities will recruit snags for future habitat suitability

Recommendations

The results of this study provide some preliminary information on stand densities and structure of nesting sites. We recommend three potential goals/objective for future research that would address the most important information needs for Flammulated Owls.

1) <u>Continue inventories to increase the sample size of known nests.</u>

Further inventory is needed to provide a larger sample of nest sites over a wider distribution in the East Kootenay. A large sample of nest sites is necessary for rigorous analyses of nesting productivity and habitat selection (recommendations 2,3). Survey transects should be stratified among habitat types to better determine owl abundance and to eliminate bias in sampling and nest search efforts. Basic information on the slope, elevation, aspect, age class and Biogeoclimatic site associations could be used to map potential nesting habitat and stratify habitat into simple rankings (high medium low).

2) <u>Collect data on owl productivity to determine habitat quality.</u>

There is a need for detailed information on the characteristics of nesting stands and landscapes needed for Flammulated Owls. Ideally information on owl productivity (Reproductive success, survival) would be used in conjunction with nesting habitat data to determine habitat quality. Reproductive success could be determined by monitoring known nest sites later into the breeding season. Observations made at night could determine if hatching is successful based on feeding visits and fledgling numbers. An alternative to monitoring reproductive success is to monitor territory occupancy. When measured over long time frames territory occupancy was linked with habitat quality (Linkhart et.al. 1998). Re-use of territories is apparent for the two FLOW nests found in 1999 and 2000 (T. Antifeau pers. comm.). Many owl locations from the 2003 survey overlap with 2000 and 2001 survey results. The use of consistent survey locations and annual mapping of owl detections would facilitate territory occupancy determinations. Observation protocols that minimize disturbance to nesting owls should be developed for productivity assessments.

3) <u>Test for nesting habitat selectivity across a range of spatial scales</u>

Habitat sampling can be structured to determine use vs. availability of habitat features. This generally involves sampling habitat at nests and at random sites. Patch level selectivity can be measured by comparing nest plots to randomly located plots within the nest stand. Stand level selectivity can be measured by comparing stands with nests to randomly selected adjacent stands. Air-photo interpretation may be appropriate for measuring the complexities of forest structure at the stand level (Waterhouse et. al. 2002). Landscape level selection can be tested by measuring the proportion of habitat types within 500m radius circles centred on nest sites and random sites. Habitat selection across these spatial scales can identify structures required for nesting and the habitat contexts they should be provided in. This information can then contribute to management strategies or activities operating at each spatial scale.

Literature Cited

Addison, C. and D. Christie 2002 Final Report, Year 2: 2001 Flammulated Owl Inventory of the Rocky Mountain Trench.

Braumandl, T.F. and M. Curran 1992. A Field Guide for site identification and interpretation for the Nelson Forest Region. Ministry of Forests Land and Management Handbook No. 20

Cannings, R.J. and A.M van Woudenberg 2003. Flammulated Owl Species Account. Standards for managing Identified Wildlife.

Committee on the status of Endangered Wildlife in Canada (COSEWIC) 2003. Canadian Species at Risk

Howie, R.R. and R. Ritcey. 1987. Distribution, habitat selection, and densities of flammulated owls in British Columbia. P. 249-254 *in* Biology and conservation of northern forest owls. USDA For. Serv. GTR-RM-142. Fort Collins, CO.

Linkhart, Brian D., and Richard T. Reynolds. 1997. Territories of flammulated owls: is occupancy a measure of habitat quality? Pages 250-254 in James R. Duncan, David H. Johnson, and Thomas H. Nichols, editors. Biology and conservation of owls of the northern hemisphere. USDA Forest Service General Technical Report NC-190.

Ministry of Forests and Ministry of Environment 1998. Field Manual for describing Terrestrial Ecosystems. Land and Management Handbook No. 25

Resources Inventory Committee. 2001 Inventory methods for raptors (Standards for components of British Columbia's biodiversity; no. 11). http://www.for.gov.bc.ca/ric

Reynolds, R.T. and B.D. Linkhart. 1992. Flammulated owls in ponderosa pine: evidence of preference for old growth. P. 166-169 in the Southwest and Rocky Mountain regions: Proceedings of a workshop (M.R. Kaufmann, W.H. Moir, and R.L. Bassett, tech. coords.). USDA For. Serv. General Technical Report RM-213.

van Woudenberg, A. 1999. Status of the Flammulated Owl in British Columbia. BC Ministry of Environment Lands and Parks, Wildlife Branch WR-95

van Woudenberg, A., D. A. Christie and D. Erikson. 2000. Progress Report:2000 Flammulated Owl Inventory of the Rocky Mountain Trench. Report to HCTF and MELP Region 4 December 2000 12 pp.

Waterhouse, F.L.; R. Bradley; J. Markila; F. Cooke; and L. Lougheed. 2002. *Use of Airphotos to Identify, Describe, and Manage Forest Structure of Marbled Murrelet Nesting Habitat at a Coastal British Columbia Site*. Technical Report TR-016. Research

Section, Vancouver Forest Region, BC Ministry of Forests. Nanaimo, BC.
http://www.for.gov.bc.ca/rco/research/wildlifereports/tr016.pdf

Table 1. Summary of Flammulated Owl Call playback survey effort and owls detections.

Transect	**Date surveyed**	**Transect** length km # of stations	**FLOW** Total detections	**FLOW** Spontaneous detections	**FLOW** Elicited detections	**Other Owls** Species-#	**Estimated Flammulated Owl** number density
Maus Creek	May 22	4.5 km 10 stations	11	11	0	0	6 1.3
Lakit	May 27	8 km 15 stations	11	7	4	BAOW-1	8 1.0
Brewery	May 28	4.5 km 10 stations	16	16	0	BAOW-2	9 2.0
Wolf Creek	May 26	5.5 km 11 stations	11	9	2	0	6 1.09
Lazy Lake	May 23	7.5 km 13 stations	13	12	1	BAOW-1 NPOW-1	6 0.8
Mt Swansea	May 30 June 6	4.5km 10stations	13	10	3	0	9 2.0
Whiteswan	May 29 June 3	12 km 24 stations	25	20	5	BAOW-3 GHOW-2 NPOW-2	16 1.3
Columbia Lake	June 6	2.5km 4 stations	5	1	4	0	4 1.6

Table 2. Summary of nest search efforts for Flammulated owls, East Kootenay 2003

Location	Date	Habitat suitability- comments	Search outcome
Whiteswan Rd 8km site	June 19	Poor foraging roosting and nesting. Some Pp snags but few cavities	-
Lazy Lake 7.5 km site	July 3	Very good foraging and security habitat. Douglas-fir snags for nesting are patchy	**Nest** WHA recommended
Lazy Lake 6 km site	July 3	Very good nesting, foraging and security habitat	WHA recommended
Lazy Lake 4 km site	July 8	Good snags are in rocky areas, some small grassy patches but these lack snags	-
Maus Creek upper site	July 9	Excellent habitat, large Pp snags and abundant cavities	**Nest** WHA recommended
Maus Creek lower site	July 14	Excellent habitat. Pp snags abundant cavities	WHA recommended
Mt Swansea 1 km site	July 10	Excellent foraging habitat, security and nesting habitat are present but not abundant	**Nest** WHA recommended
Mt Swansea 1.5 km site	July 10	Very good nesting, roosting, foraging and security habitat	WHA recommended
Mt Swansea 2.5 km site	July 10	Poor foraging and nesting habitat in this area	-
Brewery Creek 1km site	July 11	Very good nesting, roosting foraging and security habitat	**Nest** WHA recommended
Brewery Creek 4.5 km site	July 11	Lower slope is wetter with thick shrub under story. Upper slope lacks suitable snags and cavities	-
***Columbia Lake** 2km site	July 15	Good nesting, foraging and security habitat	WHA recommended
***Columbia Lake** 3.5 km site	July 15	Good nesting habitat, very good foraging and security habitat	WHA recommended

* These sites are within a Provincial park and WHAs may not apply to this area

Table 3. Time and dates of Owl observations at nest cavities

Nest site	Date	Time	Owl behaviour
***Whiteswan upper**	June 19	~ 10:30	Visible after scratching snag
***Whiteswan Lower**	June 19	~09:00	No owl present
Lazy Lake	July 3	~9:20	Visible after scratching snag
Lazy Lake	July 16	08:30-12:00	Visible briefly during habitat plot
Maus Creek	July 9	12:30	Visible after scratching snag
Maus Creek	July 14	13:00-16:30	Visible briefly during habitat plot
Mt Swansea	July 10	8:20	Visible after scratching snag
Mt Swansea	July 15	13:30-16:00	Visible briefly during habitat plot
Brewery Creek	July 11	8:55	Owl visible on hearing our approach
Brewery Creek	July 14	08:00-12:00	Visible briefly during habitat plot

* nests discovered prior to 2003

Table 4. Characteristics of 4 Flammulated Owl nest trees in the located in the East Kootenay 2003.

	Lazy Lake	Brewery Creek	Maus Creek	Mt. Swansea
Tree Species	Douglas-fir	Douglas-fir	Ponderosa Pine	Douglas-fir
Tree dbh (cm)	42	58	53.4	36.9
Nest tree height (m)	7.2	9	22	5.5
Nest cavity height (m)	7	8.5	10.5	3.9
Nest cavity Orientation (degrees)	192	310	210	98
Decay Class	3	3	5	2
Appearance code	4 top broken	4 top broken	5 top intact	2 top broken
Crown Condition	4	4	4	2
Bark	3 5-25% lost	2 <5% lost	4 26-50% lost	2 <5% lost
Wood Condition	4 mostly hard but decay spreading	3 limited decay	5 balance of hard and soft wood	3 limited decay

* For definitions of Appearance, Crown condition, Bark retention and Wood condition see Field Manual for Describing Terrestrial Ecosystems' (Ministry of Forests and Ministry of Environment 1998).

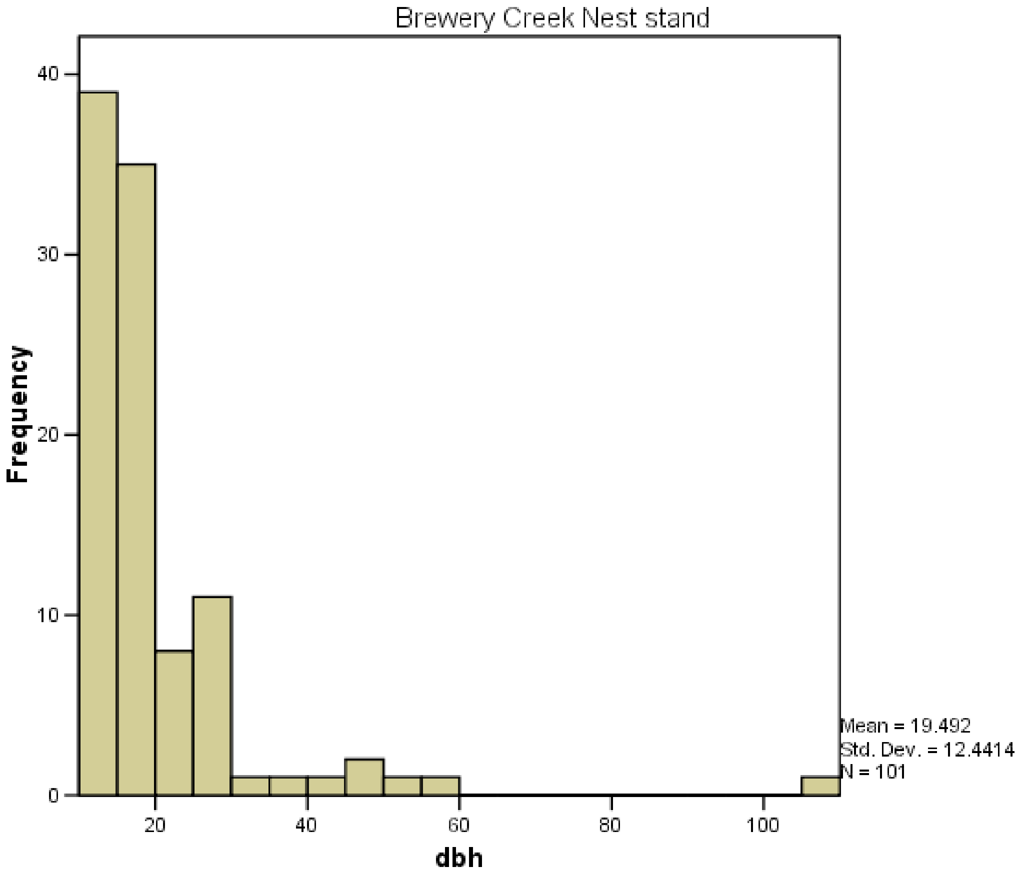
Table 5. Characteristics of 4 Flammulated Owl nest sites in the located in the East Kootenay 2003.

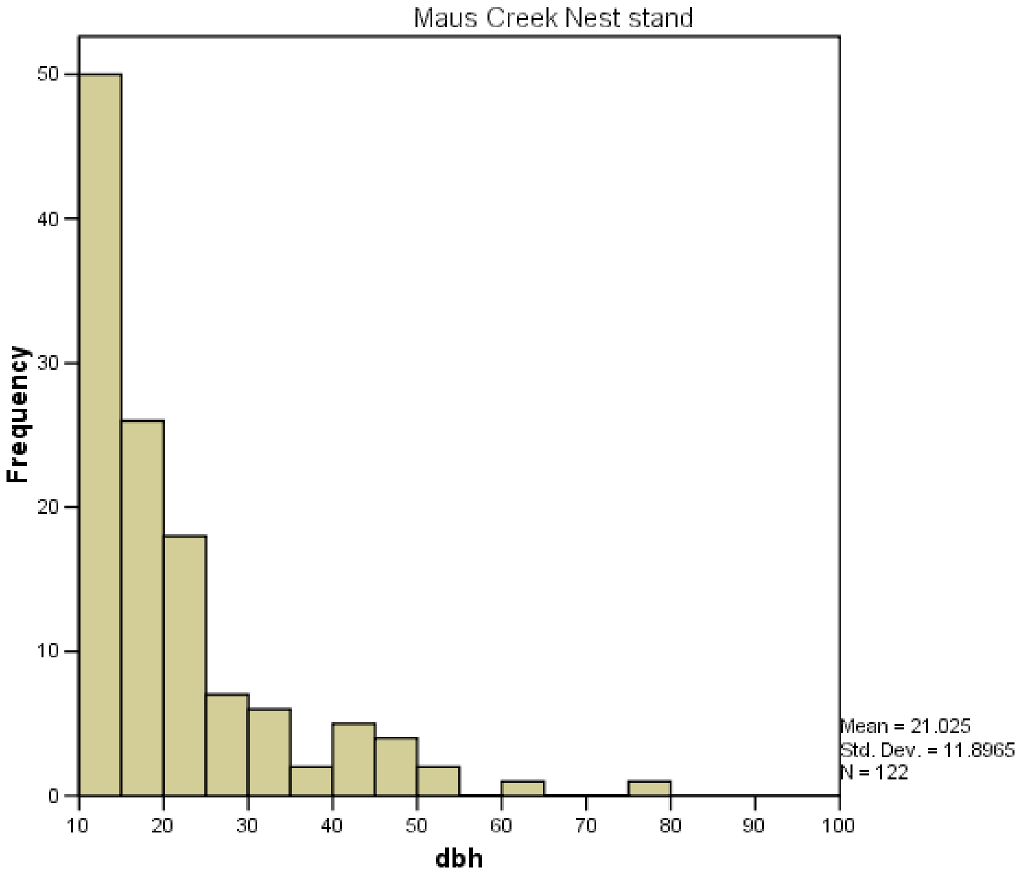
	Lazy Lake	Brewery Creek	Maus Creek	Mt. Swansea
UTM ZEN	11U 599730 5520293	11U 601346 5502920	11U 602338 5449262	11U 574358 5593290
Slope %	70	78	62	73
Aspect (degrees)	260	180	230	190
Elevation (m)	1115	1156	1068	1100
Structural Stage	6	7	7	6
BGC subzone variant	IDF dm2-03	IDF dm2-03	IDF dm2-03	IDF dm2-02
Forest cover	Fd	Fd	Fd	Fd
Polygon	78	210	210	78
Leading species and estimated **Stand age**				
Crown Closure %	15	25	32	10
Shrub % Cover	66	17	14	32
Herb layer % Cover	14	41	25	20

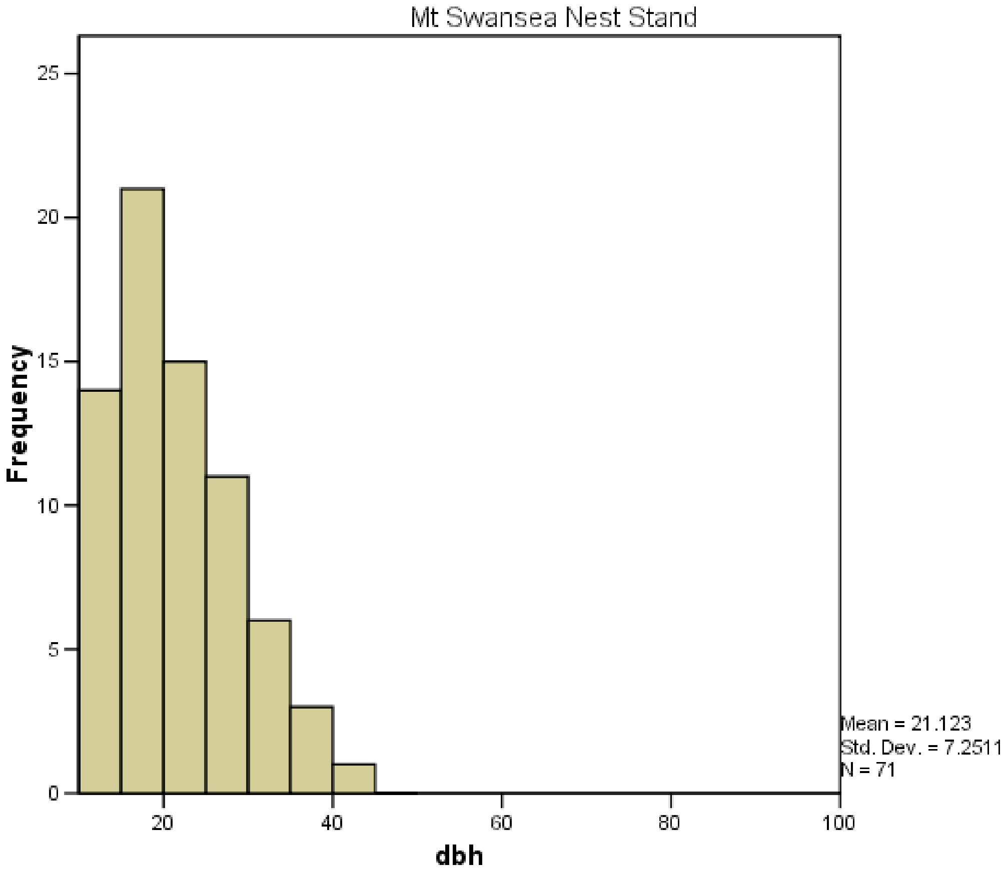
Table 6. Characteristics of 4 Flammulated Owl nest Stands located in the East Kootenay 2003.

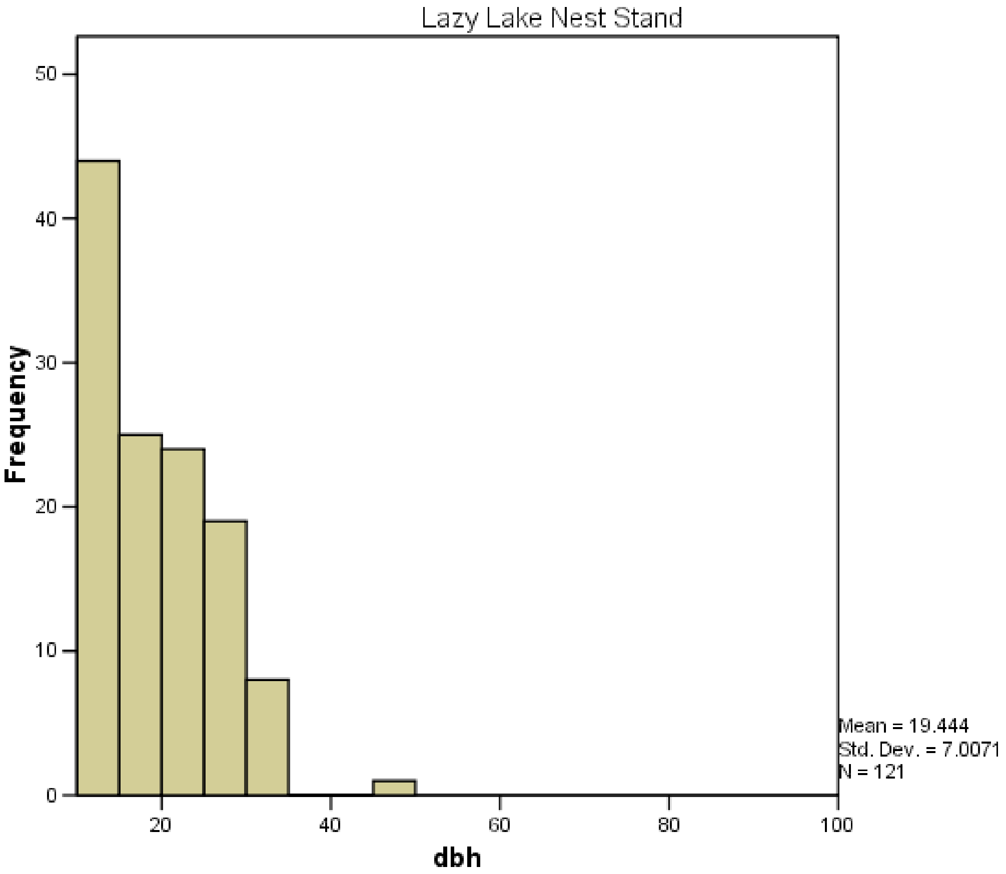
	Lazy Lake	**Brewery Creek**	**Maus Creek**	**Mt. Swansea**
Stems/ha <10 cm dbh	489	550	1065	15
Stems/ha >10cm dbh	616	514	621	361
Stems/ha 10-29 cm dbh	560	474	504	305
Stems/ha 30-49 cm dbh	46	25	87	51
Stems/ha 50-69cm dbh	0	10	15	0
Stems/ha >70 cm dbh	0	5	5	0
Tree height Mean (St.dev.)	10.9 ± (3.6)	10.6 ± (3.6)	12.7 ± (4.5)	8.9 ± (2.4)
Tree dbh Mean (St.dev.)	19.4 ± (7.0)	19.5 ± (12.4)	21.0 ± (11.9)	21.1 ± (7.3)
Snags /ha	25	36	20	82
Snag dbh Mean (St.dev.)	21.9 (12.4)	30.9 (21.8)	47.9 (13.4)	28.9 (8.9)
Snag ht Mean	4.5	8.7	18.7	7.9
Large Snags/hectare >35cm dbh	5	10	20	20
Large Snags Dbh Mean (St.dev.)	42	61.5 (4.9)	47.8 (13.4)	38.6 (3.4)
Coarse woody debris 25m transect	1piece 25cm dbh decay class 6	2pieces 15, 40 cm dbh decay class 4	2pieces 10 cm dbh decay class 3-5	1 piece 20 cm dbh decay class 3

Figure 9. Tree diameter frequency histograms (trees >10cm dbh) for four Flammulated Owl nest sites in the East Kootenay 2003.




Maus Creek Nest stand
Frequency
50
40
30
20
10
0
10
20
30
40
50
60
70
80
90
100
dbh
Mean = 21.025
Std. Dev. = 11.8965
N = 122


Mt Swansea Nest Stand
Frequency
25
20
15
10
5
0
20
40
60
80
100
dbh
Mean = 21.123
Std. Dev. = 7.2511
N = 71


Lazy Lake Nest Stand
Frequency
0
10
20
30
40
50
20
40
60
80
100
dbh
Mean = 19.444
Std. Dev. = 7.0071
N = 121

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How important is dead wood for woodpeckers foraging in eastern North American boreal forests?

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ABSTRACT

Dead and decaying trees may be a limited resource for woodpeckers in managed forests, especially for species that rely on dead wood for nesting and foraging. Whereas recent nest web studies greatly increased our understanding of nest tree use by woodpeckers, knowledge on woodpeckers foraging requirements is much less developed. We quantified and compared tree selection patterns and foraging behavior of six bark-foraging woodpeckers – downy woodpecker (*Picoides pubescens*), hairy woodpecker (*Picoides villosus*), American three-toed woodpecker (*Picoides dorsalis*), black-backed woodpecker (*Picoides arcticus*), yellow-bellied sapsucker (*Sphyropicus varius*) and pileated woodpecker (*Dryocopus pileatus*) – that co-occur in eastern boreal forests of North America. A total of 271 observation bouts and more than 600 foraging trees were recorded at three study sites characterized as mixedwood, conifer, and burn. Our results show that dead wood represents an important foraging substrate for most bark-foraging woodpeckers in Canadian eastern boreal forests. However, significant differences in individual species were found with regard to substrate use patterns, foraging behavior and associated prey. Woodpeckers were categorized according to their selection for specific stages of tree degradation, with the yellow-bellied sapsucker and the pileated woodpecker representing opposite ends of this gradient. The black-backed woodpecker showed the highest use of dead wood and was very specific in its tree selection by using mostly recently dead trees. We emphasize that providing foraging substrates for most woodpecker species not only requires maintaining dead wood but also paying heed to the underlying dynamics of dead wood (e.g. recruitment and degradation) in managed boreal forest landscapes.

1. Introduction

Dead wood is a key component of biodiversity in forest ecosystems worldwide. Decaying and dead trees provide habitat resources for thousands of species such as wood-inhabiting fungi, saproxylic invertebrates and cavity-nesting vertebrates (Raphael and White, 1984; Grove, 2002; Cockle et al., 2011; Stokland et al., 2012). Saproxylic species – defined as "species that depend, during some part of their life cycle, upon wounded or decaying woody material from living, weakened or dead trees" (Stokland et al., 2012) – show strong affinities to specific tree hosts, decay stages, tree sizes and microhabitat conditions and are sensitive to the abundance of their preferred dead wood substrates in both managed and unmanaged forests (Siitonen, 2001; Stokland et al., 2012).

In the boreal forest, forestry practices still include the extensive use of low tree retention clearcuts and short harvest rotations, which result in a significant decrease in the abundance and diversity of dead wood as well as its associated biodiversity (Siitonen, 2001; Grove, 2002; Jonsson and Siitonen, 2012). In European boreal forests, intensive forest management has led to the decline or to the local extirpation of several saproxylic species (Angelstam and Mikusiński, 1994; Berg et al., 1994; Siitonen, 2012). In North American boreal forests, maintaining dead wood in managed forests is often identified as a critical issue given the extent of even-aged management and the increase of salvage logging after natural disturbances (Hannon and Drapeau, 2005). In different regions of the North American boreal forest, ecosystem-based management strategies are now aimed at providing an adequate representation of cover types and stand age structure at landscape scales (Bergeron et al., 2002; Gauthier et al., 2009), with harvesting practices such as variable retention harvest or partial cutting that maintain variable amounts of dead trees as well as significant green-tree retention in harvested blocks (Sullivan et al., 2001; Serrouya and D'Eon, 2004; Fenton et al., 2009). Although these

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0378-1127/

new approaches likely contribute to the conservation of biodiversity, their efficiency to maintain saproxylic species still needs to be assessed (but see Cooke and Hannon, 2012). Specifically, decisions regarding management targets and prescriptions (e.g. amount of old-growth forests at landscape scales, levels and types of retention of live and dead trees in harvested blocks) will likely influence the persistence of saproxylic species populations in managed landscapes. Knowledge on these species habitat requirements as well as their dependence to dead wood may help identify the species most sensitive to the effects of forest management (focal species *sensu* Lambeck, 1997) and may be used to improve conservation planning of saproxylic species assemblage in managed landscapes.

Woodpeckers play an important ecological role in forest ecosystems by providing cavities to a broad range of vertebrate and invertebrate species ("nest-web"; Martin and Eadie, 1999; Wesolowski, 2011). These keystone species may be particularly important in conifer-dominated boreal forests where natural cavities are much less abundant (e.g. Aitken and Martin, 2007; Cockle et al., 2011). Dead wood is often identified as a critical habitat attribute for woodpeckers nesting, given that many species are known to prefer snags or living trees with decaying heartwood for their nest cavities (e.g. Raphael and White, 1984; Blanc and Martin, 2012). Yet, dead wood may also be a critical component of woodpeckers' food web. Indeed, snags are critical habitats for saproxylic insects (Saint-Germain et al. 2004, 2007), which are important prey of many woodpecker species (Murphy and Lehnhausen, 1998; Imbeau and Desrochers, 2002; Nappi et al., 2003). Given the number of trees required for foraging, woodpecker populations could be much more limited by the availability of suitable foraging substrates than by potential nest trees, and may thus in turn be more sensitive to the reduction of decaying and dead trees in managed landscapes (Imbeau et al., 2001). A decrease in the abundance of woodpeckers may thus have a cascading effect on the abundance of cavity-nesting species and on the nest-web community structure. Knowledge on woodpecker foraging requirements, in addition to nesting habitat features, may thus be crucial for setting dead wood conservation targets that could maintain the complex ecological network associated with dead wood (i.e. saproxylic food and nest webs).

Use and partitioning of foraging resources among sympatric woodpecker species has received much attention in North America and Europe (e.g. Hogstad, 1971; Bull et al., 1986; Török, 1990). Although use of decaying and dead trees has often been reported, few studies have documented the selection *per se* (disproportionate use of resources as compared to their availability; Johnson, 1980) of dead wood by foraging woodpeckers and their differential tree selection patterns. In the North American boreal forest, the few studies on foraging ecology of woodpeckers have mostly focused on single species in one habitat type and were restricted to coniferous landscapes (e.g. Imbeau and Desrochers, 2002; Tremblay et al., 2010; Nappi and Drapeau, 2011). Foraging requirements and the relative importance of dead wood as a foraging substrate have yet to be quantified for most boreal woodpeckers.

We studied the foraging ecology of the six bark-foraging woodpecker species that co-occur in the eastern North American boreal forest: downy woodpecker (*Picoides pubescens*), hairy woodpecker (*Picoides villosus*), American three-toed woodpecker (*Picoides dorsalis*), black-backed woodpecker (*Picoides arcticus*), yellow-bellied sapsucker (*Sphyropicus varius*) and pileated woodpecker (*Dryocopus pileatus*). Foraging ecology was examined by analyzing foraging tree selection, foraging behavior and woodpeckers' prey. Our study was conducted in different forest cover types – mixedwood, conifer and burned conifer stands – representative of the natural forest landscape in eastern Canada. More specifically, our study addresses the following questions: (1) what is the relative importance of dead wood as a foraging substrate for woodpeckers in the boreal forest? and (2) how do these species differ in foraging tree selection and foraging behavior?

2. Methods

2.1. Study sites

The study area is part of the northern Clay Belt of Quebec and Ontario, a large physiographic region dominated by clay deposits. Forest composition shows a latitudinal transition from mixedwood forests in the south (*Abies balsamea–Betula papyrifera* bioclimatic domain) to conifer-dominated forests in the north (*Picea mariana*-moss bioclimatic domain; Saucier et al., 1998). Fire and insect outbreaks are the main natural disturbances in these forest landscapes. We selected one study site in the southern mixedwood and two sites in the northern coniferous forest. Whereas these three sites are part of the same physiographic region, they are spatially dispersed one from another because we were interested in studying woodpeckers' foraging in unmanaged forests that represented the range of natural forest conditions (composition and structure) in this region.

The mixedwood site ("MXW") is located at the Lake Duparquet Research and Teaching Forest (LDRTF; 48°30′ N, 79°22′ W; Fig. 1). The LDRTF is a 8045-ha forest landscape composed of mainland, islands and peninsulas. The mainland fire regime is characterized by stand-replacement fires: thirteen fires within LDRTF over the last three centuries have created a complex natural forest mosaic (Harvey, 1999). Stand composition varies according to time since fire, from early seral stands dominated by deciduous (trembling aspen (*Populus tremuloides*), paper birch (*Betula papyrifera*)), to mixed stands (with white spruce (*Picea glauca*)), to coniferous stands (balsam fir (*Abies balsamea*), eastern white cedar (*Thuya occidentalis*)) (Bergeron, 2000). Black spruce (*Picea mariana*) and jack pine (*Pinus banksiana*) occur in localized areas as well. Three spruce budworm (*Choristoneura fumiferana*) outbreaks occurred in the last century, the most recent between 1970 and 1987, an event that was especially severe in balsam fir-dominated stands (Bergeron et al., 1995). Our study took place in the eastern part of the LDRTF mainland, a conservation area that has been lightly affected by anthropogenic disturbances.

The conifer-dominated study site ("CON") is located at the Muskuuchii Hills Projected Biodiversity Reserve (50°12′ N, 78°43′ W; Fig. 1). The biodiversity reserve covers 80,100 ha, of which half consists of peat bogs on organic deposits that support black spruce stands of varying densities. The other half is composed of terraces and hills characterized by well-drained till, sand and fine sediment deposits (Gouvernement du Québec, 2008). Our study took place in a portion of the landscape dominated by mature stands (>120 years) on mesic sites. From 1998 to 2000, experimental partial cuts were conducted in a case-control manner that resulted in a mosaic of intact and partial cut stands. Black spruce and jack pine dominate forest composition. Other species include balsam fir, trembling aspen and paper birch.

The third study site is a 8-year-old coniferous burn landscape ("BURN"), located 200 km east of the "CON" site (50°30′ N, 75°43′ W; Fig. 1). Vegetation is dominated by black spruce and jack pine with scattered white birch and trembling aspen. Burn severity was highly variable, with close to 50% of the area consisting of unburned and low-severity burned stands (details in Nappi et al., 2010).

Woodpecker foraging observations were collected in predefined large sampling blocks at the MXW and CON sites (Fig. 1). At the MXW site, sampling blocks were distributed in four 60-year classes (60–120, 120–180, 180–240 and >240 years), based on fire history mapping (Dansereau and Bergeron, 1993). We selected three sampling blocks in forests of each age class (total of 12 blocks). Each

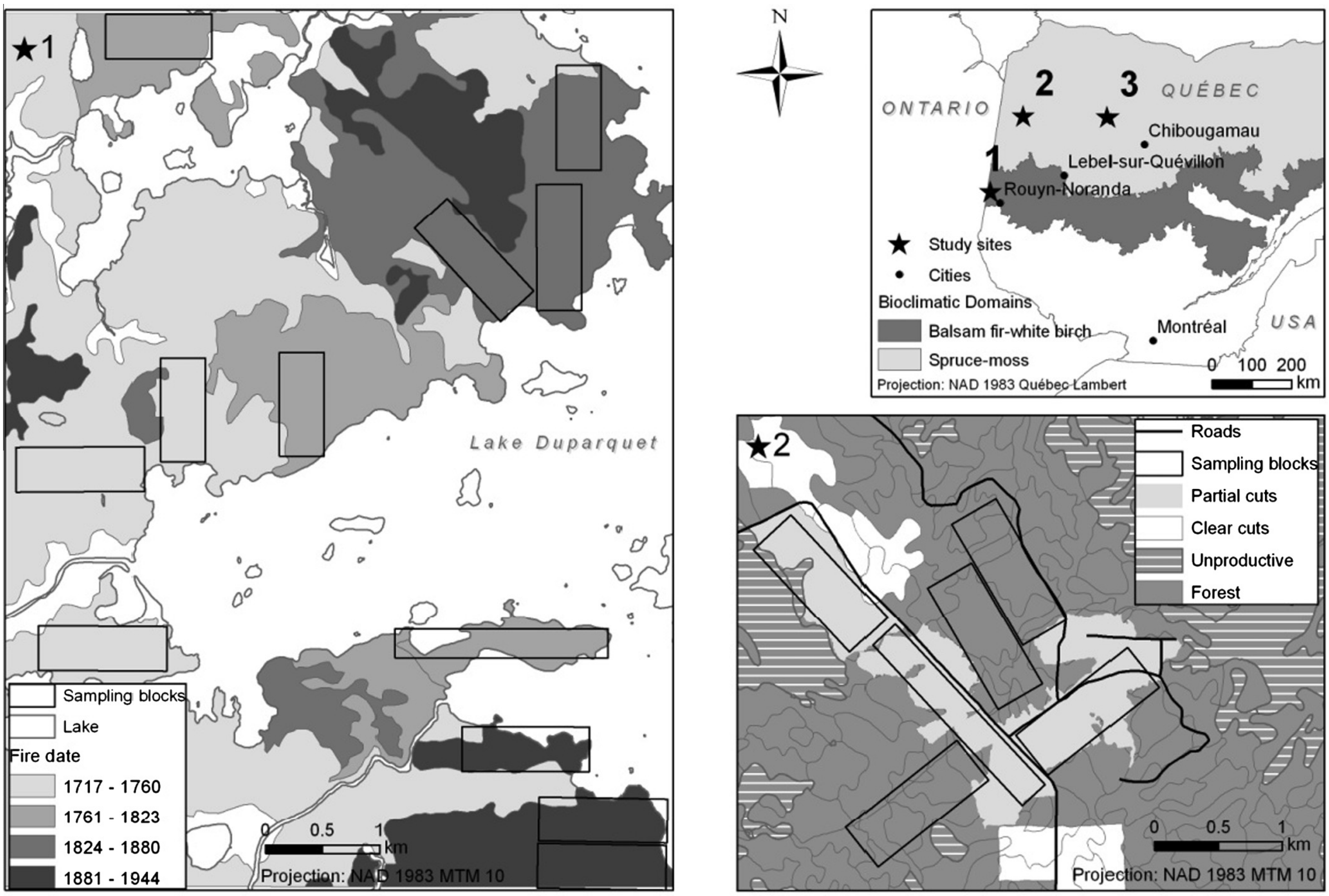

Fig. 1. Location of study sites (1 – Mixedwood site; 2 – Conifer site; 3 – Burn site) in the boreal forest of Quebec, Canada. Enlargements show location of sampling blocks at the mixedwood and conifer sites within which foraging observations were made.

sampling block consisted of one or two linear transects, generally 1 km long, with a 100-m buffer on each side within which woodpecker foraging observations were made. Blocks ranged from 24 to 40 ha in size (total of 448 ha). At the CON site, six sampling blocks ranging from 20 to 48 ha (total of 268 ha) were selected. Three of these were in unharvested mature stands whereas the other three were located in partially harvested mature stands. The blocks were designed to represent large but homogeneous areas in terms of stand age, structure and composition. At the BURN site, no sampling blocks were used and foraging activities of black-backed woodpeckers were recorded opportunistically to provide additional information on the foraging ecology of this species in burned forest habitats.

2.2. Foraging observations

Observations of foraging woodpeckers were made from mid-May to early July (breeding season of these species in our study area) in 2003 and 2004. For MXW and CON sites, observers walked systematically along the predefined linear transects. When a bird was heard or seen within 100 m of the transect line, it was followed until it flew out of sight or up to a maximum of 10 min (hereafter an “observation bout”). An observation bout had to involve at least one foraging technique (see below) and could include a single or multiple trees. Because birds were not banded, a sampling procedure was used in order to reduce the possibility of resampling the same individuals. After data was collected on a given individual, we continued walking the line transects until we found an individual of a different gender or species. Two consecutive observation bouts of the same species and gender had to be separated by at least an hour. For each species, a similar proportion of males and females were sampled. We also distributed our sampling efforts among sampling blocks so that observations of foraging birds were made at the highest number of different locations as possible. At the BURN site, although no sampling blocks were used to record foraging observations, similar sampling procedures were taken to cover different areas of the burned landscape. Observations were distributed among point count sampling stations used for another research project (Nappi et al., 2010).

During each observation bout, we recorded foraging activities continuously using a recording system. Information was later transcribed using instantaneous sampling (i.e. fixed-interval time point; Martin and Bateson, 1993). Observation bouts were divided into 5-s periods at the end of which we noted the corresponding predefined foraging activity. We recorded the following four variables related to foraging behavior: foraging height, tree section, substrate condition and foraging technique. Foraging height was recorded in four classes: lower, middle and upper third of the tree, and coarse woody debris (downed logs and stumps <1 m height). Tree section corresponded to the specific part of the tree used: trunk, branch, junction of trunk-branch and foliage. Substrate condition referred to wood condition where the bird was foraging and was noted as live or dead. We distinguished between five foraging techniques following an adaptation of classifications used by other authors (Hogstad, 1976; Murphy and Lehnhausen, 1998; Imbeau and Desrochers, 2002): gleaning (picking insects from the surface and within bark fissures); pecking (striking the wood superficially); scaling (flaking off the bark); excavating (digging holes to access deep wood-dwelling arthropods); sap licking (digging sap holes and sucking sap from ringed trees).

Each tree with recorded foraging activities was marked and was characterized (tree species, diameter at breast height (dbh), tree position (standing or fallen) and tree degradation) at the end of the observation bout. Tree degradation was classified based on visual appearance in five categories according to a modified version of Maser et al. (1979): Deg1 (‘‘live healthy’’, >20% green foliage); Deg2 (‘‘live decaying’’, <20% green foliage); Deg3 (‘‘recently dead’’, hard wood, firm bark cover, dead foliage or small twigs); Deg4 (‘‘moderately degraded’’, soft wood, some bark missing, no dead foliage or small twigs, main branches remaining, usually intact top); Deg5 (‘‘highly degraded’’, decayed wood, little bark remaining, very few branches, often broken top). Observation bouts were difficult to record for the pileated woodpecker so we searched for the typically large and deep foraging excavations of the pileated woodpecker (Lemaître and Villard, 2005) and characterized trees with recent foraging excavations (<1 year, based on fresh wood chips).

To compare characteristics of trees used for foraging with available trees, we collected vegetation data in sampling blocks (for the MXW and CON sites only). At each block, sampling plots 200 m apart were distributed along the linear transects (MXW = 112; CON = 34). Each sampling plot was 0.06 ha (60 × 10 m), a minimum size for estimating the density of the entire range of tree degradation stages (e.g. including rare degradation classes). We sampled all standing trees >5 cm of dbh and noted species, dbh and degradation stage of each tree.

2.3. Wood-dwelling arthropods

To investigate the link between tree selection, foraging behavior and prey type, we collected arthropods through wood dissection in a subset of trees that were used for foraging by the four *Picoides* species. Wood dissection is a useful technique for inspecting potential woodpecker prey as it gives an instant and exact portrait of the arthropod assemblage, including deep wood-dwelling insects, present in selected trees (Saint-Germain et al., 2007; Nappi et al., 2010). For each woodpecker species, we selected the trees most intensively used in 2004 (based on observed foraging time). A total of 47 foraging trees were cut down and dissected to collect and identify wood-dwelling arthropods (10 for downy and 10 for hairy woodpeckers at the MXW site; 10 for black-backed and 10 for American three-toed woodpeckers at the CON site; 10 for the black-backed woodpecker at the BURN site).

Tree cutting and dissection were conducted during the last two weeks of June 2004 to obtain a representative sample of the prey species present in the wood when foraging observations were made. From each tree, two 1-m bole segments were taken, the first at the base of the tree (0–1 m) and the second at 4 m (conifers) or at half of the tree height (deciduous trees). These wood samples were taken to the laboratory for wood dissection and all arthropods were identified to family, genus or species depending on available identification criteria (see methods in Saint-Germain et al., 2007). All specimens were classified by their length (> or <1 cm) and by the portion of the bole in which they were found (‘‘bark-associates’’: within or under the bark; ‘‘wood-associates’’: within sapwood or heartwood).

2.4. Statistical analyses

Foraging tree selection analyses were based on observations made at the MXW and CON sites. For each woodpecker species, foraging tree selection was assessed by comparing trees used for foraging with available trees in neighboring 0.6 ha sampling plots. We restricted our analyses to standing trees since these comprised the vast majority of substrates used and these could be directly compared to available standing tree data. Foraging tree selection was assessed using random-effect discrete-choice logit models (Cooper and Millspaugh, 1999; Güthlin et al., 2011; Kneib et al., 2011). This approach models the probability of a tree being used for foraging by identifying important characteristics of tree selection with regard to tree species, tree diameter and tree degradation stage. Tree availability may also be defined separately for each individual or location, so that trees identified as available were likely accessible to the animal. In our analyses, foraging trees were compared to available trees characterized at the closest sampling plot. We also used the vegetation sampling plot as our ‘‘observational unit’’ for the analyses: all observations close to (or within) a sampling plot for the same species were assumed to involve the same individual. This decision was applied to the yellow-bellied sapsucker and to the downy, black-backed and American three-toed woodpeckers and was based on location of nests (*P. Drapeau, unpublished data*) which suggested that observations made close to different vegetation sampling plots were from different individuals. For hairy and pileated woodpeckers, we used the sampling block (instead of the sampling plot) as the observational unit, given the much larger home-ranges of these species in our study area. In our analyses, the observational unit is considered as a random effect allowing for each unit specific deviations in selection preferences from the general model. Random effects are assumed to follow a normal distribution. Analyses were performed with the R package (Viton, 2014).

Differences in the foraging ecology of woodpeckers were investigated using discriminant analysis. Foraging behaviors recorded during observation bouts were compiled for each variable as a percentage of time per observation bout (Pechacek, 2006). For instance, if a given bird was observed excavating a total of 6 time points during a 60-s observation bout, the percentage of time spent excavating would have been 50% (6 of 12 time points). We also included tree use variables in the analysis (tree species (deciduous or coniferous) and dbh). Tree species was compiled as percentage of time per observation bout whereas dbh corresponded to the mean dbh of all trees used during each observation bout. Some variables were excluded from the analysis because of high multicollinearity (e.g. dead vs live substrates). Each observation bout was weighted so that all observation bouts of the same species and gender at a given sampling plot contributed to only one degree of freedom in the analysis. The pileated woodpecker was not included in the analysis because we had no foraging observations. Also, a preliminary analysis involving the yellow-bellied sapsucker showed a very distinct foraging behavior of this species that obscured the comparison of the other woodpecker species. Therefore, we restricted our analysis to the four *Picoides* species. Discriminant analysis was performed using SPSS 15.0. Mean proportions of foraging time for variables presented in tables and figures are based on weighted means (weighting by sampling station).

Wood-dwelling arthropods were compared among trees used by the different *Picoides* species. Individuals were pooled by tree (sum of two bole segments) and density was calculated as the number of individuals per square meter of bark sampled. Densities for each wood-dwelling arthropod group (e.g. size, wood association or family) with at least 20 individuals were compared between trees used by co-occurring species at each site (downy and hairy woodpeckers at the MXW site; black-backed and American three-toed woodpeckers at the CON site) or by the black-backed woodpecker at the CON and BURN sites using Mann–Whitney non-parametric tests.

3. Results

Foraging activities of woodpeckers were recorded during 271 observation bouts (Table 1), for a total of 9592 foraging observations (5-s time points). For all species combined, standing trees ($n = 627$) represented 82% of all substrates used for foraging (Table 1, last column). Standing trees represented the majority of foraging substrates, except for the black-backed woodpecker which used logs and stumps in higher proportion.

3.1. Selection of foraging trees

Discrete-choice models revealed clear and distinct patterns of tree selection among species (Table 2). Tree degradation was an important selection criterion for most species (Table 2, Fig. 2a). The yellow-bellied sapsucker avoided snags and showed a preference for live decaying trees. Downy woodpeckers used trees in different degradation stages and only avoided highly degraded snags. The American three-toed woodpecker showed no significant preference for either tree degradation class. The black-backed woodpecker showed a high preference toward live decaying and recently dead trees. Recently dead trees alone represented 70% of all foraging trees of the black-backed woodpecker (Fig. 2a). The hairy woodpecker avoided live healthy trees and selected other degradation stages, from live decaying trees to highly degraded snags. The pileated woodpecker showed a preference for highly degraded snags; these represented 56% of all foraging trees of this species.

The yellow-bellied sapsucker, the downy woodpecker and the hairy woodpecker showed a clear preference for deciduous trees (paper birch and/or trembling aspen) (Table 2, Fig. 2b). Pileated woodpeckers used both deciduous (mostly dead trembling aspen) and conifer trees (mostly live eastern white cedar and dead balsam fir). American three-toed woodpeckers selected conifers (mainly black spruce) whereas black-backed woodpeckers used conifers in the same proportion than their availability. Three species – the black-backed, the hairy and the pileated woodpeckers – selected larger trees (Table 2, Fig. 2c). Mean dbh of foraging trees were respectively 18.5, 26.7 and 26.9 cm for these species.

Tree selection was further investigated for the hairy woodpecker at the MXW and CON sites separately. We found similar patterns in tree selection with regard to degradation, tree species or dbh (the only difference being the lack of preference for Deg4 at the CON site).

3.2. Differences in foraging behavior

Picoides species showed a clear partitioning of their foraging behavior based on both substrate use and foraging technique (Fig. 3, Table 3). The first two discriminant functions accounted for 94% of the explained variance. The first discriminant function mainly partitioned *Picoides* based on their relative use of conifers and the scaling foraging technique. The second discriminant function was mainly associated with the relative use of dead wood and excavation technique.

The downy woodpecker showed the less variation in foraging behavior in comparison with other *Picoides* species (Fig. 3). This species foraged mainly on deciduous trees, made extensive use of branches and used pecking and gleaning as its main foraging techniques (Figs. 3 and 4, Table 3). The hairy woodpecker occupied an intermediate position on the discriminant function scatter plot but overlapped broadly with the downy woodpecker. The main differences between these two species were the higher use of dead wood and the broader range of foraging techniques used by the hairy

Table 1
Number of observation bouts and foraging trees recorded for six woodpecker species at the mixedwood (MXW), conifer (CON) and burn (BURN) sites.

Species	Observation bouts				Foraging trees				
	MXW	CON	BURN	Total	MXW	CON	BURN	Total	% of all substrates
Yellow-bellied sapsucker (YBSA)	74	5		79	202	13		215	100
Downy woodpecker (DOWO)	64			64	159			159	99
Hairy woodpecker (HAWO)	33	12		45	84	19		103	82
American three-toed woodpecker (ATWO)		34		34		49		49	68
Black-backed woodpecker (BBWO)		37	12	49		43	15	58	35
Pileated Woodpecker (PIWO)[a]					41	2		43	100
Total	171	88	12	271	486	126	15	627	82

[a] Based on recent foraging excavations (see methods).

Table 2
Woodpecker species preferences for different degradation stages, tree species and tree diameter (dbh). Reference degradation class corresponds to live healthy trees (DEG1). Tree species refers to preference (positive estimate) or avoidance (negative estimate) of deciduous trees. The number of sampling plots or blocks used as "observational units" in the discrete-choice model analyses are indicated (n). Odd ratios provide information on the relative preference for a woodpecker species to forage on a specific class of tree degradation or tree species whereas in the case of a continuous variable such as tree diameter, odd ratios indicate the relative increase in the probability of a tree to be used with one unit increment in dbh.

Variable	Yellow-bellied sapsucker ($n = 35$)		Downy woodpecker ($n = 36$)		Hairy woodpecker ($n = 11$)		American three-toed woodpecker ($n = 12$)		Black-backed woodpecker ($n = 9$)		Pileated woodpecker ($n = 12$)	
	β	Odd ratio	β	Odd ratio	β	Odd ratio	β	Odd ratio	β	Odd ratio	β	Odd ratio
Degradation												
Live decaying (DEG2)	0.916[***]	2.499	0.551	1.735	3.000[***]	20.070	0.898	2.454	3.966[***]	52.792	1.037	2.821
Recently dead (DEG3)	−1.608[**]	0.200	−0.902	0.406	2.981[***]	19.716	0.727	2.068	2.717[***]	15.140	0.317	1.373
Moderately degraded (DEG4)	−2.060[***]	0.127	−0.719	0.487	2.485[***]	12.000	−17.057	<0.001	2.254[**]	9.526	1.251	3.493
Highly degraded (DEG5)	−3.715[***]	0.024	−1.730[***]	0.177	0.834[*]	2.302	−1.302	0.272	0.877	2.403	3.302[***]	27.166
Tree species (deciduous)	1.015[***]	2.759	4.224[***]	68.285	2.164[***]	8.707	−1.974[**]	0.139	−1.725	0.178	−0.220	0.803
Tree diameter	0.067	1.069	0.063	1.065	0.085[***]	1.089	0.046	1.047	0.322[***]	1.380	0.135[***]	1.144

Significance level.

[*] $P < 0.05$.

[**] $P < 0.01$.

[***] $P < 0.001$.

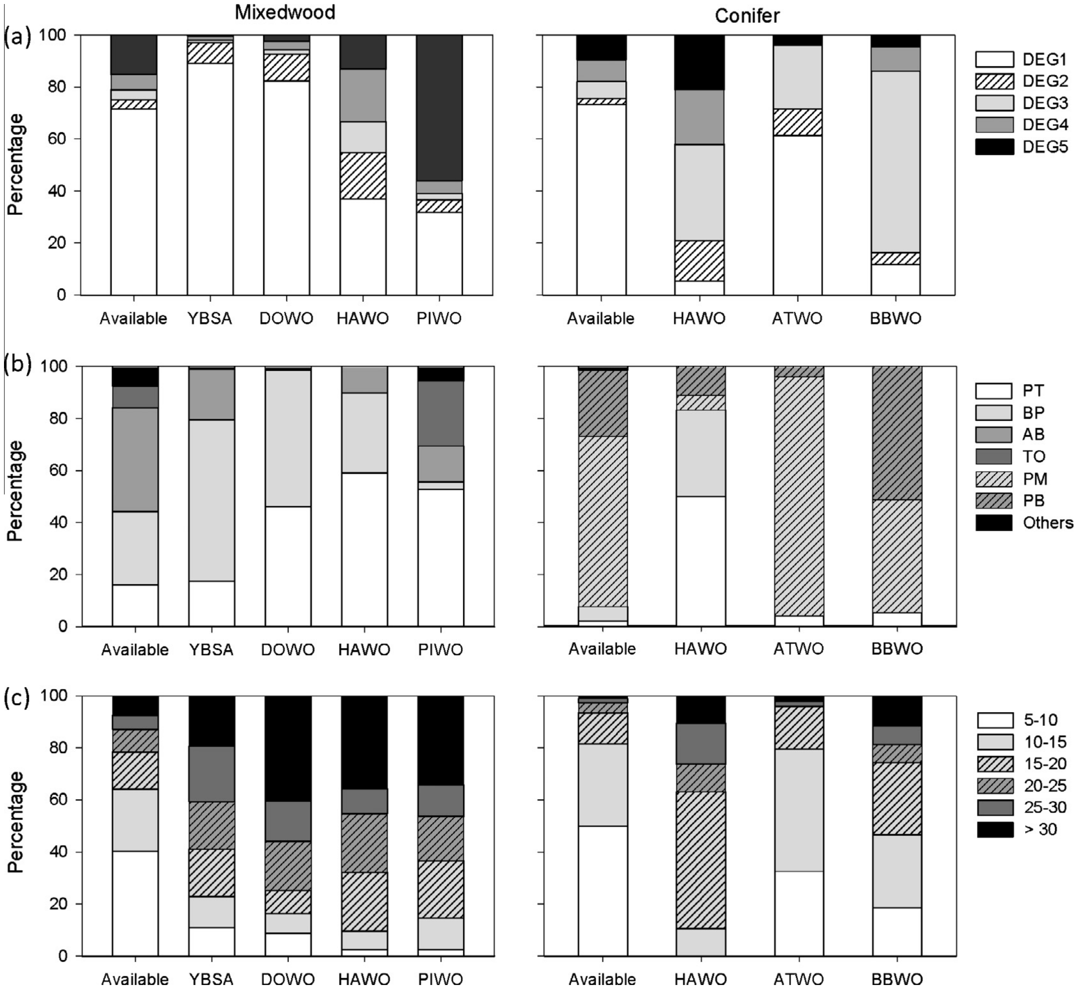

Fig. 2. Proportion of available trees and trees used by foraging woodpeckers based on (a) degradation stage, (b) tree species and (c) diameter at breast height (dbh) class (see Table 1 for woodpecker species codes). Degradation classes: Deg1 = live healthy, Deg2 = live decaying, Deg3 = recently dead, Deg4 = moderately degraded, Deg5 = highly degraded. Tree species: PT = *Populus tremuloides*, BP = *Betula papyrifera*, AB = *Abies balsamea*, TO = *Thuya occidentalis*, PM = *Picea mariana*, PB = *Pinus banksiana*. Results are presented for species with more than 15 foraging trees at the mixedwood or conifer site. "Availability" is based on the overall availability of tree substrates based on all vegetation sampling plots.

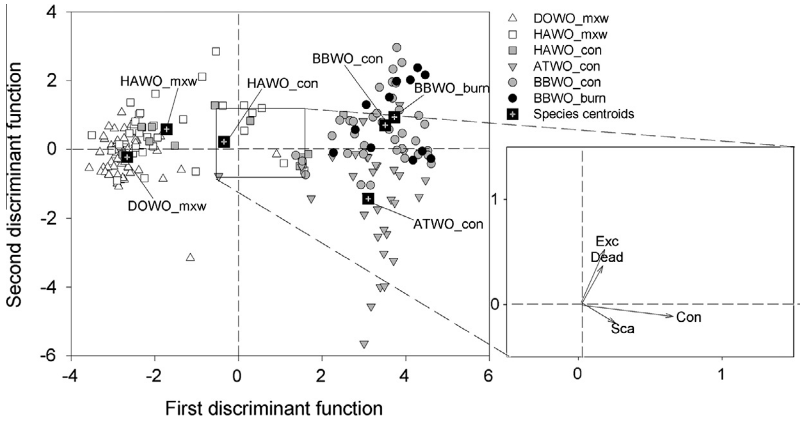

Fig. 3. Partitioning of *Picoides* woodpeckers' foraging behavior on a discriminant analysis diagram. Group centroids are the mean discriminant scores for each group of species and sites (MXW = Mixedwood site; CON = Conifer site; BURN = burn site). See Table 1 for species codes and Table 3 for foraging variables. The two variables most highly correlated with each discriminant function are shown.

woodpecker at both the mixedwood and conifer sites (Figs. 3 and 4).

The American three-toed woodpecker was mainly associated with the use of conifers and scaling (Figs. 3 and 4). It was the only woodpecker to use all bark-foraging techniques as well as the only *Picoides* species to use sap-licking frequently. The black-backed woodpecker was the species most associated with dead wood, including coarse woody debris, and was the species that used most

Table 3
List of explanatory variables included in the discriminant analysis to compare foraging behavior of *Picoides* woodpeckers. Some variables were excluded from the analysis because of high multicollinearity (e.g. dead vs live substrates). Values indicate correlations of these variables with discriminant functions (structure matrix).

Variable	Code	Discriminant function	
		First	Second
Tree species			
Conifer	Con	0.77	−0.12
Tree diameter	DBH	−0.23	0.32
Foraging height			
Upper third	Upp	−0.21	−0.03
Middle third	Mid	−0.02	−0.23
Coarse woody debris	CWD	0.22	0.23
Tree section			
Branches	Bra	−0.27	−0.01
Substrate condition			
Dead wood	Dead	0.20	0.45
Foraging technique			
Scaling	Sca	0.31	−0.23
Excavating	Exc	0.20	0.60
Pecking	Pec	−0.25	0.00
Gleaning	Gle	−0.18	−0.05

frequently the excavating foraging technique (Figs. 3 and 4, Table 3). A great overlap in the foraging behavior of this species was observed at the burned and unburned sites (Fig. 3).

The yellow-bellied sapsucker showed little overlap in foraging with *Picoides* species, as it was the only species to use sap-licking as its main foraging technique (not included in the discriminant analysis; Fig. 4).

Although they differed in several aspects of their foraging behavior, all four *Picoides* species used dead wood substrates for a substantial portion of their foraging time (Fig. 5). At the MXW and CON sites, mean percentage of time spent foraging on dead wood varied from 33% for the downy woodpecker to 89% for the black-backed woodpecker. Black-backed woodpeckers foraged exclusively on dead wood at the BURN site. Dead wood included snags but also dead portions of live trees such as dead branches. For instance, when foraging on branches of live trees, downy and hairy woodpeckers spent respectively 44% and 56% of their time on dead branches.

3.3. Wood-dwelling arthropods in foraging trees

The foraging trees used for wood dissection were representative of the woodpeckers' species-specific tree selection patterns described above (Table 4). Foraging trees of downy woodpeckers corresponded to live deciduous trees (trembling aspen and paper

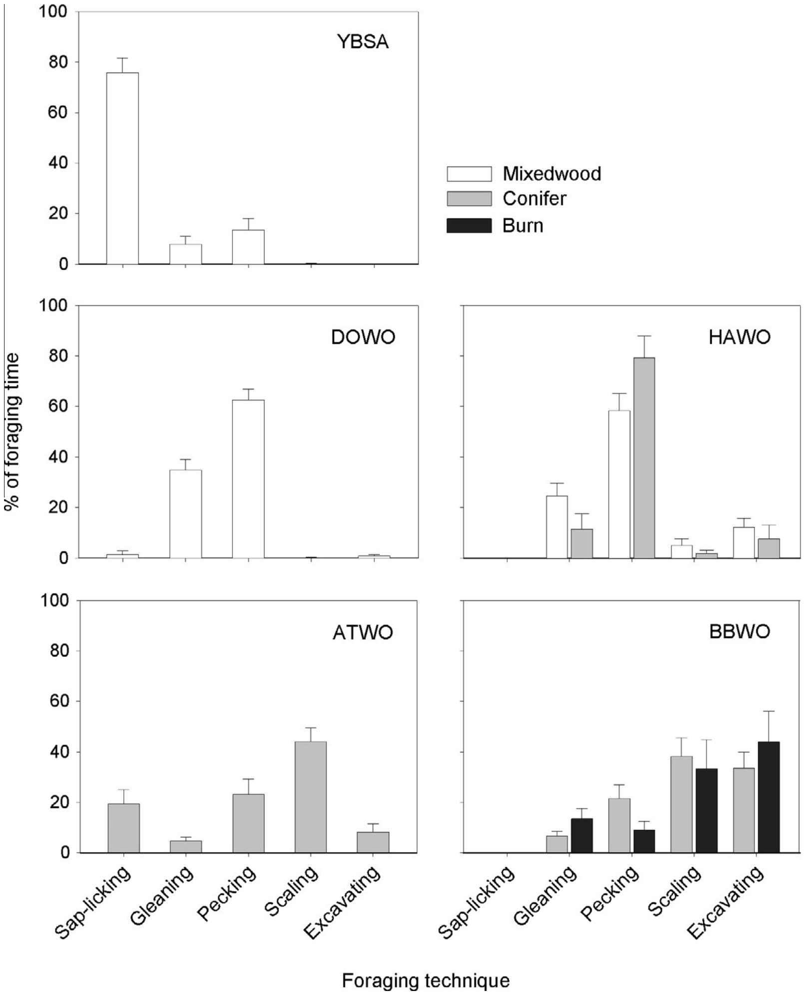

Fig. 4. Mean proportion of foraging time spent using different foraging techniques by five woodpecker species at the mixedwood, conifer and burn sites (weighed means + 1 SE). See Table 1 for species code.

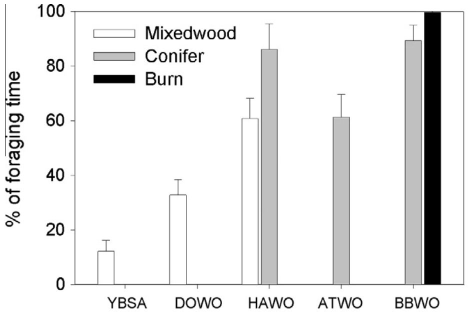

Fig. 5. Mean proportion of foraging time spent on dead wood substrates (snags, coarse woody debris or dead portions of live trees) by five woodpecker species at the mixedwood, conifer and burn sites (weighed means + 1 SE). See Table 1 for species code.

Table 4
Characteristics of trees used by foraging woodpeckers and sampled to examine wood-dwelling arthropod composition and abundance at the mixedwood (MXW), conifer (CON) and burn (BURN) sites.

Tree characteristics	DOWO (MXW)	HAWO (MXW)	ATWO (CON)	BBWO (CON)	BBWO (BURN)
Tree species[a]					
Populus tremuloides	5	5			
Betula papyrifera	5	5			
Picea mariana			10	10	6
Pinus banksiana					1
Diameter at breast height[b]	24.8 (13.3–31.8)	15.1 (12.5–31.3)	9.5 (6.2–14.3)	17.8 (13.2–22.0)	15.1 (10.7–20.7)
Degradation class[b]	1 (1–1)	3 (1–5)	3 (2–4)	3 (3–4)	4 (4–4)

[a] Number of trees for each tree species.
[b] Median (Min–Max).

birch) whereas those of hairy woodpeckers were deciduous trees of different degradation stages. Foraging trees of both American three-toed and black-backed woodpeckers consisted mainly of recently dead spruce trees, although conifer trees used by black-backed woodpeckers at the BURN site were slightly more degraded than those used at the CON site. Foraging trees of the American three-toed were smaller in diameter than those of black-backed woodpeckers.

We collected 773 arthropods from the bole segments, of which 32% and 24% were Cerambycidae and Scolytinae (sub-family of Curculionidae) respectively. At the MXW site, foraging trees of hairy woodpeckers contained significantly higher densities of large wood-dwelling arthropods (from various taxons) than trees used by downy woodpeckers (Fig. 6). At the CON site, foraging trees of black-backed woodpeckers supported higher densities of wood-associated arthropods (e.g. *Monochamus* species from the Cerambycidae family) than trees used by American three-toed woodpeckers. In contrast, Scolytinae (mostly bark-associates) were exclusively found in trees selected by American three-toed woodpeckers. Foraging trees of black-backed woodpeckers at the CON and BURN sites supported similar densities of Cerambycidae, large and wood-associated arthropods. However, foraging trees of black-backed woodpeckers at the CON site contained higher densities of small and bark-associated arthropods from Cerambycidae and other taxons.

4. Discussion

4.1. Importance of dead wood for foraging

This study is the first to quantify and compare the foraging requirements of all bark-foraging woodpeckers that co-occur in the eastern North-American boreal forest. By looking simultaneously at tree selection, foraging behavior and potential prey, this study provides a comprehensive portrait of the foraging ecology of these woodpeckers in natural-dominated boreal landscapes.

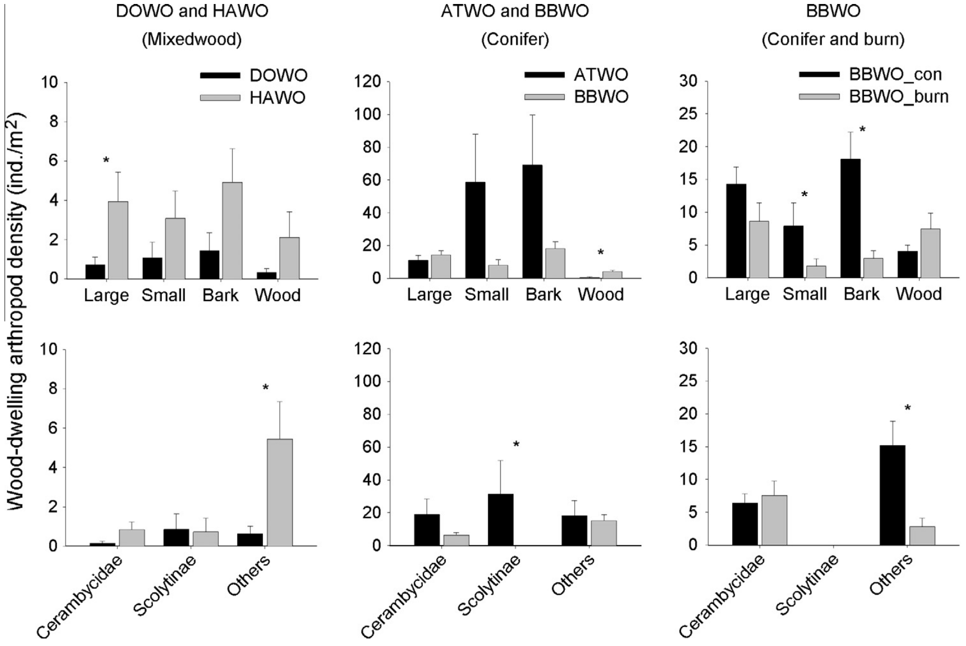

Fig. 6. Wood-dwelling arthropod densities in trees used for foraging by co-occurring species at each site (downy and hairy woodpeckers at the mixedwood site; black-backed and American three-toed woodpeckers at the conifer site) and by the black-backed woodpecker at the conifer and burn sites. Categories of wood-dwelling arthropods are: large (>1 cm in length), small (<1 cm in length), bark (found in or under bark), wood (found within sapwood or heartwood). Stars indicate significant differences between species or sites ($P < 0.05$).

Based on both tree selection and foraging behavior analyses, our results clearly demonstrate that dead wood (i.e. live decaying trees and snags in various stages of decay) represent an important foraging substrate for boreal woodpeckers as a group. The hairy, the black-backed and the pileated woodpeckers showed strong selection for live decaying trees and/or snags. In complement with tree selection analyses, foraging observations revealed that most woodpeckers used dead wood substrates for a substantial portion of their foraging time. Indeed, in addition to their time spent on standing snags, we observed that several species increased their foraging time on dead wood by using logs and stumps (American three-toed, black-backed and hairy woodpeckers) or by using dead portions of live deciduous trees such as dead branches or other defects (downy and hairy woodpeckers, yellow-bellied sapsucker). Among the species analyzed, the black-backed woodpecker showed the highest association with dead wood. Our results also revealed clear patterns of foraging tree preferences (degradation stage, tree species or tree diameter) among boreal woodpeckers, which were linked to species-specific differences in foraging behavior and related prey.

4.2. Associations with degradation stages

We found that woodpecker species could be categorized according to their preference for specific tree degradation stages, a pattern that was linked to their foraging behavior. The yellow-bellied sapsucker and the pileated woodpecker occupied opposite ends of the tree degradation gradient. The yellow-bellied sapsucker, well-known for its sap-licking foraging behavior and the importance of sap in its summer diet (Tate, 1973; Eberhardt, 2000), was the species most strongly associated with live trees. However, this species showed a preference for decaying over live healthy trees, a pattern that has also been reported by Eberhardt (2000).

At the opposite end of this degradation gradient was the pileated woodpecker, a species that selected mostly highly degraded snags. This woodpecker's diet is composed primarily of forest-dwelling ants, in particular carpenter ants (*Camponotus* species), but may also include bark and wood-boring beetles and other arthropods (Bull et al., 1992). Given our observations for the pileated woodpecker were based on this species' typical large foraging excavations, our results are restricted to its use of carpenter ants. Large snags, logs and stumps with a certain amount of decay are known as important nesting sites for carpenter ants (Torgersen and Bull, 1995) and selection of snags by pileated woodpeckers has been documented in different studies (Bull and Holthausen, 1993; Lemaître and Villard, 2005; Newell et al., 2009). However, to our knowledge, our study is the first to show a preference for more degraded snags, suggesting a higher abundance of carpenter ants in such substrates. These snags were mainly trembling aspens but also included balsam fir snags created after the most recent spruce budworm outbreak.

Although all *Picoides* species made extensive use of dead wood, our results highlight significant differences in each species patterns of substrate use, foraging behavior and associated wood-dwelling arthropods. The downy woodpecker selected deciduous trees and avoided highly degraded snags. This species was often seen pecking on dead branches of live trees or snags to feed on sub-cortical prey. In his detailed account of the feeding behavior of downy woodpeckers on paper birch, Kilham (1970) noted trees with broken branches and other defects to be the most attractive to downy woodpeckers. The relatively high use of dead wood by the downy woodpecker in our study (more than 30% of its foraging time) is consistent with this pattern. It suggests that partial mortality in live deciduous trees, in addition to snags in early stages of degradation, may play an important role in providing foraging substrates for this species.

In contrast to the downy woodpecker, the hairy woodpecker showed a clear preference for live decaying trees to highly degraded deciduous snags and was seen foraging more often on dead wood (more than 60% of its foraging time). The majority of hairy woodpecker foraging trees were dead (50% and 80% at mixedwood and conifer sites respectively), a pattern similar to what has been reported for this species elsewhere (Raphael and White, 1984; Weikel and Hayes, 1999). The hairy woodpecker was also the species that made the highest use of moderately degraded trees. In our study area, Saint-Germain et al. (2007) showed wood-feeding insects to be more abundant in middle to late than in earlier stages of decay in aspens. In our study, the more degraded deciduous trees used by hairy woodpeckers supported large wood-dwelling arthropods from many different guilds including wood-feeders, fungivores and sub-cortical insect predators. This prey diversity may explain the wider foraging strategy used by this species for capturing wood-dwelling arthropods, in comparison with the downy woodpecker.

Black-backed and American three-toed woodpeckers were only observed in northern coniferous forests where they specialized on conifers for foraging. The black-backed woodpecker foraged almost exclusively on dead wood (89% of foraging time at the conifer site). It showed a strong selection for live decaying and recently dead conifers and used scaling and excavation frequently to feed on wood-boring beetles. This selection pattern may be explained by the higher abundance of wood-feeding Coleoptera associated with conifers in the early stages of decay (Saint-Germain et al., 2007). Preferred foraging trees of the black-backed woodpecker differed from those of the American three-toed woodpecker by the higher abundance of wood-boring arthropods. We also noted a similar foraging behavior of black-backed woodpeckers at unburned and burned conifer sites, suggesting a consistent foraging niche across different habitat types in the boreal forest.

The American three-toed woodpecker made extensive use of both live and dead conifers in different stages of degradation. The use of decaying and recently dead trees and the associated use of scaling and pecking as predominant foraging techniques may be explained by its feeding specialization on bark beetles (Murphy and Lehnhausen, 1998; Imbeau and Desrochers, 2002). Indeed, we found that foraging trees of American three-toed woodpeckers contained more Scolytinae beetles (mostly bark-associates) than black-backed woodpecker foraging trees. Our results also emphasize the use of live healthy spruces for sap-licking, a foraging behavior that has been generally reported to be marginal for the American and European (*Picoides tridactylus*) three-toed woodpeckers. For instance, sap-licking represented less than 3% of foraging time in studies conducted in North America (Villard, 1994; Imbeau and Desrochers, 2002) and Europe (Pechacek, 2006). In contrast, we found that about 20% of foraging time was devoted to sap-licking (i.e. sap licking and related drilling behaviors), suggesting that this foraging behavior may be much more important during the breeding season than previously reported.

Picoides woodpeckers generally experience significant increases following recent natural disturbances and associated insect outbreaks in conifer forests (Murphy and Lehnhausen, 1998; Fayt et al., 2005; Covert-Bratland et al., 2006; Nappi and Drapeau, 2009; Rota et al., 2014b). Foraging specializations found in our study emphasize that the abundance of these woodpeckers (in particular the black-backed woodpecker) is tightly linked to the abundance of saproxylic insects and thus to the presence of dying and recently dead conifers in both disturbed (Murphy and Lehnhausen, 1998; Nappi and Drapeau, 2009; Rota et al., 2014b) and undisturbed coniferous forests (Imbeau and Desrochers, 2002; Tremblay et al., 2009, 2010; this study). Moreover, differences in the foraging ecology of these species may explain the response patterns observed following different disturbance agents.

Foraging specialization on wood-boring beetles by the black-backed woodpecker is consistent with its propensity to occupy burned coniferous forests, where these insects are generally abundant (Saint-Germain et al., 2004; Hannon and Drapeau, 2005). Conversely, the more pronounced response of the American three-toed woodpecker in forests disturbed by bark beetle outbreaks is consistent with its foraging specialization (Fayt et al., 2005). Natural disturbances that create large amount of snags and logs have also been shown to provide suitable habitats for the pileated woodpecker for several decades after disturbance (Bull et al., 2007). Indeed, in our study area, the most recent spruce budworm outbreak that occurred 20–30 years ago provided a significant amount of well-decayed snags that were highly used by foraging pileated woodpeckers.

4.3. Tree diameter

Although all woodpecker species showed a tendency to use larger diameter trees, this pattern was significant only for the black-backed, the hairy and the pileated woodpeckers. These three species often excavate to extract large wood-boring beetles or, in the case of the pileated woodpecker, carpenter ants. Larger trees typically contain higher densities of wood-boring insect larvae. This pattern may be attributed to the requirements of late-instar larvae that excavate deep galleries into sapwood and heartwood, to the thicker bark that increases protection against desiccation, and to the thicker phloem which is of higher nutritional quality for first-instar larvae (Gardiner, 1957; Rose, 1957; Saint-Germain et al., 2004). Selection of larger trees and logs by pileated woodpeckers has been well documented throughout its range and has been linked to the nesting preference of carpenter ants, its main prey (Torgersen and Bull, 1995; Lemaître and Villard, 2005; Newell et al., 2009).

4.4. Conservation and management implications

We found that most bark-foraging woodpeckers in eastern North American boreal forests show strong associations with live decaying and/or snags for foraging. Among the six species examined, the black-backed woodpecker showed the strongest association with dead wood and was very specific in its tree selection with regard to degradation stage. This finding suggests a high dependence of this species on recently dead wood and likewise a high vulnerability to the reduction of this tree degradation stage in managed boreal forests. For woodpeckers as a guild, their high use of dead wood for foraging suggests that they may be much more limited by the availability of foraging trees than nest trees. Limitation of foraging substrates for woodpeckers may have a cascading effect on the structure of the nest web in these forests, especially in conifer stands, because of their key ecological roles in providing nest cavities to secondary-cavity users. This study thus reiterates the importance of planning the conservation of dead wood for a wide range of tree age, species and decay classes to ensure a steady supply of suitable foraging trees (Drapeau et al., 2009b) in addition to cavity-bearing tree requirements (Edworthy and Martin, 2013) in managed boreal forest landscapes.

Providing trees of high forage value for woodpeckers in managed boreal forests may require different strategies for species associated with deciduous trees (e.g. downy or hairy woodpecker) compared to those associated with conifers (e.g. American three-toed or black-backed woodpeckers). For deciduous trees, suitable dead wood substrates may be provided by both the partial mortality present in live decaying trees (e.g. dead branches) and snags of different degradation stages. In contrast, the temporal window of foraging opportunities appears to be much shorter in conifers, which generally support high densities of saproxylic insects mainly in their declining and recently dead stages (Saint-Germain et al., 2007). Hence, in conifer landscapes, maintaining foraging habitat in managed forests implies ensuring a continuous recruitment of recent conifer snags.

In the boreal ecosystem, old-growth and post-disturbance forests are the two most important sources of dead wood in natural forest landscapes. Late seral stands are shaped by small-scale mortality processes that may provide a constant recruitment of recent snags for woodpeckers (Imbeau and Desrochers, 2002; Tremblay et al., 2009). Yet, at the landscape scale, natural stand-replacement disturbances such as fire and insect outbreaks may represent important sources of dead wood that provide high quality habitats for woodpeckers (Saab et al., 2005). For the black-backed woodpecker for instance, high concentrations of recently dead trees such as the ones created following stand-replacement wildfires clearly represent optimal habitats for the species (Nappi and Drapeau, 2009; Rota et al., 2014a). Although these foraging habitats are ephemeral (because all snags are created more or less simultaneously), the recurrence of these events at regional scales provide a continuous source of foraging habitats for woodpeckers. In North American boreal forests where old-growth forests are decreasing because of short harvest rotations and post-disturbance forests are increasingly salvage-logged, both habitats are thus of conservation concern for the persistence of woodpecker populations (Nappi et al., 2004; Schmiegelow et al., 2006; Drapeau et al., 2009a).

Harvesting practices such as variable retention or partial cutting may provide woodpecker foraging habitat depending on the abundance and quality of dead wood that is left as well as the level of green-tree retention that is maintained for future dead wood recruitment. At our conifer site, partial cutting provided short-term suitable foraging substrates for black-backed and American three-toed woodpeckers, as was evidenced by the numerous foraging observations within partial cut stands. These partial cuts, while maintaining live trees, also resulted in mortality of standing trees and in high amounts of downed logs. Given our study took place 3–6 years after partial cutting, downed logs and snags were still in early stages of degradation and supported high abundance of wood-dwelling insects. By creating a combination of sun-exposed habitat and a high abundance of recently dead trees and downed logs, partial cutting may also resembled post-disturbance forests. Indeed, many of the saproxylic insects found during wood dissection (e.g. *Monochamus* spp., *Acmaeops proteus*) are well-known for their high abundance following fire (Saint-Germain et al., 2004; Boulanger and Sirois, 2007).

Woodpeckers are often identified as species of interest in forest management because of their role as ‘‘keystone’’ species in nest-webs (Martin and Eadie, 1999) and because they are generally good indicators of forest bird diversity (Mikusiński et al., 2001; Roberge and Angelstam, 2006; Drever et al., 2008). Our study emphasizes the strong association of woodpeckers with dead wood for foraging, which makes them good indicators of the abundance of this habitat resource in managed forests. Because use of foraging trees is mainly linked to prey abundance, woodpeckers may also represent good indicators of the presence of saproxylic insect species in managed boreal forests (see also Martikainen et al., 1998). Moreover, because of their association with specific degradation stages and snag recruitment, woodpeckers as a group could be used as indicators of snag dynamics and therefore be considered as ‘‘process-limited’’ species (*sensu* Lambeck, 1997). Maintaining the woodpecker guild in managed forests would thus benefit the conservation of a diversity of saproxylic plant and animal species associated with distinct stages of wood degradation.

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References

Aitken, K.E.H., Martin, K., 2007. The importance of excavators in hole-nesting communities: availability and use of natural tree holes in old mixed forests of western Canada. J. Ornithol. 148 (suppl. 2), S425–S434.

Angelstam, P., Mikusiński, G., 1994. Woodpecker assemblages in natural and managed boreal and hemiboreal forest – a review. Ann. Zool. Fennici 31, 157–172.

Berg, Å., Ehnstrom, B., Gustafsson, L., Hallingbäck, T., Jonsell, M., Weslien, J., 1994. Threatened plant, animal, and fungus species in Swedish forests: distribution and habitat associations. Conserv. Biol. 8, 718–731.

Bergeron, Y., 2000. Species and stand dynamics in the mixed woods of Quebec's southern boreal forest. Ecology 81, 1500–1516.

Bergeron, Y., Leduc, A., Morin, H., Joyal, C., 1995. Balsam fir mortality following the last spruce budworm outbreak in north-western Québec. Can. J. For. Res. 25, 1375–1384.

Bergeron, Y., Leduc, A., Harvey, B., Gauthier, S., 2002. Natural fire regime: a guide for sustainable management of the Canadian boreal forest. Silva Fennica 36, 81–95.

Blanc, L.A., Martin, K., 2012. Identifying suitable woodpecker nest trees using decay selection profiles. For. Ecol. Manage. 286, 192–202.

Boulanger, Y., Sirois, L., 2007. Postfire succession of saproxylic arthropods, with emphasis on Coleoptera, in the north boreal forest of Quebec. Environ. Entomol. 36, 128–141.

Bull, E.L., Holthausen, R.S., 1993. Habitat use and management of pileated woodpeckers in northeastern Oregon. J. Wildlife Manage. 51, 335–345.

Bull, E.L., Peterson, S.R., Thomas, J.W., 1986. Resource partitioning among woodpeckers in northeastern Oregon. USDA Forest Service, Pacific Northwest Research Station, Research Note PNW-444, Lagrande, OR.

Bull, E.L., Beckwith, R.C., Holthausen, R.S., 1992. Arthropod diet of pileated woodpeckers in northeastern Oregon. Northwestern Nat. 73, 42–45.

Bull, E.L., Nielsen-Pincus, N., Wales, B.C., Hayes, J.L., 2007. The influence of disturbance events on pileated woodpeckers in northeastern Oregon. For. Ecol. Manage. 243, 320–329.

Cockle, K.L., Martin, K., Wesolowski, T., 2011. Woodpeckers, decay, and the future of cavity-nesting vertebrate communities worldwide. Front. Ecol. Environ. http://dx.doi.org/10.1890/110013.

Cooke, H.A., Hannon, S.J., 2012. Nest-site selection by old boreal forest cavity excavators as a basis for structural retention guidelines in spatially-aggregated harvests. For. Ecol. Manage. 269, 37–51.

Cooper, A.B., Millspaugh, J.J., 1999. The application of discrete choice models to wildlife resource selection studies. Ecology 80 (2), 566–575.

Covert-Bratland, K.A., Block, W.M., Theimer, T.C., 2006. Hairy woodpecker winter ecology in ponderosa pine forests representing different ages since wildfire. J. Wildlife Manage. 70, 1379–1392.

Dansereau, P.-R., Bergeron, Y., 1993. Fire history in the southern boreal forest of northwestern Quebec. Can. J. For. Res. 23, 25–32.

Drapeau, P., Leduc, A., Bergeron, Y., 2009a. Bridging ecosystem and multiple species approaches for setting conservation targets in managed boreal landscapes. In: Villard, M.-A., Gunnar-Jonsson, B.G. (Eds.), Setting Conservation Targets in Managed Forest Landscapes. Cambridge University Press, Cambridge, UK, pp. 129–160, 426 p.

Drapeau, P., Nappi, A., Imbeau, L., Saint-Germain, M., 2009b. Standing deadwood for keystone bird species in the eastern boreal forest: managing for snag dynamics. Forest. Chron. 85, 227–234.

Drever, M.C., Aitken, K.E.H., Norris, A.R., Martin, K., 2008. Woodpeckers as reliable indicators of bird richness, forest health and harvest. Biol. Conserv. 141, 624–634.

Eberhardt, L.S., 2000. Use and selection of sap trees by yellow-bellied sapsuckers. Auk 117, 41–51.

Edworthy, A.B., Martin, K., 2013. Persistence of tree cavities used by cavity-nesting verterbrates declines in harvested forests. J. Wildlife Manage. 77, 770–776.

Fayt, P., Machmer, M.M., Steeger, C., 2005. Regulation of spruce bark beetles by woodpeckers – a literature review. For. Ecol. Manage. 206, 1–14.

Fenton, N., Bescond, H., Imbeau, L., Boudreault, C., Drapeau, P., Bergeron, Y., 2009. Silvicultural and ecological evaluation of partial harvest in the boreal forest on the clay belt, Québec. In: Gauthier, S., Vaillancourt, M.-A., Leduc, A., De Grandpré, L., Kneeshaw, D., Morin, H., Drapeau, P., Bergeron, Y. (Eds.), Ecosystem Management in the Boreal Forest. Presses de l'Université du Québec, Québec, Québec, Canada, pp. 373–393, 539 p.

Gardiner, L.M., 1957. Deterioration of fire-killed pine in Ontario and causal wood-boring beetles. Can. Entomol. 6, 241–263.

Gauthier, S., Vaillancourt, M.-A., Leduc, A., De Grandpré, L., Kneeshaw, D., Morin, H., Drapeau, P., Bergeron, Y., 2009. Ecosystem management in the boreal forest. Presses de l'Université du Québec, Québec, Québec, Canada, 539 p.

Gouvernement du Québec, 2008. Proposed Muskuuchii Hills Biodiversity Reserve: Conservation Plan. Gouvernement du Québec, Québec strategy for protected areas, Québec, Québec, Canada, 11 p. on-line <http://www.mddelcc.gouv.qc.ca/biodiversite/reserves-bio/muskuchii/PSC_Muskuchii-en.pdf>.

Grove, S.J., 2002. Saproxylic insect ecology and the sustainable management of forests. Annu. Rev. Ecol. Syst. 33, 1–23.

Güthlin, D., Knauer, F., Kneib, T., Küchenhoff, H., Kaczensky, P., Rauer, G., Jonozovič, M., Mustoni, A., Jerina, K., 2011. Estimating habitat suitability and potential population size for brown bears in the Eastern Alps. Biol. Conserv. 144, 1733–1741.

Hannon, S.J., Drapeau, P., 2005. Bird responses to burning and logging in the boreal forest of Canada. Stud. Avian Biol. 30, 97–115.

Harvey, B., 1999. The Lake Duparquet Research and Teaching Forest: Building a foundation for ecosystem management. Forest. Chron. 75, 389–393.

Hogstad, O., 1971. Stratification in winter feeding of the great spotted woodpecker (*Dendrocopos major*) and the three-toed woodpecker (*Picoides tridactylus*). Ornis Scandinavica 2, 143–146.

Hogstad, O., 1976. Sexual dimorphism and divergence in winter foraging behavior of three-toed woodpeckers *Picoides tridactylus*. Ibis 118, 41–50.

Imbeau, L., Desrochers, A., 2002. Foraging ecology and use of drumming trees by three-toed woodpeckers. J. Wildlife Manage. 66, 222–231.

Imbeau, L., Mönkkönen, M., Desrochers, A., 2001. Long-term effects of forestry on birds of the eastern Canadian boreal forest: a comparison with Fennoscandia. Conserv. Biol. 15, 1151–1162.

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

Jonsson, B.G., Siitonen, J., 2012. Dead wood and sustainable forest management. In: Stokland, J.N., Siitonen, J., Jonsson, B.G. (Eds.), Biodiversity in Dead Wood. Cambridge University Press, Cambridge, UK, pp. 302–337, 509 p.

Kilham, L., 1970. Feeding behavior of downy woodpeckers. I. Preference for paper birches and sexual differences. Auk 87, 544–556.

Kneib, T., Knauer, F., Küchenhoff, H., 2011. A general approach to the analysis of habitat selection. Environ. Ecol. Stat. 18, 1–25.

Lambeck, R.J., 1997. Focal species: a multi-species umbrella for nature conservation. Conserv. Biol. 11 (4), 849–856.

Lemaître, J., Villard, M.-A., 2005. Foraging patterns of pileated woodpeckers in a managed Acadian forest: a resource selection function. Can. J. For. Res. 35, 2387–2393.

Martikainen, P., Kaila, L., Haila, Y., 1998. Threatened beetles in white-backed woodpecker habitats. Conserv. Biol. 12, 293–301.

Martin, P., Bateson, P., 1993. Measuring Behaviour: An Introductory Guide, second ed. Cambridge University Press, Cambridge, UK, 222 p.

Martin, K., Eadie, J.M., 1999. Nest webs: a community-wide approach to the management and conservation of cavity-nesting birds. For. Ecol. Manage. 115, 243–257.

Maser, C., Anderson, R.G., Cromack Jr., K., Williams, J.T., Martin, R.E., 1979. Dead and down woody material. In: Thomas, J.W. (Ed.), Wildlife Habitats in Managed Forests: The Blue Mountains of Oregon and Washington. USDA Forest Service, Agriculture Handbook No.533, pp. 78–95, 512 p.

Mikusiński, G., Gromadzki, M., Chylarecki, P., 2001. Woodpeckers as indicators of forest bird diversity. Conserv. Biol. 15, 208–217.

Murphy, E.C., Lehnhausen, W.A., 1998. Density and foraging ecology of woodpeckers following a stand-replacement fire. J. Wildlife Manage. 62, 1359–1372.

Nappi, A., Drapeau, P., 2009. Reproductive success of the black-backed woodpecker (*Picoides arcticus*) in burned boreal forests: are burns source habitats? Biol. Conserv. 142, 1381–1391.

Nappi, A., Drapeau, P., 2011. Pre-fire forest conditions and fire severity as determinants of the quality of burned forests for deadwood-dependent species: the case of the black-backed woodpecker. Can. J. For. Res. 41, 994–1003.

Nappi, A., Drapeau, P., Giroux, J.-F., Savard, J.-P.L., 2003. Snag use by foraging black-backed woodpeckers (*Picoides arcticus*) in a recently burned eastern boreal forest. Auk 120, 505–511.

Nappi, A., Drapeau, P., Savard, J.-P., 2004. Salvage logging after wildfire in the boreal forest: is it becoming a hot issue for wildlife? Forest. Chron. 80, 67–74.

Nappi, A., Drapeau, P., Saint-Germain, M., Angers, V.A., 2010. Effect of fire severity on long-term occupancy of burned boreal conifer forests by saproxylic insects and wood-foraging birds. Int. J. Wildland Fire 19, 500–511.

Newell, P., King, S., Kaller, M., 2009. Foraging behavior of pileated woodpeckers in partial cut and uncut bottomland hardwood forest. For. Ecol. Manage. 258, 1456–1464.

Pechacek, P., 2006. Foraging behavior of Eurasian three-toed woodpeckers (*Picoides tridactylus alpinus*) in relation to sex and season in Germany. Auk 123, 235–246.

Raphael, M.G., White, M., 1984. Use of snags by cavity-nesting birds in the Sierra Nevada. Wildlife Monogr. 86, 1–66.

Roberge, J.-M., Angelstam, P., 2006. Indicator species among resident forest birds – a cross-regional evaluation in northern Europe. Biol. Conserv. 130, 134–147.

Rose, A.H., 1957. Some notes on the biology of *Monochamus scutellatus* (Say) (Coleoptera: Cerambycidae). Can. Entomol. 89, 547–553.

Rota, C.T., Millspaugh, J.J., Rumble, M.A., Lehman, C.P., Kesler, D.C., 2014a. The role of wildfire, prescribed fire, and mountain pine beetle infestations on the population dynamics of black-backed woodpeckers in the Black Hills, South Dakota. PLoS ONE 9 (4), e94700. http://dx.doi.org/10.1371/journal.pone.0094700.

Rota, C.T., Rumble, M.A., Millspaugh, J.J., Lehman, C.P., Kesler, D.C., 2014b. Space-use and habitat associations of black-backed woodpeckers (*Picoides arcticus*) occupying recently disturbed forests in the Black Hills, South Dakota. For. Ecol. Manage. 313, 161–168.

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. Stud. Avian Biol. 30, 76–96.

Saint-Germain, M., Drapeau, P., Hébert, C., 2004. Xylophagous insects species composition and substratum use patterns on fire-killed black spruce in central Quebec. Can. J. For. Res. 34, 677–685.

Saint-Germain, M., Drapeau, P., Buddle, C.M., 2007. Host-use patterns of saproxylic phloeophagous and xylophagous Coleoptera adults and larvae along the decay gradient in standing dead black spruce and aspen. Ecography 30, 737–748.

Saucier, J.-P., Bergeron, J.-F., Grondin, P., Robitaille, A., 1998. Les régions écologiques du Québec méridional (3e version). L'Aubelle, pp. 1–12.

Schmiegelow, F.K.A., Stepnisky, D.P., Stambaugh, C.A., Koivula, M., 2006. Reconciling salvage logging of boreal forests with a natural-disturbance management model. Conserv. Biol. 20, 971–983.

Serrouya, R., D'Eon, R., 2004. Variable retention forest harvesting: research synthesis and implementation guidelines. Sustainable Forest Management Network, University of Alberta, Edmonton, Alberta, Canada 48 p. <http://www.sfmn.ales.ualberta.ca/en/Publications/~/media/sfmn/Publications/ResearchNotes/Documents/E25_Variable_retention.ashx>.

Siitonen, J., 2001. Forest management, coarse woody debris and saproxylic organisms: Fennoscandian boreal forests as an example. Ecol. Bull. 49, 11–41.

Siitonen, J., 2012. Threatened saproxylic species. In: Stokland, J.N., Siitonen, J., Jonsson, B.G. (Eds.), Biodiversity in Dead Wood. Cambridge University Press, Cambridge, UK, pp. 356–379, 509 p.

Stokland, J.N., Siitonen, J., Jonsson, B.G., 2012. Biodiversity in Dead Wood. Cambridge University Press, Cambridge, UK, 509 p.

Sullivan, T.P., Sullivan, D.S., Lindgren, P.M.F., 2001. Influence of variable retention harvests on forest ecosystems. I. Diversity of stand structure. J. Appl. Ecol. 38, 1221–1233.

Tate, J.J., 1973. Methods and annual sequence of foraging by the sapsucker. Auk 90, 840–856.

Torgersen, T.R., Bull, E.L., 1995. Down logs as habitat for forest-dwelling ants: the primary prey of pileated woodpeckers in northeastern Oregon. Northwest Sci. 69, 294–303.

Török, J., 1990. Resource partitioning among three woodpecker species *Dendrocopos spp.* during the breeding season. Holartic Ecol. 13, 157–264.

Tremblay, J.A., Ibarzabal, J., Dussault et, C., Savard, J.-P.L., 2009. Habitat requirements of breeding black-backed woodpeckers (*Picoides arcticus*) in managed, unburned boreal forests. Avian Conservation and Ecology – Écologie et conservation des oiseaux 4(1), 2, <http://www.ace-eco.org/vol4/iss1/art2/>.

Tremblay, J.A., Ibarzabal, J., Savard, J.-P.L., 2010. Foraging ecology of black-backed woodpeckers (*Picoides arcticus*) in unburned eastern boreal forest stands. Can. J. For. Res. 40, 991–999.

Villard, P., 1994. Foraging behavior of black-backed and three-toed woodpeckers during spring and summer in a Canadian boreal forest. Can. J. Zool. 72, 1957–1959.

Viton, P.A., 2014. Discrete-Choice Logit Models with R. Ohio State University, Columbus, OH, 38 p. <http://facweb.knowlton.ohio-state.edu/pviton/courses2/crp5700/5700-mlogit.pdf>.

Weikel, J.M., Hayes, J.P., 1999. The foraging ecology of cavity-nesting birds in young forests of the northern Coast Range of Oregon. Condor 101, 58–66.

Wesolowski, T., 2011. "Lifespan" of woodpecker-made holes in a primeval temporate forest: a thirty year study. For. Ecol. Manage. 262, 1846–1852.

Flammulated Owl (Otus flammeolus) breeding habitat abundance in ponderosa pine forests of the United States

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FLAMMULATED OWL (*OTUS FLAMMEOLUS*) BREEDING HABITAT ABUNDANCE IN PONDEROSA PINE FORESTS OF THE UNITED STATES

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Abstract. Flammulated Owl (*Otus flammeolus*) inhabits mid-elevation montane forests of ponderosa pine (*Pinus ponderosa*) and Jeffrey pine (*Pinus jeffreyi*) across western North America during the breeding season. We employed data from the USDA Forest Service's Forest Inventory and Analysis (FIA) program to assess historic and current extent of ponderosa and Jeffrey pines comprising the majority of Flammulated Owl habitat within 11 western states of the USA. We cross-referenced breeding habitat characteristics to FIA data attributes; then produced estimates and maps of forest land area and potential habitat abundance from FIA data and made comparisons with other published data. We estimated area of current ponderosa and Jeffrey pine forest types on forest land as 98 633 km^2 and 83 000 km^2, from FIA and LANDFIRE data, respectively. Area of ponderosa and Jeffrey pine forest types on timberland (a subset of forest land) decreased from 136 200 km^2 to 86 827 km^2 (36%,) between 1953 and 2007. Within the past two decades these forest types decreased by 14% overall; increased 16% within stand-ages of 100–149 years and decreased 49% within stand-ages of 150 years or older; increased 22% within diameters of 30–49 cm and decreased 28% within diameters of 50 cm and larger; and increased 115% within stocking classes of 10–59%. We estimated area of potential breeding habitat abundance for Flammulated Owl at about 48 000 km^2 from FIA data, 50 000 km^2 from LANDFIRE data, and 522 000 km^2 from GAP data sources. FIA provides data and information for producing estimates of Flammulated Owl breeding habitat abundance.

Key Words: Flammulated Owl, *Otus flammeolus*, ponderosa pine, *Pinus ponderosa*, habitat, forest inventory, FIA.

TECOLOTE OJO OSCURO (*OTUS FLAMMEOLUS*) ABUNDANCIA DEL HÁBITAT REPRODUCTIVO EN BOSQUES DE PINO PONDEROSA EN LOS ESTADOS UNIDOS

Resumen. El tecolote ojo oscuro (*Otus flammeolus*) habita en elevaciones medias de bosques de montaña de pinos ponderosa (*Pinus ponderosa*) y de Jeffrey (*Pinus jeffreyi*) en el oeste de Norte America durante la temporada reproductiva. Empleamos datos del Inventario Forestal y Análisis (FIA) del Servicio de Bosques de USDA para evaluar la extensión histórica y actual de los bosques de pino ponderosa y de Jeffrey que componen la mayoría del hábitat del tecolote ojo oscuro en 11 estados del oeste de los Estados Unidos. Referenciamos las características del hábitat reproductivo con atributos de FIA e hicimos estimaciones y mapas del área forestal y abundancia potencial del hábitat de datos de FIA y los comparamos con otros datos publicados. Estimamos el área actual de los tipos de bosque de poderosa y de Jeffrey como 98 633 km^2 y 83 000 km^2, de FIA y datos de LANDFIRE, respectivamente. El área de los bosques de pino ponderosa y de Jeffrey en áreas maderables (una división de área boscosa) decreció de 136 200 km^2 a 86 827 km^2 (36%,) entre 1953 y 2007. En las pasadas dos décadas estos bosques decrecieron un 14% en toda el área; incrementaron un 16% dentro de los parches de 100–149 años y decrecieron un 49% dentro de los parches de 150 o mas años; incrementó un 22% dentro de diámetros de 30–49 cm y decreció un 28% dentro de diámetros de 50 cm y mayores; e incrementó 115% dentro de las clases en reserva de 10–59%. Estimamos la abundancia el área potencial del hábitat reproductivo para el tecolote ojo oscuro de alrededor de 48 000 km2 de los datos de FIA, 50 000 km^2 de los datos de LANDFIRE, y 522 000 km^2 de los datos de GAP. La FIA provee datos e información para producir estimaciones de la abundancia del hábitat reproductivo del tecolote ojo oscuro.

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INTRODUCTION

Ponderosa pine (*Pinus ponderosa*) is the most wide-ranging conifer species in the United States and Canada (Graham and Jain 2005). Prior to Euro–American settlement, dry ponderosa pine and mixed conifer forests of the Inland Northwest burned frequently from low- or mixed-severity surface fires (Hessburg et al. 2005), which favored fire-tolerant trees, such as ponderosa pine and maintained low and variable tree densities, light and patchy ground fuels, simplified forest structure, and a low and patchy cover of associated fire-tolerant shrubs and herbs (Hessburg et al. 2005).

Ponderosa pine forest has decreased in abundance and has changed in forest structure during the past century, attributed to European settlement and resulting land management practices of timber harvest and wildfire suppression, introduction of domestic livestock, and climate cycles (Cooper 1960, Graham and Jain 2005). These "dry forests" now are dominated not only by ponderosa pine, but also by dry mixed conifer forests of grand fir (*Abies grandis*), white fir (*Abies concolor*), and Douglas-fir (*Pseudotsuga menziesii*) zones. Furthermore, these forests now comprise large landscapes that are more homogeneous in their composition and structure, and are more susceptible to severe, large fire and insect disturbance events (Hessburg et al. 2005).

Concerns over these changes in ponderosa pine forest, especially as they relate to habitat for cavity-nesting birds, led to a special session on this topic at the 4th International Partners In Flight (PIF) conference, held 13–16 February 2008, McAllen, Texas, USA. This paper addresses Inventory and Monitoring needs for producing a PIF Bird Conservation Needs Assessment pertaining to ponderosa pine restoration and bird habitat.

HABITAT TERMINOLOGY

Hall et al. (1997) define 'habitat' as "…the resources and conditions present in an area that produce occupancy—including survival and reproduction by a given organism." Animals select habitat through a process of hierarchical spatial scaling at four levels: (1) geographic range of a species during a particular season, which may be genetically determined; (2) home ranges of individual animals; (3) specific sites or components within a home range; and (4) resources procured within microsites (Johnson 1980, Hutto 1985). Habitat 'abundance' is a term preferred over habitat 'availability', because the latter implies a measure of "…accessibility and procurability of physical and biological components of a habitat by animals"—attributes which are difficult to quantify (Hall et al. 1997).

Our study focused on assessing the abundance of potential breeding habitat for the Flammulated Owl (*Otus flammeolus*) over the United States portion of the owl's range. Our rationale was that (1) PIF sets population goals for each bird species based in part on abundance of breeding habitat, (2) sufficient studies of breeding habitat have been conducted to characterize the vegetation component of species-habitat relationships for the Flammulated Owl, and (3) sufficient forest inventory data on vegetation composition and structure are available across nearly the entire breeding range of this species, allowing for estimates of potential breeding habitat abundance.

FLAMMULATED OWL

The Flammulated Owl is a cavity-nesting, insectivorous, Neotropical migratory bird. During the breeding season, it typically occupies mid-elevation montane forest with seasonably temperate climates, primarily within forests of western yellow pine–ponderosa pine, Jeffrey pine (*Pinus jeffreyi*), and Washoe pine (*P. washoensis*)—and submontane Douglas fir (*Pseudotsuga menziesii*). The range of this owl extends from northern Mexico into southern British Columbia, Canada, including the states of Washington, Oregon, California, Idaho, Utah, Nevada, Arizona, Montana, Colorado, New Mexico, and a small portion of western Texas (McCallum 1994, Dunham et al. 1996, van Woudenberg 1998) (Fig. 1). Barnes (2007) provides an estimate of 300 000 adult Flammulated Owls in the United States, based on the following assumptions: 24 million ha of habitat (Morgan 1994), a ratio of adult females to adult males of 0.65 (Reynolds and Linkhart 1987), and a mean density of 0.3 singing males per 40 ha.

Habitat

Barnes (2007) described spatial scales of Flammulated Owl habitat selection for an Idaho study area—terms corresponding to spatial scaling levels in Johnson (1980) and Hutto (1985) are shown in parentheses: landscape (geographic range); home range (home range); and nest vegetation, nest tree, and day roost (specific sites or components of a home range). Landscape-scale habitat selection was strongly influenced by slope position and aspect variables—mid to upper slopes, often with east- or south-facing aspects. In colder regions like Idaho, south and east-facing upper slopes may warm faster,

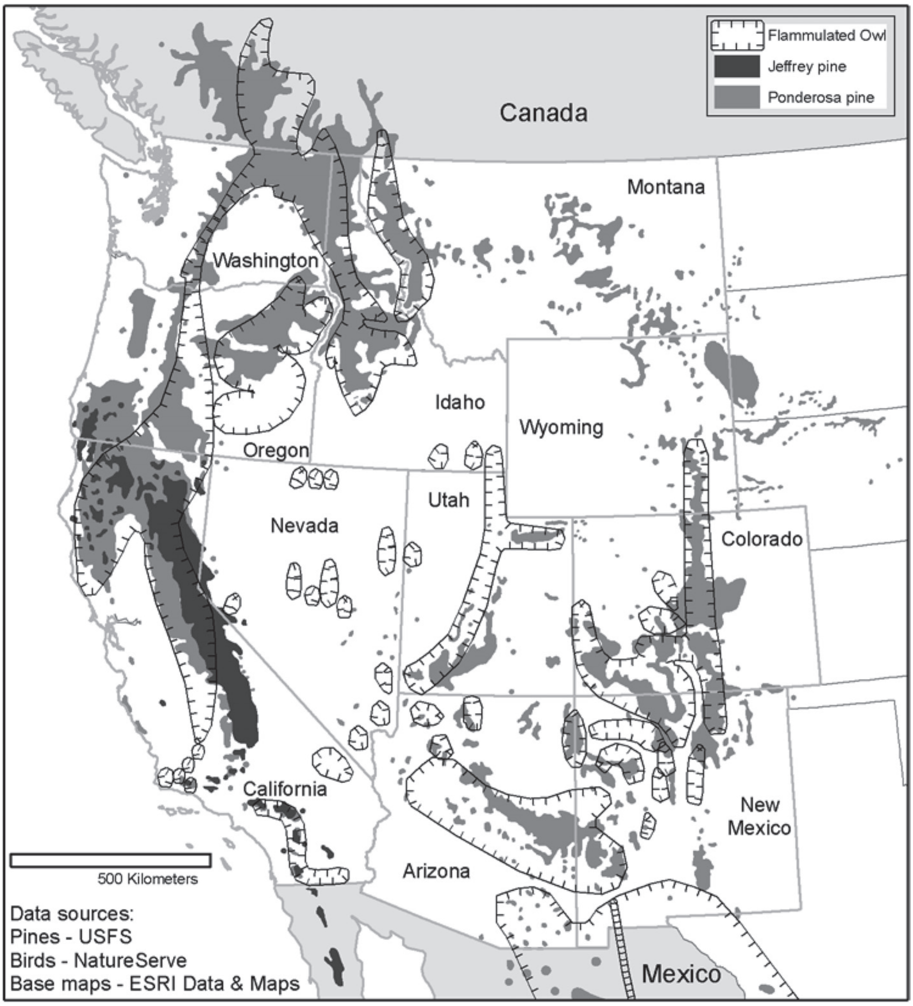

FIGURE 1. Geographic distribution of Flammulated Owl (*Otus flammeolus*), ponderosa pine (*Pinus ponderosa*), and Jeffrey pine (*Pinus jeffreyi*) within Canada, Mexico, and 11 western states, USA.

thereby creating more favorable microclimates for insects, the primary prey of Flammulated Owls. Forest cover exhibited moderate influence on selection at the landscape scale—forest cover of 25–75% being selected most. At the home-range scale, areas with high density of snags were more likely to be occupied compared with adjacent areas with lower densities of snags. But the proportion of forest and clearings was highly variable within home ranges. Habitat selection at the nest-vegetation and nest-tree scale showed no differences between used and available sites for any measured variable or between used and available cavity trees in the Idaho study, suggesting that availability of nest cavities at the home-range scale constrained owl use to certain areas, whereas foraging habitat may have been adequate throughout the entire study area. Ponderosa pines used as day-roosts were 55–70 cm diameter at breast height (dbh; 1.37 m) (peak of the frequency distribution); ponderosa pine was selected disproportionately more than available, and Douglas fir disproportionately less than available (Barnes 2007). In a Colorado study, Flammulated Owls occupied home ranges that contained more old ponderosa pine/Douglas fir than other types of overstory vegetation, likely a result of a corresponding abundance of lepidopteran prey (Linkhart et al. 1998).

Western yellow pines do not typify Flammulated Owl breeding habitat within the Great Basin. In Nevada, yellow pines are limited to the eastern slope of the Sierra Nevadas and adjacent large mountain ranges, and to mountain ranges in the extreme eastern and southern portions of the state. Where western yellow pines are absent, montane conifer forests comprised of white fir (*Abies concolor*), subalpine fir (*A. lasiocarpa*), and limber pine (*Pinus flexilis*) provide breeding habitat for Flammulated Owls in Nevada (Dunham et al. 1996).

Although Flammulated Owl nest sites are not limited to cavities in snags (standing dead trees), snags provide an important source of both nest and roost sites for the species (Scott et al. 1977). Limited abundance of nesting substrates constrains nest-site selection for most owl species (Marks 2001). In Arizona's ponderosa pine forests, for example, secondary cavity-nesters, including the Flammulated Owl, comprise one-third of breeding bird species, most of which nest in snags (Balda 1975).

METHODS

We employed data from the USDA Forest Service's Forest Inventory and Analysis (FIA) program and the Forest and Rangeland Renewable Resources Planning Act of 1974 (RPA) forest resource assessments database to assess historic and current amount and structure of two western yellow pine tree species in the United States: ponderosa pine and Jeffrey pine. These pine types encompass the majority of the breeding habitat of the Flammulated Owl within 11 western states of the USA (Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, Washington, and Wyoming). We compiled literature descriptions of Flammulated Owl breeding habitat characteristics, cross-referenced these to FIA data attributes of forest composition and structure, and estimated and mapped habitat abundance. We compared the FIA-based results with estimates and maps of owl habitat distribution obtained from other sources.

Vegetation Characteristics of Habitat

We characterized vegetation within Flammulated Owl breeding habitat at or near nest sites in the context of data attributes available in inventory tables and geospatial datasets. Specifically, we compiled data from 16 papers and reports that described Flammulated Owl breeding habitat at 185 territories, including 153 nest sites in seven states, from which we tabulated specific habitat characteristics and summarized the following characteristics of forest type, size, and density: (1) type—presence of ponderosa pine or Jeffrey pine forest type—referred to collectively as ponderosa pine forest type group—or trees of these species within associated forest types (Fig. 2); (2) size—presence of one or more trees at least 30 cm diameter at breast height (dbh) or canopy height of at least 10 m; and (3) density—tree density of 300–700 trees per ha (tph) or canopy cover of 30–80%.

Data and Analyses

Geographic range maps

We used data from NatureServe (Ridgely et al. 2005) to produce a map of the Flammulated Owl geographic range. We used polygon delineations digitized from Little (1971) to map Ponderosa pine and Jeffrey pine geographic ranges, which we overlaid with the owl range map and a state boundary dataset to visualize the correspondence among these ranges.

FIA

The FIA program conducts detailed surveys of the nation's forests across all ownerships. FIA collects data from field plots, with a sampling intensity of approximately one plot per 2 400 ha (Reams et al. 2005). FIA defines *forest land* as land with 10% minimum tree stocking level or, for several western woodland types where stocking cannot be determined, 5% minimum tree canopy cover; minimum area of 0.4047 ha; and a minimum continuous canopy width of 36.58 m (Bechtold and Patterson 2005). *Timberland* is a designation referring to a subset of forest land that "…is producing or capable of producing in excess of 20 cubic feet per acre per year [1.4 m^3 ha^{-1} yr^{-1}] of wood at culmination of mean annual increment", excluding *reserved forest land*, which "…is permanently reserved from wood products utilization through statute or administrative designation" (Bechtold and Patterson 2005). Estimates of forest land and timberland area within each state were obtained by multiplying total area inventoried by the mean proportion estimated from forest inventory plot observations (Scott et al. 2005).

We used draft tables of the 2007 RPA report (Smith et al. 2009) and historic RPA data to produce area estimates of current and past forest land and ponderosa and Jeffrey pine forest type for each state. For some attributes, RPA estimates represent groups of forest types: ponderosa and Jeffrey pines types are combined into the ponderosa pine forest type group. For some attributes, historic RPA estimates apply only to timberland, which captures more than 87% of forest land area in the ponderosa pine forest type group and thus is assumed to be

FIGURE 2. Ponderosa pine (*Pinus ponderosa*) forest with vegetation composition and structure typical of breeding habitat for Flammulated Owl (*Otus flammeolus*), Colorado, USA. Photograph by Brian Linkhart.

representative of historic trends in forest land for this forest type group.

We produced estimates of the area of Flammulated Owl habitat abundance for 11 western states by querying the FIA database (U.S. Department of Agriculture Forest Service 2008) and constraining estimates to ponderosa pine forest type group on forest land with tree diameters of at least 30 cm and trees per ha in the range 300–700. Using a hexagon sampling array developed by the U.S. Environmental Protection Agency, Environmental Monitoring and Assessment Program (EMAP) (White et al. 2005), we produced estimates for each hexagon within the 11 western states. EMAP provides a sampling grid with each hexagon measuring approximately 648 square kilometers in area (U.S. Environmental Protection Agency 2002). We used a geographic information system (GIS) to produce hexagon-based maps for all forest land in ponderosa pine forest type group, and for Flammulated Owl potential breeding habitat abundance under the same conditions used for producing state-wide estimates.

LANDFIRE

The Landscape Fire and Resource Planning Tools Project, or LANDFIRE, is a project for producing maps and data describing vegetation, wildland fuel, and fire regimes across the United States (Rollins and Frame 2006). LANDFIRE geospatial data are produced as 30-m spatial resolution raster datasets and distributed by mapping zones in Albers Conical Equal Area Projection (USGS parameters), North American Datum of 1983. LANDFIRE data products analyzed in this study include attributes corresponding to forest type, size and density: (1) Existing Vegetation Type (EVT) (Comer 2003), and linked to a classification system from the Society of American Foresters and Society for Range Management (Eyre 1980); (2) Canopy Height (CH), which is the average height (m) to the top of the vegetation canopy; and (3) Canopy Cover (CC), which is the percent cover of tree canopy. We obtained these geospatial datasets for all mapping zones that encompass any portion of the 11-state study area and used a GIS to constrain the LANDFIRE data to the 11-state geographic extent, and attribute each pixel with the state code in which the pixel is located. We used the following filters to label LANDFIRE forest pixels as potential habitat: EVT = SAF237 (Interior Ponderosa Pine) or SAF247 (Jeffrey Pine), CH ≥ 10 m, and CC ≥ 30% and CC < 80%. We estimated abundance of Flammulated Owl breeding habitat by tabulating the number of selected pixels and weighting by the per-pixel area (900 m^2).

Gap Analysis Program (GAP)

Gap Analysis Program (GAP) geospatial datasets representing known or likely habitat for animal species are based on a combination of land cover classes associated with habitats and counties or hexagons for which animal species occurrence information is recorded (Gap Analysis Program 2008). GAP data are appropriate for use in regional analyses or planning applications, but not for site-specific analyses. Stockenberg et al. (2008) describe the role of GAP data for setting biological objectives for habitats and associated bird species within a Bird Conservation Region. GAP data were obtained for each of the 11 western states in the study area. We used a GIS to process and analyze GAP data including: converting data to 90-m spatial resolution raster format, merging datasets from 11 states to produce a single geospatial dataset, masking out non-'habitat' pixels, reprojecting the dataset to Albers 'USGS' projection (as described for LANDFIRE, above), and attributing each pixel with its corresponding state code. We produced estimates of habitat abundance by tabulating the number of 'habitat' pixels and weighting by the per-pixel area (8100 m^2).

RESULTS

Geographic Range Maps

Flammulated Owl breeding range corresponds closely with ponderosa pine and Jeffrey pine range, illustrating the strong positive association of this owl with the two western yellow pines (see Fig. 1). Exceptions include the extension beyond the owl range of ponderosa pine in western Oregon, central and eastern Montana, northeastern Wyoming and in the Black Hills of western South Dakota, which is partially outside the 11-state study area; and of Jeffrey pine in southcentral California and southwestern Oregon. Also, portions of the owl range encompass areas beyond the geographic range of the two yellow pines (i.e., southern British Columbia and Mexico).

Forest Land and Timberland Area

Current estimates of ponderosa and Jeffrey pine forest types on forest land were 98 633 km^2 and 83 000 km^2 from FIA and LANDFIRE data, respectively (Fig. 3). RPA estimates of ponderosa pine forest type group on timberland declined by 36% from about 136 200 km^2 in 1953 to 86 827 km^2 in 2007, with modest fluctuations

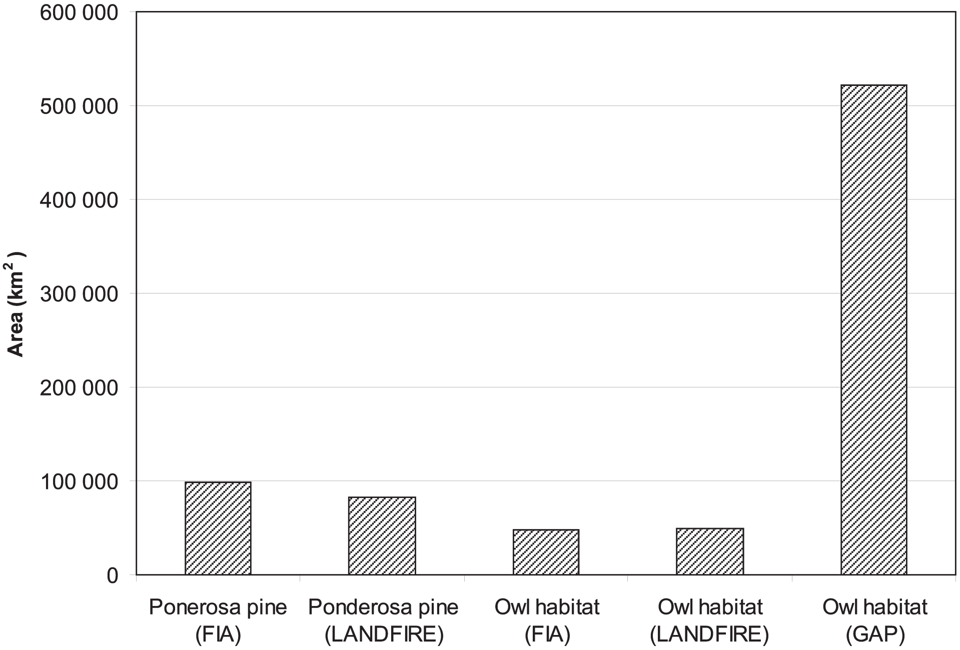

FIGURE 3. Area estimates of ponderosa pine (*Pinus ponderosa*) forest type group on forest land, derived from FIA and LANDFIRE data; and estimates of Flammulated Owl (*Otus flammeolus*) potential breeding habitat, derived from FIA, LANDFIRE, and GAP data, within 11 western states, USA.

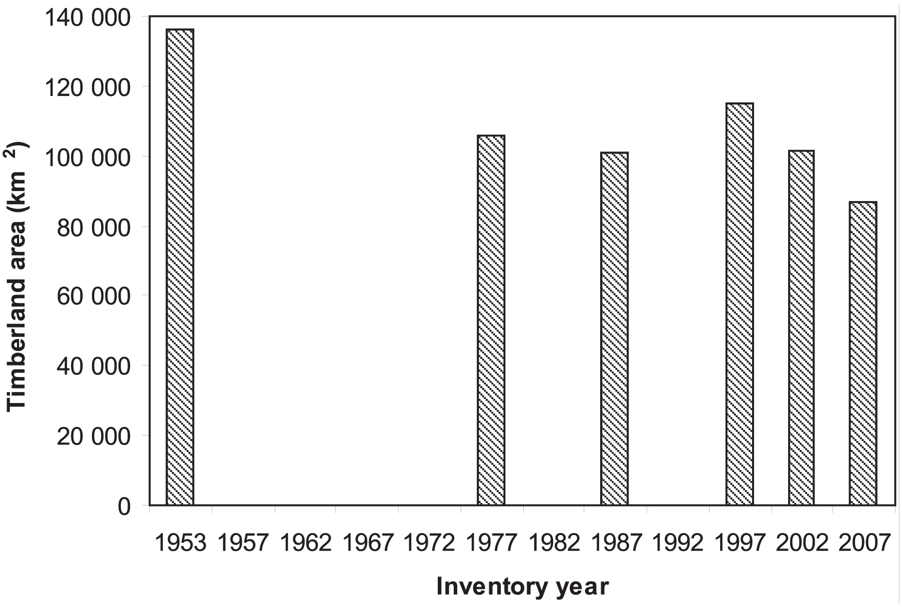

FIGURE 4. Historical area estimates of ponderosa pine (*Pinus ponderosa*) forest type group on timberland within 11 western states, USA, 2007.

during intermediate years (Fig. 4). Between 1987 and 2007, a decrease in these forest types on timberland was estimated at 14%, characterized by a 16% increase within stand-ages of 100–149, but a 49% decrease within stand-ages of 150 or older (Fig. 5); a 22% increase within diameter classes of 30–49 cm, but a 28% decrease for diameters of 50 cm and larger (Fig. 6); and a 115% increase within the poorly stocked and medium stocked classes (10–59%) (Fig. 7). Figure 8A reveals an FIA-based map of ponderosa and Jeffrey pine spatial distribution that is very similar to Little's range maps for these tree species (Fig. 1).

Extent of Breeding Habitat

The extent of Flammulated Owl breeding habitat was estimated to be 47 890 km^2 from FIA data, 49 740 km^2 from LANDFIRE data, and 521 548 km^2 from GAP data (Fig. 3). Figure 8B portrays the spatial distribution of Flammulated Owl potential breeding habitat, which encompasses most of the geographic range of ponderosa and Jeffrey pine (Fig. 8A), but at lower densities per hexagon across the range and with fewer hexagons containing habitat, especially within the eastern extent of the range.

DISCUSSION

Literature reports tend to have high agreement in how they characterize vegetation compositional and structural components of Flammulated Owl breeding habitat. Minimum area requirements and other landscape metrics were not emphasized in the Flammulated Owl literature, although these features are known to affect habitat quality of many other species of wildlife. Thus, we felt justified in using FIA data to assess forest vegetation relating to the extent of Flammulated Owl potential breeding habitat across the United States portion of this owl's geographic range.

The geographic range of Flammulated Owl corresponds closely, but not perfectly, with ponderosa and Jeffrey pine distribution, illustrating the strong positive association between this owl and these two western yellow pines (Fig. 1). This is expected, given the generalized nature of species range maps, which are not intended for estimating area of vegetation, abundance of habitat, or population size. Large areas of western yellow pine range appear to be unoccupied by the owl, primarily in the northeast extent of ponderosa pine range. Recent field surveys in Montana (Cilimburg 2005) resulted in additional observations of Flammulated Owls, east of the previously delineated owl range. We recommend that the owl geographic range map be revised to capture these new records. To date, Flammulated Owls have not been observed during field surveys of the Black Hills, an area visible in Figure 1 as a large island of ponderosa forest in eastern Wyoming and western South

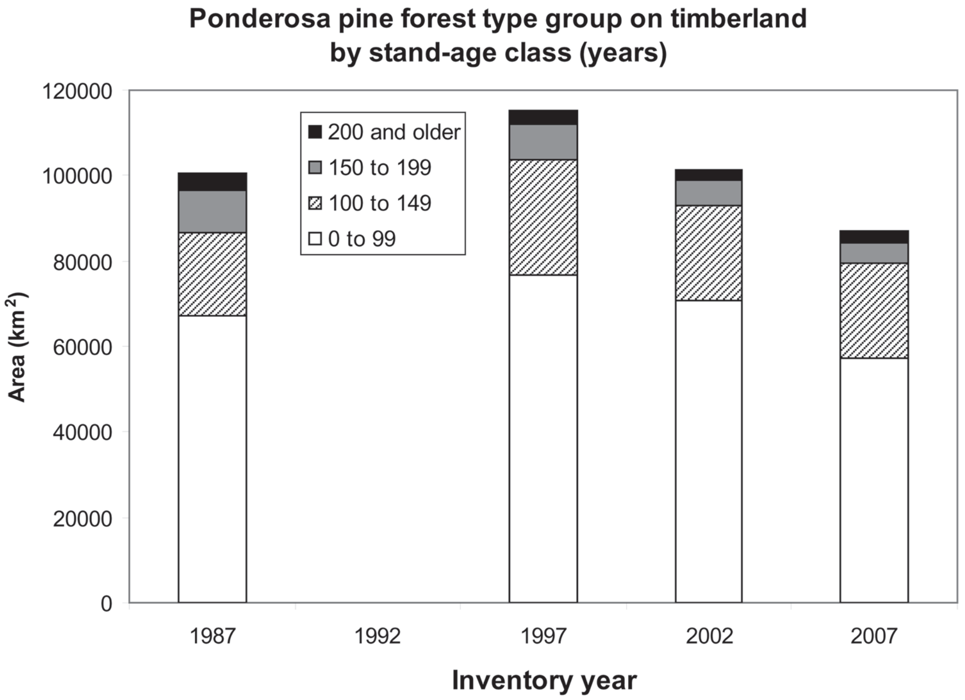

FIGURE 5. Area estimates of ponderosa pine forest type group on timberland by stand-age class within 11 western states, USA, 2007.

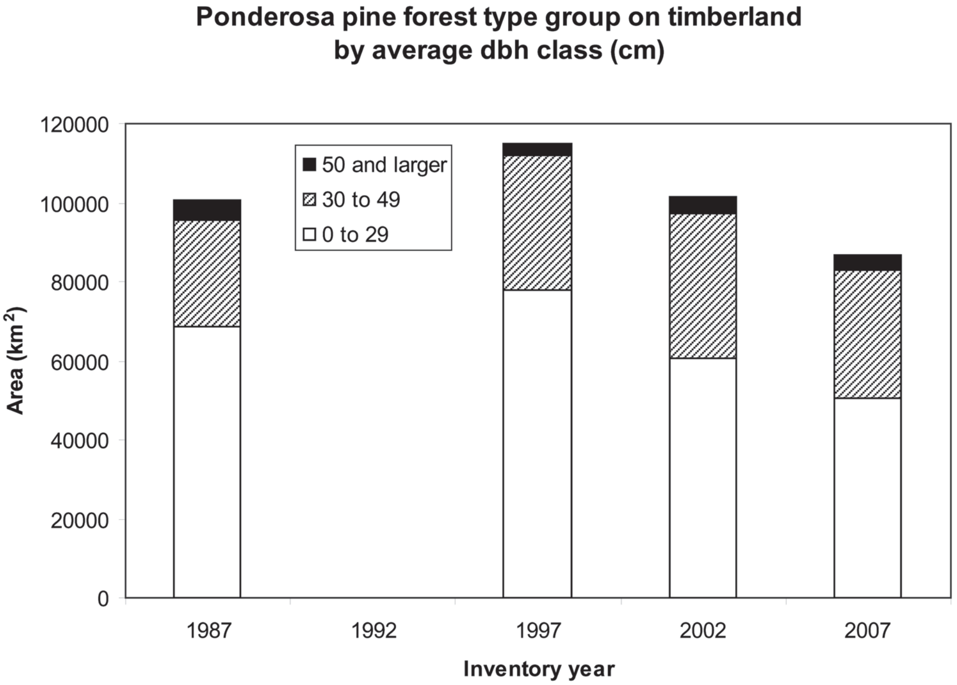

FIGURE 6. Area estimates of ponderosa pine forest type group on timberland by average dbh class within 11 western states, USA, 2007.

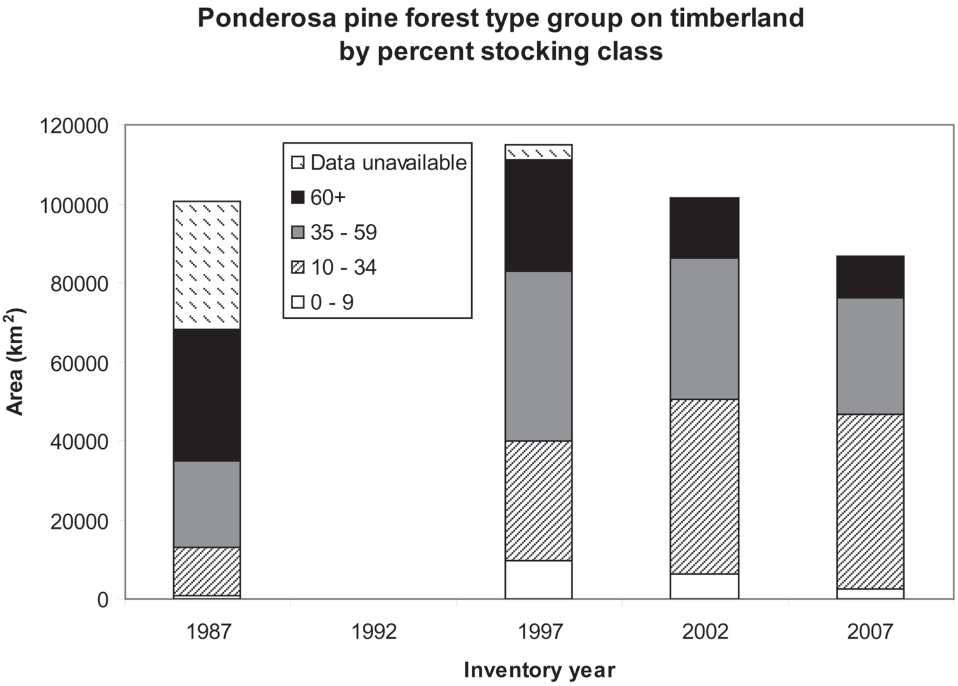

FIGURE 7. Area estimates of ponderosa pine forest type group on timberland by stocking class within 11 western states, USA, 2007.

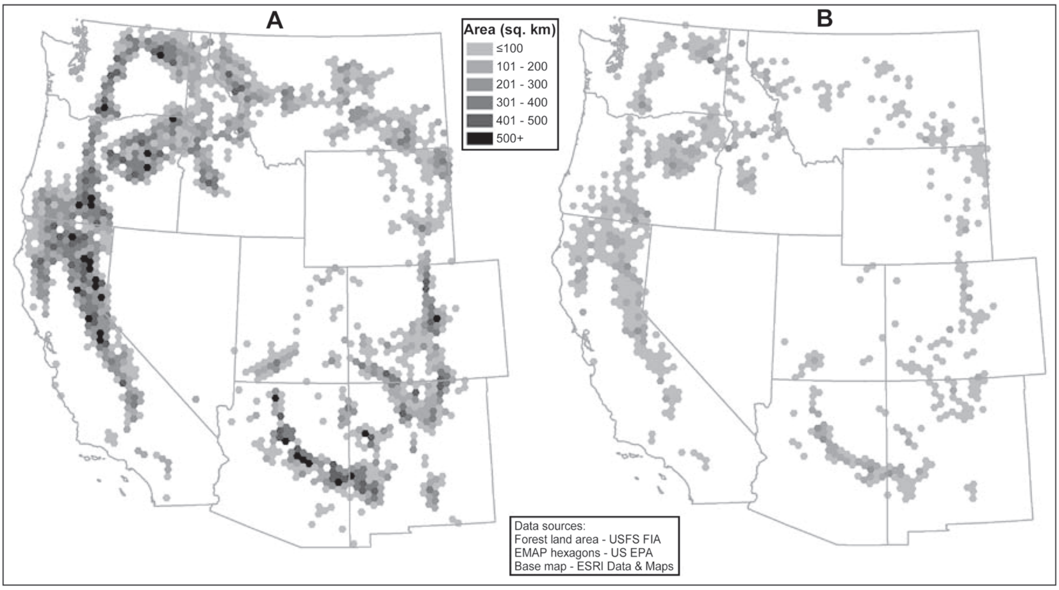

FIGURE 8. Geographic distribution within 11 western states, USA of (a) ponderosa pine (*Pinus ponderosa*) or Jeffrey pine (*Pinus jeffreyi*) forest land; and (b) potential abundance of Flammulated Owl (*Otus flammeolus*) breeding habitat, where one or more ponderosa pine or Jeffrey pine trees are ≥ 30 cm dbh, and tph of all trees are between 300 and 700 stems/ha.

Dakota (Rocky Mountain Bird Observatory, Online database).

RPA statistical reports, derived from FIA data, provided historical estimates of forest land and timberland area, by common attributes of forest type, age, size, and density. Forest types of ponderosa and Jeffrey pines comprise about 10% of all forest land in the West, and the area of forest in these types has decreased by 36% over the past 5 decades. Little of this forest is in stand-age classes older than 100 years, dbh classes larger than 30 cm, and intermediate stocking classes, attributes that are typical of Flammulated Owl breeding habitat. Ganey and Vojta (2007) predict that densities of snags in ponderosa pine forests in Northern Arizona will increase over the next 30 years, but densities of large snags will remain below target densities for management and conservation of wildlife.

Compared with FIA estimates, LANDFIRE-based estimates of forestland area in ponderosa pine forest type group and potential breeding habitat abundance appear slightly lower, but GAP-based habitat estimates appear to greatly overestimate habitat abundance, likely due to GAP's more general land cover classes.

Although the FIA- and LANDFIRE-based estimates of habitat area are constrained to ponderosa pine forest type group and omit habitat within other forest types, we suggest that these estimates capture the core resources needed by breeding Flammulated Owls. However, the thresholds of forest size and density selected for defining habitat may have led to overestimates of habitat abundance. Our minimum threshold of 30 cm dbh characterizes the minimum size of trees recorded as nesting sites, but the mean dbh of these nesting trees was about 50 cm. Constraining our estimates to trees at least 50 cm would have reduced both the geographic extent and the total area of our estimates of habitat abundance. Therefore, our estimates are reported as potential habitat abundance, which includes habitat of low quality.

CONCLUSIONS

Abundance of Flammulated Owl potential breeding habitat in ponderosa and Jeffrey pines declined substantially over the past five decades. Current estimates of Flammulated Owl potential breeding habitat area were about 48 000 km^2 from FIA data and about 50 000 km^2 from LANDFIRE data—a consistency that suggests reliability of these data sources. Additional analyses are recommended to assess spatial consistency between these two datasets. A GAP-based estimate of known/likely Flammulated Owl habitat (522 000 km^2) was considerably greater than from FIA- and LANDFIRE-based estimates, likely due to the less specific land cover classes available in the GAP data. Recent enhancements to GAP, i.e., the Southwest Regional Gap Analysis Project (SWReGAP), may provide for improved estimates and we recommend that these data be assessed.

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LITERATURE CITED

Balda, R. P. 1975. The relationship of secondary cavity nesters to snag densities in western coniferous forests, p. 37. Wildlife Habitat Technical Bulletin 1. USDA Forest Service, Southwestern Region, Albuquerque, NM.

Barnes, K. P. 2007. Ecology, habitat use, and probability of detection of Flammulated Owls in the Boise National Forest. M.S. thesis, Boise State University, Boise, ID.

Bechtold, W. A., and P. L. Patterson. 2005. The enhanced Forest Inventory and Analysis Program—national sampling design and estimation procedures, p. 85. General Technical Report SRS-80. U.S. Department of Agriculture Forest Service, Asheville, NC.

Cooper, C. F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecological Monographs 30:129–164.

Dunham, S., L. Butcher, D. Charlet, and J. M. Reed. 1996. Breeding range and conservation of Flammulated Owls (*Otus flammeolus*) in Nevada. Journal of Raptor Research 30:189–193.

Ganey, J. L., and S. C. Vojta. 2007. Modeling snag dynamics in northern Arizona mixed-conifer and ponderosa pine forests, p. 15. RMRS-RP-66WWW. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Graham, R. T., and T. B. Jain. 2005. Ponderosa pine ecosystems, pp. 1–32. *In* M. W. Ritchie, D. A. Maguire, and A. Youngblood, [eds.], Proceedings of the Symposium on Ponderosa Pine: Issues, Trends, and Management, Gen. Tech. Rep. PSW-GTR-198. Albany CA:

Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture, Klamath Falls, OR.

Hall, L. S., P. R. Krausman, and M. L. Morrison. 1997. The habitat concept and a plea for standard terminology. Wildlife Society Bulletin 25:173–182.

Hessburg, P. F., J. K. Agee, and J. F. Franklin. 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.

Hutto, R. L. 1985. Habitat selection by non-breeding, migratory land birds, pp. 455–476. *In* M. L. Cody [ed.], Habitat selection in birds. Academic Press, Orlando, FL.

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

Linkhart, B. D., R. T. Reynolds, and R. A. Ryder. 1998. Home range and habitat of breeding Flammulated Owls in Colorado. Wilson Bulletin 110:342–351.

Little, E. L., Jr. 1971. Atlas of United States trees, volume 1, conifers and important hardwoods, p. 9, 200 maps. U.S. Department of Agriculture, Washington, D.C.

Marks, J. S. 2001. Assessing nest-site selection in owls: random is not always better. Journal of Field Ornithology 72:462–464.

McCallum, D. A. 1994. Flammulated Owl (*Otus flammeolus*). *In* A. Poole, and F. Gill, [eds.], The Birds of North America, 93. Philadelphia: The Academy of Natural Sciences; Washington, D.C.: The American Ornithologists' Union.

Morgan, P. 1994. Dynamics of Ponderosa and Jeffrey Pine forests, pp. 47–73. *In* G. D. Hayward, and J. Verner [eds.], Flammulated, boreal, and great gray owls in the United States: A technical conservation assessment. Gen. Tech. Rep. RM-253, U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Reams, G. A., W. D. Smith, M. H. Hansen, W. A. Bechtold, F. A. Roesch, and G. G. Moisen. 2005. The Forest Inventory and Analysis sampling frame, pp. 11–26. *In* P. L. Patterson, [ed.], The enhanced Forest Inventory and Analysis Program—national sampling design and estimation procedures. U.S. Department of Agriculture Forest Service, Asheville, NC.

Reynolds, R. T., and B. D. Linkhart. 1987. The nesting biology of Flammulated Owls in Colorado, pp. 239–248. *In* R. W. Nero, R. J. Clark, R. J. Knapton, and R. H. Hamre [eds.], Biology and conservation of northern forest owls. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Ridgely, R. S., T. F. Allnutt, T. Brooks, D. K. McNicol, D. W. Mehlman, B. E. Young, and J. R. Zook. 2005. Digital Distribution Maps of the Birds of the Western Hemisphere, version 2.1. NatureServe. Arlington, VA.

Rollins, M. G., and C. K. Frame [eds.] 2006. The LANDFIRE prototype project: nationally consistent and locally relevant geosptatial data for wildland fire management. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Scott, C. T., W. A. Bechtold, G. A. Reams, W. D. Smith, J. A. Westfall, M. H. Hansen, and G. G. Moisen. 2005. Sample-based estimators used by the Forest Inventory and Analysis National Information Management System, pp. 43–67. *In* P. L. Patterson [ed.], The enhanced Forest Inventory and Analysis Program—national sampling design and estimation procedures. U.S. Department of Agriculture Forest Service, Asheville, NC.

Scott, V. E. 1978. Characteristics of ponderosa pine snags used by cavity-nesting birds in Arizona. Journal of Forestry 76:26–28.

Scott, V. E., K. E. Evans, D. R. Patton, and C. P. Stone. 1977. Cavity nesting birds of North American forests, p. 112. USDA Agricultural Handbook 511. U.S. Department of Agriculture, Washington, D.C.

Smith, W. B., P. D. Miles, C. H. Perry, and S. A. Pugh. 2009. Forest resources of the United States, 2007. U.S. Department of Agriculture, Forest Service, Washington, D.C.

U.S. Department of Agriculture Forest Service. 2008. The Forest Inventory and Analysis database: database description and users manual, version 3.0 for phase 2, revision 1, p. 243. U.S. Department of Agriculture, Forest Service, Washington Office, Internal report. On file with: U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis, 201 14th St., Washington, D.C., 20250.

van Woudenberg, A. M. 1998. Status of the Flammulated Owl in British Columbia, p. 29. Wildlife Working Report No. WR-95. Ministry of Environment, Lands and Parks, Wildlife Branch, Victoria, BC.

Colorado's worst methane polluter is an Arch Coal mine

By Kelsey Ray - May 3, 2016

Somerset, CO — If you know what to look for, you can spot the old well pads in the hills above West Elk Mine. The reclaimed patches of land, carefully restored to their original contours, stand out only as a lighter green against the dark, dense forest. The trees, the Forest Service promises, will grow back later.

Otherwise, to the casual hiker, this tract of Gunnison National Forest offers few hints of the massive coal mine hundreds of feet below. The landscape is lush, the sky clear blue. Only the low thrum of chorus frogs and the occasional passing mule deer break the forest's silence.

But hike along a certain route, several miles up muddy access roads high in the hills, and you'll find West Elk's dirty secret. On a well pad cut flat into the sloping earth sits a loud, smelly exhauster. Day and night, this machine perpetually is pulling methane gas from deep underground and thrusting it into the sky.

Methane vent well pads dot the terrain above the West Elk Mine in Gunnison County, Colorado.

Coal mines account for nearly 10 percent of all methane gas emissions nationwide. In 2014, the most recent year data is available from the U.S. Environmental Protection Agency, West Elk Mine was the single largest methane polluter in Colorado, venting more of the greenhouse gas than even the state's most methane-polluting oil and gas operator, Encana Oil & Gas.

Unlike methane from oil and gas drilling, coal mine methane remains unregulated at both state and federal levels. As climate change activists see it, this makes neither environmental nor economic sense: In addition to being many times more active a heat-trapping gas than carbon dioxide, methane is a valuable fuel source. West Elk's gassy coal seams leak enough of it to potentially heat almost the entire city of Longmont.

But rather than working to capture the thousands of tons of methane West Elk emits annually, or at least destroying it to significantly reduce its global warming impact, the mine's operators simply vent it into the atmosphere, where the only thing it's heating is the planet.

Gov. John Hickenlooper has passed strict, groundbreaking methane regulations for the oil and gas industry that are a model for the country. But even though West Elk is the state's worst methane offender, Hickenlooper supports reinstating a legal exception that would likely allow the mine to significantly expand. If approved, the exception would grant access to nearly 20,000 acres and 170 million more tons of coal, meaning Colorado's single largest methane polluter could continue venting even more of the potent gas for years to come.

"I hope Governor Hickenlooper and leaders in the Obama administration recognize that this mine would roll back a large portion of the progress we've made in tackling methane on the state, national, and even international levels," said Conservation Colorado Executive Director Pete Maysmith.

"There are myriad reasons not to expand the West Elk Coal Mine," he added. "But perhaps the most compelling is that this mine would unleash a torrent of methane that threatens to undermine Colorado's efforts to lead the nation in reducing methane pollution and climate change."

The buzz

You can hear the mechanical exhauster long before your legs, weighed down by the thick April mud, make it up the final hill. The destination: well vent E6-6, one of the latest spots where Arch Coal, the corporation that owns West Elk, is currently venting methane from the mine. From afar, the forest sounds almost like a fluorescent-lit room, a faint, droning hum, until you hike closer to the source, where the buzz is overwhelming.

This is public land, but the exhauster is caged behind a chain-link fence on which signs warn of fire hazard. The machine itself is connected to a hole that bores deep into the mine.

A methane exhauster above West Elk Mine. The gas it's venting can't be seen by the naked eye.

Coal mining and methane are inextricably linked. Extracting the carbon-rich fuel requires breaking up hard earth hundreds of feet below ground, creating cracks and fissures that release the long-stored gas into the mineshaft. The older, deeper and blacker the coal, the more methane. The super rich mine seams of Colorado's North Fork Valley, as it happens, meet that description.

"West Elk is particularly gassy," explained Scott Braden, an energy advocate with Conservation Colorado.

Methane is nontoxic, but highly flammable — the source of generations of catastrophic explosions that made mining such a dangerous vocation. In the old days, mine operators would send one unlucky worker underground to check for gas pockets, outfitting him with only a torch and a wet blanket to hide under. Now, methane venting is mandatory and highly regulated. Gas meters offer a far less fatal detection tool than human torch-bearers.

The gas that poses a safety risk underground is also harmful when released. Scientists now recognize that methane is between 20 and 25 times more potent as a greenhouse gas than the better-known warming culprit, carbon dioxide. Though methane only remains in the atmosphere for little over a decade, while CO2 lingers much longer, methane's heat-trapping properties are more devastating in the short run.

As its climate impacts are becoming better known, methane is finally, albeit slowly, becoming subject to regulation.

Gov. Hickenlooper was hailed as an environmental champion in 2013 when he passed his "zero tolerance" policy on methane emissions from oil and gas operations. Colorado became the first state to impose such a rule, which requires drillers to capture 95 percent of all methane and to find and fix all gas leaks.

But the governor's low tolerance for methane doesn't apply to the coal industry. Colorado, like the federal government, doesn't regulate coal mine methane beyond mandating that large polluters report their emissions to the EPA.

The proposed expansion of West Elk, which would line up 10 million more tons of methane-rich coal for mining, has the potential to undo much of the climate change benefits of the state's methane rule. While Hickenlooper's oil and gas regulations are expected to cut methane emissions equivalent to 2.34 million tons of CO2 each year, the coal mine expansion could add 1.2 million CO2-equivalent tons of the gas annually. And none of that, of course, factors in the additional methane emissions that would come from burning the coal.

Said Braden: "The back-of-the-napkin calculation is that you essentially lose the savings of the oil and gas rule by allowing a new coal-based methane source."

Cases for and against expansion

As an odorless and colorless gas, methane is invisible and imperceptible. That's partly why it's allowed to leak in such massive amounts without consequence into the skies above Somerset: No one can see it.

At least, not without an infrared camera.

Earthworks, an environmental advocacy group based in Washington, D.C., bought its forward-looking infrared, or FLIR, camera in 2014. A $100,000 price tag and the need for specialized thermography training usually keep this technology out of the hands of average folks. But Sharon Wilson, a former oil and gas executive who embraced environmental activism after her own land in Texas was fracked in 2008, is fully certified.

Wilson mostly shoots oil and gas leaks and temporary industry disasters. But in late April, she took Earthworks' FLIR camera into the hills above West Elk to get a look, for the first time ever, at coal mine methane.

Sharon Wilson films a methane exhauster above West Elk Mine with her infrared camera.

West Elk is located in Gunnison County, a mile east of Somerset and about 10 miles east of Paonia, on a combination of private and federal land. Its owner, St. Louis-based Arch Coal, is the nation's second-largest coal supplier. Notably, Arch filed for bankruptcy in January, citing low coal prices and steep competition from natural gas.

But the company has no plans to stop mining. Since 2012, Arch Coal's subsidiary, Mountain Coal Company, has sought permission to add 1,721 acres to its existing federal coal leases. Although the acres in question are on U.S. Forest Service land, the Mineral Leasing Act requires that the Bureau of Land Management (BLM) approve all new coal lease requests on federal land.

Environmentalists had hoped the West Elk expansion would be halted when, in January this year, President Obama ordered a three-year moratorium on all new federal coal leases in order to more carefully consider the environmental impacts of the coal industry. But as it turns out, a legal loophole grandfathering in lease applications that have already been ruled on means the pause doesn't apply to West Elk.

Still, other obstacles have blocked the mine's expansion.

West Elk isn't just asking to dig underground. In order to vent methane from the mine, as safety regulations require, Arch Coal would need to build roads in the new lease areas — paths that would allow the drilling of vent holes and facilitate the transport of the mobile exhausters. That's a problem because the expanded West Elk leases in question fall on land within Colorado's Sunset Roadless Area.

For city folk unfamiliar with the designation, a roadless area is a swath of public land that the government says must remain, well, without roads. The BLM can still approve new coal leases in these areas, but mine operators cannot build access roads for transport — or for the generators that vent methane. For its purposes, West Elk needs roads.

When former President Bill Clinton passed the federal roadless rule in 2001, just as he was leaving office, the policy was predictably controversial. Environmentalists praised it, while conservatives and industries that operate on federal land feared losing access to the back-country would cause economic losses. When George W. Bush took over, he suggested that the rule may not last, urging interested states to formulate their own policies instead.

In the end, the federal roadless rule stayed on the books. But Colorado is one of two states – along with Idaho – that passed its own roadless policy, which environmentalists say is comparatively less protective of public lands than its federal counterpart.

The Colorado Roadless Rule governs 4.2 million acres of land, but grants some industries privileges the federal rule doesn't allow, such as logging in remote areas and backcountry skiing access. Even more contentious, it includes a 19,100-acre exception in the North Fork Valley, where West Elk is located, that allows mining companies to build temporary roads.

In 2012, just before Colorado's roadless rule passed, Jane Danowitz, then-director of the public lands program at the Washington-based Pew Environment Group, told *The New York Times* it "would give Colorado, at the end of day, fewer protections than any other state."

Plenty of supporters countered that the balance between environment and industry was the right choice for Colorado.

"…We believe it's better for Colorado – that we are able to address our unique environmental circumstances, and our unique economic circumstances, in a way that the 2001 rule simply couldn't and didn't," Mike King, then the executive director of Colorado's Department of Natural Resources, said in 2012.

When the feds approved the passage of the Colorado roadless rule that year, it seemed that King and coal supporters were victorious.

But environmental groups like WildEarth Guardians and the Sierra Club weren't giving up without a fight. They filed a lawsuit accusing multiple federal agencies, including the BLM and Forest Service, of approving both the roadless rule exception in the North Fork Valley and West Elk's lease modifications without adequately considering the environmental impacts of increased coal mining in the area.

U.S. District Judge R. Brooke Jackson sided with the environmentalists. In 2014, he halted both the roadless rule exception and West Elk's lease expansion, citing the Forest Service's failure to adequately consider the climate impacts in its environmental assessments.

The Forest Service, fighting the court's ruling, is preparing secondary environmental impact statements for both the West Elk expansion and the roadless rule exception. The final reports for both are expected this spring or summer. The coal mine's expansion, of course, hinges on federal permission to build roads in the roadless area.

Hick weighs in

Land reclamation efforts in progress above West Elk Mine

In January of this year, Hickenlooper's administration voiced support for the North Fork exemption to the roadless rule.

"…The North Fork coal mining exception was… included in the State of Colorado's petitions for rulemaking to ensure that the coal mines in that area would be able to expand and continue to provide critical jobs for Coloradans," King, then Hickenlooper's top natural resources official, wrote in a letter to the Forest Service.

Though the letter is widely considered an endorsement of the West Elk expansion, the administration won't comment specifically on the mine. In a statement on the Governor's support of the roadless exception, Todd Hartman of the Department of Natural Resources called it "part of a balanced rule that respected the economic needs of coal communities and allowed for mining-related road construction."

But what about West Elk's already massive methane emissions? And what about the inconsistency between the administration being ahead-of-the-curve tough on oil and gas methane, while letting coal mines off with a free pass to pollute?

When asked about this contradiction, the administration said that soon, coal mines like West Elk may not get away with methane emissions as easily as they have so far.

Hartman said that the administration will weigh in on future projects attempting to take advantage of the North Fork exception, including West Elk's expansion. He also said that Gov. Hickenlooper "has directed the administration to take certain steps to mitigate the impacts of continued coal mining in the North Fork Valley," including "ways in which methane emissions from proposed projects can be minimized."

This would be an important step in curbing Colorado's biggest source of methane pollution. Notably, the state in March commissioned a report through the Colorado Energy Office analyzing the commercial possibilities for coal mine methane in Colorado.

When asked for specifics about which kinds of methane regulations the administration might impose, spokeswoman Kathy Green said only that the Governor would "apply the Colorado Division of Reclamation Mining and Safety regulations that are on the books."

But the regulatory documents on that agency's website don't mention methane. And as long as coal mine methane capture and destruction remain voluntary, the coal industry will continue to be able to legally justify inaction.

'A wonderful neighbor'?

Earthworks' Wilson, a single mom in her 60s, is a flatlander – a Texan unaccustomed to altitude hiking, especially with her five-pound infrared camera. But, having never before "seen" methane from a coal mine, she was determined not to let health problems or exhaustion keep her from trudging way up the muddy mountains behind West Elk to get the shot.

Upon approaching the methane exhauster, Wilson snapped to attention. Quickly, determinedly, she went to work with her camera at multiple angles and settings, capturing images of the methane plume environmentalists and locals in these parts have long heard about, but can't see.

Why the fascination with infrared technology? "It exposes the dirty secrets that these fossil fuel industries want to keep hidden," Wilson said.

Infrared footage shows methane emissions from West Elk Mine. Video by Sharon Wilson, Earthworks.

Arch Coal said West Elk wasn't able to devote company resources for a tour of the mine and its methane exhausters. Mine employees refused to even respond to calls, emails and multiple voicemails requesting interviews.

Logan Bonacorsi, Arch's spokeswoman, stressed the economic importance of the mine. She said West Elk currently has over 300 employees. The annual pay and benefits packages of these jobs, she noted, averages over $100,000.

But the coal market is struggling, and Bonacorsi said the proposed expansion is less an opportunity for major growth than a chance simply to keep the mine's lights on.

"It is important to point out that while it is often discussed as an expansion, it's really additional reserves to allow the operation to continue mining in the future," she said in an email. "The addition of mineable coal reserves could extend the life of West Elk's current mining operations."

Locals in these parts know how key mining jobs are to their rural economy. Ed Marston, an environmentalist, longtime resident of nearby Paonia and former editor of *High Country News*, calls himself "a huge fan of coal mining in my valley." He's particularly supportive of coal compared to other fossil fuels.

"Coal has coexisted well with communities around it for many decades," he said. "Coal has been a wonderful neighbor."

In addition to the jobs it provides, West Elk has been lauded for its adherence to rules. Over the past decade, the mine has won several awards for its safety record and commitment to restoring land disturbed by mining operations.

But financial records suggest the mine's owner, Arch Coal, may not always have its employees' and shareholders' best interests in mind. As Arch Coal's fortunes have dropped steeply over the past few years, it has cut wages and suspended dividends. Meanwhile, the company's CEO, John Eaves, saw his pay nearly double from $3.9 million in 2012 to $7.3 million in 2014. And U.S. Securities Exchange Commission filings show that the company paid out $8 million in executive bonuses just one business day before it filed for bankruptcy in January.

The invisible plume

Methane pollution made international news late last year when a natural gas storage facility in Southern California's Aliso Canyon began pouring thousands of tons of methane into the air each day. The leak forced residents of a massive housing development called Porter Ranch to flee their homes, many complaining of health problems. The blowout was the worst single man-made greenhouse gas accident in history, spewing a total 97,100 metric tons of methane over only four months.

Earthworks' infrared footage of the leak — which made the invisible gas visible — garnered worldwide attention and alarm. Headlines described the situation with words like "catastrophe" and "disaster."

Triggering nowhere near the same sense of urgency or concern, West Elk legally and allowably is polluting half of what accidentally leaked from the massive Aliso Canyon blowout — 58,000 tons of methane in 2011, according to the 2012 final environmental impact statement for the expansion. And that's before the proposed expansion. A governor who claims to be tough on methane and proactive against climate change is supporting the legal exception that would allow the mine to grow and its methane pollution to significantly swell.

A map showing the proposed expansion to West Elk Mine

Ted Zukoski, an attorney with environmental law firm EarthJustice, doesn't deny that coal mining is economically important for Colorado. But he says the environmental impacts of Colorado's roadless rule exemption are too significant to ignore.

Allowing roads to be built on 20,000-acre of purportedly roadless areas in the North Fork Valley would give coal companies like Arch access to 170 million more tons of coal. That much, Zukoski says, would yield additional methane emissions equivalent to 1 to 2 million tons of carbon dioxide a year. Carbon dioxide equivalency is often used as a standard measure in order to simplify the calculation of greenhouse gas emissions. To put that amount in perspective, West Elk released the equivalent of about 650,000 metric tons of CO2 in 2014, according to the EPA. Zukoski expects the additional accessible coal would allow mining for an additional 30 years or more, depending on the coal market, with methane emissions at or above current levels.

Just because methane is a natural byproduct of coal mining doesn't mean companies have to simply spit it into the atmosphere. Companies could destroy it by flaring it, which is still a waste of potential energy, but far less polluting. Or, even better, they could capture it and harness it as an energy source.

Coal mine methane was added to Colorado's Renewable Energy Standard as an "eligible energy source" in 2013. That means utility companies get credit for using it when calculating the percentage of renewable energy they use in their operations. Investor-owned utility companies will be required to use at least 30 percent renewables starting in 2020.

But, as Zukoski tells it, West Elk hasn't properly considered those alternatives. In 2008, he and his team – which included the environmental group WildEarth Guardians – filed a lawsuit against the U.S. Forest Service for approving a previous West Elk Mine expansion without examining options beyond simply spilling the valuable methane into the sky.

WildEarth lost its case when the court ruled that capturing methane wasn't economically feasible, and that therefore the Forest Service was justified in not requiring it as an alternative to methane venting.

Zukoski says the Forest Service made its determination without considering relevant scientific and economic evidence from the Environmental Protection Agency.

"We suggested several ways the Forest Service could have limited methane emissions for the lease expansions But the Forest Service declined to require or even analyze seriously any of these measures," he said.

Zukoski called the agency's excuses for not considering the measures "bogus." In his mind, the Forest Service is ignoring the problem of coal mine methane pollution while pretending to take action.

In a 2008 op-ed in the Denver Post, Charlie Richmond, former Forest Service supervisor for the Gunnison National Forest, promised that the agency would continue to "lead the charge" on methane use. "Finding just the right solution for utilization of methane, thereby reducing greenhouse gas emissions, will not happen overnight but is on the nearby horizon," Richmond wrote.

"Eight years after that statement, the official Forest Service position remains that there is nothing to be done about methane at the West Elk Mine," Zukoski said.

"So much for 'leading the charge,'" he said.

Venting frustrations

To the naked eye, the methane perpetually spewing from West Elk's E6-6 exhauster is indiscernible. But on the infrared camera's viewing screen, the gas is starkly clear. Two black plumes billow from vents, not in bursts, but in dark, thick clouds that can be seen wafting through pine trees into sky above the West Elk Mountains and upwards into the atmosphere where science tells it will trap the sun's heat within the earth's atmosphere, slowly and steadily heating the planet.

"It's horrifying," said Wilson.

According to a recent report released by the Colorado Energy Office, West Elk currently employs a small amount of waste methane to heat the mine. In 2013, heat generation utilized 3.7 percent of the mine's total methane emissions.

West Elk Mine may not be interested in harnessing its remaining waste methane. But one Denver-based company, Vessels Coal Gas, is.

Vessels has researched and furthered the development of methane capture and flaring technologies in order to prevent valuable waste methane from wafting into the sky.

The company has a partnership with another Somerset-area mining operation, Oxbow's Elk Creek Mine. Though the mine itself went idle in 2013, a methane capture system implemented the year before is still running. Methane from Elk Creek currently generates three megawatts of electricity, which is powering all operations at Aspen Ski Company.

Electricity generation operation at Oxbow Mining's Elk Creek Mine. Aside from methane capture, Elk Creek is currently idle. Photo credit: Vessels Coal Gas

In addition to climate-savvy ski companies, capturing waste methane might make sense for some rural communities near coal mines. But existing regulations and contracts typically prevent local utilities from adding waste methane to their portfolios, which activists are working to remedy.

If captured, methane from West Elk could power almost 30,000 homes — or all the houses in Longmont.

But price is an obstacle. Given the high cost of cleaning methane, "it's not super economic" as a direct fuel source, admitted Evan Vessels, an administrator and son of company founder Tom Vessels. He said that Aspen Ski Company, which paid a premium to set up its methane capture operation at Elk Creek, was more interested in broadening its energy portfolio than in cost savings.

Another way to reduce methane emissions is to simply flare, or burn, the leaking gas. Though flaring doesn't capture any energy or electricity potential, and still emits greenhouse gases, it converts methane into much smaller amounts of CO2. The net global warming impact of flared methane is about 80 percent less than vented gas.

California encourages methane flaring with its carbon credit cap-and-trade market. Flared methane earns credits, which Vessels Coal Gas sells for about $10 each on California's market. If Colorado created such a market — or if Congress implemented a nationwide price on carbon — monetizing waste methane would be even easier.

"The reason why it makes business sense to vent methane is because we don't put a price on climate pollution," said Earthworks Executive Director Jennifer Krill. "That's got to change."

Krill said it's time for the state to reckon with the troubling reality of coal mine methane.

"The global climate shouldn't be the dumping ground for any industry, and Colorado recognizes that when it comes to both the oil and gas industry, and when it comes to burning coal."

Now, she added, "the challenge is to stop letting coal mining dump its pollution in our global climate."

But as things stand, even methane flaring, which is a comparatively easy and inexpensive practice — "It's pennies," Evan Vessels said — fails to win over coal companies in Colorado.

"They're just uncooperative," Vessels said.

The problem, according to both owner and son, is that coal companies feel entitled to all revenue generated from what they feel is "their" resource. Why pay a middleman to capture or flare your methane when you can do it yourself for free?

But the reality is that most coal companies aren't doing it themselves because they're typically too busy with daily operations. Oxbow's Elk Creek Mine, in its partnership with Aspen Ski Company, is the only mine in the state to make use of the technology.

Arch Coal told *The Independent* that it "continue[s] to explore opportunities for the potential future viability of the commercial use of methane."

But Evan Vessels counters that's simply not true.

"They're sitting around, and we are begging them to let us make them money," he said.

In the meantime, it's with impunity that West Elk and other underground coal mines keep spewing their invisible of methane from exhausters so remote that nobody notices or thinks much about their effect on the planet.

And why not? Without regulations like the kind Colorado has set for oil and gas, there's nothing stopping them.

Correction May 4, 2016: *This story orginally stated Hickenlooper's oil and gas regulations are expected to cut methane emissions equivalent to 2.34 tons of CO2 each year. In fact, they are expected to cut methane emissions by 2.34 million tons of CO2.*

Cover photo credit: Sharon Wilson, Earthworks; story photos by Kelsey Ray

Accessibility

Kelsey Ray

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Drill, baby, drill: the influence of woodpeckers on post-fire vertebrate communities through cavity excavation

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Abstract

Several studies have addressed the importance of woodpeckers as ecological engineers in forests due to their excavation of cavities. Although research in green, unburned forests has identified the influence of different excavators on secondary use by cavity-dependent species, little is known about the relative importance of cavities created by woodpeckers in recently burned forests. By excavating cavities, woodpeckers create habitat for secondary cavity users that can facilitate post-fire regeneration through seed dispersal, seed germination and regulation of insect populations that affect vegetative growth. In this study, we monitored 77 cavities created by three species of *Picoides* woodpeckers for use by secondary cavity species in a fire that burned in the Sierra Nevada, California. At each cavity we measured nest tree and site-specific parameters to determine if these characteristics could explain differential use by secondary cavity users. We found substantial overlap in cavity characteristics between woodpecker species, with the white-headed woodpecker differing most notably in their placement of cavities in larger diameter, shorter and more decayed trees in less dense stands than either hairy or black-backed woodpeckers. These differences in cavity placement may have resulted in the high diversity and large number of detections of secondary cavity species in white-headed woodpecker cavities. Black-backed and hairy woodpeckers were similar in the number of detections of secondary cavity use, although black-backed woodpecker cavities were used by more species than hairy woodpecker cavities. Secondary cavity use was high (86%) suggesting these woodpeckers, and the white-headed woodpecker in particular, can have an accelerating affect effect on ecological succession by providing valuable habitat features for seed dispersing birds and mammals, insectivorous birds, and small predators, thereby impacting ecological processes and functions.

Introduction

Ecological succession describes changes in plant and wildlife community composition over time and is often illustrated by changes in habitat following disturbance (Clements, 1904; Connell & Slatyer, 1977). The path of ecological succession is unpredictable and may depend on the facilitative effects of ecosystem engineers. Ecosystem engineers have the ability to influence succession by altering or creating habitat (Lawton & Jones, 1995) and may facilitate colonization or re-colonization by other species after disturbance (Andersen & MacMahon, 1985; Nummi & Holopainen, 2014). Many species of woodpeckers may act as ecosystem engineers that affect successional dynamics following fire, the dominant source of disturbance in mixed conifer forests of western North America (Skinner & Chang, 1996). They may colonize burned areas immediately after fire by exploiting the abundance of food (bark and wood-boring beetles) and nest resources (snags), utilizing habitat unsuitable to many other bird and mammal species (Kotliar *et al*., 2002). As ecosystem engineers, woodpeckers excavate cavities that provide nesting, roosting, denning and resting sites for secondary cavity users (SCUs), species dependent on cavities, but unable to excavate them (Raphael & White, 1984). In many coniferous habitats cavities that are not created by woodpeckers are rare (Aitken & Martin, 2007) and competition for cavities has been shown to limit population growth of SCUs (Holt & Martin, 1997), creating a strong dependence of SCUs on woodpeckers (Martin & Eadie, 1999). In the Sierra Nevada alone, there are over 50 SCUs from several functional groups including seed and spore dispersers, insectivores and small carnivores (Raphael & White, 1978, 1984).

In green, unburned forests, Northern flickers *Colaptes auratus* are considered keystone excavators because their cavities are abundant and used by many species of SCUs (Martin & Eadie, 1999; Martin, Aitken & Wiebe, 2004; Robles & Martin, 2013, 2014); however, this species is relatively rare in burned forests in the first years after fire. This suggests that immediately after fire, woodpeckers that are early colonizers of burned habitat will provide the majority of cavities that support SCUs. Whereas abundance patterns and reproductive success of woodpeckers colonizing recent burns is well-documented (Saab, Dudley & Thompson, 2004; Nappi & Drapeau, 2009), little attention has been given to their contribution as ecosystem engineers for SCUs in burned forests, although they may be especially important after fire has consumed previously existing snags with cavities (Horton & Mannan, 1988; Bagne, Purcell & Rotenberry, 2008).

In coniferous forests of the western US, three members of the *Picoides* genus can be commonly found in burned habitat: the black-backed woodpecker *Picoides arcticus*, hairy woodpecker *Pic. villosus* and white-headed woodpecker *Pic. albolarvatus*. These three species rarely re-use cavities and are characterized by strong excavation abilities (Garrett, Raphael & Dixon, 1996; Dixon & Saab, 2000; Jackson, Ouellet & Jackson, 2002). Because these species are similar in size, their cavities have roughly the same dimensions, with nearly identical minimum diameter at entrance, depth and internal cavity diameter (Raphael & White, 1984). Although it is likely that these species differ in aspects of nest characteristics, there is a limited understanding of how they differ and the influence those differences may have on the structure and composition of SCUs utilizing post-fire habitat.

The objective of this study was to investigate cavity use by SCUs in recently burned coniferous forest. In the second and third year following fire we located cavities, recorded the excavating species and collected data on nest tree and site characteristics. Each cavity was monitored for subsequent use by SCUs in the following breeding season. We compared cavity characteristics between these species to determine if any observed differences were associated with different SCUs. In this way, we were able to estimate the influence or importance of each of these woodpecker species in providing habitat to SCUs.

Methods

Study area

We conducted our study on the south-west shore of Lake Tahoe (38.91°N, 120.04°W), *c.* 140 km east of Sacramento, California, where the human-caused Angora fire burned *c.* 1245 ha of coniferous forest in June and July 2007. The fire occurred in an area with a high degree of intermixed private and public land and adjacent to large expanses of undeveloped public land. Some public lands around the fire had been treated in the past to reduce fuels. Burn severity varied within the area, resulting in a mosaic of post-fire conditions. Pre-fire vegetation was predominately Jeffrey pine *Pinus jeffreyi* and white fir *Abies concolor* forest with red fir *A. magnifica* found at the highest elevations and incense cedar *Calocedrus decurrens*, sugar pine *Pin. lambertiana* and lodgepole pine *Pin. contorta* found at the lower elevations. For a more complete site description and characterization of the fire, see Safford, Schmidt & Carlson (2009).

Woodpecker nest searching

The US Forest Service established a systematic grid of points spaced 400-m apart across the fire area. From this grid, we selected a sample of survey points that were roughly balanced across burn severity classes. In order to increase our sample size and help balance the design, we selected a limited number of points on other public lands that had also burned in the fire. A total of 98 unique points were sampled over the 2-year period, with 41 of these sites sampled in both years. We categorized each point into one of four burn severity classes based on satellite-derived per cent tree mortality (0, 1–20, 20–70, >70%). The total area surveyed was *c.* 308 ha with 72% of the surveyed area within the fire perimeter and 18% in unburned forest surrounding the fire area.

Cavity searches were conducted between May and July in 2009 and 2010, 2 and 3 years after the fire burned. At each survey point, observers first thoroughly canvassed a 60-m radius (*c.* 1 ha) area for active cavity nests (Martin & Geupel, 1993) and cavity nesters (minimum of 15 min). Then observers moved out into the area between 60 and 100 m from the survey point (*c.* 2-ha area), and spent a minimum of 1 h searching this larger area. If a cavity or bird was observed, no maximum time was set to determine if the cavity was active or to locate the bird's nest. Nests that were encountered while moving to and from sites were also included in our sample. Survey points were searched a minimum of three times per season, with at least 1 week between visits.

Woodpecker nest tree and nest site characteristics

When an active nest was confirmed, the bird species, stage of nest development and location of the nest were recorded. After the nesting attempt ended we recorded tree species, diameter at breast height (DBH) and tree height, cavity height, per cent scorch of the bole and decay class (Cline, Berg & Wight, 1980). To quantify vegetation structure at the nest site, we established an 11.3-m radius plot (0.04-ha) surrounding the cavity tree (James & Shugart, 1970; Martin *et al.*, 1997). Although this area does not represent the home range for the woodpecker species, it does represent the scale of habitat that may influence use by SCUs. At each nest site, we collected data on the density of snags and live trees and the per cent cover of coarse woody debris (CWD), shrubs and herbs. Stems of trees and snags were categorized as medium (28–61 cm DBH) or large (>61 cm DBH) and densities were calculated for each size class as the number of stems per hectare. Smaller diameter trees and snags were rare within the fire and were not included in this measure. Because burn severity varied within each 0.04 ha plot, we weighted each burn severity value by the per cent area it represented. To determine if human infrastructures influenced use of nest sites, we used an impervious surfaces (houses, roads, buildings) data layer (Manley *et al*., 2009) to calculate a per cent cover of urban constructs around each nest.

We used discriminant function analysis to identify the nest characteristics that best differentiated the species at both the nest tree and nest site scale. At the scale of the nest tree we included cavity height, tree height and DBH, decay class, and per cent of the bole that was scorched. At the nest site scale we analyzed medium- and large-snag densities, total tree density, per cent cover of CWD, per cent cover of shrubs, per cent cover of herbs, per cent cover of impervious surfaces and burn severity. Explanatory variables were standardized and transformed prior to analysis.

Secondary cavity use

Because our interests were in understanding how different species of woodpeckers may influence vertebrate assemblages following fire, we monitored woodpecker cavities for one breeding season following excavation to quantify use by SCUs. Remote-triggered digital cameras (Leaf River Outdoor Products, Taylorsville, MS, USA) were placed at cavities to monitor use for two, 7-day sessions in the breeding season of 2010 and 2011, allowing for detection of elusive, diurnal and nocturnal wildlife. All individuals detected in photographs were identified to species when possible. In addition, we used a Treetop Peeper (Sandpiper Technologies, Manteca, CA, USA) to observe the interior of cavities twice during each season to check for active nests or dens, nesting material and other evidence of use. Data on SCUs were used to create nest webs to compare the influence of each woodpecker species on the SCU community (Martin & Eadie, 1999).

To quantify and compare the biodiversity supported by each woodpecker species, we calculated an effective species number (ESN) based on the number of species of SCUs and the number of individual sightings of each species found in cavities excavated by each woodpecker (Jost, 2006). The ESN is derived from the Shannon Diversity Index (H); however, the ESN is measured in units of number of species and is linearly scaled such that communities with ESNs that differ by a factor of two represent an actual difference in diversity such that one community is twice as diverse as the other.

$$ENS = \exp\left(H = -1 \times \sum_{i=1}^{S} p_i \, ln \, p_i\right)$$

where S is the total number of species and p is the proportion of species i relative to the total number of species (p_i).

The ENS represents the number of species in a community given that each species is equally abundant.

Results

Woodpecker nest tree and nest site characteristics

A total of 257 cavities were found during the two surveyed breeding seasons (110 in 2009, 147 in 2010). The majority of cavities found were attributed to the three target woodpecker species including 39 black-backed woodpecker nests (15 in 2009 and 24 in 2010), 80 hairy woodpecker nests (37 in 2009 and 43 in 2010) and 48 white-headed woodpecker nests (18 in 2009 and 30 in 2010). Nests were found in roughly equal proportions for each species in each stage; therefore, we assume that our sample is not biased toward successful nests or biased by species-specific detection probabilities related to parental behavior. Nests of the black-backed woodpecker occurred within the fire and were almost exclusively located in *Pinus* species. In contrast, white-headed woodpeckers (five nests) and hairy woodpeckers (two nests) built nests in unburned areas and utilized the different tree species in similar proportions. Mean values for nest tree and site characteristics for each species of woodpecker in the burned area are shown in Table 1, nests located in unburned areas were omitted from analysis. Other cavities excavated within the burned survey area included 10 Northern flicker, three pileated woodpecker *Dryocopus pileatus*, two Williamson's sapsucker *Sphyrapicus thyroideus*, two red-breasted nuthatch *Sitta canadensis* and 16 pygmy nuthatch *Si. pygmaea* nests, plus one cavity that was created by a broken branch and rot. We also located 41 woodpecker-excavated nests that were already occupied by SCUs. To ensure a robust sample size, we focused our monitoring efforts on cavities created by the *Picoides* species.

The variance in individual nest tree and site characteristics across the focal species overlapped substantially (Table 1). The results of the discriminant function analysis indicate that the mean values of the tree characteristics were more effective at differentiating nests of woodpecker species than the mean site characteristics (Table 2). At the tree scale we found that both canonical axes were significant (Table 2a), with the first axis explaining 75% of variation among species. The first axis represented a gradient of short, large diameter, decayed snags to tall, smaller diameter, hard snags. This first axis primarily separated the mean value of white-headed woodpecker nests from the mean values of hairy and black-backed woodpecker nests. On average, white-headed woodpeckers tended to utilize larger, shorter and more decayed snags than the other two species (Fig. 1). The second axis primarily represented cavity height (Table 2a), with increasing values along the axis. This axis primarily separated the black-backed woodpecker from the hairy and white-headed woodpeckers (Fig. 1). Black-backed woodpeckers excavated cavities lower on the bole than hairy woodpeckers and in higher trees than white-headed woodpeckers. However, despite the separation in canonical variate means, the probability of correctly assigning a cavity to a focal species based on nest tree characteristics was relatively low, with only the most extreme values having predictive power (Fig. 1).

Table 1 Summary of means and standard deviations of nest characteristics for three woodpecker species

	Black-backed woodpecker	Hairy woodpecker	White-headed woodpecker
Nest tree			
Pinus species (%)	82	55	45
Cavity height (m)	4.72 ± 3.01	7.23 ± 4.10	4.03 ± 2.35
DBH (cm)	34.48 ± 8.64	38.73 ± 8.69	50.62 ± 48.11
Tree height (m)	16.49 ± 6.24	16.58 ± 7.52	9.20 ± 7.41
Decay (1–5)	1.56 ± 1.02	1.49 ± 0.85	2.85 ± 1.33
Scorch on tree (%)	93 ± 15	93 ± 13	99 ± 4
Nest site			
Herb cover (%)	14 ± 13	13 ± 16	17 ± 20
Shrub cover (%)	25 ± 18	30 ± 26	26 ± 23
CWD cover (%)	2 ± 3	3 ± 2	2 ± 2
Trees (stems per hectare)	5.02 ± 16.7	3.34 ± 14.2	10.04 ± 24.84
Small snags (stems per hectare)	178.69 ± 95.08	139.96 ± 90.98	94.69 ± 75.66
Large snags (stems per hectare)	19.37 ± 20.61	23.64 ± 33.39	16.50 ± 24.55
Burn severity (%)	93 ± 9	93 ± 10	91 ± 10
Impervious cover (%)	7 ± 7	5 ± 7	9 ± 8

CWD, coarse woody debris; DBH, diameter at breast height.

Table 2 Tests of dimensionality and standardized discriminant coefficients for discriminant function analysis of nest characteristics of black-backed woodpecker, hairy woodpecker and white-headed woodpecker at two scales: (a) nest tree and (b) nest site

	Canonical dimensions	
(a) Nest tree		
Predictors	1	2
Tree height	0.495	–0.710
DBH	–0.339	0.144
Decay	–0.603	0.002
Cavity height	0.297	1.180
Scorch	–0.145	0.191
Canonical correlation	0.57	0.38
	$F_{10,302} = 9.64$	$F_{4,152} = 6.30$
	$P < 0.001$	$P < 0.001$
(b) Nest site		
Predictors	1	2
Herb cover	0.141	–0.001
Shrub cover	–0.057	–0.194
Coarse woody debris	–0.079	–0.386
Total tree density	0.099	0.258
Small snag density	0.978	0.413
Large snag density	0.031	–0.442
Burn severity	0.120	0.274
Impervious cover	–0.346	0.752
Canonical correlation	0.358	0.658
	$F_{16,296} = 1.89$	$F_{7,149} = 1.25$
	$P = 0.021$	$P = 0.279$

Standardized coefficients indicate the relationship of each variable to one standard deviation of change in the dimension.
DBH, diameter at breast height.

At the site scale, only the first canonical axis was statistically significant ($P = 0.02$; Table 2b; Fig. 2) and eigenvalues indicate that it explained 71% of the variation that existed among the species. The single most influential variable in classifying nests by species was density of medium-diameter snags, which increased along the axis. White-headed woodpecker nests were characterized by the lowest density of medium snags surrounding the nest, black-backed woodpeckers had the highest medium snag density and hairy woodpeckers nested in sites with intermediate densities of medium snags. Again, we found that the substantial degree of overlap in the use of site characteristics between species made it difficult to distinguish nests of one species from another.

Secondary cavity use

Despite the abundance of cavities found in the burned forest, a large proportion of nests were lost during the first winter following excavation as a result of snags falling or breaking at or below the cavity. Of the 158 nests monitored for secondary cavity use, 81 fell prior to the subsequent breeding season. Black-backed and hairy woodpecker nests had the greatest attrition, with losses of 54 and 65%, respectively. In contrast, only 24% of white-headed woodpecker nests fell 1 year after excavation. The remaining 77 woodpecker cavities were available for secondary cavity use observations the following breeding season: 18 black-backed (2010 = 8, 2011 = 10), 27 hairy (2010 = 13, 2011 = 14) and 32 white-headed (2010 = 11, 2011 = 21).

A total of 111 detections of secondary cavity use were observed across the 77 nests monitored with a high percentage (86%) of cavities receiving visits by one or more SCUs. The white-headed woodpecker had the highest proportion of their cavities visited (94%), followed by the black-backed woodpecker (89%) and the hairy woodpecker (73%). Cavities of all three species of woodpeckers were utilized by both birds and small mammals. Ten species of SCUs were detected: seven bird species, two small mammal species, plus chipmunks (representing multiple *Tamias* species; Fig. 3). White-headed woodpecker cavities had both the largest number of individual

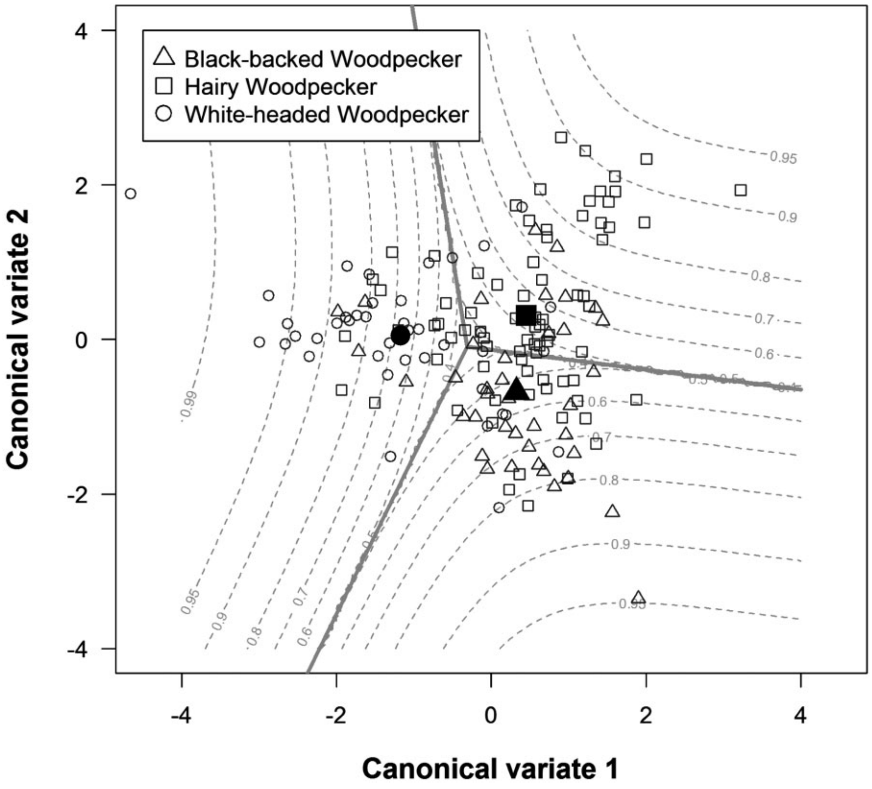

Figure 1 Canonical dimension scores of the three species of woodpecker at the tree scale. Small symbols are the individual sample points and the larger symbols (shown in the legend) represent the location of the canonical variate means. The gray lines show the separation of the predictions from the linear discriminant function. The associated prediction is the same as that of the species where the location of the canonical variate mean resides. The contour lines represent the posterior probability of assignment for each of the three species.

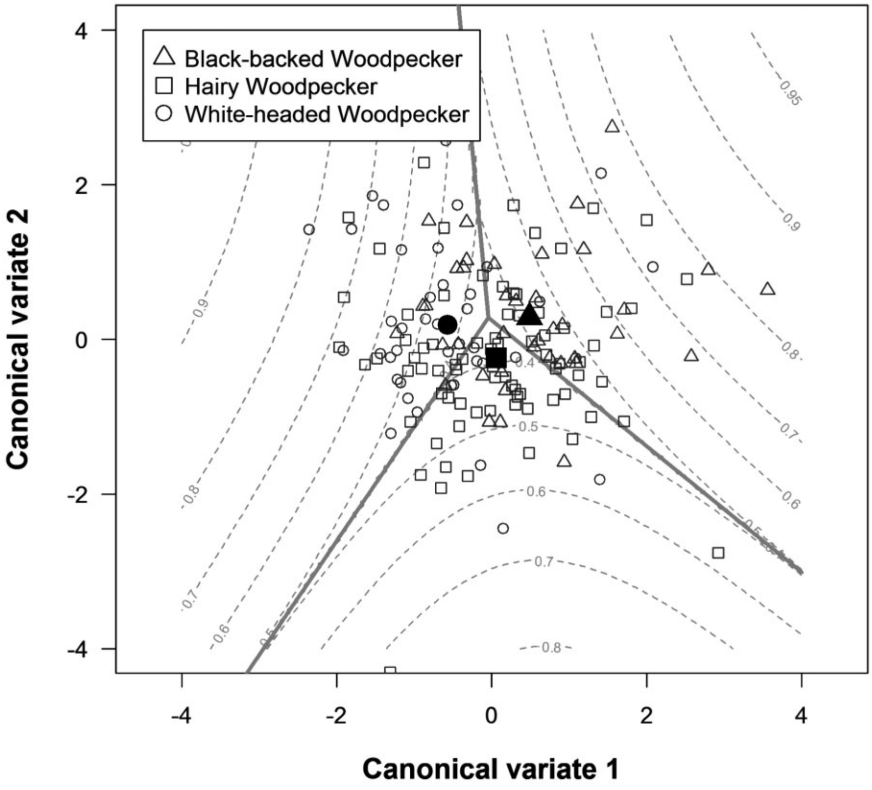

Figure 2 Canonical dimension scores of the three species of woodpecker at the site scale. Small symbols are the individual sample points and the larger symbols (shown in the legend) represent the location of the canonical variate means. The gray lines show the separation of the predictions from the linear discriminant function. The associated prediction is the same as that of the species where the location of the canonical variate mean resides. The contour lines represent the posterior probability of assignment for each of the three species.

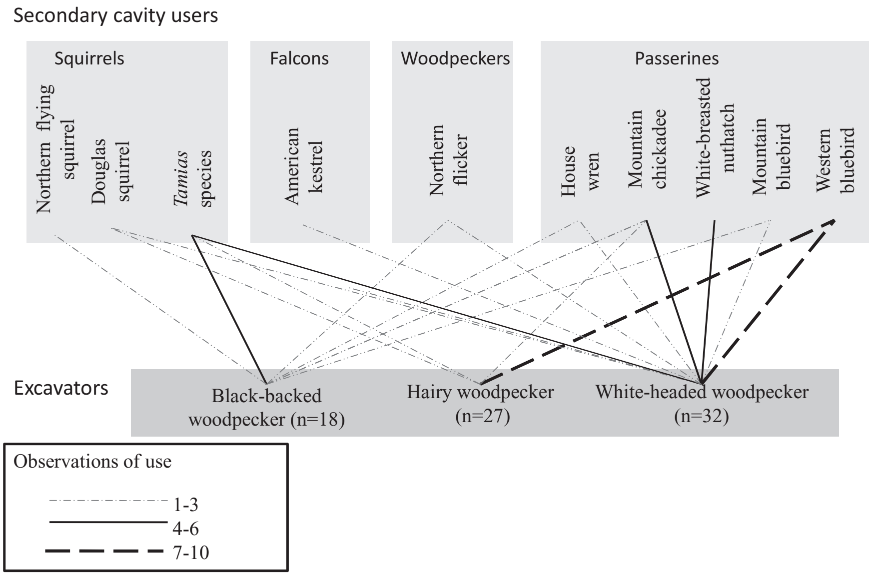

Figure 3 Nest web of cavity-dependent community after the Angora Fire. Lines indicate the number of observations of species using a particular resource. 'Use' occurred when an organism was observed in the cavity or observed entering the cavity.

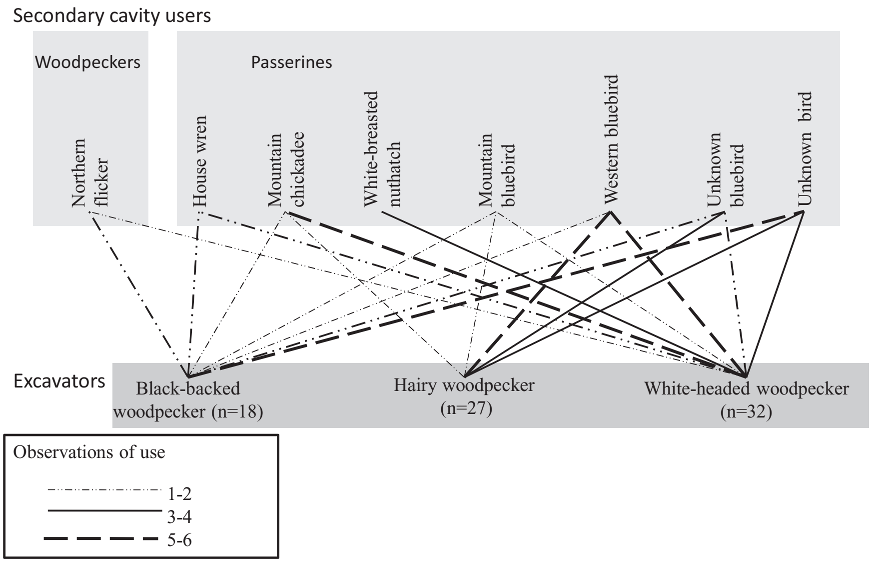

Figure 4 Nest web of breeding cavity nesting birds in the Angora Fire. Lines indicate the number of observations of species using a particular resource. 'Use' occurred when a bird was observed building a nest, incubating eggs or feeding nestlings or mate.

sightings and highest diversity of SCUs detected. White-headed woodpecker cavities were used by every species of SCU, with the exclusion of the Northern flying squirrel. Black-backed and hairy woodpeckers were similar in the number of detections in their cavities, although cavities excavated by the black-backed woodpecker were used by more species. Subsequently, the diversity index for white-headed woodpeckers had the highest value (ESN = 8 species), followed by the black-backed woodpecker (ESN = 6 species) and the hairy woodpecker (ESN = 4 species). This indicates that white-headed woodpeckers supported two more species than the black-backed woodpecker and a community that was twice as diverse as that supported by the hairy woodpecker. Of the 111 detections of SCUs, 52 were classified as breeding attempts by cavity nesting birds. We observed denning material in seven cavities used by small mammals, but could not confirm that these cavities were used specifically for breeding. White-headed woodpecker cavities were used by every breeding bird in the study and had the most total detections, followed by black-backed and hairy woodpeckers, respectively (Fig. 4).

Discussion

Many species of woodpeckers play an important role in forests by creating cavities that are used by other species that rely on them for aspects of their life history (Martin & Eadie, 1999). This ecosystem engineering by woodpeckers may be paramount post-fire when some species of woodpeckers can rapidly colonize burned forests that are less accessible to other animals (Kotliar *et al.*, 2002). We found a high density of cavities in the first few years following fire indicating that several woodpecker species, and in particular the white-headed, hairy and black-backed woodpeckers, capitalized on this newly created habitat. Nearly all cavities that were monitored had detections of use, indicating that by excavating cavities, woodpeckers allowed SCUs to take advantage of the ephemeral resources provided by the fire. By facilitating movement into the burned area, woodpeckers may accelerate forest succession by providing habitat for seed and spore dispersing small mammals, insectivorous birds and small predators, thereby impacting ecological processes and functions (Otvos, 1979; Maser & Maser, 1988; Carey & Harrington, 2001). Although this study does not address whether these species would enter the burned area in the absence of cavities, engineering by woodpeckers undoubtedly facilitates the colonization or re-colonization of burned areas by SCUs.

Although all three species of woodpecker play a role in facilitating the presence of SCUs and could potentially impact forest succession, the objective of this study was to investigate whether different species of woodpeckers support unique communities of SCUs in burned forests. We found the amount of overlap in SCUs commensurate with the amount of overlap in the characteristics at the nest and site scales of these excavating species. However, there appeared to be some differences in the communities that the excavators supported. In particular, white-headed woodpecker cavities were utilized the most, supported the highest diversity of SCUs and were used in the highest proportion by nesting birds.

White-headed woodpeckers may provide cavities that are preferred by SCUs due to their nest tree or site characteristics. White-headed woodpecker cavities differed most notably from the other two woodpeckers in their placement in larger DBH, shorter and more decayed snags in less dense stands. In our study, white-headed woodpecker cavities, although they occurred in more decayed snags, were more likely to remain standing after excavation (at least 1 year more) because the cavities were low on the bole of large diameter snags. Cavities placed lower on the bole may be more persistent because the snag is unlikely to break at or below this height. In comparison, hairy woodpecker cavities, located highest on the bole in smaller diameter snags, fell the most frequently. Due to their relative persistence, white-headed woodpecker cavities were the most abundant in the fire area and may have been used more regularly by SCUs based on the formation of a search image.

Although not as readily used as white-headed woodpecker nests, black-backed woodpecker cavities supported a higher diversity than hairy woodpecker cavities. Black-backed woodpecker cavities were similar to white-headed woodpecker cavities in their low placement on the bole of the tree. Cavity height may affect the likelihood of predation and may therefore influence whether cavities are occupied by SCUs. Although in green forests, lower cavities are reportedly depredated more than higher cavities (Li & Martin, 1991; Fisher & Wiebe, 2006), this relationship has not been described in burned forests. Overall, predation in burned forests tends to be lower than in green forests (Saab & Vierling, 2001) and burned forests may be dominated by visual predators such as corvids (Jones *et al.*, 2002) that may depredate higher nests that are visible to these aerial predators.

In green forests, Northern flickers are the most influential woodpecker, providing abundant cavities and supporting a diverse community of SCUs (Martin & Eadie, 1999; Martin *et al.*, 2004; Robles & Martin, 2013, 2014). However, in the first two years of nest searching following the fire, we only located 10 Northern flicker nests within the burned area. This is likely to change over time as snags fall, shrubs and herbs dominate the live cover and arthropods and ants re-populate the area, providing forage for flickers. Northern flickers create larger holes and may provide habitat for larger SCUs, increasing the overall diversity in the burned area. However, Northern flickers often enlarge cavities created by other species rather than creating their own nest hole. Although the direct role of the *Picoides* woodpeckers may diminish as time since fire increases, their cavities may indirectly support the movement of Northern flickers and larger SCUs into the burned area.

Despite the common classification of black-backed woodpeckers as fire specialists and white-headed woodpeckers as green forest specialists, burn severity was not a factor that differentiated nest sites of the three species of woodpeckers. This discordance may be partially explained by the smaller habitat scale considered in this study, however, when investigating larger scale habitat features (7.04-ha scale) at our survey points we found a lack of variability in burn severity (Tarbill, 2010), suggesting that for a fire of this size with densely packed resources, smaller scales may be more important. The relatively small size and the elongated shape of the fire also resulted in highly burned areas near green edge habitat. We observed that the burned forests, even in the first few years after fire, supported a wide array of species not commonly considered burn specialists including chipmunks, Northern flying squirrels and mountain chickadees. Forest edge habitat has been associated with increases in avian species richness and abundance (Sisk & Battin, 2002). However, cavities in burned areas may be ecological sinks (Pulliam & Danielson, 1991) or traps (Dwernychuk & Boag, 1972) for SCUs. If cavities in the surrounding green forest are rare, SCUs may be forced to use cavities in burned areas or forego reproduction.

Although new snags are created by fire and new cavities are created by these early colonizing species of woodpeckers, cavities may continue to be limited due to the ephemeral nature of burned snags. Most fire-killed trees die within 2 years of fire, although delayed mortality occurs at least 10 years post-fire (Angers *et al.*, 2011). Snags with cavities are structurally compromised and may be more susceptible to decay (Farris & Zack, 2005), leading to increased fall rates. We observed a loss of 51% of snags with excavated cavities in the second year following the fire, supporting the notion that cavities could be limited even in areas with high-snag densities. Communities of SCUs in burned forests will depend on continued presence of woodpeckers to replenish the supply of cavities. As time since fire increases, the rate of snag fall will quickly outpace the rate of tree death and the forest will slowly regenerate, producing areas rich in shrubs and small trees (Bock, Raphael & Bock, 1978). Snags will again become rare and the engineering of woodpeckers will continue to be important to SCUs.

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References

Aitken, K.E.H. & Martin, K. (2007). The importance of excavators in hole-nesting communities: availability and use of natural tree holes in old mixed forests of western Canada. *J. Ornithol.* **148**, S425–S434.

Andersen, D.C. & MacMahon, J.A. (1985). Plant succession following the Mount St. Helens volcanic eruption: facilitation by a burrowing rodent, *Thomomys talpoides*. *Am. Midl. Nat.* **114**, 62–69.

Angers, V.A., Gauthier, S., Drapeau, P., Jayen, K. & Bergeron, Y. (2011). Tree mortality and snag dynamics in North American boreal tree species after a wildfire: a long-term study. *Int. J. Wildland Fire* **20**, 751–763.

Bagne, K.E., Purcell, K.L. & Rotenberry, J.T. (2008). Prescribed fire, snag population dynamics, and avian nest site selection. *For. Ecol. Manage.* **255**, 99–105.

Bock, J.H., Raphael, M. & Bock, C.E. (1978). A comparison of planting and natural succession after a forest fire in the northern Sierra Nevada. *J. Appl. Ecol.* **15**, 597–602.

Carey, A.B. & Harrington, C.A. (2001). Small mammal in young forests: implications for sustainability. *For. Ecol. Manage.* **154**, 289–309.

Clements, F.E., 1904. Development and structure of vegetation. Botanical Seminar.

Cline, A.B., Berg, A.B. & Wight, H.M. (1980). Snag characteristics and dynamics in Douglas fir forests, Western Oregon. *J. Wildl. Mgmt.* **44**, 773–786.

Connell, J.H. & Slatyer, R.O. (1977). Mechanisms of succession in natural communities and their role in community stability and organization. *Am. Nat.* **111**, 1119–1144.

Dixon, R.D. & Saab, V.A. (2000). Black-backed woodpecker *Picoides arcticus*. In *The birds of North America online*. Poole, A. (Ed.). Ithaca: Cornell Lab of Ornithology. http://bna.birds.Cornell.edu/bna/species/509, doi: 10.2173/bna.509.

Dwernychuk, L.W. & Boag, D.A. (1972). Ducks nesting in association with gulls – an ecological trap? *Can. J. Zool.* **50**, 559–563.

Farris, K.L. & Zack, S. (2005). Woodpecker-snag interactions: an overview of current knowledge in ponderosa pine systems. USDA Forest Service. PSW-GTR-198.

Fisher, R.J. & Wiebe, K.L. (2006). Nest site attributes and temporal patterns of northern flicker nest loss: effects of predation and competition. *Oecologia* **147**, 744–753.

Garrett, K.L., Raphael, M.G. & Dixon, R.D. (1996). White-headed woodpecker *Picoides albolarvatus*. In *The birds of North America online*. Poole, A. (Ed.). Ithaca: Cornell Lab of Ornithology. http://bna.birds.cornell.edu/bna/species/252, doi: 10.2173/bna.252.

Holt, R.F. & Martin, K. (1997). Landscape modification and patch election: the demography of two secondary cavity nesters colonizing clearcuts. *Auk* **114**, 443–455.

Horton, S.P. & Mannan, R.W. (1988). Effects of prescribed fire on snags and cavity-nesting birds in southeastern Arizona pine forests. *Wildl. Soc. Bull.* **16**, 37–44.

Jackson, J.A., Ouellet, H.R. & Jackson, B.J. (2002). Hairy woodpecker (*Picoides villosus*). In *The birds of North America online*. Poole, A. (Ed.). Ithaca: Cornell Lab of Ornithology. http://bna.birds.cornell.edu/bna/species/702, doi: 10.2173/bna.702.

James, F.C. & Shugart, H.H. (1970). A quantitative method of habitat description. *Ecology* **59**, 871–883.

Jones, D.D., Conner, L.M., Warren, R.J. & Ware, G.O. (2002). The effect of supplemental prey and prescribed fire on success of artificial nests. *J. Wildl. Mgmt.* **66**, 1112–1117.

Jost, L. (2006). Entropy and diversity. *Oikos* **113**, 363–375.

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. *Stud. Avian Biol.* **25**, 49–64.

Lawton, J.H. & Jones, C.G. (1995). Linking species and ecosystems: organisms as ecosystem engineers. In *Linking species and ecosystems*: 141–150. Lawton, J.H. & Jones, C.G. (Eds). New York: Chapman and Hall.

Li, P. & Martin, T.E. (1991). Nest-site selection and nesting success of cavity-nesting birds in high elevation forest drainages. *Auk* **108**, 405–418.

Manley, P.N., Parks, S.A., Campbell, L.A. & Schlesinger, M.D. (2009). Modeling urban land development as a continuum to address fine-grained habitat heterogeneity. *Landscape Urban Plan.* **89**, 28–36.

Martin, K. & Eadie, J.M. (1999). Nest webs: a community-wide approach to the management and conservation of cavity-nesting forest birds. *Forest Ecol. Manage.* **115**, 243–257.

Martin, K., Aitken, K. & Wiebe, K.L. (2004). Nest sites and nest webs for cavity-nesting communities in interior British Columbia, Canada: nest characteristics and niche partitioning. *Condor* **106**, 5–19.

Martin, T.E. & Geupel, G.R. (1993). Nest-monitoring plots: methods for locating nests and monitoring success. *J. Field Ornithol.* **64**, 507–519.

Martin, T.E., Paine, C., Conway, C.J., Hochachka, W.M., Allen, P. & Jenkins, W. (1997). BBIRD field protocol. Biological Resources Division, Montana Cooperative Research Unit, Missoula, MT.

Maser, C. & Maser, Z. (1988). Interactions among squirrels, mycorrhizal fungi, and coniferous forests in Oregon. *Great Basin Nat.* **48**, 358–369.

Nappi, A. & Drapeau, P. (2009). Reproductive success of black-backed woodpeckers (*Picoides arcticus*) in burned boreal forests: are burns source habitat? *Biol. Conserv.* **142**, 1381–1391.

Nummi, P. & Holopainen, S. (2014). Whole community facilitation by beaver: ecosystem engineer increases waterbird diversity. *Aquat. Conserv.* **24**, 623–633.

Otvos, I.S. (1979). The effects of insectivorous bird activities in forest ecosystems: an evaluation. In *The role of insectivorous birds in forest ecosystems*: 341–374. Dickson, J.G., Conner, E.N., Fleet, R.R., Kroll, J.C. & Jackson, J.A. (Eds). New York: Academic Press.

Pulliam, H.R. & Danielson, B.J. (1991). Sources, sinks and habitat selection: a landscape perspective on population dynamics. *Am. Nat.* **137**, S50–S66.

Raphael, M.G. & White, M. (1978). Snags, wildlife, and forest management in the Sierra Nevada. *Cal-Neva. Wildl.*, 23–41.

Raphael, M.G. & White, M. (1984). Use of snags by cavity-nesting birds in the Sierra Nevada. *Wildl. Monogr.* **86**, 1–66.

Robles, H. & Martin, K. (2013). Resource quantity and quality determine the inter-specific associations between ecosystem engineers and resource users in a cavity-nest web. *PLoS ONE* **8**, e74694.

Robles, H. & Martin, K. (2014). Habitat-mediated variation in the importance of ecosystem engineers for secondary cavity nesters in a nest web. *PLoS ONE* **9**, e90071.

Saab, V.A. & Vierling, K.T. (2001). Reproductive success of Lewis's woodpecker in burned pine and cottonwood riparian forests. *Condor* **103**, 491–501.

Saab, V.A., Dudley, J. & Thompson, W.L. (2004). Factors influencing occupancy of nest cavities in recently burned forests. *Condor* **106**, 20–36.

Safford, H.D., Schmidt, D.A. & Carlson, C.H. (2009). Effects of fuel treatments on fire severity in an area of wildland–urban interface, Angora Fire, Lake Tahoe Basin, California. *Forest Ecol. Manag.* **258**, 773–787.

Sisk, T.D. & Battin, J. (2002). Habitat edges and avian habitat: geographic patterns and insights for western landscapes. *Stud. Avian Biol.* **25**, 30–48.

Skinner, C.N. & Chang, C., 1996. Fire regimes, past and present. Sierra Nevada Ecosystem Project: Final report to Congress, vol. II, Assessments and scientific basis for management options. UC Davis, Centers for Water and Wildland Resources Center Report, 38, 1041–1069.

Tarbill, G.L., 2010. Nest site selection and influence of woodpeckers in a burned forest of the Sierra Nevada. M.S. Sacramento State University. Sacramento, CA.

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General Technical
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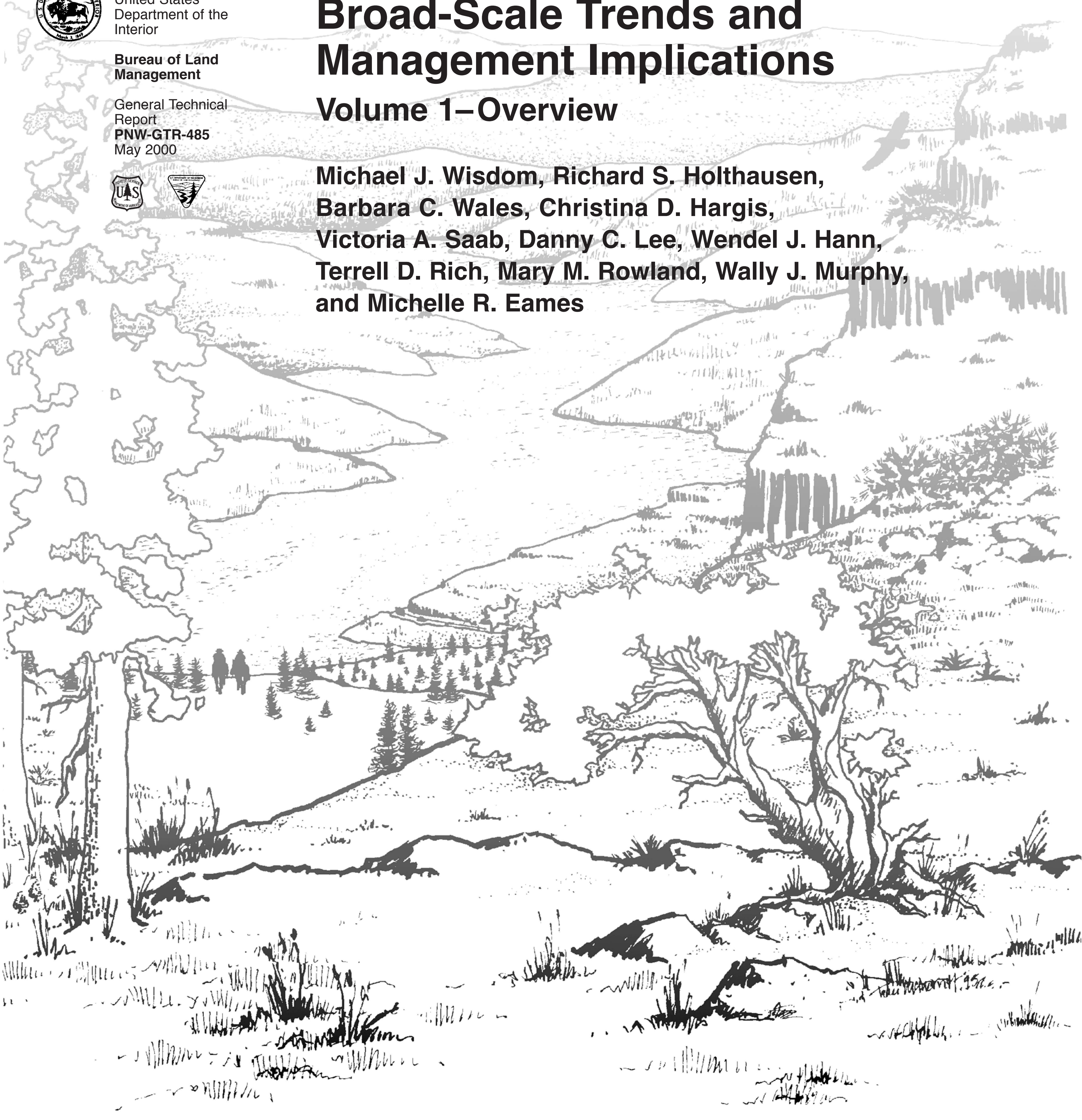
Source Habitats for Terrestrial Vertebrates of Focus in the Interior Columbia Basin: Broad-Scale Trends and Management Implications

Volume 1–Overview

**Michael J. Wisdom, Richard S. Holthausen,
Barbara C. Wales, Christina D. Hargis,
Victoria A. Saab, Danny C. Lee, Wendel J. Hann,
Terrell D. Rich, Mary M. Rowland, Wally J. Murphy,
and Michelle R. Eames**



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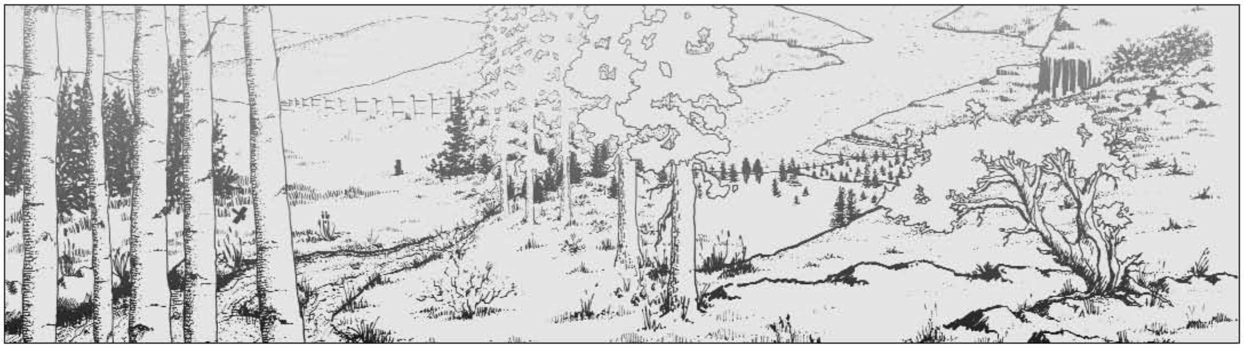

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Interior Columbia Basin Ecosystem Management Project: Scientific Assessment

Thomas M. Quigley, Editor

Volume 1 contains pages 1 through 156.

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Abstract

Wisdom, Michael J.; Holthausen, Richard S.; Wales, Barbara C.; Hargis, Christina D.; Saab, Victoria A.; Lee, Danny C.; Hann, Wendel J.; Rich, Terrell D.; Rowland, Mary M.; Murphy, Wally J.; Eames, Michelle R. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia basin: broad-scale trends and management implications. Volume 1—Overview. Gen. Tech. Rep. PNW-GTR-485. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 3 vol. (Quigley, Thomas M., tech. ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

We defined habitat requirements (source habitats) and assessed trends in these habitats for 91 species of terrestrial vertebrates on 58 million ha (145 million acres) of public and private lands within the interior Columbia basin (hereafter referred to as the basin). We also summarized knowledge about species-road relations for each species and mapped source habitats in relation to road densities for four species of terrestrial carnivores. Our assessment was conducted as part of the Interior Columbia Basin Ecosystem Management Project (ICBEMP), a multiresource, multidisciplinary effort by the USDA Forest Service (FS) and the USDI Bureau of Land Management (BLM) to develop an ecosystem-based strategy for managing FS and BLM lands within the basin. Our assessment was designed to provide technical support for the ICBEMP and was done in five steps. First, we identified species of terrestrial vertebrates for which there was ongoing concern about population or habitat status (species of focus), and for which habitats could be estimated reliably by using a large mapping unit (pixel size) of 100 ha (247 acres) and broad-scale methods of spatial analysis. Second, we evaluated change in source habitats from early European settlement (historical, circa 1850 to 1890) to current (circa 1985 to 1995) conditions for each species and for hierarchically nested groups of species and families of groups at the spatial scales of the watershed (5th hydrologic unit code [HUC]), subbasin (4th HUC), ecological reporting unit, and basin. Third, we summarized the effects of roads and road-associated factors on populations and habitats for each of the 91 species and described the results in relation to broad-scale patterns of road density. Fourth, we mapped classes of the current abundance of source habitats for four species of terrestrial carnivores in relation to classes of road density across the 164 subbasins and used the maps to identify areas having high potential to support persistent populations. And fifth, we used our results, along with results from other studies, to describe broad-scale implications for managing habitats deemed to have undergone long-term decline and for managing species negatively affected by roads or road-associated factors.

Our results indicated that habitats for species, groups, and families associated with old-forest structural stages, with native grasslands, or with native shrublands have undergone strong, widespread decline. Implications of these results for managing old-forest structural stages include consideration of (1) conservation of habitats in subbasins and watersheds where decline in old forests has been strongest; (2) silvicultural manipulations of mid-seral forests to accelerate development of late-seral stages; and (3) long-term silvicultural manipulations and long-term accommodation of fire and other disturbance regimes in all forested structural stages to hasten development and improvement in the amount, quality, and distribution of old-forest stages. Implications of our results for managing rangelands include the potential to (1) conserve native grasslands and shrublands that have not undergone large-scale reduction in composition of native plants; (2) control or eradicate exotic plants on native grasslands and shrublands where invasion potential or spread of exotics is highest; and (3) restore native plant communities by using intensive range practices where potential for restoration is highest.

Our analysis also indicated that >70 percent of the 91 species are affected negatively by one or more factors associated with roads. Moreover, maps of the abundance of source habitats in relation to classes of road density suggested that road-associated factors hypothetically may reduce the potential to support persistent populations of terrestrial carnivores in many subbasins. Management implications of our summarized road effects include the potential to mitigate a diverse set of negative factors associated with roads. Comprehensive mitigation of road-associated factors would require a substantial reduction in the density of existing roads as well as effective control of road access in relation to management of livestock, timber, recreation, hunting, trapping, mineral development, and other human activities.

A major assumption of our work was that validation research will be conducted by agency scientists and other researchers to corroborate our findings. As a preliminary step in the process of validation, we found high agreement between trends in source habitats and prior trends in habitat outcomes that were estimated as part of the habitat outcome analysis for terrestrial species within the basin. Results of our assessment also were assumed to lead to finer scale evaluations of habitats for some species, groups, or families as part of implementation procedures. Implementation procedures are necessary to relate our findings to local conditions; this would enable managers to effectively apply local conservation and restoration practices to support broad-scale conservation and restoration strategies that may evolve from our findings.

Keywords: Cluster analysis, conservation, forest management, habitat, habitat condition, habitat management, habitat trend, interior Columbia basin, Interior Columbia Basin Ecosystem Management Project, landscape ecology, landscape analysis, population viability, rangeland management, terrestrial vertebrates, spatial analysis, species of focus, sink, sink environment, source, source environment, source habitat, source habitats, restoration, species groups, monitoring, validation research, viability, wildlife, wildlife-habitat relations.

Foreword

This publication consists of three volumes so that our findings—which consist of hundreds of tables, figures, pages of text, and supporting citations—could be presented in a manner most usable to resource managers, biologists, and the public. Volume 1 is designed as an overview of objectives, methods, key results, and management implications. Volumes 2 and 3 contain increasingly detailed results that support and complement results in volume 1. We believe that resource managers may find sufficient detail in the generalized results and implications presented in volume 1, but that management biologists and other users of the results and supporting data will want to refer to all three volumes. Results, management implications, and supporting citations provided in volume 2 are especially important to consider as part of step-down implementation procedures and related management conducted by field units within the interior Columbia basin. By contrast, information in volume 1 may be particularly useful in serving broad-scale planning issues, objectives, and strategies for the interior Columbia basin as a whole. Regardless of application, all three volumes are intended to function together as a comprehensive assessment of habitat trends and a summary of other environmental factors affecting terrestrial vertebrates whose population or habitat status is of ongoing concern to resource managers. Data underlying most tables presented in the three volumes also are available at the web site for the ICBEMP: http://www.icbemp.gov/spatial/metadata/databases.

Preface

The Interior Columbia Basin Ecosystem Management Project was initiated by the Forest Service and the Bureau of Land Management to respond to several critical issues including, but not limited to, forest and rangeland health, anadromous fish concerns, terrestrial species viability concerns, and the recent decline in traditional commodity flows. The charter given to the project was to develop a scientifically sound, ecosystem-based strategy for managing the lands of the interior Columbia River basin administered by the Forest Service and the Bureau of Land Management. The Science Integration Team was organized to develop a framework for ecosystem management, an assessment of the socioeconomic and biophysical systems in the basin, and an evaluation of alternative management strategies. This paper is one in a series of papers developed as background material for the framework, assessment, or evaluation of alternatives. It provides more detail than was possible to disclose directly in the primary documents.

The Science Integration Team, although organized functionally, worked hard at integrating the approaches, analyses, and conclusions. It is the collective effort of team members that provides depth and understanding to the work of the project. The Science Integration Team leadership included deputy team leaders Russell Graham and Sylvia Arbelbide; landscape ecology—Wendel Hann, Paul Hessburg, and Mark Jensen; aquatic—Jim Sedell, Kris Lee, Danny Lee, Jack Williams, and Lynn Decker; economic—Richard Haynes, Amy Horne, and Nick Reyna; social science—Jim Burchfield, Steve McCool, Jon Bumstead, and Stewart Allen; terrestrial—Bruce Marcot, Kurt Nelson, John Lehmkuhl, Richard Holthausen, Randy Hickenbottom, Marty Raphael, and Michael Wisdom; spatial analysis—Becky Gravenmier, John Steffenson, and Andy Wilson.

Thomas M. Quigley
Editor

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Interior Columbia
Basin Ecosystem
Management Project

Executive Summary

Introduction

Habitat for terrestrial wildlife is declining rapidly worldwide. Declines are due to various human causes; increasing urbanization, conversion of lands to agriculture, and intensive management of forests, rangelands, and other biomes to meet human demands for food, shelter, and leisure are key examples. In the United States, declines in habitat during the past century are largely responsible for the dramatic increase in the number of species listed as candidate, threatened, or endangered under the Endangered Species Act (ESA). This rate of habitat loss has led to an accelerated rate of species listings.

In response to such problems, managers of Federal lands are moving increasingly toward broad-scale, ecosystem-based strategies for conserving and restoring habitats. Examples include the Northwest Forest Plan, the Southern Appalachian Assessment, and the Sierra Nevada Assessment. In this paper, we present results of an ecosystem-based analysis of habitat change and a synthesis of road-associated effects on selected terrestrial vertebrates in support of the Interior Columbia Basin Ecosystem Management Project (ICBEMP). The ICBEMP was established in January 1994 through a charter signed by the Chief of the USDA Forest Service (FS) and the Director of the USDI Bureau of Land Management (BLM). The charter directed that work be undertaken to develop and adopt an ecosystem-based strategy for all lands administered by the FS and BLM within the interior Columbia basin (hereafter referred to as the basin). This area extends over 58 million ha[1] (145 million acres) in Washington, Oregon, Idaho, Montana, and small portions of Wyoming, Nevada, California, and Utah. Fifty-three percent of the basin is public land administered by the FS or BLM.

Our purpose for analysis was to (1) develop an understanding of changes in habitats that have occurred across the basin since early European settlement; (2) assess effects of these changes on source habitats for species of terrestrial vertebrates for which there is ongoing concern about population or habitat status (species of focus); (3) summarize effects of roads and associated factors on populations and habitats of these species; (4) display broad-scale patterns of road density as a spatially explicit measure of road effects on terrestrial vertebrates, particularly in relation to four species of terrestrial carnivores; and (5) synthesize results from these evaluations into major patterns, implications of which could be addressed by managers in the form of broad-scale strategies and practices.

Objectives and Methods

Within our purpose framework, we had six objectives that formed the basis for our methods:

1. Identify species of terrestrial vertebrates whose habitats might require further assessment and management at broad spatial scales within the basin; these species are referred to as broad-scale species of focus. Broad-scale species of focus are vertebrate species whose population size is known or suspected to be declining in response to habitat decline or to nonhabitat effects of human activities, and whose habitats can be estimated reliably by using a large mapping unit (pixel size) of 100 ha (247 acres) and broad-scale methods of spatial analysis. Because our results were targeted for use in broad-scale, ecosystem-based management, our process of identifying species was designed to include all species for which there might be even moderate concern. Our process was not designed to highlight just those species critically in need of attention. Use of an inclusive rather than an exclusive list of species assures that all associated habitats in need of restoration are addressed. Moreover, use of an inclusive list facilitates a holistic approach to maintaining animal communities rather than single species.

2. Determine species relations with source habitats. Source habitats are those characteristics of macrovegetation that contribute to stationary or positive population growth for a species in a specified area and time. Source habitats contribute to source environments, which represent the composite of all environmental conditions that results in stationary or positive population growth for a species in a specified area and time. The distinction between

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[1] See "Abbreviations" table p. 137, for definitions of abbreviated units of measure.

source habitats and source environments is important for understanding our evaluation and its limitations. For example, source habitats for a bird species during the breeding season would include those characteristics of vegetation that contribute to successful nesting and rearing of young but would not include nonvegetative factors such as the effects of pesticides on thinning of eggshells, which also affect production of young.

3. Conduct a spatial assessment of source habitats for all broad-scale species of focus, including an assessment of change in source habitats from early European to current conditions. Our spatial assessment was based on the composition and structure of vegetation estimated to exist during early European settlement (historical, circa 1850 to 1890) and current (circa 1985 to 1995) conditions, based on prior ICBEMP landscape assessments. Specifically, we wanted to relate historical and current estimates of vegetation characteristics to source habitats deemed to contribute to sustainable populations of the species of focus, and to assess changes in those habitats from historical to current periods.

4. Develop a system to evaluate source habitats for individual species as well as for groups of species. Our system was designed to nest evaluations of individual species hierarchically within evaluations conducted for groups of species and for multiple groups (families of groups). Our system was developed to enable managers to identify broad-scale, robust patterns of habitat change that affect multiple species in a similar manner, and to allow managers to address the needs of all species efficiently, accurately, and holistically with the use of broad-scale strategies and practices. Moreover, our system was designed to enable managers to address the needs of either single- or multi-species, depending on objectives, and to check how well an evaluation of a group of species or a family of groups represents evaluations conducted for each species within the group or family. Finally, our system was designed to evaluate source habitats at multiple spatial scales and across time, thus providing maximum flexibility in the conduct of spatial and temporal analyses.

5. Identify species whose populations or habitats may be negatively affected by roads and associated factors, summarize the effects, display the broad-scale patterns of road density as an index of these effects, and map areas that contain both abundant source habitats and low road densities for selected species of terrestrial carnivores. Areas containing abundant source habitats may not support persistent populations of some species because of the negative effects of factors associated with roads; that is, source habitats may contribute to positive or stationary population growth, but the road effect may override the habitat effect, thereby resulting in a sink environment. Knowledge about the negative effects of factors associated with roads is therefore an important, complementary component to proper management of vegetation for terrestrial vertebrates.

6. Describe the broad-scale implications for managing terrestrial vertebrates whose source habitats have undergone long-term decline, or for terrestrial vertebrates whose habitats or populations are negatively affected by one or more factors associated with roads. Management implications are broad-scale considerations about the potential to conserve or restore source habitats, or to manage human access and human activities, on FS- and BLM-administered lands in response to habitat decline or to negative effects of human disturbance. Describe these implications from results of our assessment, from the scientific literature, and from results of prior assessments conducted as part of the ICBEMP. Whenever possible, link these implications to specific geographic areas of the basin based on our spatial assessment of source habitats and our mapping of broad-scale patterns of road density.

Source Habitats for Families of Groups

We identified 91 species of birds, mammals, and reptiles (broad-scale species of focus) for analysis, based on criteria that indexed various habitat or population problems regarding the current status of each species. Placement of the 91 species into 40 groups, and the further placement of 37 of the groups into 12 families, by using a combination of cluster analysis and empirical knowledge of the similarities of species in habitat requirements, resulted in distinct differences among families in the number of terrestrial community types and source habitats used. Family 4 had the most restricted number of terrestrial community types and source habitats used by species of any family, with habitats restricted to early-seral forests. Species in family 1 also were restricted to a small number of terrestrial community types, and in this case, the types were composed of low-elevation, late-seral forests. By contrast, species in family 2 used a higher number and variety of terrestrial community types that encompassed all elevations of late-seral forests. Species in family 3 used an even greater variety of forested conditions; habitats encompassed the highest number and type of source habitats within the highest number of terrestrial community types of any family dependent on forested habitats.

Species dependent strictly on rangelands were placed in families 10, 11 and 12. Species in families 11 and 12 were restricted to a relatively small number of terrestrial community types, with family 11 primarily dependent on sagebrush, and family 12 dependent on grassland and open-canopy sagebrush habitats. Species in family 10 used a broader set of terrestrial communities, consisting of various grassland, shrubland, woodland, and related cover types in comparison to families 11 and 12.

Species in families 5, 6, 7, 8, and 9 were associated with various terrestrial community types, but the set of source habitats for each family was distinctly different from the others. Habitats for species in family 9 were restricted to relatively few source habitats within the upland woodland and upland shrubland types. By contrast, species in family 5 used habitats that encompassed nearly all terrestrial community types. Species in family 6 also used various terrestrial communities, with the types composed of forests, woodlands, and montane shrubs. Terrestrial community types used by family 7 were similar to family 6, with the main difference being the use of sagebrush types instead of montane shrubs. Finally, habitats for family 8 spanned a fairly restrictive but unusual combination of terrestrial community types composed of both early- and late-seral forests, as well as woodland, shrubland, and grassland types.

These differences in terrestrial communities and source habitats among the families resulted in distinctly different habitat trends for each family. In the following section, results are summarized in terms of major habitat trends and key implications for management. Also included is a summary of species-road relations, based on a survey of species-roads literature.

Major Findings and Implications

1. Source habitats for most species declined strongly from historical to current periods across large areas of the basin. Strongest declines were for species dependent on low-elevation, old-forest habitats (family 1), for species dependent on combinations of rangelands or early-seral forests with late-seral forests (family 8), and for species dependent on native grassland and open-canopy sagebrush habitats (family 12). Widespread but less severe declines also occurred for most species dependent on old-forest habitats present in various elevation zones (family 2); for species dependent on early-seral forests (family 4); for species dependent on native herbland, shrubland, and woodland habitats (family 10); and for species dependent on native sagebrush habitats (family 11). Source habitats for all of the above-named families have become increasingly fragmented, simplified in structure, and infringed on or dominated by exotic plants.

2. Primary causes for decline in old-forest habitats (families 1 and 2) are intensive timber harvest and large-scale fire exclusion. Additional causes for decline in low-elevation, old-forest habitats are conversion of land to agriculture and to residential or urban development. Intensive timber harvest and large-scale fire exclusion also are primarily responsible for the large decline in early-seral habitats (family 4).

3. Primary causes for decline in native herbland, woodland, grassland, and sagebrush habitats (families 10, 11, and 12) are excessive livestock grazing, invasion of exotic plants, and conversion of land to agriculture and residential and urban development. Altered fire regimes also are responsible for decline in native grassland and shrubland habitats.

4. Various road-associated factors negatively affect habitats or populations of most of the 91 species of focus. Effects of road-associated factors can be direct, such as habitat loss and fragmentation because of road construction and maintenance. Effects also can be indirect, such as displacement or increased mortality of populations in areas near roads in relation to motorized traffic and associated human activities. Because of the high density of roads present across large areas of the basin,

effects from road-associated factors must be considered additive to that of habitat loss. Moreover, many habitats likely are underused by some species because of the effects of roads and associated factors; this may be especially true for species of carnivorous mammals, particularly gray wolf and grizzly bear.[2]

5. Implications of our results for managing old-forest structural stages include the potential to conserve old-forest habitats in subbasins and watersheds where decline has been strongest; manipulate mid-seral forests to accelerate development of late-seral stages when such manipulations can be done without further reduction in early- or late-seral forests; and restore fire and other disturbance regimes in all forested structural stages to hasten development and improvement in the amount, quality, and distribution of old-forest stages. Many of the practices designed to restore old-forest habitats also can be designed to restore early-seral habitats. For example, long-term restoration of more natural fire regimes will hasten development of both early- and late-seral structural conditions, and minimize area of mid-seral habitats, which few if any species depend on as source habitat.

6. Implications of our results for managing rangelands include the potential to conserve native grasslands and shrublands that have not undergone large-scale reduction in composition of native plants; control or eradicate exotic plants on native grasslands and shrublands where invasion potential or spread of exotics is highest; and restore native plant communities, by using intensive range practices, where potential for restoration is highest. Restoration includes the potential to manipulate livestock grazing systems and stocking rates where existing or past grazing practices have contributed to the decline of native grasslands and shrublands.

7. Implications of our summary of road-associated effects include the potential to mitigate a diverse set of negative factors associated with roads. Comprehensive mitigation of road-associated factors would require a substantial reduction in the density of existing roads as well as effective control of road access in relation to management of livestock, timber, recreation, hunting, trapping, mineral development, and other human activities. Efforts to restore habitats without simultaneous efforts to reduce road density and control human disturbances will curtail the effectiveness of habitat restoration, or even contribute to its failure; this is because of the large number of species that are simultaneously affected by decline in habitat as well as by road-associated factors.

8. Implications of all our results, when considered at multiple spatial scales ranging from the basin, ecological reporting unit, subbasin, and watershed, provide spatially explicit opportunities for conservation and restoration of source habitats across various land ownerships and jurisdictions. Moreover, our results provide temporally explicit opportunities for design of long-term efforts to restore source habitats that have undergone strong, widespread decline, with simultaneous design of efforts to conserve these same habitats where they exist currently. Use of our findings to conduct effective spatial and temporal prioritization of restoration and conservation efforts for terrestrial species and habitats represents a major opportunity for resource managers in the interior Columbia basin.

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[2] See table 1 for common and scientific names of the vertebrate broad-scale species of focus, and appendix 3, volume 3, for scientific names of plants and animals not addressed as terrestrial vertebrates of focus.

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Volume 3

Introduction

Habitat for terrestrial wildlife is declining rapidly worldwide. Declines are due to several human causes; increasing urbanization, conversion of lands to agriculture, and intensive management of forests, rangelands, and other biomes to meet human demands for food, shelter, and leisure are key examples (Alverson and others 1994, Noss and others 1995, Western and Pearl 1989). In the United States, declines in habitat during the past century are largely responsible for the dramatic increase in the number of species listed as candidate, threatened, or endangered under the Endangered Species Act (ESA) (Easter-Pilcher 1996; Flather and others 1994, 1998) (See "Glossary," vol. 3, for terms used in this paper). This rate of habitat loss has led to an accelerated rate of species listings. For example, based on the apparent exponential rate at which species have been listed under ESA during the past 11 years (Flather and others 1994, 1998), the number of species in the United States that may warrant listing early in the 21st century may double, or perhaps triple, the number already listed.

In response to such projections, managers of Federal lands are moving increasingly toward broad-scale, ecosystem-based strategies for conserving and restoring habitats. Examples include the Northwest Forest Plan (USDA Forest Service and USDI Bureau of Land Management 1994), the Southern Appalachian Assessment (SAMAB 1996), and the Sierra Nevada Assessment (Anonymous 1996). Such ecosystem-based strategies are needed to sustain habitats for all species in a holistic manner by using broad-scale methods intended to prevent further listings under ESA. Such strategies also are designed to comply with additional laws regarding maintenance of viable populations, such as the National Forest Management Act (NFMA).

In this paper, we present results of an ecosystem-based analysis of habitat change and a synthesis of road-associated effects on selected terrestrial vertebrates in support of the Interior Columbia Basin Ecosystem Management Project (ICBEMP). The ICBEMP was established in January 1994 through a charter signed by the Chief of the USDA Forest Service (FS) and the Director of the USDI Bureau of Land Management (BLM) (USDA Forest Service 1996). The charter directed that work be undertaken to develop and adopt an ecosystem-based strategy for managing all lands administered by the FS and BLM within the interior Columbia basin (hereafter referred to as the basin). This area extends over 58 million ha (145 million acres) in Washington, Oregon, Idaho, Montana, and small portions of Nevada, California, Utah, and Wyoming (fig. 1A). Fifty-three percent of the basin is public land administered by the FS or BLM.

The work of the ICBEMP has resulted in new understanding of both the biological and social systems in the basin (Quigley and others 1996, USDA Forest Service 1996). Of most significance to terrestrial vertebrates are the changes in terrestrial habitats and disturbance processes that have occurred since the time of early European settlement. Chief among these changes are dramatic shifts in fire regimes, reductions in area of native grasslands and shrublands, declines in the early and late stages of forest development, reduction in wetland area, deterioration of riparian habitat conditions, and increases in road density (Hann and others 1997, Quigley and others 1996, USDA Forest Service 1996). These changes have reduced habitat for many species within the basin (Lehmkuhl and others 1997, Noss and others 1995), and populations of several vertebrates have declined (Saab and Rich 1997).

This knowledge of biological and social systems has been used to craft ecosystem-based management strategies, and the basis for those strategies has been reported in scientific publications (for example, Hann and others 1997, Hessburg and others 1999; Lee and others 1997, Lehmkuhl and others 1997), as well as in draft environmental impact statements (DEIS) (USDA Forest Service and USDI Bureau of Land Management 1997a, 1997b). These documents detail how current management of Federal lands not only seems inadequate to maintain sufficient habitat for many terrestrial vertebrates, but how the continuation of such management is projected to result in further deterioration of habitats (Lehmkuhl and others 1997). New strategies that are likely to be more favorable to terrestrial vertebrates are those that provide for active restoration of habitats and ecosystem processes. These new strategies are projected to result in maintenance or improvement of habitat for many species and continued deterioration of habitat for fewer species (Lehmkuhl and others 1997).

Although strategies that include an active restoration component hold promise for reversing the deterioration of habitat conditions within the basin, there are many unknowns concerning implementation of those

Figure 1—Assessment boundaries of, and land ownership within, the Interior Columbia Basin Ecosystem Management Project (A), and the 13 ecological reporting units (B).

B

VIEW

MAP

Figure 1—Assessment boundaries of, and land ownership within, the Interior Columbia Basin Ecosystem Management Project (A), and the 13 ecological reporting units (B).

strategies. Restoration practices are well understood for some environments but not adequately studied or understood for other habitats (Hann and others 1997). In addition, spatial priorities for implementation of these practices, from the standpoint of terrestrial vertebrates, have received little attention as part of the ICBEMP thus far. The information needed to establish such priorities is spatially explicit knowledge of change in habitat conditions throughout the basin and of resulting effects on vertebrate species. This information, linked with an understanding of the processes that have caused changes and effects on other components of the ecosystem, would facilitate the development of spatially explicit management strategies that span a full range of ecological and social concerns. That was the motivation for our analysis of habitat change and synthesis of road-associated effects on selected terrestrial vertebrates presented here.

Objectives

The purpose for an analysis was to (1) develop an understanding of changes in habitats that have occurred across the basin since early European settlement; (2) assess effects of these changes on source habitats for species of terrestrial vertebrates for which there is ongoing concern about population or habitat status (species of focus); (3) summarize effects of roads and associated factors on populations and habitats of these species; (4) display broad-scale patterns of road density as a spatially explicit measure of road effects on terrestrial vertebrates, particularly in relation to four species of terrestrial carnivores; and (5) synthesize results from these evaluations into major patterns, implications of which could be addressed by managers in the form of broad-scale strategies and practices. Within this framework, we had six specific objectives:

1. Identify species of terrestrial vertebrates whose habitats might require further assessment and management at broad spatial scales within the basin; these species are referred to as broad-scale species of focus. Broad-scale species of focus are vertebrate species whose population size is known or suspected to be declining in response to habitat decline or to nonhabitat effects of human activities, and whose habitats can be estimated reliably by using a large mapping unit (pixel size) of 100 ha (247 acres) and broad-scale methods of spatial analysis. Because our results were targeted for use in broad-scale, ecosystem-based management, our process of identifying species was designed to include all species for which there might be even moderate concern. Our process was not designed to highlight just those species critically in need of attention. Use of an inclusive rather than an exclusive list of species assures that all associated habitats in need of restoration are addressed. Moreover, use of an inclusive list facilitates a holistic approach to maintenance of animal communities rather than single species.

2. Determine species relations with source habitats. Source habitats are those characteristics of macrovegetation that contribute to stationary or positive population growth for a species in a specified area and time. Source habitats contribute to source environments (Pulliam 1988, Pulliam and Danielson 1991), which represent the composite of all environmental conditions that results in stationary or positive population growth for a species in a specified area and time. The distinction between source habitats and source environments is important for understanding our evaluation and its limitations. For example, source habitats for a bird species during the breeding season would include those characteristics of vegetation that contribute to successful nesting and rearing of young, but would not include nonvegetative factors, such as the effects of pesticides on thinning of eggshells, which also affect production of young.

Consideration of both vegetative and nonvegetative factors that contribute to population persistence requires an evaluation of source environments, which is beyond the purpose and scope of our evaluation. As part of the process of identifying and evaluating vegetation characteristics that contribute to stationary or positive population growth, however, we defined and identified source habitats as being distinctly different from habitats that are simply associated with species occurrence, which may or may not contribute to viable, long-term population persistence. That is, in contrast to source habitats, those habitats in which species occur can contribute to either source or sink environments (Pulliam and Danielson 1991). Consequently, species occurrence by itself indicates little or nothing about the capability of the associated environment to support long-term persistence of populations (Conroy and Noon 1996, Conroy and others 1995).

Consequently, data based strictly on species occurrence did not meet our objective to identify those characteristics of vegetation that contribute to long-term population persistence, which we defined as source habitats.

3. Conduct a spatial assessment of source habitats for all broad-scale species of focus, including an assessment of change in source habitats from early European to current conditions. Our spatial assessment was based on the composition and structure of vegetation estimated to exist during early European settlement (historical, circa 1850 to 1890) and current (circa 1985 to 1995) conditions, based on work by Hann and others (1997) and methods of Keane and others (1996). Specifically, we wanted to relate historical and current estimates of vegetation characteristics to source habitats deemed to contribute to sustainable populations of the species of focus, and to assess changes in those habitats from historical to current periods.

4. Develop a system to evaluate source habitats for individual species as well as for groups of species. Our system was designed to nest evaluations of individual species hierarchically within evaluations conducted for groups of species and for multiple groups (families of groups). Our system specifically was developed to enable managers to identify broad-scale, robust patterns of habitat change that affect multiple species in a similar manner, and to allow managers to address the needs of all species efficiently, accurately, and holistically with the use of broad-scale strategies and practices. Moreover, our system was designed to enable managers to address the needs of either single or multiple species, depending on objectives, and to allow managers to check how well an evaluation of a group of species or a family of groups represents evaluations conducted for each species within the group or family. Finally, our system was designed to evaluate source habitats at multiple spatial scales and across time, thus providing maximum flexibility in the conduct of spatial and temporal analyses.

Use of hiearchically nested single- and multi-species evaluations, conducted at multiple spatial scales, is considered a requirement for managers who need information at different levels of resolution and complexity. Our use of both single- and multi-species evaluations was designed to provide maximum flexibility in how managers address different issues of habitat management. Our rationale for using both single- and multi-species evaluations, each nested hiearchically within one another, was that each habitat issue requires a different level of detail and knowledge for effective management.

5. Identify species whose populations or habitats may be negatively affected by roads and associated factors, summarize the effects, display the broad-scale patterns of road density as an index of these effects, and map areas that contain both abundant source habitats and low road densities for selected species of terrestrial carnivores. It is possible that areas containing abundant source habitats may not support persistent populations of some species because of the negative effects of factors associated with roads; that is, source habitats may contribute to positive or stationary population growth, but the road effect may override the habitat effect, thereby resulting in a sink environment. Knowledge about the negative effects of factors associated with roads is therefore an important, complementary component to proper management of vegetation for terrestrial vertebrates.

6. Describe the broad-scale implications for managing terrestrial vertebrates whose source habitats have undergone long-term decline, or for terrestrial vertebrates whose habitats or populations are negatively affected by one or more factors associated with roads. Management implications are broad-scale considerations about the potential to conserve or restore source habitats, or to manage human access and human activities, on FS- and BLM-administered lands in response to habitat decline or to negative effects of human disturbance. Describe these implications from results of our assessment, from the scientific literature, and from results of prior assessments conducted as part of the ICBEMP. Whenever possible, link these implications to specific geographic areas of the basin based on our spatial assessment of source habitats and our mapping of broad-scale patterns of road density.

As part of management implications, we listed broad-scale strategies and practices that may be useful to managers seeking to conserve and restore habitats that have undergone long-term decline. This list should be considered a menu of possible approaches that managers could adopt to help achieve their objectives for conservation and restoration of habitats. Before any of these approaches are adopted, they should be analyzed to determine their effectiveness, their compatibility with overall ecosystem management objectives, and their applicability to local situations. If any of these strategies are used, it is particularly important that testing and validation continue at all geographic scales of their implementation. We assumed that each local situation will be analyzed to determine if the strategies identified as part of our assessment will have the intended effects and be compatible with other land management objectives and activities.

Following this logic, our objectives did not call for an assessment of potential strategies in terms of their effect on the habitat outcomes of Lehmkuhl and others (1997) because it is expected that managers will adopt unique sets of strategies in response to various legal, social, and economic considerations that are beyond the scope of this paper. Spatially explicit strategies, developed by managers of BLM- and FS-administered lands, could be assessed at a later date in terms of their adequacy to comply with laws such as ESA and NFMA.

Related Assessments

Our assessment was designed to complement results from previous scientific assessments conducted for the ICBEMP, particularly the work by Quigley and others (1996), Hann and others (1997), Lee and others (1997), Lehmkuhl and others (1997), Marcot and others (1997), and Hessburg and others (1999). Hann and others (1997) characterized landscape conditions within the basin, historically (mid to late 19th century) and currently (late 20th century), in terms of vegetation, succession, and disturbance regimes using a large mapping unit (pixel size) of 100 ha (247 acres), broad-scale methods of spatial analysis, and complete sampling coverage. Hessburg and others (1999) also characterized landscape conditions within the basin, but did so at a finer scale (mapping unit of 4 ha [10 acres]), over a different time period (early or mid 1900s to late 1900s), and using samples taken from <5 percent of the land base. Lee and others (1997) characterized aquatic conditions within the basin, particularly in terms of cold-water fisheries. Marcot and others (1997) catalogued the terrestrial plant and animal taxa occurring within the basin, particularly in terms of the number of species, their distributions, their ecological functions and roles, and their environmental correlates. Marcot and others (1997) also mapped several broad-scale spatial patterns related to biological diversity, such as hotspots and centers of endemism. Lehmkuhl and others (1997) assessed habitat outcome of selected terrestrial plant and animal species, historically, currently, and under each of the alternatives proposed in the DEIS (USDA Forest Service and USDI Bureau of Land Management 1997a, 1997b). Finally, Quigley and others (1996) integrated results from the above assessments in a spatially explicit manner at the scale of the subbasin. Integration focused primarily on combining estimates of ecological integrity from landscape, aquatic, and socioeconomic resources, and mapping the combined results across subbasins in the form of six classes of forest and six classes of rangeland clusters, with each class depicting a different level of ecological condition (Quigley and others 1996). Concise summaries of these prior science assessments for the ICBEMP are described by Hann and others (1998), Haynes and others (1998), Lee and others (1998), and Raphael and others (1998). Noss and others (1995) also described habitat trends for the basin and other areas of the United States.

In contrast to these prior assessments, our assessment was intended to be a broad-scale analysis of macrohabitat conditions across the basin for a targeted set of terrestrial vertebrates. Results of our assessment were intended to be integrated with information on landscape conditions, aquatic resources, and socioeconomic patterns to refine our composite knowledge of ecological risk and opportunity throughout the basin. Results of our assessment also were assumed to lead to finer scale evaluations of habitats for some groups or species as part of implementation procedures. Implementation procedures were necessary to relate our findings to local conditions as part of the management application process.

Study Area

Our assessment covered the basin east of the crest of the Cascade Range and those portions of the Klamath and Great Basins within Oregon (fig. 1A). The 58-million-ha (145-million-acre) basin (fig. 1A) is stratified into four spatial scales (Gravenmier and others 1997): (1) ecological reporting unit (ERU), (2) subbasin, (3) watershed, and (4) subwatershed. Ecological reporting units, of which there are 13 (fig. 1B), range in size from about 740 000 to 6 800 000 ha (1,829,000 to 16,800,000 acres; mean size of about 2 375 000 ha [5,866,250 acres]). The 164 subbasins, or 4th hydrologic unit code (HUC), average about 345 000 ha (850,000 acres), whereas the 2,562 watersheds, or 5th HUCs, average about 22 500 ha (56,000 acres) each. The 7,654 subwatersheds (6th HUCs) average about 7700 ha (19,000 acres). Quigley and others (1996) described these spatial scales and the diverse ecological components of the basin in detail. Marcot and others (1997) further described flora and fauna occurring within the basin.

Methods

Several large-scale, ecosystem-based assessments have been completed recently (Anonymous 1996, USDA Forest Service and USDI Bureau of Land Management 1994, SAMAB 1996), yet few standard methods exist for evaluating terrestrial species and their habitats at a broad scale. Moreover, even fewer methods exist for developing an analysis framework in which broad-based management strategies can be established for holistic management of a large complex of terrestrial vertebrates. Efforts have been made to develop broad-scale methods to identify areas having little management protection and high species richness, such as gap analysis (Kiester and others 1996, Scott and others 1993). Less attention has been devoted, however, to the problems of identifying historical changes in habitats and to the challenges of developing spatially explicit themes to correct problems caused by long-term, negative changes in those habitats. Consequently, our methods were designed to meet unique objectives. Previous, broad-scale methods of habitat assessment, such as those used by Kiester and others (1996), Marcot and others (1997), and Scott and others (1993), relied on estimates of species occurrence in relation to current habitat conditions. Our methods build on these but were also designed to meet objectives that called for identifying only those habitats that presumably contribute to stationary or positive population growth (source habitats), and that required measurement of temporal change in such habitats from historical to current conditions. Consequently, our broad-scale methods differ from broad-scale approaches adopted elsewhere.

Given this background, the major steps of our analysis were (1) identifying species on which to focus the analysis; (2) delineating species ranges; (3) determining the relation of species with source habitats; (4) designing a hierarchical system of single- and multi-species assessment; (5) clustering the species into groups, based on similarities in source habitats; (6) assessing change in source habitats from historical to current conditions for species and groups; (7) forming families of groups to summarize results among multiple groups; (8) correlating change in source habitats among species within groups and families to verify how well group and family trends reflected trends of individual species; (9) summarizing knowledge about species-road relations; (10) mapping road density in relation to abundance of source habitats for selected species; (11) interpreting results and identifying broad-scale management implications for those species, groups, and families whose source habitats have undergone long-term decline, or for those species whose populations or habitats are negatively affected by factors associated with roads; and (12) validating agreement between change in source habitats and trends in viability that were projected by Lehmkuhl and others (1997). Following are the specific methods used for each step.

Identifying Species of Focus

We used seven criteria to develop an initial list of species that were the focus of our assessment. Most of these criteria were based on results of the assessment of species-habitat conditions under planning alternatives (Lehmkuhl and others 1997) that were developed for the basin as part of the DEIS (USDA Forest Service and USDI Bureau of Land Management 1997a, 1997b). The process used by Lehmkuhl and others (1997) defined five classes of habitat outcome that were possible for each species (fig. 2). The five outcome classes were defined as follows: outcome 1—habitat broadly distributed with opportunity for nearly continuous distribution of the species; outcome 2—habitat broadly distributed but with gaps; patches large or close enough to permit dispersal; outcome 3—habitat primarily in patches, some of which are small or isolated, causing limitations in species dispersal; outcome 4—habitat in isolated patches with strong limitations on dispersal; some likelihood of local extirpation; and outcome 5—habitat scarce with little or no opportunity for dispersal among patches and strong likelihood of extirpation.

Expert panels were used to assess the likelihood that these conditions existed for each species historically, currently, and under the future scenarios projected for

Figure 2—Conceptual diagram of the five habitat outcome classes developed by Lehmkuhl and others (1997) to assess effects of planning alternatives on selected plants and animals within the interior Columbia basin. Classes were defined as follows: outcome 1—habitat broadly distributed with opportunity for nearly continuous distribution of the species; outcome 2—habitat broadly distributed but with gaps; patches large or close enough to permit dispersal; outcome 3—habitat primarily in patches, some of which are small or isolated, causing limitations in species dispersal; outcome 4—habitat in isolated patches with strong limitations on dispersal; some likelihood of local extirpation; and outcome 5—habitat scarce with little or no opportunity for dispersal among patches and strong likelihood of extirpation. Results of the habitat outcome-based analysis by Lehmkuhl and others (1997) were used as part of the criteria by which to select vertebrate species (broad-scale species focus) for analysis of source habitats.

each planning alternative. Results were expressed as both a distribution of 100 likelihood points across the five outcome classes (fig. 2) and as a weighted mean outcome of these likelihood points. Lehmkuhl and others (1997) presented results of this analysis and provided further details about the methods described above.

For our analysis of source habitats, species were included in an initial list if they met any of the following criteria:

1. Species for which there is at least moderate likelihood of population isolation resulting from habitat conditions. These were identified from the assessment of Lehmkuhl and others (1997) as species with <90 total likelihood points in the combined categories of habitat outcomes 1, 2, and 3, either for historical conditions, for current conditions, or for any DEIS planning alternative.

2. Species for which a significant increase or decrease in habitat outcome was projected from current to future conditions under any environmental impact statement (EIS) alternative. These were identified from the assessment of Lehmkuhl and others (1997) as species whose weighted mean habitat outcome changed by a value of 0.5 or more.

3. Species for which Lehmkuhl and others (1997) adjusted results of habitat outcomes from those assigned by the expert panels. This included 25 species for which Lehmkuhl and others (1997) judged that the expert panel findings are inconsistent with projected habitat trends or with the standards and guidelines of the planning alternatives.

4. Species for which Lehmkuhl and others (1997) did not complete an analysis because of their restricted distribution within the basin. These species were recommended for “fine-scale” analysis.

5. Species that were the subject of the petition filed by the Natural Resources Defense Council with the Regional Forester of the Pacific Northwest Region, USDA Forest Service, on March 30, 1993. Other species that were the subject of repeated appeals to either the FS or the BLM within the jurisdictions of the basin also were included.

6. Species for which The Nature Conservancy assigned a Global ranking of 1 or 2.

7. Species added by the expert panel process that was conducted for terrestrial habitat assessment during September 1997. Some of the species added during this process were not evaluated by Lehmkuhl and others (1997).

We reviewed the initial species list developed from this set of criteria to ensure that it included species associated with all source habitats that were declining, or thought to be declining. We also reviewed the initial list to ensure that it included species whose source habitats were not only declining, but whose population or habitat status was identified as requiring coordination across administrative units of the FS and BLM. The list was reviewed again by panels of species experts to ensure that it included all species of potential concern within the basin as part of criterion 7 described above.

Application of these seven criteria resulted in a final list of 91 species whose source habitats could be mapped reliably by using a pixel size of 100 ha (247 acres), as determined by expert panels (table 1). These species, referred to as broad-scale species of focus, composed our broad-scale analysis. Additional species (>80), most of which were deemed to be dependent on riparian or water habitats, also met the seven criteria (table 1); source habitats for these species, however, were identified by experts as needing mapping units smaller than 100 ha (247 acres) to reliably estimate their habitat abundance.

Again, it is important to note that our species list (table 1) was intended to be inclusive rather than exclusive and to help focus our analysis on ecosystem conditions. It should not be interpreted as a list of species representing some critical legal or biological threshold.

Delineating Species Ranges

We used range maps developed by Marcot and others (in prep.) to estimate the inclusive geographic area that was occupied historically and currently by each species of focus. Range maps were drawn by using the following criteria:

- For broadly distributed species, range maps were drawn to simply reflect the outer extent of the occurrence of the species. Consequently, these maps include large areas of both used and unused habitats.

- For common species with disjunct populations, range maps were drawn to reflect the outer extent of each individual population.

- For locally endemic species or species with small, scattered populations, range maps were drawn to reflect known and potential areas occupied by the species.

- For species whose range is known to have shifted significantly from historical conditions (as defined by Marcot and others, in prep.), separate maps were developed for current and historical range. For all other species, maps that delineate the current range by definition also denote the historical range.

- Maps of each species range were drawn only for areas within the boundaries of the basin because our evaluation was restricted to the basin. When interpreting results of our analyses, however, in combination with population and habitat data available from other studies, we typically considered the entire range of a species if it potentially affected our interpretations.

Information used to develop range maps included previously published maps and published and unpublished location data (Marcot and others, in prep.). Maps were drawn with the help of species experts and subsequently reviewed by these experts to ensure that the final map of the range of each species adhered to the above criteria.

Table 1—Common and scientific names of 173 terrestrial vertebrate species of focus brought forward for additional analysis and the associated criteria by which each species was selected

				Criteria								
Class[a]	Common name	Scientific name	Scale[b]	<90H[c]	<90C[d]	<90A[e]	EIS SIG[f]	Fine[g]	NRDC[h]	G1G2[i]	Adjust[j]	Add[k]
R	Mojave black-collared lizard	*Crotaphytus bicinctores*	BS			X						
R	Longnose leopard lizard	*Gambelia wislizenii*	BS	X	X	X						
R	Sharptail snake	*Contia tenuis*	BS	X	X	X	X				X	
R	Striped whipsnake	*Masticophis taeniatus*	BS	X	X	X						
R	California mountain kingsnake	*Lampropeltis zonata*	BS					X				
R	Longnose snake	*Rhinocheilus lecontei*	BS					X				
R	Ground snake	*Sonora semiannulata*	BS					X				
B	Northern goshawk	*Accipiter gentilis*	BS		X	X			X			
B	Ferruginous hawk	*Buteo regalis*	BS		X	X	X					
B	Blue grouse	*Dendrogapus obscurus*	BS									X
B	Sage grouse	*Centrocercus urophasianus*	BS		X	X	X					
B	Columbian sharp-tailed grouse	*Tympanuchus phasianellus columbianus*	BS		X	X	X					
B	Mountain quail	*Oreortyx pictus*	BS	X	X	X						
B	Flammulated owl	*Otus flammeolus*	BS		X	X	X		X			
B	Burrowing owl	*Athene cunicularia*	BS		X	X						
B	Great gray owl	*Strix nebulosa*	BS	X	X	X	X		X			
B	Long-eared owl	*Asio otus*	BS	X	X	X						
B	Short-eared owl	*Asio flammeus*	BS									X
B	Boreal owl	*Aegolius funereus*	BS	X	X	X	X					
B	Vaux's swift	*Chaetura vauxi*	BS	X	X	X	X		X		X	
B	Rufous hummingbird	*Selasphorus rufus*	BS			X	X					X
B	Black-chinned hummingbird	*Archilochus alexandri*	BS			X						
B	Broad-tailed hummingbird	*Selasphorus platycercus*	BS		X	X						
B	Lewis' woodpecker	*Melanerpes lewis*	BS		X	X	X					
B	Williamson's sapsucker	*Sphyrapicus thyroideus*	BS		X	X	X					
B	White-headed woodpecker	*Picoides albolarvatus*	BS		X	X	X		X		X	
B	Three-toed woodpecker	*Picoides tridactylus*	BS			X	X		X			
B	Black-backed woodpecker	*Picoides arcticus*	BS		X	X	X		X			
B	Pileated woodpecker	*Dryocopus pileatus*	BS	X	X	X	X		X		X	
B	Olive-sided flycatcher	*Contopus cooperi*	BS		X	X						
B	Hammond's flycatcher	*Empidonax hammondii*	BS		X	X	X					

Table 1—Common and scientific names of 173 terrestrial vertebrate species of focus brought forward for additional analysis and the associated criteria by which each species was selected (continued)

Class[a]	Common name	Scientific name	Scale[b]	Criteria								
				<90H[c]	<90C[d]	<90A[e]	EIS SIG[f]	Fine[g]	NRDC[h]	G1G2[i]	Adjust[j]	Add[k]
B	Ash-throated flycatcher	*Myiarchus cinerascens*	BS			X	X					
B	Chestnut-backed chickadee	*Parus rufescens*	BS	X	X	X	X					
B	Bushtit	*Psaltriparus minimus*	BS	X		X	X					
B	White-breasted nuthatch	*Sitta carolinensis*	BS			X	X		X			
B	Pygmy nuthatch	*Sitta pygmaea*	BS			X	X		X			
B	Brown creeper	*Certhia americana*	BS									X
B	Winter wren	*Troglodytes troglodytes*	BS		X	X	X					
B	Golden-crowned kinglet	*Regulus satrapa*	BS									X
B	Western bluebird	*Sialia mexicana*	BS			X	X					
B	Varied thrush	*Ixoreus naevius*	BS									X
B	Sage thrasher	*Oreoscoptes montanus*	BS			X	X					
B	Loggerhead shrike	*Lanius ludovicianus*	BS			X	X					
B	Hermit warbler	*Dendroica occidentalis*	BS					X				
B	Lazuli bunting	*Passerina amoena*	BS/FS			X						
B	Clay-colored sparrow	*Spizella pallida*	BS					X				
B	Brewer's sparrow	*Spizella breweri*	BS			X	X					
B	Vesper sparrow	*Pooecetes gramineus*	BS				X					
B	Lark sparrow	*Chondestes grammacus*	BS			X	X					
B	Black-throated sparrow	*Amphispiza bilineata*	BS					X				
B	Sage sparrow	*Amphispiza belli*	BS				X					
B	Lark bunting	*Calamospiza melanocorys*	BS									X
B	Grasshopper sparrow	*Ammodramus savannarum*	BS	X	X	X						
B	Western meadowlark	*Sturnella neglecta*	BS									X
B	Brown-headed cowbird	*Molothrus ater*	BS									X
B	Black rosy finch	*Leucosticte atrata*	BS	X	X	X						
B	Gray-crowned rosy finch	*Leucosticte tephrocotis*	BS	X	X	X						
B	White-winged crossbill	*Loxia leucoptera*	BS			X						
B	Pine siskin	*Carduelis pinus*	BS									X
M	Preble's shrew	*Sorex preblei*	BS					X				
M	Pygmy shrew	*Sorex hoyi*	BS					X				
M	Yuma myotis	*Myotis yumanensis*	BS					X				
M	Long-eared myotis	*Myotis evotis*	BS		X	X					X	

Table 1—Common and scientific names of 173 terrestrial vertebrate species of focus brought forward for additional analysis and the associated criteria by which each species was selected (continued)

				Criteria								
Class[a]	Common name	Scientific name	Scale[b]	<90H[c]	<90C[d]	<90A[e]	EIS SIG[f]	Fine[g]	NRDC[h]	G1G2[i]	Adjust[j]	Add[k]
M	Fringed myotis	*Myotis thysanodes*	BS		X	X	X				X	
M	Long-legged myotis	*Myotis volans*	BS		X	X	X				X	
M	Western small-footed myotis	*Myotis ciliolabrum*	BS		X	X	X				X	
M	Silver-haired bat	*Lasionycteris noctivagans*	BS		X	X	X				X	
M	Hoary bat	*Lasiurus cinereus*	BS		X	X	X				X	
M	Spotted bat	*Euderma maculatum*	BS	X	X	X					X	
M	Townsend's big-eared bat	*Corynorhinus townsendii pallescens*	BS	X	X	X					X	
M	Pallid bat	*Antrozous pallidus*	BS									X
M	Pygmy rabbit	*Brachylagus idahoensis*	BS	X	X	X						
M	Idaho ground squirrel	*Spermophilus brunneus*	BS					X				
M	White-tailed antelope squirrel	*Ammospermophilus leucurus*	BS					X				
M	Washington ground squirrel	*Spermophilus washingtoni*	BS					X		G2		
M	Wyoming ground squirrel	*Spermophilus elegans*	BS									X
M	Uinta ground squirrel	*Spermophilus armatus*	BS					X				
M	Western gray squirrel	*Sciurus griseus*	BS					X				
M	Northern flying squirrel	*Glaucomys sabrinus*	BS		X	X	X				X	
M	Sagebrush vole	*Lemmiscus curtatus*	BS									X
M	Gray wolf	*Canis lupus*	BS									X
M	Kit fox	*Vulpes macrotis*	BS					X				
M	Grizzly bear	*Ursus arctos*	BS		X	X						
M	American marten	*Martes americana*	BS		X	X	X		X		X	
M	Fisher	*Martes pennanti*	BS	X	X	X	X		X		X	
M	Wolverine	*Gulo gulo*	BS		X	X			X		X	
M	Lynx	*Lynx canadensis*	BS		X	X			X		X	
M	Woodland caribou	*Rangifer tarandus caribou*	BS	X	X	X						
M	Pronghorn	*Antilocapra americana*	BS		X	X	X				X	
M	Mountain goat	*Oreamnos americanus*	BS									X
M	California bighorn sheep	*Ovis canadensis californiana*	BS	X	X	X						
M	Rocky Mountain bighorn sheep	*Ovis canadensis canadensis*	BS									X
A	Coeur d'Alene salamander	*Plethodon idahoensis*	FS	X	X	X					X	
A	Larch mountain salamander	*Plethodon larselli*	FS					X		G2		

Table 1—Common and scientific names of 173 terrestrial vertebrate species of focus brought forward for additional analysis and the associated criteria by which each species was selected (continued)

Class[a]	Common name	Scientific name	Scale[b]	Criteria <90H[c]	<90C[d]	<90A[e]	EIS SIG[f]	Fine[g]	NRDC[h]	G1G2[i]	Adjust[j]	Add[k]
A	Idaho giant salamander	*Dicamptodon aterrimus*	FS					X				
A	Tailed frog	*Ascaphus truei*	FS	X	X	X	X					
A	Western toad	*Bufo boreas*	FS		X	X	X				X	
A	Woodhouse's toad	*Bufo woodhousii*	FS	X	X	X	X					
A	Northern leopard frog	*Rana pipiens*	FS	X	X	X	X				X	
A	Columbian spotted frog	*Rana luteiventris*	FS	X	X	X					X	
A	Oregon spotted frog	*Rana pretiosa*	FS		X	X					X	
R	Western pond turtle	*Clemmys marmorata*	FS			X						
R	Painted turtle	*Chrysemys picta*	FS			X	X					
R	Rubber boa	*Charina bottae*	FS	X	X	X	X					
R	Common garter snake	*Thamnophis sirtalis*	FS			X	X					
B	Common loon	*Gavia immer*	FS	X	X	X						
B	Red-necked grebe	*Podiceps grisegena*	FS	X	X	X						
B	Western grebe	*Aechmophorus occidentalis*	FS	X	X	X						
B	Clark's grebe	*Aechmophorus clarkii*	FS	X	X	X						
B	American white pelican	*Pelecanus erythrorhynchos*	FS	X	X	X						
B	American bittern	*Botaurus lentiginosus*	FS	X	X	X						
B	Western least bittern	*Ixobrychus exilis hesperis*	FS	X	X	X						
B	Great blue heron	*Ardea herodias*	FS	X	X	X						
B	Great egret	*Ardea alba*	FS	X	X	X						
B	Snowy egret	*Egretta thula*	FS	X	X	X						
B	Black-crowned night heron	*Nycticorax nycticorax*	FS	X	X	X						
B	White-faced ibis	*Plegadis chihi*	FS	X	X	X						
B	Trumpeter swan	*Cygnus buccinator*	FS	X	X	X						
B	Wood duck	*Aix sponsa*	FS	X	X	X						
B	Green-winged teal	*Anas crecca*	FS		X	X						
B	Mallard	*Anas platyrhynchos*	FS		X	X						
B	Northern pintail	*Anas acuta*	FS		X	X						
B	Blue-winged teal	*Anas discors*	FS		X	X						
B	Cinnamon teal	*Anas cyanoptera*	FS		X	X						
B	Northern shoveler	*Anas clypeata*	FS		X	X						
B	Gadwall	*Anas strepera*	FS		X	X						

Table 1—Common and scientific names of 173 terrestrial vertebrate species of focus brought forward for additional analysis and the associated criteria by which each species was selected (continued)

Class[a]	Common name	Scientific name	Scale[b]	Criteria								
				<90H[c]	<90C[d]	<90A[e]	EIS SIG[f]	Fine[g]	NRDC[h]	G1G2[i]	Adjust[j]	Add[k]
B	American wigeon	*Anas americana*	FS		X	X						
B	Canvasback	*Aythya valisineria*	FS	X	X	X						
B	Redhead	*Aythya americana*	FS	X	X	X						
B	Ring-necked duck	*Aythya collaris*	FS	X	X	X						
B	Lesser scaup	*Aythya affinis*	FS	X	X	X						
B	Harlequin duck	*Histrionicus histrionicus*	FS	X	X	X	X					
B	Common goldeneye	*Bucephala clangula*	FS	X	X	X						
B	Barrow’s goldeneye	*Bucephala islandica*	FS	X	X	X						
B	Bufflehead	*Bucephala albeola*	FS	X	X	X						
B	Hooded merganser	*Lophodytes cucullatus*	FS	X	X	X						
B	Common merganser	*Mergus merganser*	FS	X	X	X						
B	Ruddy duck	*Oxyura jamaicensis*	FS	X	X	X						
B	Bald eagle	*Haliaeetus leucocephalus*	FS		X	X	X					
B	Yellow rail	*Coturnicops noveboracensis*	FS					X				
B	Virginia rail	*Rallus limicola*	FS	X	X	X						
B	Sora	*Porzana carolina*	FS	X	X	X						
B	Greater sandhill crane	*Grus canadensis tabida*	FS	X	X	X						
B	Western snowy plover	*Charadrius alexandrinus nivosus*	FS	X	X	X						
B	Black-necked stilt	*Himantopus mexicanus*	FS	X	X	X						
B	American avocet	*Recurvirostra americana*	FS	X	X	X						
B	Willet	*Catoptrophorus semipalmatus*	FS	X	X	X						
B	Spotted sandpiper	*Actitis macularia*	FS									X
B	Upland sandpiper	*Bartramia longicauda*	FS		X	X						
B	Long-billed curlew	*Numenius americanus*	FS	X	X	X						
B	Common snipe	*Gallinago gallinago*	FS		X	X						
B	Wilson’s phalarope	*Phalaropus tricolor*	FS									X
B	Forster’s tern	*Sterna forsteri*	FS	X	X	X						
B	Black tern	*Chlidonias niger*	FS	X	X	X						
B	Yellow-billed cuckoo	*Coccyzus americanus*	FS	X	X	X						
B	Western screech owl	*Otus kennicottii*	FS			X						
B	Red-naped sapsucker	*Sphyrapicus nuchalis*	FS		X	X	X					
B	Downy woodpecker	*Picoides pubescens*	FS			X	X					

Table 1—Common and scientific names of 173 terrestrial vertebrate species of focus brought forward for additional analysis and the associated criteria by which each species was selected (continued)

Class[a]	Common name	Scientific name	Scale[b]	Criteria								
				<90H[c]	<90C[d]	<90A[e]	EIS SIG[f]	Fine[g]	NRDC[h]	G1G2[i]	Adjust[j]	Add[k]
B	Willow flycatcher	*Empidonax traillii*	FS		X	X						
B	Least flycatcher	*Empidonax minimus*	FS					X				
B	Veery	*Catharus fuscescens*	FS		X	X						
B	Red-eyed vireo	*Vireo olivaceus*	FS	X	X	X						
B	Virginia’s warbler	*Vermivora virginiae*	FS									X
B	Yellow warbler	*Dendroica petechia*	FS		X	X						
B	American redstart	*Setophaga ruticilla*	FS									X
B	Wilson’s warbler	*Wilsonia pusilla*	FS	X	X	X						
B	Yellow-breasted chat	*Icteria virens*	FS		X	X						
B	Fox sparrow	*Passerella iliaca*	FS									X
B	Bobolink	*Dolichonyx oryzivorus*	FS	X	X	X						
B	Tricolored blackbird	*Agelaius tricolor*	FS					X				
B	Brewer’s blackbird	*Euphagus cyanocephalus*	FS									X
M	Water shrew	*Sorex palustris*	FS					X				
M	Water vole	*Microtus richardsoni*	FS									X
M	Northern bog lemming	*Synaptomys borealis*	FS					X				

[a] B = bird, M = mammal, R = reptile, and A = amphibian.

[b] BS = broad-scale species of focus assessed in this paper. Ninety-one species were identified as broad-scale vertebrates of focus, whose source habitats could reliably be evaluated by using a mapping unit (pixel size) of 100 ha (254 acres). FS = fine-scale species of focus whose source habitats require mapping units <100 ha (254 acres).

[c] <90H = habitat outcome score (from Lehmkuhl and others 1997) <90 points in the added scores of outcomes 1, 2, and 3 for the historical time period, BLM and FS lands only.

[d] <90C = habitat outcome score (from Lehmkuhl and others 1997) <90 points in the added scores of outcomes 1, 2, and 3 for the current time period, BLM and FS lands only.

[e] <90A = habitat outcome score (from Lehmkuhl and others 1997) <90 points in the added scores of outcomes 1, 2, and 3 for any of the 7 alternatives (BLM and FS lands only) described in either the draft eastside EIS (USDA Forest Service and USDI Bureau of Land Management 1997a) or draft upper Columbia River basin EIS (USDA Forest Service, USDI Bureau of Land Management 1997b).

[f] EIS SIG = the weighted mean outcome score in any one of the alternatives (BLM and FS lands only) increased or decreased by more than 0.5 points from the current outcome score, a significant change according to the EIS teams.

[g] Fine = species for which Lehmkuhl and others (1997) did not complete an analysis for the outcome assessment because of the restricted distribution of these species within the basin. These species ranges are predominately outside the basin.

[h] NRDC = species that were the subject of the petition filed by the Natural Resources Defense Council with the Regional Forester, Pacific Northwest Region, USDA Forest Service, on March 30, 1993.

[i] G1G2 = species listed by the Natural Heritage program as Global Rank 1 or Global Rank 2.

[j] Adjust = species for which panelists’ scores were adjusted by the science team (Lehmkuhl and others 1997). Scores were adjusted when considered to reflect a misinterpretation or incomplete understanding of the management alternatives or their outcomes, or the species’ ecology.

[k] Add = species added by terrestrial habitat panelists (vol. 3, appendix 2) during September 1997 due to concerns about habitat or population status. Some of these species were not evaluated in the prior outcome assessment by Lehmkuhl and others (1997).

Determining Species Relations With Source Habitats

Vegetation classification system used to define source habitats—We used the vegetation classification system of cover types and structural stages that was derived for broad-scale vegetation assessments of the ICBEMP (Hann and others 1997) as the basis for defining source habitats for each species of focus. We used this system because (1) it is the standard classification system that was developed to characterize the composition and structure of vegetation at the broad scale within the basin; (2) this system was created specifically to characterize broad-scale patterns of disturbance regimes and succession dynamics over a diverse array of forest and rangeland conditions, at large spatial scales, and over long periods of time; and (3) our results are intended to be integrated with results from all other broad-scale scientific assessments of the ICBEMP, all of which have used this system (for example, see assessments for landscape ecology [Hann and others 1997] and aquatic resources [Lee and others 1997]). Below is a detailed description of the methods used to estimate composition and structure of vegetation under this broad-scale system of classification.

Estimating and validating occurrence of cover types and structural stages for broad-scale assessment—Maps of vegetation cover types (CT) and structural stages (ST) were derived originally as part of the Columbia River basin succession model (CRBSUM) (Keane and others 1996) for broad-scale assessment of vegetation in the basin. The CRBSUM specifically was built to map the composition and structure of vegetation for historical and current periods, accounting for coarse-scale disturbance regimes and succession dynamics (Hann and others 1997, Keane and others 1996). As part of this process, cover types were developed to estimate the plant species that characterize the vegetative composition of a mapping unit, with the mapping unit defined as a pixel or cell of 1 km^2 (0.4-mi^2) at the broad scale. Examples of cover types mapped at this cell size include lodgepole pine, western larch, and whitebark pine for forested environments and big sagebrush, native forb, and juniper/sagebrush for rangeland environments (Hann and others 1997). By contrast, structural stages were developed to estimate the structural conditions of plant species that characterize a mapping unit of 1 km^2 (0.4-mi^2). Examples of structural stages mapped at this scale include stand initiation, understory reinitiation, and old-forest single-story for forested environments and open herbland, closed low-medium shrub, and open tall shrub for rangeland environments (Hann and others 1997).

Methods for deriving the initial estimates of the cover types and structural stages were described by Hann and others (1997) and Menakis and others (1996). Initial estimates of CT and ST were then mapped and rectified with each other and with the CRBSUM potential vegetation type (PVT) map as part of the classification and modeling process (Menakis and others 1996). The PVTs are classes of biophysical environments based on combinations of climate, terrain, and soil that are labeled by plant species, with the labels serving as indicators of the kind of environmental conditions present (Hann and others 1997). Indicator plants used to name the PVT are often not the plant species name of the CT because of disturbances, succession, and exotic plant invasions that result in dominance by other species. For example, ponderosa pine is a common CT in the Douglas-fir PVTs in environments where fire has been frequent historically, which is part of the native regime. Cheatgrass, an exotic plant species, is a common CT in sagebrush PVTs in dry environments, typically in conjunction with a combination of excessive livestock grazing and increased frequency of fire (Hann and others 1997), which is not part of the native regime. The PVTs have been grouped into potential vegetation groups (PVGs) such as forest, dry shrub, and agriculture.

Rectification among CT, ST, and PVT estimates was conducted to ensure that CTs and STs would only occur on sites that had the successional potential to produce those CTs and STs (Menakis and others 1996). This not only improved broad-scale accuracy, but also met the logic conditions for simulating succession and disturbance dynamics with the CRBSUM. For example, if a ponderosa pine CT occurred with an open herbland ST on a whitebark pine/subalpine larch north PVT, an obvious problem existed with the input data. Many combinations of CT/ST/PVT, however, had potential errors that were more subtle. The CRBSUM contained a logic-checking routine that compared the CT/ST/PVT combinations with the successional pathways of combinations of CT/ST that could occur in a given PVT. A rule set was established for correcting logic errors. In general, the PVT input map was more accurate than the CT and ST maps because of its direct relation to biophysical characteristics. Consequently, if an error was detected, the CT or ST typically was changed to be consistent with the PVT. In some instances, however, certain CTs had high predictive reliability; in these cases, the CTs were used to identify a need to correct some of the PVTs.

The CRBSUM maps for the current period were designed to reflect average conditions for the decade 1985 to 1995 (Hann and others 1997). Two input maps were used to develop the CRBSUM CT map. Hardy and others (1996) provided a broad classification of cover types through use of 1-km^2 (0.4-mi^2) satellite imagery. A land cover characterization (LCC) map for the United States provided an additional source for broad cover types (Eidenshink 1992, Loveland and others 1991). These two maps were refined by ecologists during several ICBEMP workshops and used to develop the final input map (Menakis and others 1996). This final map was then refined based on the CRBSUM logic-checking process described above and in Menakis and others (1996).

The current period CRBSUM ST map was developed by using a statistical analysis of current mid-scale subwatershed sample data from Hessburg and others (1999) that was aggregated to a 1-km^2 (0.4-mi^2) scale (Menakis and others 1996). The ST data from the subwatershed sample were correlated with other 1-km^2 (0.4 mi^2) scale data, such as CT, PVT, ownership, and road density, and then extrapolated with a statistical function across all 1-km^2 (0.4-mi^2) cells of the basin. This ST map was then refined based on the CRBSUM logic-checking process (Menakis and others 1996).

The CRBSUM maps for the historical period were designed to reflect average conditions for the latter half of the 19th century (1850 to 1900) (Hann and others 1997). The CT input map for historical conditions was a vector map from Losensky (1994), derived from a compilation of late 1800s and early 1900s vegetation survey, potential land use, and military expedition maps. This CT map was then refined based on the CRBSUM logic-checking process (Menakis and others 1996).

The CRBSUM ST map for the historical period was developed from survey data supplied by Losensky (1994). These data were used to determine a ST composition by CT for each of the survey areas, and were then extrapolated across the basin within cover type and ecoregion stratifications (Menakis and others 1996). This ST map was then refined based on the CRBSUM logic-checking process for combinations of CT, ST, and PVT described earlier (Menakis and others 1996).

The current and historical period CT, ST, and PVT data were compared with maps of cover types and structural stages estimated at the mid-scale (cell size of 4 ha [10 acres]) from aerial photos taken during the current period (1990s) and a more recent historical period (1930s to 1950s) that was the basis for the mid-scale analysis of Hessburg and others (1999) (Hann and others 1997, Menakis and others 1996). The more recent historical data from Hessburg and others (1999), which represent the mid-20th century estimate of CTs and STs at the mid-scale (4-ha [10-acre] cell size), do not represent the same time period as the historical period for broad-scale data; thus the mid-scale and broad-scale estimates of CTs and STs could not be compared directly. The mid-scale and broad-scale data used to estimate the current period, however, represent comparable periods. Results of comparisons between mid- and broad-scale estimates of CTs and STs for the current period are reported in Hann and others (1997) and Menakis and others (1996). Additional data used for assessment of accuracy of the broad-scale mapping included paired historic-current oblique photographs from Losensky (1995) and plot data that were used for the assessment of succession-disturbance regimes and general landscape patterns (Hann and others 1997).

Because maps of cover types and structural stages were produced at a 1-km^2 (0.4-mi^2) (or 100 ha) scale as part of the development of CRBSUM, users should be aware of the implications of this large mapping scale. A 1-km^2 (0.4-mi^2) cell is about 250 acres [somewhat larger than a 1/4 section (160 acres)]. Linear features such as roads, narrow riparian vegetation, and streams cannot be mapped at this scale. Cover types that occur in small patches of <4 ha (10 acres) and that have an average patch size less than one-fourth of the area of a 1-km^2 (0.4-mi^2) cell also are not mapped. Cover types that occur in either large or small patches and that have an average patch size greater than one-fourth the area of a 1-km^2 (0.4-mi^2) cell (that is, >25 ha or 62 acres), however, are typically mapped because some of these patches will be large enough to dominate a 1-km^2 (0.4-mi^2) cell. Any 1-km^2 (0.4-mi^2) cell has only about a 10- to 30-percent chance of being correctly typed, but about 65 to 95 percent of a large number of cells (for example, 100 or more cells) of the same type or of a group of types typically are mapped correctly. The phenomenon of low probability of any one cell being correctly typed, but high probability of correctly typing a large number of cells of the same type, occurs for four reasons:

1. **High variation in number of types within the cell.** Mapping units composed of 1-km^2 (0.4-mi^2) cells typically contain three to five different cover types that occur in patch sizes of about 4 ha (10 acres) or larger. A patch size of 4 ha (10 acres) is equivalent to the mapping unit used by Hessburg and others (1999) for the mid-scale landscape analysis of the basin, and is the size patch that generally can be detected as part of mapping at the broad-scale of 1 km^2 (0.4-mi^2). Typically, the cover type with the largest area or greatest biomass dominates the characteristics of the cell. In many cases, the named type only covers 20 to 30 percent of the cell area, but it has the largest area and thus dominates the reflectance shown in the remote-sensed data source. In other cases, a forest type may compose less area than a nonforest type, but because of the large amount of biomass in forest types, the spectral reflectance may be dominated by the forest type. Accurate mapping of these types is dependent on the summary of many cells, which dampens the effect of high variation in cell composition.

2. **High variation in type distribution within cells.** Cover types that typically occur in small patches but are distributed abundantly and scattered throughout the cell also may dominate the characteristics of the cell. Accurate mapping of these types is dependent on summary of many cells or grouping of cover types, which again dampens the effect of high variation in type distribution within cells.

3. **Small sample size.** Cover types that occur in large patches, but that do not occur in many cells, will dominate the characteristics of those cells. Accurate mapping of these types is dependent on grouping of related types, which dampens the effect of small sample size.

4. **Cover types with similar characteristics.** Two or more cover types that have similar characteristics may dominate the characteristics of many cells. Accurate mapping of these types is dependent on finding accurate correlations with other mapped biophysical and human-caused characteristics. This dampens effects of errors in misclassification to other cover types that have similar prediction characteristics.

These points provide context for understanding results of a formal assessment of mapping accuracy that was conducted to estimate the minimum-sized area (for example, subbasin or ERU) at which broad-scale data could be summarized to ±10 percent confidence of the true estimate of the percentage of area occupied by cover types and structural stages (table 2). In general, groups of subbasins or an ERU were found to be appropriate levels at which to summarize the 1-km^2 (0.4-mi^2) CT and ST data. Hann and others (1997) demonstrated that grouping similar CT and ST into physiognomic types or terrestrial communities substantially increased this accuracy. Results of this accuracy assessment (table 2) imply that use of CT and ST combinations to analyze source habitats for terrestrial vertebrates is not sufficiently accurate for making summaries at an individual subwatershed or watershed scale. Sufficient accuracy can be achieved, however, when base data for individual subwatersheds or watersheds are summarized to the larger scales of subbasin, ERU, or basin, by using base data from collections of subwatersheds or watersheds (table 2).

Building species-source habitat matrices—Marcot and others (1997) originally developed matrices of habitat associations for 547 vertebrate species occurring within the basin. These matrices included species associations with macrohabitats based on species occurrence, as well as species use of finer scale or nonvegetative features termed key environmental correlates. We used these data as a starting point to define source habitats and special habitat features for each species of focus. As noted earlier, source habitats are those characteristics of macrovegetation that contribute to stationary or positive population growth. Special habitat features are those nonvegetative factors or finer scale characteristics of vegetation that also contribute to stationary or positive population growth.

The habitat matrices of Marcot and others (1997) were based on slightly modified combinations of cover types and structural stages that were defined for macrovegetation of the basin (tables 3 and 4); methods used to estimate these cover types and structural stages at the broad scale were described in the previous section and described in further detail by Keane and others (1996), Menakis and others (1996), and

Table 2—Current and historical broad-scale cover type and structure vegetation maps with estimated accuracy

Map	Representative period	Method	Minimum area to achieve acceptable accuracy for codominant types	Minimum area to achieve acceptable accuracy across all types
Current cover type	1985-95	Correlation of ground truth with 1-km 1991 AVHRR satellite spectral class	Subbasin	Ecological reporting unit
Current structure	1985-95	Prediction model from correlation of mid-scale samples with broad-scale attributes	2-4 subbasins	Ecological reporting unit
Historical cover type	1850-1900	Vector mapping from late 1800s and early 1900s maps and records	3-6 subbasins	Ecological reporting unit
Historical structure	1850-1900	Cover type and ecoregion section random allocation of structure distribution from late 1800s and early 1900s records	5-10 subbasins	Ecological reporting unit
Current physiognomic types	1985-2005	Grouping of current cover types and structures based on similar response to succession and disturbance	Subbasin	2-3 subbasins
Historical physiognomic types	1800-1900	Grouping of historical cover types and structures based on similar response to succession and disturbance	2-3 subbasins	4-6 subbasins
Current physiognomic groups by PVG	1985-2005	Grouping of current physiognomic types by PVG	Watershed	2-3 watersheds
Historical physiognomic groups by PVG	1800-1900	Grouping of historical physiognomic types by PVG	2-3 watersheds	4-6 watersheds

Table 2—Current and historical broad-scale cover type and structure vegetation maps with estimated accuracy (continued)

Map	Representative period	Method	Minimum area to achieve acceptable accuracy for codominant types	Minimum area to achieve acceptable accuracy across all types
Current physiognomic group by PVG dominant patterns	1985-2005	Classes of uniform, mosaic, or mixed dominant composition patterns of physiognomic groups by PVG	Subwatershed	NA
Historical physiognomic group by PVG dominant patterns	1800-1900	Classes of uniform, mosaic, or mixed dominant composition patterns of physiognomic groups by PVG	Subwatershed	NA
Current terrestrial communities	1985-2005	Grouping of current cover types and structures based on similar terrestrial habitat characteristics	Subbasin	3-4 subbasins
Historical terrestrial communities	1800-1900	Grouping of historical cover types and structures based on similar terrestrial habitat characteristics	3-4 subbasins	6-8 subbasins

NA = not applicable.

Hann and others (1997). We expanded these estimates of macrovegetation to include two different types of structural stages for young forests: managed young forest and unmanaged young forest (tables 3 and 4). This expansion was important because the young-forest structural stage can differ widely in the density of large snags and legacy trees (Hann and others 1997). Moreover, differences in the densities of snags and legacy trees presumably affect survival of several cavity- and snag-dependent species (Thomas and others 1979), many of which we identified as species of focus. Managed young-forests, which we defined quantitatively in table 4, are young-forest structural stages within areas that are roaded and with some history of timber harvest and fire exclusion (table 3.178, Hann and others 1997); these stands contain relatively few large snags and trees >53 cm (21 in) in diameter at breast height (d.b.h.) (table 3.178, Hann and others 1997). By contrast, unmanaged young forests, which we also defined quantitatively in table 4, are young-forest structural stages within areas that are unroaded, with fire exclusion and no history of timber harvest (table 3.178, Hann and others 1997); these stands contain relatively higher densities of large snags and trees (table 3.178, Hann and others 1997). In addition, for the purpose of our evaluation, we lumped the six structural stages of woodlands into one (table 4).

Table 3—Terrestrial community groups and terrestrial community types and their included cover types and structural stages as adapted from Hann and others (1997)

Terrestrial community group/type	Included cover types	Included structural stage codes[a]
Alpine:		
Alpine	Alpine tundra	Olms, Clms
Subalpine forest		
Late-seral subalpine single-layer forest	Whitebark pine	Ofs
Late-seral subalpine single-layer forest	Mountain hemlock	Ofs
Late-seral subalpine multi-layer forest	Whitebark pine	Ofm
Late-seral subalpine multi-layer forest	Whitebark pine-alpine larch	Ofm
Late-seral subalpine multi-layer forest	Engelmann spruce-subalpine fir	Ofm
Late-seral subalpine multi-layer forest	Mountain hemlock	Ofm
Mid-seral subalpine forest	Whitebark pine	UYf, MYf, Ur, Sec
Mid-seral subalpine forest	Whitebark pine-alpine larch	UYf, MYf, Ur, Seo
Mid-seral subalpine forest	Engelmann spruce-subalpine fir	UYf, MYf, Ur, Sec
Mid-seral subalpine forest	Mountain hemlock	UYf, MYf, Ur, Sec
Early-seral subalpine forest	Whitebark pine	Si
Early-seral subalpine forest	Whitebark pine-alpine larch	Si
Early-seral subalpine forest	Engelmann spruce-subalpine fir	Si
Early-seral subalpine forest	Mountain hemlock	Si
Montane forest:		
Late-seral montane single-layer forest	Western redcedar-western hemlock	Ofs
Late-seral montane single-layer forest	Interior Douglas-fir	Ofs
Late-seral montane single-layer forest	Western larch	Ofs
Late-seral montane single-layer forest	Lodgepole pine	Ofs
Late-seral montane single-layer forest	Grand fir-white fir	Ofs
Late-seral montane single-layer forest	Sierra Nevada mixed conifer	Ofs
Late-seral montane single-layer forest	Western white pine	Ofs
Late-seral montane multi-layer forest	Pacific silver fir-mountain hemlock	Ofm
Late-seral montane multi-layer forest	Western redcedar-western hemlock	Ofm
Late-seral montane multi-layer forest	Interior Douglas-fir	Ofm
Late-seral montane multi-layer forest	Western larch	Ofm
Late-seral montane multi-layer forest	Lodgepole pine	Ofm
Late-seral montane multi-layer forest	Red fir	Ofm
Late-seral montane multi-layer forest	Grand fir-white fir	Ofm
Late-seral montane multi-layer forest	Sierra Nevada mixed conifer	Ofm
Late-seral montane multi-layer forest	Western white pine	Ofm
Mid-seral montane forest	Pacific silver fir-mountain hemlock	UYf, MYf, Ur, Sec
Mid-seral montane forest	Western redcedar-western hemlock	UYf, MYf, Ur, Sec
Mid-seral montane forest	Interior Douglas-fir	UYf, MYf, Ur, Sec
Mid-seral montane forest	Western larch	UYf, MYf, Ur, Sec
Mid-seral montane forest	Lodgepole pine	UYf, MYf, Ur, Sec
Mid-seral montane forest	Red fir	UYf, MYf, Ur, Sec
Mid-seral montane forest	Grand fir-white fir	UYf, MYf, Ur, Sec
Mid-seral montane forest	Sierra Nevada mixed conifer	UYf, MYf, Ur, Sec
Mid-seral montane forest	Western white pine	UYf, MYf, Ur, Sec
Early-seral montane forest	Pacific silver fir-mountain hemlock	Si
Early-seral montane forest	Western redcedar-western hemlock	Si
Early-seral montane forest	Interior Douglas-fir	Si

Table 3—Terrestrial community groups and terrestrial community types and their included cover types and structural stages as adapted from Hann and others (1997) (continued)

Terrestrial community group/type	Included cover types	Included structural stage codes[a]
Early-seral montane forest	Western larch	Si
Early-seral montane forest	Lodgepole pine	Si
Early-seral montane forest	Red fir	Si
Early-seral montane forest	Grand fir-white fir	Si
Early-seral montane forest	Sierra Nevada mixed conifer	Si
Early-seral montane forest	Western white pine	Si
Early-seral montane forest	Shrub or herb/tree regeneration	Ots, Olms, Clms, Ch
Lower montane forest:		
Late-seral lower montane single-layer forest	Pacific ponderosa pine	Ofs
Late-seral lower montane single-layer forest	Interior ponderosa pine	Ofs
Late-seral lower montane multi-layer forest	Pacific ponderosa pine	Ofm
Late-seral lower montane multi-layer forest	Interior ponderosa pine	Ofm
Mid-seral lower montane forest	Pacific ponderosa pine	UYf, MYf, Ur, Sec
Mid-seral lower montane forest	Interior ponderosa pine	UYf, MYf, Ur, Seo, Sec
Early-seral lower montane forest	Pacific ponderosa pine	Si
Early-seral lower montane forest	Interior ponderosa pine	Si
Riparian woodland:		
Riparian woodland	Aspen	Ofm, UYf, MYf, Ur, Sec, Si
Riparian woodland	Cottonwood/willow	Ofm, UYf, MYf, Ur, Sec, Si
Upland woodland:		
Upland woodland	Limber pine	Wdl
Upland woodland	Juniper woodlands	Wdl
Upland woodland	Mixed-conifer woodlands	Wdl
Upland woodland	Juniper/sagebrush	Wdl
Upland woodland	Oregon white oak	Wdl
Upland shrubland:		
Upland shrubland	Chokecherry-serviceberry-rose	Ots, Olms, Clms
Upland shrubland	Mountain mahogany	Olms, Clms
Upland shrubland	Big sagebrush	Olms, Clms, Ch
Upland shrubland	Mountain big sagebrush	Olms, Clms
Upland shrubland	Low sage	Olms, Clms
Upland shrubland	Salt desert shrub	Olms, Clms
Upland shrubland	Antelope bitterbrush/bluebunch wheatgrass	Clms
Upland herbland:		
Upland herbland	Wheatgrass bunchgrass	Ch, Oh
Upland herbland	Fescue-bunchgrass	Ch, Oh
Upland herbland	Native forb	Ch, Oh
Riparian shrubland:		
Riparian shrubland	Shrub wetlands	Cts, Olms, Clms
Riparian herbland:		
Riparian herbland	Herbaceous wetlands	Ch, Oh
Exotic herbland:		
Exotic herbland	Exotic forbs-annual grass	Ch, Oh

Table 3—Terrestrial community groups and terrestrial community types and their included cover types and structural stages as adapted from Hann and others (1997) (continued)

Terrestrial community group/type	Included cover types	Included structural stage codes[a]
Agriculture:		
Agricultural	Cropland-hay-pasture	Ch, Oh
Rock:		
Rock/barren	Barren	
Urban:		
Urban	Urban	
Water:		
Water	Water	

[a] Structural stage codes are defined in table 4.

The inclusion of these refined structural stages (table 4) with previous estimates of macrovegetation (Hann and others 1997) resulted in 157 cover type-structural stage combinations nested within 15 terrestrial community groups (table 3, fig. 3). Only those combinations of cover types and structural stages that plausibly occurred historically or that occur currently were used.

We also refined the seasons of use identified by Marcot and others (1997) because source habitats can function as breeding, rearing, migratory, or wintering areas. Consequently, source habitats were classified according to the seasonal functions that such habitats provide in supporting population persistence by using several broad categories. Species were first characterized as being either migratory or year-long residents of the basin. Migratory species were defined as species that spend part of the year outside the basin. Resident species were defined as species that live year-long within the basin.

For migratory species, we established three seasonal categories of habitat function: (1) **migrant breeding habitat**, defined as source habitat used for breeding or rearing in the basin by species that migrate seasonally to areas outside the basin; (2) **migrant wintering habitat**, defined as source habitat used for winter survival by species that reside within the basin during winter but breed elsewhere; and (3) **migrant migratory habitat**, defined as source habitat used for survival during migration through the basin by species that breed or winter elsewhere.

For resident species, we also established three categories of habitat function: (1) **resident summer habitat**, defined as source habitats used for survival or reproduction or rearing, or all three, late spring through early fall, by species who live year-long within the basin; (2) **resident winter habitat**, defined as source habitats used for survival during late fall through early spring by species that live year-long within the basin; and (3) **resident year-long habitat**, defined as source habitats used commonly throughout the year by a species to meet all seasonal life functions.

For species that depend on different source habitats in different seasons, a separate set of source habitat designations was indicated for each season based on the above system of classification. For resident species that depend on the same source habitats year-round, only one entry, resident year-long, was identified. For migrant species, those that were known to breed within the basin were always evaluated under the category of migrant breeding habitat; either of the other two categories (migrant wintering and migrant migratory habitats) was used only if the species was known not to breed within the basin, or if wintering or migratory habitat was deemed to constitute a different set of source habitats than those for breeding habitat.

Another variation in seasonal habitat function was used for one species, the Lewis' woodpecker. Experts identified two distinct populations, one migratory, the other resident. Accordingly, the migratory population of Lewis' woodpeckers was evaluated under the category of migrant breeding habitat; this population was deemed to occur throughout the range of the species

Figure 3—Illustration of forest structural stages defined in table 3 and in Hann and others (1997) that were used as part of methods to determine species relations with source habitats for 91 broad-scale species of focus.

Table 4—Structural stages defined for assessing the structural features of macrovegetation across the interior Columbia basin, as adapted from Hann and others (1997)

Structural stage	Structural stage code	Description[a]
Forest:		
Stand initiation	Si	LgT_cc <30% and SS_cc ≥10% and [(PT_cc + SmT_cc + MedT_cc <20%) or (PT_cc + SmT_cc + MedT_cc ≤60% and PT_cc + SmT_cc + MedT_cc ≥20% and SmT_cc + MedT_cc <10%)]
Stem-exclusion open canopy	Seo	LgT_cc <30% and SS_cc <10% and PT_cc + SmT_cc + MedT_cc ≤70%
Stem-exclusion closed canopy	Sec	LgT_cc <30% and SS_cc <10% and PT_cc + SmT_cc + MedT_cc >70%
Understory reinitiation	Ur	LgT_cc <30% and SS_cc ≥10% and PT_cc + SmT_cc + MedT_cc >60%
Managed young multi-story	MYf	LgT_cc <30% and SS_cc ≥10% and PT_cc + SmT_cc + MedT_cc ≤60% and SmT_cc ≥10% or MedT_cc ≥10%. Has undergone some form of silivicultural treatment, salvage, or roading; contain relatively few large snags and trees (>53.2 cm d.b.h.)
Unmanaged young multi-story	UYf	LgT_cc <30% and SS_cc ≥10% and PT_cc + SmT_cc + MedT_cc ≤60% and SmT_cc ≥10% or MedT_cc ≥10%. Has not undergone active forms of management; contain relatively higher densities of large snags and trees (>53.2 cm d.b.h.)
Old multi-story	Ofm	LgT_cc ≥30% and SS_cc + PT_cc + SmT_cc + MedT_cc >20%
Old single story	Ofs	LgT_cc ≥30% and SS_cc + PT_cc + SmT_cc + MedT_cc ≤20%
Woodland:	WDL	All structural stages of the woodland community group were combined as one for this assessment
Stand initiation		PT_cc + SmT_cc + MedT_cc + LgT_cc <10% and SS_cc ≥10%
Stem exclusion		LgT_cc <10% and PT_cc + SmT_cc + MedT_cc ≥10% and SS_cc <10%
Understory reinitiation		LgT_cc <10% and PT_cc + SmT_cc + MedT_cc ≥10% and SS_cc ≥10%
Young multi-story		LgT_cc <10% and SmT_cc + MedT_cc ≥10% and PT_cc ≥10% and SS_cc ≥10%
Old multi-story		LgT_cc ≥10% and SS_cc + PT_cc + SmT_cc + MedT_cc ≥10%
Old single story		LgT_cc ≥10% and SS_cc + PT_cc + SmT_cc + MedT_cc <10%

Table 4—Structural stages defined for assessing the structural features of macrovegetation across the interior Columbia basin, as adapted from Hann and others (1997) (continued)

Structural stage	Structural stage code	Description[a]
Nonforest-nonwoodland:[b]		
Open herbland	Oh	A canopy of herbaceous vegetation with <66% projected canopy cover; <10% cover each of shrubs or trees; ≥1 stratum
Closed herbland	Ch	A canopy of herbaceous vegetation with ≥66% projected canopy cover; <10% cover each of shrubs or trees; ≥1 stratum
Open low-medium shrub	Olms	A canopy of low (<50 cm) or medium-sized (50 cm - 2 m) shrubs with <66% projected canopy cover; shrubs dominate; tree cover <10%; ≥2 strata, ≥2 cohorts possible
Closed low-medium shrub	Clms	A canopy of low (<50 cm) or medium-sized (50 cm - 2 m) shrubs with ≥66% projected canopy cover; shrubs dominate; tree cover <10%; ≥2 strata, ≥2 cohorts possible
Open tall shrub	Ots	A canopy of tall (2 - 5 m) shrubs with <66% projected canopy cover; shrubs dominate; tree cover <10%; ≥2 strata, ≥2 cohorts possible
Closed tall shrub	Cts	A canopy of tall (2 - 5 m) shrubs with ≥66% projected canopy cover; shrubs dominate; tree cover <10%; ≥2 strata, ≥2 cohorts possible
Agricultural		Dominated by crop and pasture land use
Urban		Dominated by rural and urban buildings and facilities
Water		Large bodies of water
Rock		Large areas of rock with <5% vegetative canopy cover

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[a] Structural stage descriptions include the following abbreviations:

- tree size class: SS = seedlings and saplings [<12.6 cm diameter at breast height (d.b.h.)]; PT = pole trees (12.7 - 22.6 cm d.b.h.); SmT = small trees (22.7 - 40.4 cm d.b.h.); MedT = medium trees (40.5 - 53.1 cm d.b.h.); and LgT = large trees (>53.2 cm d.b.h.).
- cc = crown cover. Crown cover was interpreted in 10-percent increments, and class percentages were expressed as midpoints, for example, 10 percent = 5 to 14 percent, and 20 percent = 15 to 24 percent.

[b] Canopy cover related to herblands and shrubs is based on the definition and measurement technique reported in Hann and others (1997; Appendix 3-G, p. 1007) and in Hessburg and others (1999). This technique uses photo interpretation methods at a scale of about 1:12,000, which is not applicable to the fine-scale techniques typically used by Forest Service and BLM field staff on the ground. These agencies typically measure on-the-ground cover at a 1:1 scale, often by a line-intercept technique for shrubs, or by a quadrat microplot for herbaceous plants.

A comparison of the two techniques and scales (1:1 versus 1:12,000) reveals a ratio of about 1:4; i.e., canopy cover thresholds using the photo interpretation (1:12,000) scale will be about 4 times higher than canopy cover thresholds using the line intercept (1:1) scale (S. Bunting, University of Idaho Range Science Department). For example, a 15-percent canopy cover of shrubs using line intercept at a 1:1 on-the-ground scale will be comparable to a 60- to 70-percent canopy cover using photo-interpretation dot-grid techniques at a 1:12,000 scale.

This table uses the definition for canopy cover that is consistent with that used in photo interpretation (i.e., 1:12,000).

Figure 4—Conceptual framework for the hierarchical system of species, groups, and families as part of a systems ecology approach to identify habitat requirements and habitat trends for 91 broad-scale species of focus within the interior Columbia basin.

within the basin. The resident population was evaluated under the category of resident year-long habitat; this population was identified as occurring primarily in oak woodlands within a narrow band along the western boundary of the basin, immediately south and north of the Columbia River.

We then refined the species-habitat matrices of Marcot and others (1997) by asking experts to identify each cover type-structural stage combination that presumably contributes to positive or stationary population growth for a given species (source habitat) and for a given season of habitat function. We also asked experts to identify nonvegetative factors or fine-scale vegetative characteristics that presumably contribute to stationary or increasing rate of population growth (see special habitat features in vol. 3, appendix 1, table 2). Specifically, the experts (1) identified the seasonal function of source habitat represented in the existing matrix of Marcot and others (1997), (2) converted the existing species-habitat associations to species relations with source habitats, (3) created a separate record of species-source habitat relations for any additional seasonal habitats needed to fully represent disparate seasons of source habitat function, and (4) refined information as appropriate from the key environmental correlates (Marcot and others 1997) to identify special habitat features.

For a given species, experts assigned a value of one to each combination of cover type-structural stage that was designated as source habitat, and a value of zero to each combination that was designated as nonsource habitat. These same binary codes were used to identify special habitat features deemed to contribute to stationary or positive population growth (value of one) versus those features determined not to contribute to stationary or positive growth (value of zero).

Designations of source habitats and special habitat features for each of the 91 broad-scale species of focus were summarized and stored in two Paradox[3] databases (vol. 3, appendix 1, tables 1 and 2). Data in table 1, appendix 1, volume 3, were used as the basis for our analysis of change in source habitats for species and groups. Appendix 2 in volume 3 provides a list of all experts, their professional affiliation, and the associated taxonomic groups of species that each expert addressed in the process described above.

Designing a Hierarchical System of Single- and Multi-Species Assessment

We wanted to develop a system of single- and multi-species assessment that would enable managers to (1) address either single- or multi-species needs, depending on objectives; (2) identify broad-scale, robust patterns of habitat change that affect multiple species in a similar manner; (3) address the needs of many species efficiently, accurately, and holistically with the use of broad-scale strategies and practices; (4) determine how well an evaluation of a group of species or a set of multiple groups of species indexed evaluations conducted for individual species within the groups; and (5) consider dynamics in source habitats at multiple spatial scales and across time to facilitate maximum flexibility in the design and implementation of spatially and temporally explicit strategies.

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[3] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

In response to these criteria, we established a hierarchical system to evaluate source habitats for individual species, for groups of species, and for families of groups (fig. 4). Species selected for analysis were clustered into groups based on similarities in source habitats. Likewise, groups of species were placed within families based on similarities in source habitats. Each species within a group, and each group within a family, was nested completely within each higher level grouping (fig. 4). That is, each species was assigned to one group, and each group assigned to one family.

This hierarchical nesting allowed for analysis to be flexible and adaptive. For example, managers often must generalize or blend the habitat requirements of many species to accommodate the composite needs of all species under ecosystem management. Each species, however, occupies its own niche and therefore has a unique set of habitat requirements, thereby suggesting that broad-scale, ecosystem-based management strategies may address the needs of some species better than others (Marcot and others 1994). Under our system, the degree to which a given set of management strategies meets the needs of each species can be quantified by evaluating the efficacy of the management strategies at all three levels: species, group, and family. Often, results of the family or group evaluations likely reflect the species evaluations accurately; in such cases, the higher levels of generalization (group or family) index the species-level phenomenon more efficiently than a species-by-species approach. When the requirements of a given species are not reflected well at the level of the group or family, however, evaluations of individual species can be used to complement the group- or family-level evaluations. For example, a species listed as federally threatened or endangered may have specialized or stringent habitat requirements that dictate specific consideration within a broader, ecosystem-based approach. Under our hierarchical system of species-, group-, and family-level evaluations, managers can choose multiple levels of display regarding habitat trends for species, groups, or families, depending on objectives and the level of generalization desired.

In essence, our system of single- and multi-species assessment represents the combined use of coarse-filter and fine-filter approaches described by Noss (1987) and Hunter (1991). Coarse-filter species management assumes that managing an appropriate amount and arrangement of all representative land areas and habitats will provide for the needs of all associated species. By contrast, fine-filter species management provides habitats for a single or a few species only. To date, biologists and managers have argued in favor of one approach over another (for example, Hunter 1991), with few or no efforts made to combine coarse- and fine-filter species management in a hierarchical framework (but see Hansen and others [1993] as one attempt to hybridize coarse- and fine-filter approaches). Our hierarchical system of single- and multi-species assessment represents one of the first attempts to combine past, seemingly disparate approaches at evaluating single versus multiple species, and to apply our new method at multiple spatial scales and periods.

In addition to the lack of methods available to managers for conducting multi-species assessment efficiently and accurately, vertebrate ecologists have largely been unsuccessful in developing methods of multi-species assessment that accurately reflect the habitat needs of individual species (Mannan and others 1984), particularly in terms of addressing population persistence (Conroy and Noon 1996). Consequently, we used our assessment of trends in source habitats that were conducted at all three levels—species, group, and family—to evaluate how well the group- and family-level assessments reflected the species-level assessments from an ecological view. We did this by calculating correlation coefficients of habitat trends among species within groups and within families, and comparing those coefficients with coefficients calculated for species among groups and families. (See “Correlating Change in Source Habitats Between Species within Groups and Families” below). Our hierarchical approach therefore is different from past attempts to index the needs of a large set of species by using shortcut methods that typically did not test how well such indices actually represent the needs of the larger, targeted set of species (Marcot and others 1994). Examples of such shortcut methods include the use of coarse filters (Hunter 1991, Noss 1987), management indicator species (Landres and others 1988, Marcot and others 1994), umbrella or flagship species (Landres and others 1988, Marcot and others 1994), species or indicator guilds (Morrison and others 1992, Szaro 1986, Verner 1984), and measures of species diversity such as hotspots, gaps, and centers of endemism (Marcot and others 1997, Scott and others 1993). Intended or empirical applications of these shortcut methods generally do not evaluate the needs of individual species in relation to the index but instead simply presume that the method correctly indexes the needs of a larger set of species (Hunter 1990, Morrison and others 1992, Noss 1987). Moreover, users of the shortcut methods often fail to reference the larger set of species presumably being indexed (Morrison and others 1992).

Although our hierarchical system may have advantages over previous attempts to index the needs of many species with a few indicators, our system may not perform well under assessments of microhabitats, or for evaluations of fine-scale changes in microhabitats (Mannan and others 1984). For example, two species of birds that each depend on the same old-forest habitat may respond similarly to clearcutting of an old-forest stand; that is, if the stand is eliminated, habitat for both species is removed. Each species, however, may respond differently to the selective removal of large snags, while maintaining the large overstory trees in the stand, if one species depends on large snags and the other does not. Szaro (1986) makes this distinction in his evaluation of guilds as predictive tools and cautions biologists not to simply declare a tool as either flawed or successful without applying and judging the tool at the proper spatial scale. We advise biologists to consider this same context when using our hierarchical system: it was intended for broad-scale, coarse-level evaluations, not as a fine-scale tool to evaluate microhabitats. Consequently, use of higher level groupings of species may not always be appropriate when conducting fine-scale, local evaluations of within-stand or microhabitat changes for multiple species of vertebrates

Clustering the Species into Groups

To begin building our hierarchical system of habitat evaluation for species, groups, and families, we used hierarchical cluster analysis to form 40 groups (table 5) of the 91 broad-scale species of focus. Composite groups were identified by using a hierarchical clustering algorithm based on pairwise similarities in source habitats between species (vol. 3, appendix 1, table 1). For each pair of species, similarity was estimated by using the Ochiai index of similarity (*OI*) (Ludwig and Reynolds 1988):

$$OI_{ij} = \frac{a_{ij}}{\sqrt{a_{ij}+b_i}\,\sqrt{a_{ij}+c_j}} \quad ,$$

where a_{ij} is the number of source habitats shared by species i and j, and b_i and c_j are the number of source habitats unique to species i and j, respectively. The *OI* index can range from a minimum value of zero (no shared habitats) to a maximum of one (identical use of habitats). Relative to other similarity measures (Krebs 1989, Romesburg 1984), the *OI* index is more heavily weighted by the number of habitats in common, rather than those habitats not shared by each pair. The complement of similarity, or dissimilarity (D_{ij}), is defined as one minus the similarity.

Dissimilarities between each pair of species were used to generate a distance matrix that was used in the clustering procedure. We used a hierarchical clustering procedure (Proc Clus, SAS Inc. 1989a, p. 519-614) that began with 91 species and then sequentially joined species and groups of species into progressively fewer clusters until all species were joined in a single cluster. We identified various numbers of clusters (Proc Tree, SAS Inc. 1989c, p. 1613-1632) that statistically provided the best fit to the data based on the pseudo t^2 and F-statistics generated by the cluster procedure (Proc Clus, SAS Inc. 1989a, p. 519-614). We then examined species membership within each set of clusters, looking for a degree of aggregation that would be consistent with our ecological understanding of species relations. Based on this examination, we chose the smallest number of groups that allowed aggregation without loss of important, unique patterns in source habitats for particular species. Experts then reviewed our initial groups and made recommendations for refining species membership and the number of groups to bring forward for analysis. We reviewed the experts' recommended changes, made additional refinements, and obtained additional review from experts to arrive at the final list of 40 groups (table 5).

Assessing Change in Source Habitats From Historical to Current Conditions for Species and Groups

Species-level change—We calculated the change in abundance of source habitats from early European to current periods for each of the 91 broad-scale species of focus. Change in source habitats was evaluated by using a combination of species range maps (Marcot and others, in prep.), historical and current broad-scale vegetation maps (Hann and others 1997), and the

Table 5—Membership of 91 broad-scale species of focus in 40 groups and their associated residency and season of habitat function

Class[a]	Group	Common name	Common name code	Season evaluated
B	1	White-headed woodpecker	WHWDPECK	Resident year-long
B	1	White-breasted nuthatch	WBNUTHAT	Resident year-long
B	1	Pygmy nuthatch	PNUTHAT	Resident year-long
B	2	Lewis' woodpecker (migrant population)	LWDPCKMI	Migrant breeding
M	3	Western gray squirrel	WESQUIRR	Resident year-long
B	4	Blue grouse (winter)	BLGRSEWI	Resident winter
B	5	Northern goshawk (summer)	GOSHKSU	Resident summer
B	5	Flammulated owl	FLAMMOWL	Migrant breeding
M	5	American marten	MARTEN	Resident year-long
M	5	Fisher	FISHER	Resident year-long
B	6	Vaux's swift	VSWIFT	Migrant breeding
B	6	Williamson's sapsucker	WSAPSUC	Migrant breeding
B	6	Pileated woodpecker	PWDPECK	Resident year-long
B	6	Hammond's flycatcher	HFLYCAT	Migrant breeding
B	6	Chestnut-backed chickadee	CBCHICKD	Resident year-long
B	6	Brown creeper	BCREEPER	Migrant breeding
B	6	Winter wren	WWREN	Resident summer
B	6	Golden-crowned kinglet	GCKINGLT	Resident summer
B	6	Varied thrush	VTHRUSH	Resident summer
M	6	Silver-haired bat	SILVBAT	Resident summer
M	6	Hoary bat	HOARYBAT	Resident summer
B	7	Boreal owl	BOREOWL	Resident year-long
B	8	Great gray owl	GRGROWL	Resident year-long
B	9	Black-backed woodpecker	BBWDPECK	Resident year-long
B	10	Olive-sided flycatcher	OSFLYCAT	Migrant breeding
B	11	Three-toed woodpecker	TTWDPECK	Resident year-long
B	11	White-winged crossbill	WWCROSSB	Migrant winter
M	12	Woodland caribou	WCARIBOU	Resident year-long
M	13	Northern flying squirrel	NOSQUIR	Resident year-long
B	14	Hermit warbler	HEWARB	Migrant breeding
M	15	Pygmy shrew	PYGSHREW	Resident year-long
M	15	Wolverine	WOLVERIN	Resident year-long
M	16	Lynx	LYNX	Resident year-long
B	17	Blue grouse (summer)	BLGRSESU	Resident summer
B	17	Mountain quail (summer)	MTQUAIL	Resident summer
B	18	Lazuli bunting	LZBNTNG2	Migrant breeding
M	19	Gray wolf	GRAYWOLF	Resident year-long

Table 5—Membership of 91 broad-scale species of focus in 40 groups and their associated residency and season of habitat function (continued)

Class[a]	Group	Common name	Common name code	Season evaluated
M	19	Grizzly bear	GRBEAR	Resident year-long
M	20	Mountain goat	MTGOAT	Resident year-long
B	21	Long-eared owl	LEOWL	Resident year-long
M	22	California bighorn sheep	CBISHEEP	Resident year-long
M	22	Rocky Mountain bighorn sheep (summer)	RBISHEPSU	Resident summer
M	22	Rocky Mountain bighorn sheep (winter)	RBISHEPWI	Resident winter
B	23	Rufous hummingbird	RHUMBIRD	Migrant breeding
B	23	Broad-tailed hummingbird	BTHUMBRD	Migrant breeding
R	24	Sharptail snake	SHSNAKE	Resident year-long
R	24	California mountain kingsnake	CALSNAKE	Resident year-long
B	24	Black-chinned hummingbird	BCHUMBRD	Migrant breeding
B	25	Northern goshawk (winter)	GOSHKWI	Resident winter
M	26	Yuma myotis	YUMYOTIS	Resident year-long[b]
M	26	Long-eared myotis	LEMYOTIS	Resident year-long[b]
M	26	Fringed myotis	FRMYOTIS	Resident year-long[b]
M	26	Long-legged myotis	LLMYOTIS	Resident year-long
B	27	Pine siskin	PSISKIN	Migrant breeding
M	27	Townsend’s big-eared bat	PALEBAT	Resident year-long
M	28	Western small-footed myotis	WEMYOTIS	Resident year-long
M	28	Spotted bat	SPOBAT	Resident year-long[b]
M	28	Pallid bat	PALLBAT	Resident year-long[b]
B	29	Western bluebird	WBLUEBRD	Migrant breeding
B	30	Ash-throated flycatcher	ATFLYCAT	Migrant breeding
B	30	Bushtit	BSHTIT	Resident year-long
B	31	Ferruginous hawk	FERRHWK	Migrant breeding
B	31	Burrowing owl	BURROWL	Migrant breeding
B	31	Short-eared owl	SEOWL	Resident year-long
B	31	Vesper sparrow	VESPARRO	Migrant breeding
B	31	Lark sparrow	LASPARRO	Migrant breeding
B	31	Western meadowlark	WMEDLRK	Migrant breeding
M	31	Pronghorn	PRONGHOR	Resident year-long
R	32	Mojave black-collared lizard	MOLIZARD	Resident year-long
R	32	Longnose leopard lizard	LOLIZARD	Resident year-long
R	32	Striped whipsnake	STWSNAKE	Resident year-long
R	32	Longnose snake	LONSNAKE	Resident year-long
R	32	Ground snake	GROSNAKE	Resident year-long
M	32	Preble’s shrew	PRESHREW	Resident year-long

Table 5—Membership of 91 broad-scale species of focus in 40 groups and their associated residency and season of habitat function (continued)

Class[a]	Group	Common name	Common name code	Season evaluated
M	32	White-tailed antelope squirrel	WHSQUIR	Resident year-long
M	32	Washington ground squirrel	WGRSQUIR	Resident year-long
M	32	Wyoming ground squirrel	WYGRSQUI	Resident year-long
M	32	Uinta ground squirrel	UGRSQUIR	Resident year-long
B	33	Sage grouse (summer)	SGRSESU	Resident summer
B	33	Sage grouse (winter)	SGRSEWI	Resident winter
B	33	Sage thrasher	STHRASH	Migrant breeding
B	33	Brewer's sparrow	BRSPARRO	Migrant breeding
B	33	Sage sparrow	SASPARRO	Migrant breeding
B	33	Lark bunting	LRKBUNT	Migrant breeding
M	33	Pygmy rabbit	PYRABBIT	Resident year-long
M	33	Sagebrush vole	SAGEVOLE	Resident year-long
B	34	Black-throated sparrow	BTSPARRO	Migrant breeding
M	34	Kit fox	KITFOX	Resident year-long
B	35	Loggerhead shrike	LSHRIKE	Migrant breeding
B	36	Columbian sharp-tailed grouse (summer)	STGRSESU	Resident summer
B	37	Clay-colored sparrow	CCSPARRO	Migrant breeding
B	37	Grasshopper sparrow	GRSPARRO	Migrant breeding
M	37	Idaho ground squirrel	IDGRSQUI	Resident year-long
B	38	Black rosy finch	BRFINCH	Resident summer
B	38	Gray-crowned rosy finch	GCRFINCH	Resident summer
B	39	Lewis' woodpecker (resident population)	LWDPCKRE	Resident year-long
B	40	Brown-headed cowbird	BHCOWBRD	Migrant breeding

[a] B = bird, M = mammal, and R = reptile.

[b] It is not known whether these bat species hibernate within the basin or leave the basin during winter. In the absence of migratory information, we have assumed that source habitats for these species include winter hibernacula, in addition to nonwinter habitat.

species-source habitats information that we generated. The change in available source habitats from early European settlement to the present was estimated in a six-step process:

1. The inclusive area over which a species occurs currently was estimated by using range maps developed by Marcot and others (in prep.), as described earlier. If the current range of a species had contracted significantly from its historical range, we used its historical range (Marcot and others, in prep.). Range maps were digitized and translated into a grid map composed of 1-km^2 (0.4-mi^2) pixels, consistent with the vegetation grids prepared by Hann and others (1997).
2. Overlaying the species range grid maps and the current and historical vegetation grid maps (from Hann and others 1997), we then used the species-source habitats information (vol. 3, appendix 1, table 1) to identify individual pixels within the range of a species that were designated as source habitats, historically and currently.

3. For a given species and subwatershed, the percentage of area deemed to be source habitat was calculated as the number of pixels designated as source habitats divided by the total number of pixels in the subwatershed, multiplied by 100. For areas larger than a subwatershed (basin, ERU, subbasin, or watershed), the percentage of area (also referred to as aerial extent, abundance, or extent) deemed to be source habitat historically (H_S) or currently (C_S) for a species was calculated as the number of pixels in source habitat divided by the total number of pixels in the specified area, multiplied by 100, but excluding those subwatersheds that both historically and currently contained no pixels of source habitat.

It is important to note that at least one pixel of source habitat had to be present, either historically or currently, for a subwatershed to be included in our estimate of H_S and C_S at scales of the watershed, subbasin, ERU, or basin. For example, if one of three subwatersheds composing a watershed contained no pixels of source habitat, both historically and currently, this subwatershed was excluded from the calculation of percentage of area for both H_S and C_S for the species in the watershed. Exclusion of subwatersheds that contained no source habitats ensured that large areas of nonhabitat would not dilute the calculation of habitat change that was estimated to occur from historical to current periods for each species at scales larger than a subwatershed. In essence, this exclusion of subwatershed-sized areas of nonhabitat from our calculations is a fine-scale correction for situations where the range of a species was erroneously mapped to include such areas of nonhabitat, particularly along peripheries of a range map.

4. The absolute change in percentage of area of source habitats from historical to current periods, for a given species for a specified area larger than a subwatershed (ACH_S), was calculated as $ACH_S = C_S - H_S$.

5. The relative change in percentage of area of source habitats from historical to current periods, for a given species in a specified area larger than a subwatershed (RCH_S), was calculated as $RCH_S = [(C_S - H_S)/ (H_S)] \times 100$.

6. The values of RCH_S for each species were converted to ordinal measures of relative change in percentage of area of source habitats, referred to as trend categories (TC_S). Five trend categories were established: 2, 1, 0, -1, and -2, where 2 equals “strongly increasing,” corresponding to values of RCH_S greater than or equal to a 60-percent increase; 1 equals “increasing,” corresponding to values of RCH_S greater than or equal to a 20-percent but less than a 60-percent increase; 0 equals “no change,” corresponding to positive or negative values of RCH_S less than 20 percent; -1 equals “decreasing,” corresponding to values of RCH_S greater than or equal to a 20-percent but less than a 60-percent decline; and -2 equals “strongly decreasing,” corresponding to values of RCH_S greater than or equal to a 60-percent decline.

Values of TC_S for each species were calculated for the entire basin and for each ERU within the basin, considering all land ownership (both public and private lands). Results were displayed by species, with TC_S values ordered for each species from most negative to most positive changes at the basin and the ERU scales. Because some watersheds occurred in more than one ERU, we partitioned these watersheds among the appropriate ERUs. This resulted in 23 additional watershed/ERU combinations for our calculations of TC_S.

Change in source habitats at the scale of the basin also was analyzed for public and mixed-ownership lands only; this was done by excluding all subwatersheds from the analysis that were composed entirely of private lands. This analysis allowed us to contrast the amount of relative change, or RCH_S, that was attributed to public and mixed-ownership lands versus all lands for each species. This partitioning of the contribution of public and mixed-ownership lands, exclusive of private lands, to a change in source habitats is important to managers, who need insight about differences in habitat change on public-dominated ownership versus all lands.

Group-level change—We calculated change in source habitats for each of the 40 groups using the same general steps used for individual species, but with one important difference. At the 1-km^2 (0.4-mi^2) pixel level, the percentage of area deemed to be source habitats for the group historically (H_G) or currently (C_G), or "group score" historically or currently (G_S), was calculated as:

$$G_S = \frac{\sum_{i=1}^{k} s_i}{\sum_{i=1}^{k} r_i},$$

Where s_i indicates source habitats present, either historically or currently, for species i within the range of species i, r_i indicates whether the pixel is within the range of species i, and k is the number of species within the group. Both s_i and r_i are binary (0,1) variables; group scores range in value from zero to one. Calculated in this manner, group scores at the pixel level depend only on the species whose ranges include a given pixel. Thus for a group composed of 10 species, a pixel that contains source habitat for a single member species and is within the range of only that species would have the same score as a pixel within the range of all 10 species that supports all 10. For a specified area of the basin, group scores were calculated simply as the mean of the pixel-level scores over all pixels within the specified area. As was done with the species calculations, only those subwatersheds containing at least one pixel of source habitat, either historically or currently, were included in the calculations of group scores. Group-level measures of absolute change (ACH_G), relative change (RCH_G), and trend categories of change (TC_G) from historical to current were calculated in the same manner as done for species-level changes.

The translation of the numeric measure of relative change (RCH_S) to the ordinal trend categories (TC_S) for both species and groups was intended to provide a consistent means of displaying relative change among species and groups at various scales of the basin. It should not be interpreted as a measure of statistical significance. Unfortunately, the method used to estimate change in source habitats does not lend itself to precise estimates of error. The accuracy of any given estimate depends on the combination of how well we have characterized the species range, the historical and current distribution of vegetation, and the associations between species and vegetation. Analysis of the vegetation maps suggests that the accuracy of compositional predictions increases as the scale of aggregation increases (Hann and others 1997); that is, the estimated composition of the landscape at the ERU and larger scales is likely to be more accurate than individual summaries at smaller scales, as described earlier in our methods under "Estimating and Validating Occurrence of Cover Types and Structural Stages for Broad-Scale Assessment."

Increased accuracy of vegetation estimates at ERU and basin scales versus smaller scales implies that our estimates of change in source habitats for individual species and groups are more likely to be accurate at larger scales as well. We also expect the accuracy of our predictions to be species-dependent. In general, estimates for species with broad ranges that use many source habitats are likely to be more accurate than estimates for narrowly distributed species that use few source habitats.

Forming Families of Groups to Summarize Results Among Multiple Groups

Families of groups—To complete our hierarchical system of evaluating species, groups, and families, we further generalized our group-level results by placing 37 of the 40 groups into 12 families (fig. 5, table 6). Families were defined by using the generalized vegetative themes shown in figure 5, based on a combination of formal cluster analysis (Proc Clus, SAS Inc. 1989a, p. 519-614) and empirical knowledge of the habitat requirements of each species. The clustering method used to guide placement of groups into families was identical to that used to join species into groups (see methods, "Clustering the Species into Groups"), with one exception: instead of clustering species based on similarities in cover-type structural stage combinations that explicitly define source habitats, clustering was done on similarities of species in the 24 terrestrial community types developed by Hann and others (1997).

The 24 terrestrial community types are a higher level generalization of the cover types and structural stages and provide a hierarchy within which all cover type-structural stage combinations are nested. (See Hann and others (1997) for a detailed description of the hierarchical system of nesting cover type-structural

Figure 5—Flow diagram used to place 37 groups of broad-scale species of focus into 12 families.

stage combinations within terrestrial community types and groups as the foundation for the broad-scale system of vegetation classification that was developed for the basin.) Use of the terrestrial community types for clustering allowed us to look for more generalized patterns of similarity among species habitat requirements, commensurate with our desire to generalize species and groups into the smallest number of families that could be meaningfully used by managers and biologists at the broadest scales of ecosystem management.

Thus, each family represents a collection of groups that share general similarities in source habitats, with the similarities arranged along major vegetative themes that are conventionally addressed by managers (fig. 5, table 6). For example, families one and two are composed of groups whose source habitats consist of forested environments of predominantly old-forest structural stages. By contrast, family three contains groups whose source habitats consist of forested environments that include several structural stages, whereas family four contains only one group whose source

Table 6—Membership of 37 groups and 88 broad-scale species of focus in 12 families

Family	Group	Common name	Terrestrial family name
1	1	White-headed woodpecker	Low-elevation old forest
1	1	White-breasted nuthatch	
1	1	Pygmy nuthatch	
1	2	Lewis' woodpecker (migrant population)	
1	3	Western gray squirrel	
2	4	Blue grouse (winter)	Broad-elevation old forest
2	5	Northern goshawk (summer)	
2	5	Flammulated owl	
2	5	American marten	
2	5	Fisher	
2	6	Vaux's swift	
2	6	Williamson's sapsucker	
2	6	Pileated woodpecker	
2	6	Hammond's flycatcher	
2	6	Chestnut-backed chickadee	
2	6	Brown creeper	
2	6	Winter wren	
2	6	Golden-crowned kinglet	
2	6	Varied thrush	
2	6	Silver-haired bat	
2	6	Hoary bat	
2	7	Boreal owl	
2	8	Great gray owl	
2	9	Black-backed woodpecker	
2	10	Olive-sided flycatcher	
2	11	Three-toed woodpecker	
2	11	White-winged crossbill	
2	12	Woodland caribou	
2	13	Northern flying squirrel	
3	14	Hermit warbler	Forest mosaic
3	15	Pygmy shrew	
3	15	Wolverine	
3	16	Lynx	
3	17	Blue grouse (summer)	
3	17	Mountain quail (summer)	
4	18	Lazuli bunting	Early-seral montane and lower montane
5	19	Gray wolf	Forest and range mosaic
5	19	Grizzly bear	
5	20	Mountain goat	
5	21	Long-eared owl	
5	22	California bighorn sheep	
5	22	Rocky Mountain bighorn sheep (summer)	
5	22	Rocky Mountain bighorn sheep (winter)	
6	23	Rufous hummingbird	Forests, woodlands, and montane shrubs
6	23	Broad-tailed hummingbird	
6	24	Sharptail snake	
6	24	California mountain kingsnake	

Table 6—Membership of 37 groups and 88 broad-scale species of focus in 12 families (continued)

Family	Group	Common name	Terrestrial family name
6	24	Black-chinned hummingbird	
6	25	Northern goshawk (winter)	
7	26	Yuma myotis	Forests, woodlands, and sagebrush
7	26	Long-eared myotis	
7	26	Fringed myotis	
7	26	Long-legged myotis	
7	27	Pine siskin	
7	27	Townsend's big-eared bat	
7	28	Western small-footed myotis	
7	28	Spotted bat	
7	28	Pallid bat	
8	29	Western bluebird	Rangeland and early- and late-seral forest
9	30	Ash-throated flycatcher	Woodland
9	30	Bushtit	
10	31	Ferruginous hawk	Range mosaic
10	31	Burrowing owl	
10	31	Short-eared owl	
10	31	Vesper sparrow	
10	31	Lark sparrow	
10	31	Western meadowlark	
10	31	Pronghorn	
10	32	Mojave black-collared lizard	
10	32	Longnose leopard lizard	
10	32	Striped whipsnake	
10	32	Longnose snake	
10	32	Ground snake	
10	32	Preble's shrew	
10	32	White-tailed antelope squirrel	
10	32	Washington ground squirrel	
10	32	Wyoming ground squirrel	
10	32	Uinta ground squirrel	
11	33	Sage grouse (summer)	Sagebrush
11	33	Sage grouse (winter)	
11	33	Sage thrasher	
11	33	Brewer's sparrow	
11	33	Sage sparrow	
11	33	Lark bunting	
11	33	Pygmy rabbit	
11	33	Sagebrush vole	
11	34	Black-throated sparrow	
11	34	Kit fox	
11	35	Loggerhead shrike	
12	36	Columbian sharp-tailed grouse (summer)	Grassland and open-canopy sagebrush
12	37	Clay-colored sparrow	
12	37	Grasshopper sparrow	
12	37	Idaho ground squirrel	

habitats are restricted to forests composed of early-seral stages. Additional contrast is illustrated by families five through eight; these families consist of groups whose source habitats include both forest and rangeland environments. Moreover, families 9 through 12 consist of groups whose source habitats include only rangeland-woodland environments.

Note that two groups (group 38, composed of two species of rosy finches, and group 39, composed of the resident Lewis' woodpecker) were not placed in any of the families because their source habitats were restricted to small areas of the basin and were potentially under-sampled because of the finer scale pattern at which their habitats exist. Moreover, group 40, which consists of one species, the brown-headed cowbird, also was excluded from the families because of its unique dependence on agricultural and livestock-dominated environments, and because change in its source habitats was already analyzed and shown clearly in the analysis at the group level.

Evaluating change in source habitats by family— For each of the 12 families, we summarized the change in percentage of area of source habitats from historical to current periods for each ERU using the following process. First, each watershed was assigned to one of three change classes: positive, negative, or neutral. Change classes were based on summary statistics calculated from the five trend categories of relative change for each group (TC_G) in the family. For a given family, a watershed was classified as positive if >50 percent of the groups in the watershed increased in source habitats by 20 percent or more (TC_G of 1 or 2). A watershed was classified as negative if >50 percent of the groups in the watershed declined in source habitats by 20 percent or more (TC_G of -1 or -2). Watersheds not classified positive or negative were classified as neutral. Estimates of the dominant trend in source habitats were then derived for each family for each of the 13 ERUs by (1) calculating the percentage of watersheds that were increasing, decreasing, or neutral for each family in each ERU; (2) classifying the ERU as increasing or decreasing if >50 percent of the watersheds had positive or negative trends, respectively; and (3) classifying the ERU as neutral if not classified as either increasing or decreasing.

Correlating Change in Source Habitats Between Species Within Groups and Families

Clustering of species into groups and families could result in contradictory changes in source habitats among species within a group or family. This is possible because every species except two—the black rosy finch and the gray-crowned rosy finch—is associated with a unique set of source habitats; that is, the set of source habitats for each species is different from all other species (vol. 3, appendix 1, table 1). Thus, for a given analysis area, particular source habitats that are unique to one species in a group or family could change markedly and in a different direction than another set of source habitats that are unique to one or more other species in the same group or family.

To determine if this problem existed, we calculated a nonparametric correlation coefficient, Kendalls' Tau (r_k) (Proc Corr, SAS Inc. 1989b, p. 209-235) of the relative change (RCH_S) in source habitats between each pair of species within each group and family (within-group or within-family coefficients), and among all species pairings between groups and families (between-group or between-family coefficients). Correlation coefficients were calculated on changes in source habitats that were measured at the scale of the watershed, by using all watersheds under joint occupation of each species pair. A positive coefficient (r_k values >0 and $\leq$1) for a given pair of species indicated positive agreement in direction of change in source habitats across watersheds for the pair. Values near one indicated strong positive agreement, whereas values near but above zero indicated weak positive agreement. Zero or negative coefficients (r_k values of 0 or $\leq$-1) indicated no relation or contradictory trends in source habitats between a species pair.

We interpreted positive correlation coefficients among all species pairings within a group or family as verification that the direction of change in source habitats calculated for the group or family reflected a like direction of change for all species within the group or family. Zero or negative coefficients between pairs of species within a group or family indicated that calculations of group- or family-level change might be suspect because of contradictory trends in source habitats among one or more species pairings. In the latter case, our intention was to redefine group or family membership to alleviate contradictory trends among one or more species.

To further interpret the efficacy of a group or family trend as an index of species trends within the group or family, we compared the within-group or within-family coefficients for each group or family with the mean correlation coefficient calculated for all between-group and between-family coefficients. Presumably, correlation coefficients of trend for within-group or within-family species pairings should be higher than correlation coefficients calculated for species pairings between groups or between families. If the opposite was observed, it suggested that species membership within certain groups or families could be changed to achieve a higher level of agreement in habitat trends between two or more species.

Summarizing Knowledge About Species-Road Relations

Many species of vertebrates are negatively affected by roads and the human activities associated with roads (for example, see Bailey and others 1986, Bashore and others 1985, Cole and others 1997, Fraser 1979, Hodgman and others 1994, Mattson and others 1996b, Mech and others 1988, Scott and Servheen 1985, Singer 1978, Thiel 1985). Moreover, human presence and activities are facilitated by increased access provided by roads (Hann and others 1997). Consequently, we summarized knowledge about species-road relations for each of the 91 broad-scale species of focus using the following steps. First, we conducted a literature search, and from that, identified 13 factors that consistently are associated with the nagative impact of roads on populations or habitats of terrestrial vertebrates. We then characterized the potential effects of each factor on each species of focus in one of four ways: (1) a documented effect of the factor, with explicit association of roads as a facilitator of the effect, that was demonstrated in one or more studies on the species; (2) a documented effect of the factor, but without explicit association of roads as a facilitator of the effect, that was demonstrated in one or more studies on the species; and (3) a presumed effect of the factor, based on documented effects of the factor and of roads as a facilitator of the effect, that was demonstrated in one or more studies on species of similar life history or taxa; (4) a presumed effect of the factor, based on documented effects of the factor and of roads as a facilitator of the effect, in causing declines in habitat condition on which the species depends.

To provide spatial context for road-associated effects on terrestrial vertebrates, we portrayed the broad-scale pattern of road density across the basin using a pixel-based prediction of six classes of road density that was derived originally by Menakis and others (1996) and discussed in Hann and others (1997). We then identified and discussed potential management actions that could mitigate some or all of the negative effects associated with the spatial pattern of roading. The six classes of road density predicted by Menakis and others (1996) are (1) zero (0 to 0.02 mi of road per mi^2) (0 to 0.01 km per km^2); very low (>0.02 to 0.1 mi per mi^2) (0.01 to 0.06 km per km^2); low (>0.1 to 0.7 mi per mi^2) (>0.06 to 0.44 km per km^2); moderate (>0.7 to 1.7 mi per mi^2) (>0.44 to 1.06 km per km^2); high (>1.7 to 4.7 mi per mi^2) (>1.06 to 2.94 km per km^2); and very high (>4.7 mi per mi^2) (>2.94 km per km^2). Methods used to predict these spatially explicit road classes are described in the following section.

Characterizing road density—A data set composed of continuous, mapped coverage of roads was not available for the basin. Consequently, a geographical information system (GIS) layer of predicted road density was developed at 1-km^2 (0.4-mi^2) resolution with a statistical rule set (Menakis and others 1996). This layer was summarized to the subwatershed level by using an average based on the six classes of road density identified above. The rule set for extrapolation of road density classes to create the broad-scale road density map was developed from a statistical correlation calculated between road density estimated from a sample of 1-km^2 (0.4-mi^2) cells and estimates of other variables that were available in continuous coverage of all 1-km^2 (0.4-mi^2) cells across the basin. The data set for sampled road density came from roads sampled as part of the mid-scale landscape characterization (Gravenmier and others 1997, Hessburg and others 1999, Ottmar and others 1996) and valley bottom characterization (Gravenmier and others 1997, Jensen and others 1997). Menakis and others (1996), Gravenmier and others (1997), and Hann and others (1997) described additional details about methods used to predict the classes of road density at the broad scale and limitations on use of the data.

Mapping Road Density in Relation to Abundance of Source Habitats for Selected Species

Roads hypothetically pose a direct threat to population fitness for several terrestrial carnivores by facilitating overtrapping (wolverine and lynx) or other fatal interactions with humans (gray wolf and grizzly bear). For gray wolf and grizzly bear, researchers have verified a strong, negative relation between road density and population fitness (Mace and others 1996, Mattson and others 1996b, Mech and others 1988, Thiel 1985). Similar relations have been hypothesized for wolverine and lynx within the basin (ICBEMP 1996b, 1996c), and limited research on lynx (Bailey and others 1986) outside the basin supports the hypothesis that population fitness is lower in areas characterized by increased road access (but see Ruggiero and others [1999] regarding alternative hypotheses). Because of these observed or suspected effects on population fitness, we mapped the current abundance (percentage of area or C_S, as defined earlier) of source habitats in relation to road density for each of the four species mentioned above. Mapping was intended to identify large areas of abundant source habitats that have low road density. Presumably, these areas would have highest potential to support populations that could persist without additive mortality that may be caused by road-associated factors.

Mapping involved three steps: (1) generating a map of current habitat abundance for each species at the appropriate scale; (2) generating a map of road density at the same scale as the map of habitat abundance; and (3) generating a map of the intersection of moderate to high habitat abundance with zero to low road density. Each of these maps was generated at the subbasin scale. Subbasins were used as mapping units because their large size (mean size of 345 000 ha [850,000 acres] each) is compatible with the broad scale at which lynx, wolf, wolverine, and grizzly bear function to meet their life requirements.

Generating the map of current habitat abundance for each species involved two steps. First, we calculated the current percentage of area (C_S) in each subbasin that was composed of source habitats. And second, we classified and mapped each subbasin as belonging to one of three classes—high, moderate, or low—with the highest one-third of values classified as high abundance, the middle one-third as moderate abundance, and the lowest one-third of values as lowest abundance. Maps of current abundance of source habitats were generated over the entire area estimated to be within the historical range of each species within the basin. Abundance of source habitats was mapped within historical ranges because we wanted to identify all areas of the basin that might be characterized as having moderate to high abundance of source habitats and zero to low density of roads within potential use areas for each species.

Generating the map of road density by subbasin involved four steps. First, we calculated the percentage of area in each watershed within each subbasin that had (1) zero to low road density ($\leq$0.7 mi of roads per mi^2); (2) moderate road density (0.7 to 1.7 mi of roads per mi^2); (3) high to very high road density (>1.7 mi of roads per mi^2). Second, we used these percentages to identify which of these three composite classes of road density—zero to low, moderate, or high to very high—dominated the watershed. If >50 percent of the area of the watershed was composed of one of the three composite classes of road density, that class was identified as dominant. In cases where none of the three classes made up >50 percent of the watershed, the moderate class of road density was identified as dominant. Third, we calculated the percentage of watersheds within the subbasin that had a dominant road class of zero to low, moderate, and high to very high. And fourth, we classified the subbasin as being dominated by zero to low or high to very high road density if >50 percent of the watersheds within the subbasin were dominated by these classes.

To generate the map of the intersection of habitat abundance with zero to low road density for each species by subbasin, we overlaid and then outlined the subbasins dominated by zero to low road density onto the map of habitat abundance for each species. These integrated maps were displayed for each species of terrestrial carnivore and results discussed in terms of current knowledge of the effects of roads on the habitats and populations of the species.

Interpreting Results and Describing Management Implications

Species-level interpretation and implications— Our purpose for assessment was to adopt a "systems approach" for evaluating change in source habitats for an inclusive list of terrestrial vertebrates whose habitats were suspected to have declined. We therefore focused our management implications on groups of species, and families of groups, rather than individual species. Laws such as ESA and NFMA, however, dictate that species-level needs be attended to and accounted for, regardless of the inherent problems in doing so (Hunter 1990, 1991). Moreover, if species are to be evaluated as groups, the loss of species-level accuracy must be evaluated and accounted for in making appropriate inferences for management.

For these reasons, we analyzed change in source habitats at the species level and addressed the associated management implications. Our implications focused on two subject areas: (1) identifying unique, species-level habitat requirements and habitat conditions that may be obscured by analyzing species as groups; and (2) identifying those species whose habitats have potentially declined so substantially that special management attention may be warranted.

Group-level interpretations and implications— Ecosystem management demands that robust patterns that potentially exist among multiple species be detected and accounted for, and that broad generalizations about groups of species be made without significant loss of species-level information. Accordingly, we focused our analysis, and subsequent interpretations and implications of the results, on groups rather than species. Interpretations of results at the group level were designed to (1) identify the underlying changes in cover types and structural stages that contributed to any changes observed in source habitats; (2) consider the potential effects of special habitat features not measured in our analysis, such as trends in snag densities or changes in other finer scale or nonvegetative characteristics, that may act in tandem with or independent of group-level changes in source habitats; and (3) consider the potential effects of nonvegetative factors not measured in our analysis that also may act in tandem with or independent of changes in source habitats to influence population status and trend for the broad-scale species of focus.

We did not attempt to discern the potential relation between group-level changes in source habitats and empirical trends in populations of the species within the groups. Evaluation of the change in source habitats for a group in relation to the empirical trends in populations of those species is problematic for at least four reasons. First, the spatial scale at which changes in source habitats were measured (collections of watersheds within each ERU) was not the same as that at which population data were collected. For example, population trend data often are collected by state agencies, and state boundaries do not coincide with watershed or ERU boundaries. Second, the temporal scale at which changes in source habitats are measured is far longer (>100 yr) than even the longest term data on population trends. For example, Breeding Bird Surveys (BBS) date as far back as the early 1960s, yet most or all of the large-scale changes in source habitats, such as conversion of rangelands to agriculture, may have occurred before then. Third, populations of some species may respond strongly to nonvegetative factors, such as human presence or human activities, which are not accounted for in source habitat trends. For example, the grizzly bear apparently survives well in various habitats that are characterized by little or no human disturbance but survives less well in the same habitats where human presence is high (Mattson and others 1996a, 1996b). And fourth, population trends of many species are difficult to detect without intensive monitoring, which typically has not occurred for most nongame species. Sauer and others (1996b) discuss some of these and additional problems related to analyzing and interpreting BBS data in relation to causal factors such as habitat change.

Because of these limitations, our primary basis for describing management implications focused on interpretation of changes observed in source habitats, combined with summaries of empirical literature available on conditions of special habitat features for each group. Population data that indicated widespread, negative trends or other problems with population status, however, also were considered as part of our description of management implications, regardless of how well such population data agreed with habitat trends. And, whenever possible, we attempted to identify other factors or reasons for apparent disparities between population and source habitat trends when logical or empirical explanations were evident. Accordingly, the management implications described for each group were designed to (1) identify habitat and population issues of most interest to Federal land managers in the basin; (2) list broad-scale management strategies that would be effective in addressing the issues; and (3) outline a comprehensive set of practices that would most effectively support implementation of the strategies.

When reporting population trends, we reported as much statistical detail about the trends, and the magnitude of change, as reported by the source literature. For trends obtained from results of BBS (Sauer and others 1996a), we reported the magnitude of change (percentage of change), the statistical probability of detecting a larger difference than that observed, and the sample size. We also reported BBS summaries of trends for the basin and for each of three major physiographic regions that overlap major segments of the basin (Saab and Rich 1997, Sauer and others 1996a).

Family-level interpretations and implications— Our purpose for placing groups of species into families was to further generalize the patterns of change in source habitats across subbasins and ERUs in as concise a format as possible without loss of detail. Moreover, we wanted to maintain explicit connections of families to groups, and groups to species, in making such generalizations. In this way, the more detailed group- and species-level results could be related directly and efficiently to family-level generalizations, thereby allowing managers to design and apply conservation strategies and practices at any or all of the three levels of resolution (species, groups, or families).

Thus, we drew implications about family-level results in terms of broad-scale themes of habitat change that supported species- and group-level trends. Themes described major, broad-scale changes in source habitats along major vegetative gradients that may be useful to managers, and on which strategic conservation designs can be based. Specifically, we interpreted and drew implications about family-level results to answer the following questions:

1. What source habitats have undergone the greatest decline from historical to current conditions, and which groups were associated with such declines?

2. What areas of the basin have undergone the greatest decline in source habitats, and what are the spatially explicit causes for decline?

3. What broad-scale management strategies and practices and associated ecological processes would bring about the greatest short- and long-term benefits to conservation or restoration of source habitats that have undergone long-term decline, and which species and groups of species would benefit from which strategies, practices, and ecological processes?

Answering these questions provides spatially explicit management insight about habitat status for collections of groups of species. Moreover, the answers presumably will help managers focus on broad-scale management strategies and practices that most benefit groups of species whose source habitats have undergone the greatest decline.

Validating Agreement Between Change in Source Habitats and Expert-Opinion Based Habitat Outcomes

We assume that the direction of change in source habitats reflects a like direction of trend in the associated population size of the broad-scale species of focus. Note that this is different from assuming that the magnitude of change in source habitats reflects a like magnitude of change in population size, because many factors beyond habitat can influence population trends. For all species analyzed here, however, except those for which concern is based solely on effects of nonvegetative factors such as roads, the assumption that a decline or increase in source habitats contributes to a like direction of change in population size is fundamental to development of credible management strategies and practices. If this assumption is incorrect, then management applications of our results could be misleading. This assumption can be addressed through validation research. We assume that the FS and BLM will fund broad-scale, long-term research to address the relation between our results on habitat trends and empirical estimates of population status and trend for each species analyzed in our paper.

Although broad-scale data on population status and trend have either not been synthesized or not collected at temporal and spatial scales compatible with our analysis, one set of data exists by which to assess agreement between presumed changes in habitat and populations with changes that we estimated for source habitats. Lehmkuhl and others (1997) provided expert-opinion based estimates of historical to current change in habitat amount and distribution (habitat outcomes) for 173 species of terrestrial vertebrates on FS- and BLM-administered lands within the basin. They also provided expert-opinion based estimates of historical to current change in habitat outcomes and presumed population effects based on the cumulative effects of habitat change and nonhabitat factors on all lands within the basin (cumulative effects outcomes). Estimates of change in habitat and cumulative effects outcomes were generated from a series of expert panels convened in spring 1996. Sixty-eight of these 173 species are on our list of broad-scale species of focus.

For each of these species, we characterized the change in habitat outcomes and in cumulative effects outcomes from historical to current periods from Lehmkuhl and others (1997) as being either positive or negative, and did the same for the change in source habitats at the basin scale. We then calculated the percentage of species whose change in source habitats agreed or disagreed with trends in the habitat outcomes, and with trends in the cumulative effects outcomes. Habitat and cumulative effects outcomes were estimated specifically for each of the two EIS areas (Eastside and Upper Columbia River; USDA Forest Service and USDI Bureau of Land Management 1997a, 1997b). Consequently, we calculated percentage of agreement among trends in source habitats and outcomes for both EIS areas and for a mean trend in outcomes that we calculated by pooling results from both EIS areas.

Species-Level Results and Discussion

Habitat Change by Basin and Ecological Reporting Unit

Basin-wide change—Source habitats for most species—55 of 97 species seasonal entries or 57 percent—declined strongly or very strongly from historical to current periods, based on trend categories of relative change (TC_S) at the basin scale (rank of -1 or -2, table 7). By contrast, few species (6 percent) were associated with source habitats that increased strongly or very strongly (rank of 1 or 2), but a moderate number—36 of 97 species seasonal entries or 37 percent—were associated with source habitats that showed little change (rank of 0).

In contrast to the trends based on categories of relative change, trends in source habitats were consistently more negative when expressed as continuous variables of absolute and relative change (ACH_S and RCH_S). By using these measures, 80 percent of the species were associated with a change in source habitats that was negative (table 7). Only two species (2 percent) showed no change in source habitats, and 18 percent were associated with change that was positive.

Species whose source habitats declined were associated with many forested and rangeland environments. For example, of the 20 species that underwent the strongest relative decline in source habitats (table 7), 12 are primarily dependent on forested habitats, 7 are largely dependent on rangeland habitats, and 1 is dependent on a combination of forested and rangeland habitats (vol. 3, appendix 1, table 1). This finding indicates that many source habitats have declined in the basin; in turn, this suggests that no particular species or habitats, or small set of species or habitats, are easily identified as needing priority management.

Habitat change by ecological reporting unit— Species whose source habitats declined strongly or very strongly at the basin scale (trend categories of relative change of -1 or -2, table 7) also experienced strong declines in source habitats within most ERUs (table 8; vol. 3, appendix 1, table 5). For example, the migrant population of Lewis' woodpecker, which showed the greatest relative decline in source habitats among all species at the basin scale (-83 percent, table 7), also had categories of relative change that were -1 or -2 for 100 percent of the ERUs in which the species occurred (table 8). Similarly, the grasshopper sparrow, which had the third greatest relative decline among all species in the basin (-71 percent, table 7), had categories of relative change that were -1 or -2 for 91 percent of the ERUs in which the species occurred (table 8). Other species whose source habitats underwent strong relative decline at the basin level and across most or all ERUs included the Washington ground squirrel, Columbian sharp-tailed grouse, Rocky Mountain bighorn sheep, pygmy nuthatch, flammulated owl, Williamson's sapsucker, western bluebird, white-headed woodpecker, and brown creeper. Source habitats for these species declined by more than 40 percent at the basin scale (table 7), and categories of relative change were either -1 or -2 in more than 75 percent of the ERUs in which these species occurred (table 8).

Table 7—Historical (H_S) and current (C_S) estimates of areal extent (percentage of area) of source habitats at the scale of the basin for 91 broad-scale species of focus, and resulting changes in source habitats based on three measures: absolute change (ACH_S), relative change (RCH_S), and trend categories (TC_S) of relative change[a] [b]

Family	Group	Common name	Historical estimate	Current estimate	Absolute change	Relative change	Trend category[c]
			Percentage				
1	2	Lewis' woodpecker (migrant population)	13.78	2.29	-11.49	-83.35	-2
12	37	Idaho ground squirrel	11.32	3.04	-8.28	-73.13	-2
12	37	Grasshopper sparrow	21.27	6.18	-15.09	-70.94	-2
10	32	Washington ground squirrel	71.66	22.38	-49.28	-68.77	-2
1	1	Pygmy nuthatch	20.42	6.59	-13.83	-67.73	-2
12	37	Clay-colored sparrow	18.60	6.39	-12.21	-65.65	-2
1	1	White-headed woodpecker	22.87	8.50	-14.37	-62.83	-2
2	7	Boreal owl	14.97	5.78	-9.20	-61.42	-2
2	6	Williamson's sapsucker	20.97	9.14	-11.83	-56.42	-1
2	5	Flammulated owl	22.85	10.11	-12.74	-55.76	-1
11	33	Lark bunting	54.45	24.84	-29.60	-54.37	-1
2	6	Brown creeper	22.36	11.09	-11.27	-50.40	-1
5	22	Rocky Mountain bighorn sheep (winter)	32.95	16.65	-16.29	-49.46	-1
8	29	Western bluebird	51.29	26.39	-24.90	-48.55	-1
2	6	Chestnut-backed chickadee	13.43	7.13	-6.30	-46.89	-1
2	11	White-winged crossbill	8.44	4.52	-3.92	-46.41	-1
2	6	Silver-haired bat	22.11	12.01	-10.10	-45.67	-1
12	36	Columbian sharp-tailed grouse (summer)	58.80	32.35	-26.44	-44.97	-1
2	5	Northern goshawk (summer)	22.75	12.93	-9.82	-43.16	-1
2	6	Hammond's flycatcher	22.11	12.91	-9.20	-41.59	-1
5	22	Rocky Mountain bighorn sheep (summer)	36.54	21.66	-14.88	-40.72	-1
2	5	American marten	18.82	11.54	-7.28	-38.67	-1
10	31	Short-eared owl	58.16	35.95	-22.21	-38.18	-1
10	31	Vesper sparrow	48.93	30.25	-18.68	-38.17	-1
10	32	Uinta ground squirrel	67.19	42.78	-24.41	-36.33	-1
2	4	Blue grouse (winter)	21.30	13.68	-7.62	-35.79	-1
10	31	Western meadowlark	54.80	35.23	-19.57	-35.71	-1
10	31	Lark sparrow	53.17	34.40	-18.76	-35.29	-1
2	6	Hoary bat	30.04	19.77	-10.27	-34.18	-1
2	9	Black-backed woodpecker	23.05	15.29	-7.77	-33.70	-1
10	31	Burrowing owl	72.68	48.89	-23.79	-32.73	-1
10	32	Preble's shrew	56.60	38.18	-18.42	-32.54	-1
6	25	Northern goshawk (winter)	21.37	14.59	-6.78	-31.73	-1
10	31	Ferruginous hawk	77.94	53.90	-24.04	-30.85	-1
11	33	Sage thrasher	60.90	43.56	-17.34	-28.47	-1
11	33	Brewer's sparrow	56.70	41.23	-15.47	-27.29	-1
11	33	Sage grouse (winter)	60.48	44.07	-16.41	-27.14	-1
7	28	Pallid bat	60.23	43.90	-16.33	-27.11	-1
11	33	Sage grouse (summer)	59.58	43.56	-16.02	-26.89	-1
11	33	Sagebrush vole	61.38	45.04	-16.35	-26.63	-1
11	33	Sage sparrow	77.61	57.09	-20.52	-26.45	-1

Table 7—Historical (H_S) and current (C_S) estimates of areal extent (percentage of area) of source habitats at the scale of the basin for 91 broad-scale species of focus, and resulting changes in source habitats based on three measures: absolute change (ACH_S), relative change (RCH_S), and trend categories (TC_S) of relative change[a] [b] (continued)

Family	Group	Common name	Historical estimate	Current estimate	Absolute change	Relative change	Trend category[c]
			Percentage				
10	31	Pronghorn	73.71	54.54	-19.18	-26.02	-1
1	1	White-breasted nuthatch	18.56	13.86	-4.69	-25.30	-1
5	22	California bighorn sheep	63.41	47.91	-15.50	-24.45	-1
4	18	Lazuli bunting	12.47	9.52	-2.95	-23.63	-1
2	6	Winter wren	7.86	6.01	-1.86	-23.62	-1
7	28	Western small-footed myotis	49.21	37.68	-11.53	-23.42	-1
11	33	Pygmy rabbit	63.54	48.68	-14.86	-23.38	-1
2	6	Varied thrush	11.24	8.67	-2.57	-22.86	-1
6	23	Rufous hummingbird	30.93	23.97	-6.96	-22.51	-1
2	13	Northern flying squirrel	32.26	25.26	-7.00	-21.70	-1
10	32	Ground snake	46.46	36.55	-9.91	-21.33	-1
2	6	Pileated woodpecker	10.62	8.40	-2.22	-20.88	-1
10	32	Striped whipsnake	80.20	63.68	-16.53	-20.61	-1
11	34	Black-throated sparrow	73.07	58.11	-14.96	-20.47	-1
11	35	Loggerhead shrike	47.82	38.45	-9.37	-19.60	0
2	5	Fisher	11.65	9.38	-2.27	-19.51	0
2	6	Golden-crowned kinglet	13.38	10.85	-2.54	-18.96	0
10	32	Longnose leopard lizard	74.35	60.66	-13.70	-18.42	0
7	28	Spotted bat	61.57	50.79	-10.79	-17.52	0
5	19	Grizzly bear	81.27	67.63	-13.64	-16.78	0
10	32	Wyoming ground squirrel	68.41	56.93	-11.48	-16.78	0
5	21	Long-eared owl	50.98	42.46	-8.52	-16.71	0
5	19	Gray wolf	83.82	70.71	-13.12	-15.65	0
6	23	Broad-tailed hummingbird	16.82	14.83	-1.99	-11.86	0
3	17	Blue grouse (summer)	30.41	26.94	-3.47	-11.42	0
7	26	Long-eared myotis	77.85	69.97	-7.87	-10.12	0
11	34	Kit fox	49.69	45.13	-4.56	-9.17	0
2	12	Woodland caribou	4.03	3.68	-0.36	-8.86	0
7	27	Townsend’s big-eared bat	55.71	51.21	-4.50	-8.08	0
2	6	Vaux’s swift	9.53	8.77	-0.76	-7.99	0
7	26	Yuma myotis	68.94	64.30	-4.64	-6.73	0
2	8	Great gray owl	26.53	24.94	-1.59	-5.99	0
10	32	Longnose snake	57.78	55.74	-2.04	-3.54	0
10	32	Mojave black-collared lizard	69.32	67.15	-2.17	-3.14	0
1	3	Western gray squirrel	22.43	22.03	-0.41	-1.81	0
NA	38	Gray-crowned rosy finch	8.34	8.34	-0.01	-0.09	0
10	32	White-tailed antelope squirrel	79.74	79.68	-0.05	-0.07	0
NA	38	Black rosy finch	10.87	10.87	0.00	0.00	0
NA	39	Lewis’ woodpecker (resident population)	10.25	10.25	0.00	0.00	0
6	24	California mountain kingsnake	32.50	34.92	2.42	7.44	0

Table 7—Historical (H_S) and current (C_S) estimates of areal extent (percentage of area) of source habitats at the scale of the basin for 91 broad-scale species of focus, and resulting changes in source habitats based on three measures: absolute change (ACH_S), relative change (RCH_S), and trend categories (TC_S) of relative change[a] [b] (continued)

Family	Group	Common name	Historical estimate	Current estimate	Absolute change	Relative change	Trend category[c]
			Percentage				
5	20	Mountain goat	43.25	47.50	4.24	9.81	0
3	15	Pygmy shrew	68.11	76.68	8.56	12.57	0
6	24	Black-chinned hummingbird	20.20	23.10	2.90	14.37	0
3	15	Wolverine	32.83	37.57	4.73	14.41	0
3	16	Lynx	43.30	49.58	6.28	14.49	0
3	17	Mountain quail (summer)	25.51	29.61	4.10	16.09	0
7	26	Long-legged myotis	38.55	45.17	6.62	17.16	0
7	26	Fringed myotis	43.56	51.12	7.56	17.36	0
2	10	Olive-sided flycatcher	11.38	13.37	1.99	17.50	0
7	27	Pine siskin	29.95	35.21	5.26	17.56	0
2	11	Three-toed woodpecker	6.97	8.53	1.56	22.44	1
6	24	Sharptail snake	18.93	29.39	10.46	55.23	1
9	30	Bushtit	6.43	13.01	6.58	>100.00	2
9	30	Ash-throated flycatcher	6.61	14.28	7.67	>100.00	2
3	14	Hermit warbler	6.47	21.81	15.33	>100.00	2
NA	40	Brown-headed cowbird	0.00	33.67	33.67	>100.00	2

NA = not applicable.

[a] Species are ranked by magnitude of relative change, with species whose source habitats were projected to have undergone the greatest declines listed first.

[b] Calculations of historical and current estimates of extent of source habitats for each species excluded areas outside species ranges and also excluded those subwatersheds containing no source habitats both historically and currently. See "Assessing Change in Source Habitats from Historical to Current Conditions for Species and Groups" in the "Methods" section of volume 1 for further details about calculations of areal extent of source habitats and changes.

[c] 5 trend categories were defined: -2, -1, 0, 1, and 2, where -2 = a decrease ≥60 percent; -1 = a decrease ≥20 percent and <60 percent; 0 = a decrease or increase of <20 percent; 1 = an increase ≥20 percent and <60 percent; and 2 = an increase ≥60 percent.

Source habitats for another set of species declined less strongly at the basin scale (table 7), but declines were consistent across most ERUs (table 8). Examples included the lark sparrow, short-eared owl, vesper sparrow, western meadowlark, and blue grouse (winter). Source habitats for these species declined from 35 to 38 percent basin-wide, with categories of relative change of -1 or -2 in 75 to 85 percent of the ERUs (table 8). Other species whose source habitats declined across most ERUs (table 8; vol. 3, appendix 1, table 5) included the ground snake, burrowing owl, longnose leopard lizard, Preble's shrew, Uinta ground squirrel, lark bunting, clay-colored sparrow, Hammond's flycatcher, and black-throated sparrow; source habitats for these species declined in more than 70 percent of the ERUs in which these species occurred.

Source habitats for some species also showed extremely strong declines—at or near 100 percent—for particular ERUs (vol. 3, appendix 1, table 5), even though basin-wide declines or declines across many ERUs were not as strong. For example, source habitats for summer habitat of northern goshawk declined 93 to 97 percent in the Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs (vol. 3, appendix 1, table 5), but basin-wide decline was weaker (-43 percent, table 7). Likewise, declines in source habitats for American marten and fisher ranged

Table 8—Percentage of ecological reporting units (ERUs) having various combinations of trend categories (TC_S) of relative change for each of 91 broad-scale species of focus[a b]

Family	Group	Common name	Number of ERUs	Percentage of ERUs in category -1 or -2	Percentage of ERUs in category 0	Percentage of ERUs in category 1 or 2
1	2	Lewis' woodpecker (migrant population)	11	100	0	0
10	32	Ground snake	2	100	0	0
10	32	Washington ground squirrel	4	100	0	0
12	36	Columbian sharp-tailed grouse (summer)	11	91	9	0
12	37	Grasshopper sparrow	11	91	9	0
5	22	Rocky Mountain bighorn sheep (winter)	9	89	11	0
1	1	Pygmy nuthatch	13	85	15	0
10	31	Burrowing owl	13	85	15	0
10	31	Short-eared owl	13	85	15	0
10	31	Vesper sparrow	13	85	15	0
2	5	Flammulated owl	13	85	8	8
2	6	Williamson's sapsucker	13	85	0	15
8	29	Western bluebird	11	82	18	0
1	1	White-headed woodpecker	9	78	22	0
5	22	Rocky Mountain bighorn sheep (summer)	9	78	22	0
10	31	Western meadowlark	13	77	23	0
2	4	Blue grouse (winter)	13	77	15	8
10	31	Lark sparrow	13	77	15	8
2	6	Brown creeper	13	77	8	15
10	32	Longnose leopard lizard	8	75	25	0
10	32	Preble's shrew	12	75	25	0
10	32	Uinta ground squirrel	4	75	25	0
11	33	Lark bunting	4	75	25	0
12	37	Clay-colored sparrow	4	75	0	25
11	33	Sagebrush vole	11	73	18	9
11	34	Black-throated sparrow	7	71	29	0
5	22	California bighorn sheep	10	70	30	0
7	28	Pallid bat	10	70	30	0
10	31	Ferruginous hawk	10	70	30	0
10	31	Pronghorn	10	70	30	0
10	32	Striped whipsnake	10	70	30	0
2	7	Boreal owl	10	70	20	10
2	5	Northern goshawk (summer)	13	69	15	15
2	6	Hammond's flycatcher	13	69	15	15
2	6	Silver-haired bat	13	69	15	15
11	33	Brewer's sparrow	12	67	33	0
12	37	Idaho ground squirrel	3	67	33	0
11	33	Sage grouse (summer)	12	67	25	8
11	33	Sage grouse (winter)	12	67	25	8
2	9	Black-backed woodpecker	12	67	8	25
11	33	Sage thrasher	11	64	36	0
11	33	Sage sparrow	11	64	36	0

Table 8—Percentage of ecological reporting units (ERUs) having various combinations of trend categories (TC_S) of relative change for each of the 91 broad-scale species of focus[a b] (continued)

Family	Group	Common name	Number of ERUs	Percentage of ERUs in category -1 or -2	Percentage of ERUs in category 0	Percentage of ERUs in category 1 or 2
1	1	White-breasted nuthatch	13	62	31	8
7	28	Western small-footed myotis	13	62	31	8
11	35	Loggerhead shrike	13	62	23	15
11	33	Pygmy rabbit	10	60	30	10
2	11	White-winged crossbill	10	60	10	30
2	12	Woodland caribou	5	60	0	40
6	23	Broad-tailed hummingbird	7	57	14	29
6	23	Rufous hummingbird	13	54	31	15
2	13	Northern flying squirrel	13	54	23	23
2	5	American marten	13	54	8	38
6	25	Northern goshawk (winter)	13	54	8	38
2	5	Fisher	13	54	0	46
11	34	Kit fox	4	50	50	0
2	6	Chestnut-backed chickadee	10	50	20	30
3	17	Blue grouse (summer)	13	46	15	38
4	18	Lazuli bunting	13	46	15	38
2	6	Golden-crowned kinglet	13	46	8	46
2	11	Three-toed woodpecker	13	46	0	54
1	3	Western gray squirrel	7	43	43	14
2	8	Great gray owl	12	42	17	42
2	6	Winter wren	12	42	0	58
10	32	Wyoming ground squirrel	5	40	60	0
7	27	Townsend’s big-eared bat	13	38	46	15
2	6	Varied thrush	11	36	9	55
5	19	Grizzly bear	13	31	62	8
2	6	Hoary bat	13	31	54	15
5	21	Long-eared owl	13	31	54	15
2	10	Olive-sided flycatcher	13	31	15	54
2	6	Pileated woodpecker	10	30	10	60
2	6	Vaux’s swift	11	27	0	73
7	28	Spotted bat	12	25	58	17
5	20	Mountain goat	8	25	38	38
6	24	California mountain kingsnake	4	25	25	50
5	19	Gray wolf	13	23	77	0
6	24	Black-chinned hummingbird	13	23	23	54
7	27	Pine siskin	13	23	8	69
10	32	White-tailed antelope squirrel	5	20	80	0
7	26	Long-eared myotis	13	15	85	0
3	15	Wolverine	13	15	15	69
3	16	Lynx	9	11	67	22
3	17	Mountain quail (summer)	9	11	33	56
7	26	Yuma myotis	11	9	91	0

Table 8—Percentage of ecological reporting units (ERUs) having various combinations of trend categories (TC_S) of relative change for each of 91 broad-scale species of focus[a] [b] (continued)

Family	Group	Common name	Number of ERUs	Percentage of ERUs in category -1 or -2	Percentage of ERUs in category 0	Percentage of ERUs in category 1 or 2
10	32	Mojave black-collared lizard	2	0	100	0
10	32	Longnose snake	1	0	100	0
NA	38	Black rosy finch	7	0	100	0
NA	38	Gray-crowned rosy finch	11	0	100	0
NA	39	Lewis' woodpecker (resident population)	1	0	100	0
7	26	Fringed myotis	11	0	73	27
3	15	Pygmy shrew	5	0	60	40
7	26	Long-legged myotis	13	0	54	46
9	30	Bushtit	9	0	22	78
9	30	Ash-throated flycatcher	9	0	11	89
3	14	Hermit warbler	4	0	0	100
6	24	Sharptail snake	3	0	0	100
NA	40	Brown-headed cowbird	13	0	0	100

NA = not applicable; these species not assigned to families.

[a] Trend categories were defined such that -2 = a decrease ≥60 percent; -1 = a decrease ≥20 percent and <60 percent; 0 = a decrease or increase of <20 percent; 1 = an increase ≥20 percent and <60 percent; and 2 = an increase ≥60 percent.

[b] Species are listed in descending order by percentage of ERUs with a trend category of -1 or -2. Percentages were calculated only for ERUs where the species occurred.

from 88 to 100 percent within the Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and Upper Snake ERUs (vol. 3, appendix 1, table 5), whereas basin-wide decline was less strong for both species (-39 percent for marten, -20 percent for fisher, table 7). Source habitats for sagebrush vole also declined 87 and 98 percent within the Northern Cascades and Snake Headwaters ERUs, respectively (vol. 3, appendix 1, table 5), but basin-wide decline was 27 percent (table 7).

In contrast to the large number of species whose source habitats declined across many or most ERUs, relatively few species were associated with source habitats that changed little across most ERUs. Source habitats for only 16 species had a trend category of relative change equal to 0 for most ERUs in which these species occurred (table 8). Moreover, an even smaller number of species were associated with source habitats that increased strongly across most ERUs. For example, only five species—brown-headed cowbird, sharptail snake, hermit warbler, ash-throated flycatcher, and bushtit—had source habitats that increased by >50 percent basin-wide (table 7) and had categories of relative change of 1 or 2 in >75 percent of the ERUs in which these species occurred (table 8). Cover type-structural stage combinations that contributed most to increases in source habitats for these five species were cropland-hay-pasture (associated with brown-headed cowbird), juniper woodlands (associated with ash-throated flycatcher and bushtit), various lower elevation cover types in the stem-exclusion and understory-reinitiation stages (associated with sharptail snake), and some of the lower elevation cover types in the managed young-forest stages (associated with hermit warbler here) (vol. 3, appendix 1, table 4).

Habitat Change on All Lands Versus Public and Mixed Ownership

Species whose relative change in source habitats was negative on all lands also had relative change that was negative on public and mixed ownership (figs. 6A,

Figure 6—Relative change (RCH_S) in source habitats, from historical to current periods, for each of 91 species (97 species-seasonal entries), on all lands versus public and mixed-ownership lands at the scale of the basin.

and 6B); that is, basin-wide trends in source habitats that were negative on all lands also were consistently negative on public and mixed ownership, for all species whose habitat trends had a negative sign basin-wide. The only exception was the great gray owl, which showed a slightly negative trend on all lands but a slightly positive trend on public and mixed ownership (fig. 6B). Similarly, species whose relative change in source habitats was positive on all lands also had relative change that was positive on public and mixed ownership (fig. 6B). One exception existed: the California mountain kingsnake, whose source habitats showed a slightly positive trend on all lands but a slightly negative trend on public and mixed ownership (fig. 6B).

Magnitude of relative change in source habitats on all lands versus public and mixed ownership also was highly consistent. Magnitude of decline or increase nearly always was stronger for all lands than for public and mixed ownership (figs. 6A, B), but overall differences in magnitude typically were <10 percent between all lands versus public and mixed ownership. Exceptions were chestnut-backed chickadee, broad-tailed hummingbird, woodland caribou, and western gray squirrel, whose source habitats showed a slightly stronger decline on public and mixed ownership than on all lands (fig. 6A). Additional exceptions were olive-sided flycatcher and three-toed woodpecker, whose source habitats showed a slightly stronger increase on public and mixed ownership than on all lands (fig. 6B).

Management Implications

The large number of species whose source habitats declined strongly or very strongly at the basin scale (table 7), combined with the diverse composition and structure of the source habitats of these species (vol. 3, appendix 1, table 1), suggest that no particular species or habitats, or small set of species or habitats, are easily identified as needing priority management. Rather, the large number of species undergoing decline in source habitats, combined with the diversity of habitats associated with these species, suggest that aggregations of large numbers of species and a wide array of source habitats may need management attention.

Species-level findings also suggest that it would be difficult to select a small number of management indicator or umbrella species on which to base management (see “Glossary,” Landres and others 1988, and Marcot and others 1994 for definitions and concepts of indicator and umbrella species). Moreover, the large number of species whose source habitats declined at the basin scale further suggests that any attempts to group or aggregate species must be made without losing unique, single-species trends in source habitats that could be obscured or diluted by such attempts. This potential problem has been the main criticism directed at the use of guilds (Szaro 1986) or indicator guilds (Verner 1984) for management applications. Thus, it is important that management needs of the many species undergoing a strong or very strong decline in source habitats (tables 7 and 8; vol. 3, appendix 1, table 5) be accounted for in group- and family-level methods and results that are part of our assessment. Species-level trends summarized at the ERU level (table 8; vol. 3, appendix 1, table 5) are particularly important to consider for species whose source habitats exhibited strongly different trends among ERUs.

The high consistency in direction and magnitude of change in source habitats for each species between all lands and public-mixed ownership lands further suggests that the same habitat issues likely are of interest to both public and private land managers. That is, both public and private land managers, or regulatory managers with potential jurisdiction related to both public and private lands, would be faced with the same or a similar direction and magnitude of habitat trends, regardless of land ownership. It important to note, however, that this finding may not hold at finer scales within the basin—such as subwatershed and watershed scales—where large differences in direction and magnitude of habitat trends may exist between land ownerships.

Group-Level Results and Discussion

Group Membership and Associated Source Habitats

Results are presented here for 40 groups, composed of 91 species of birds, mammals, and reptiles (table 5). With the exception of two species, the black rosy finch and gray-crowned rosy finch, each species depends on a unique set of source habitats (vol. 3, appendix 1, table 1). Species within each group, however, display strong overlap in the cover type-structural stage combinations used as source habitats, as intended by our use of cluster analysis to group species based on their degree of similarity and dissimilarity in source habitats (see "Methods," "Clustering the Species into Groups"). The specific terrestrial communities and cover type-structural stage combinations identified as source habitats for each species in each group are shown in volume 3, appendix 1, table 1.

Results and discussion presented here for the 40 groups represent an overview of more detailed results and discussion presented in volume 2. Readers should refer to volume 2 for results, by groups of species, that display (1) the geographic range of each species within each group; (2) maps of the percentage of area of source habitats, historically and currently; (3) a map of habitat change; and (4) bar charts displaying the percentage of watersheds in each ERU that have undergone positive, strongly positive, neutral, negative, and strongly negative relative change in source habitats from historical to current conditions. Discussion in volume 2 also contains detailed interpretation of habitat change in relation to associated vegetation dynamics, in relation to conditions of other habitat features, and in relation to nonvegetative factors that affect species within each group. Finally, discussion in volume 2 also includes a description of key management implications. Management implications were synthesized from results of our assessment, from the scientific literature, and from results of prior assessments conducted as part of the ICBEMP. Implications include an identification of management issues associated with species in each group, and a list of strategies and practices that might be useful in dealing with those issues. An overview of these results and their implications is described in the following sections.

Habitat Change by Basin and Ecological Reporting Unit

Basin-wide change—Fifty percent of the 40 groups of species were associated with source habitats that declined strongly or very strongly from historical to current periods, based on trend categories of relative change (TC_G) at the basin scale (rank of -1 or -2, table 9). By contrast, only four groups (10 percent) were associated with source habitats that increased strongly or very strongly (rank of 1 or 2), but a moderate number—16 groups or 40 percent—were associated with source habitats that showed little change (rank of 0).

In contrast to the trends based on categories of relative change, decline in source habitats was consistently more negative when expressed as continuous variables of absolute and relative change (ACH_G and RCH_G). By using these measures, 75 percent of the groups were associated with a decline in source habitats (table 9). Only one group showed no change in source habitats, and 23 percent of groups were associated with an increase.

As with species-level results, groups of species whose source habitats declined were associated with many forested and rangeland environments. Of the 20 groups that underwent the strongest relative decline in source habitats (table 9), 9 are primarily dependent on forested habitats, another 9 are largely dependent on rangeland habitats, and 2 are dependent on a combination of forested and rangeland habitats (vol. 3, appendix 1, table 1). Again, as with the species-level results, this finding indicates that many source habitats have declined in the basin; in turn, this suggests that no particular species or habitats, or small set of species or habitats, are easily identified as needing priority management.

Habitat change by ecological reporting unit— Groups of species whose source habitats declined strongly or very strongly at the basin scale (trend categories of relative change of -1 or -2, table 9) also experienced strong declines in source habitats across most ERUs (table 10; vol. 3, appendix 1, table 3). For example, group 36, composed of the clay-colored sparrow, grasshopper sparrow, and Idaho ground squirrel, had the second greatest relative decline among all groups of species in the basin (-71 percent, table 9) and also had categories of relative change that were -1 or -2 for 91 percent of the ERUs in which these species occurred (table 10). Other groups whose source habitats declined strongly at the basin level and across most or all ERUs included group 2 (migrant population of Lewis' woodpecker (group 2), group 36 (Columbian sharp-tailed grouse), group 31 (Ferruginous hawk, burrowing owl, short-eared owl, vesper sparrow, lark sparrow, western meadowlark, and pronghorn), group 29 (western bluebird), and group 4 (blue grouse [winter]). Source habitats for these groups declined by >35 percent at the basin scale (table 9),

Table 9—Historical (H_G) and current (C_G) estimates of areal extent (percentage of area) of source habitats at the scale of the basin for 40 groups of 91 broad-scale species of focus, and resulting changes in source habitats based on three measures: absolute change (ACH_G), relative change (RCH_G), and trend categories (TC_G) of relative change[a]

Family	Group	Common name	Historical estimate	Current estimate	Absolute change	Relative change	Trend category[b]
			Percentage				
1	2	Lewis' woodpecker (migrant population)	13.78	2.29	-11.49	-83.35	-2
12	37	Clay-colored sparrow Grasshopper sparrow Idaho ground squirrel	21.22	6.17	-15.05	-70.93	-2
2	7	Boreal owl	14.97	5.78	-9.20	-61.42	-2
1	1	White-headed woodpecker White-breasted nuthatch Pygmy nuthatch	18.37	9.01	-9.36	-50.96	-1
8	29	Western bluebird	51.29	26.39	-24.90	-48.55	-1
12	36	Columbian sharp-tailed grouse (summer)	58.80	32.35	-26.44	-44.97	-1
2	5	Northern goshawk (summer) Flammulated owl American marten Fisher	18.10	10.74	-7.37	-40.70	-1
10	31	Ferruginous hawk Burrowing owl Short-eared owl Vesper sparrow Lark sparrow Western meadowlark Pronghorn	57.06	36.55	-20.52	-35.95	-1
2	4	Blue grouse (winter)	21.30	13.68	-7.62	-35.79	-1
2	6	Vaux's swift Williamson's sapsucker Pileated woodpecker Hammond's flycatcher Chestnut-backed chickadee Brown creeper Winter wren Golden-crowned kinglet Varied thrush Silver-haired bat Hoary bat	13.94	9.07	-4.88	-34.99	-1

Table 9—Historical (H_G) and current (C_G) estimates of areal extent (percentage of area) of source habitats at the scale of the basin for 40 groups of 91 broad-scale species of focus, and resulting changes in source habitats based on three measures: absolute change (ACH_G), relative change (RCH_G), and trend categories (TC_G) of relative change[a] (continued)

Family	Group	Common name	Historical estimate	Current estimate	Absolute change	Relative change	Trend category[b]
			Percentage				
2	9	Black-backed woodpecker	23.05	15.29	-7.77	-33.70	-1
10	32	Mojave black-collared lizard Longnose leopard lizard Striped whipsnake Longnose snake Ground snake Preble's shrew White-tailed antelope squirrel Washington ground squirrel Wyoming ground squirrel Uinta ground squirrel	66.42	45.22	-21.20	-31.91	-1
6	25	Northern goshawk (winter)	21.37	14.59	-6.78	-31.73	-1
5	22	California bighorn sheep Rocky Mountain bighorn sheep (summer) Rocky Mountain bighorn sheep (winter)	50.51	34.64	-15.87	-31.42	-1
11	33	Sage grouse (summer) Sage grouse (winter) Sage thrasher Brewer's sparrow Sage sparrow Lark bunting Pygmy rabbit Sagebrush vole	54.61	39.20	-15.41	-28.21	-1
4	18	Lazuli bunting	12.47	9.52	-2.95	-23.63	-1
7	28	Western small-footed myotis Spotted bat Pallid bat	49.97	38.73	-11.24	-22.49	-1
11	34	Black-throated sparrow Kit fox	64.72	50.46	-14.25	-22.02	-1
2	13	Northern flying squirrel	32.26	25.26	-7.00	-21.70	-1
6	23	Rufous hummingbird Broad-tailed hummingbird	30.20	23.67	-6.54	-21.64	-1
11	35	Loggerhead shrike	47.82	38.45	-9.37	-19.60	0
5	21	Long-eared owl	50.98	42.46	-8.52	-16.71	0
5	19	Gray wolf Grizzly bear	82.42	69.07	-13.35	-16.20	0

Table 9—Historical (H_G) and current (C_G) estimates of areal extent (percentage of area) of source habitats at the scale of the basin for 40 groups of 91 broad-scale species of focus, and resulting changes in source habitats based on three measures: absolute change (ACH_G), relative change (RCH_G), and trend categories (TC_G) of relative change[a] (continued)

Family	Group	Common name	Historical estimate	Current estimate	Absolute change	Relative change	Trend category[b]
			Percentage				
2	12	Woodland caribou	4.03	3.68	-0.36	-8.86	0
3	17	Blue grouse (summer) Mountain quail (summer)	28.57	26.34	-2.23	-7.80	0
7	27	Pine siskin Townsend's big-eared bat	51.75	48.39	-3.36	-6.49	0
2	8	Great gray owl	26.53	24.94	-1.59	-5.99	0
7	26	Yuma myotis Long-eared myotis Fringed myotis Long-legged myotis	55.64	53.94	-1.70	-3.05	0
1	3	Western gray squirrel	22.43	22.03	-0.41	-1.81	0
NA	38	Black rosy finch Gray-crowned rosy finch	8.17	8.16	-0.01	-0.09	0
NA	39	Lewis' woodpecker (resident)	10.25	10.25	0	0	0
2	11	Three-toed woodpecker White-winged crossbill	6.91	7.53	0.62	8.90	0
5	20	Mountain goat	43.25	47.50	4.24	9.81	0
6	24	Sharptail snake California mountain kingsnake Black-chinned hummingbird	20.33	23.15	2.82	13.86	0
3	16	Lynx	43.30	49.58	6.28	14.49	0
2	10	Olive-sided flycatcher	11.38	13.37	1.99	17.50	0
3	15	Pygmy shrew Wolverine	35.87	43.08	7.21	20.11	1
9	30	Ash-throated flycatcher Bushtit	5.96	12.63	6.67	>100.00	2
3	14	Hermit warbler	6.47	21.81	15.33	>100.00	2
NA	40	Brown-headed cowbird	0	33.67	33.67	>100.00	2

NA = not applicable; these species not assigned to families.

[a] Calculations of historical and current estimates of extent of source habitats for each group excluded areas outside species ranges and also excluded those subwatersheds containing no source habitats both historically and currently. See "Assessing Change in Source Habitats From Historical to Current Conditions for Species and Groups" in the "Methods" section of volume 1 for further details.

[b] 5 trend categories were defined: -2, -1, 0, 1, and 2, where -2 = a decrease ≥60 percent; -1 = a decrease ≥20 percent and <60 percent; 0 = a decrease or increase of <20 percent; 1 = an increase ≥20 percent and <60 percent; and 2 = an increase ≥60 percent.

and categories of relative change were either -1 or -2 in >70 percent of the ERUs in which these species occurred (table 10; vol. 3, appendix 1, table 3).

Other groups whose source habitats declined strongly across most ERUs included group 32 (Mojave black-collared lizard, longnose leopard lizard, striped whipsnake, longnose snake, ground snake, Preble's shrew, white-tailed antelope squirrel, Washington ground squirrel, Wyoming ground squirrel, and Uinta ground squirrel), group 22 (California and Rocky Mountain bighorn sheep), group 33 (sage grouse, sage thrasher, Brewer's sparrow, sage sparrow, lark bunting, pygmy rabbit, and sagebrush vole), group 34 (black-throated sparrow and kit fox), group 7 (boreal owl), and group 1 (white-headed woodpecker, white-breasted nuthatch, and pygmy nuthatch). Source habitats for these groups declined in >65 percent of the ERUs in which the groups occurred (table 10; vol. 3, appendix 1, table 3).

Source habitats for some groups also exhibited extremely strong declines—at or near 100 percent—for particular ERUs (vol. 3, appendix 1, table 3), even though trends were not consistent across ERUs. For example, source habitats for group 6 (northern goshawk [summer], flammulated owl, American marten, and fisher) declined >90 percent in the Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs (vol. 3, appendix 1, table 3), but trends were neutral or increasing in almost 40 percent of the ERUs (table 10). Likewise, decline in source habitats for group 9 (black-backed woodpecker) ranged from 86 to 94 percent within the Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and Upper Snake ERUs (vol. 3, appendix 1, table 5), but trends were neutral or increasing in >30 percent of ERUs. Source habitats for group 18 (lazuli bunting) also underwent similar declines—ranging from 82 to 93 percent—within the Upper Klamath, Blue Mountains, and Lower Clark Fork ERUs (vol. 3, appendix 1, table 5), but almost 40 percent of the ERUs for this group had a neutral or increasing trend.

In contrast to the large number of groups whose source habitats declined across many or most ERUs, relatively few groups were associated with source habitats that changed little across most ERUs. Specifically, source habitats for five groups had categories of relative change of 0 for most ERUs in which the groups occurred (table 10). Similarly, six groups were associated with source habitats that increased strongly across most ERUs (table 10).

Habitat Change on All Lands Versus Public and Mixed Ownership

The direction of trends in source habitats between all lands versus public and mixed ownership for groups of species (fig. 7) was similar to that found for individual species (fig. 6); that is, basin-wide trends in source habitats that were negative on all lands also were consistently negative on public and mixed ownership, for all groups whose habitat trends had a negative sign basin-wide. One exception existed: group 8 (great gray owl), which showed a slightly negative trend on all lands but a slightly positive trend on public and mixed ownership (fig. 7). Similarly, groups whose relative change in source habitats was positive on all lands also had relative change that was positive on public and public mixed ownership (fig. 7).

Magnitude of relative change in source habitats on all lands versus public and mixed ownership also showed the same highly consistent pattern for groups of species (fig. 7) as that found for individual species (fig. 6). Magnitude of decline or increase nearly always was stronger for all lands than on public and mixed ownership (fig. 7), but overall differences in magnitude most often were <10 percent. Exceptions were group 4 (blue grouse [winter]), group 12 (woodland caribou), group 19 (gray wolf and grizzly bear), and group 3 (western gray squirrel), whose source habitats showed a slightly stronger decline on public and mixed ownership than on all lands (fig. 7). Additional exceptions were group 11 (loggerhead shrike) and group 10 (olive-sided flycatcher), whose source habitats showed a slightly stronger increase on public and mixed ownership than on all lands (fig. 7).

Correlation of Habitat Trends Among Species Within Groups

Relative change in source habitats was positively correlated ($P < 0.05$) for all of the 177 species pairings within the multi-species groups (fig. 8). Moreover, the grand mean of all correlation coefficients, calculated from the means of all within-group coefficients, was relatively high ($r = 0.66$). By contrast, the grand mean

Table 10—Percentage of ecological reporting units (ERUs) having various combinations of trend categories (TC_G) of relative change for each of the 40 groups of 91 broad-scale species of focus[a] [b]

Family	Group	Common name	Number of ERUs	Percentage of ERUs in category -1 or -2	Percentage of ERUs in category 0	Percentage of ERUs in category 1 or 2
1	2	Lewis' woodpecker (migrant population)	11	100	0	0
12	36	Columbian sharp-tailed grouse (summer)	11	91	9	0
12	37	Clay-colored sparrow Grasshopper sparrow Idaho ground squirrel	11	91	9	0
10	31	Ferruginous hawk Burrowing owl Short-eared owl Vesper sparrow Lark sparrow Western meadowlark Pronghorn	13	85	15	0
10	32	Mojave black-collared lizard Longnose leopard lizard Striped whipsnake Longnose snake Ground snake Preble's shrew White-tailed antelope squirrel Washington ground squirrel Wyoming ground squirrel Uinta ground squirrel	13	85	15	0
8	29	Western bluebird	11	82	18	0
2	4	Blue grouse (winter)	13	77	15	8
5	22	California bighorn sheep Rocky Mountain bighorn sheep (summer) Rocky Mountain bighorn sheep (winter)	13	77	23	0
11	33	Sage grouse (summer) Sage grouse (winter) Sage thrasher Brewer's sparrow Sage sparrow Lark bunting Pygmy rabbit Sagebrush vole	12	75	25	0
11	34	Black-throated sparrow Kit fox	8	75	25	0

Table 10—Percentage of ecological reporting units (ERUs) having various combinations of trend categories (TC_G) of relative change for each of the 40 groups of 91 broad-scale species of focus[a b] (continued)

Family	Group	Common name	Number of ERUs	Percentage of ERUs in category -1 or -2	Percentage of ERUs in category 0	Percentage of ERUs in category 1 or 2
2	7	Boreal owl	10	70	20	10
1	1	White-headed woodpecker White-breasted nuthatch Pygmy nuthatch	13	69	23	8
2	9	Black-backed woodpecker	12	67	8	25
2	5	Northern goshawk (summer) Flammulated owl American marten Fisher	13	62	15	23
7	28	Western small-footed myotis Spotted bat Pallid bat	13	62	38	0
11	35	Loggerhead shrike	13	62	23	15
2	12	Woodland caribou	5	60	0	40
2	6	Vaux’s swift Williamson’s sapsucker Pileated woodpecker Hammond’s flycatcher Chestnut-backed chickadee Brown creeper Winter wren Golden-crowned kinglet Varied thrush Silver-haired bat Hoary bat	13	54	23	23
2	13	Northern flying squirrel	13	54	23	23
6	23	Rufous hummingbird Broad-tailed hummingbird	13	54	31	15
6	25	Northern goshawk (winter)	13	54	8	38
2	11	Three-toed woodpecker White-winged crossbill	13	46	0	54
4	18	Lazuli bunting	13	46	15	38
1	3	Western gray squirrel	7	43	43	14

Table 10—Percentage of ecological reporting units (ERUs) having various combinations of trend categories (TC_G) of relative change for each of the 40 groups of 91 broad-scale species of focus[a b] (continued)

Family	Group	Common name	Number of ERUs	Percentage of ERUs in category -1 or -2	Percentage of ERUs in category 0	Percentage of ERUs in category 1 or 2
2	8	Great gray owl	12	42	17	42
7	27	Pine siskin Townsend's big-eared bat	13	38	23	38
3	17	Blue grouse (summer) Mountain quail (summer)	12	33	25	42
2	10	Olive-sided flycatcher	13	31	15	54
5	19	Gray wolf Grizzly bear	13	31	69	0
5	21	Long-eared owl	13	31	54	15
5	20	Mountain goat	8	25	38	38
6	24	Sharptail snake California mountain kingsnake Black-chinned hummingbird	13	23	23	54
3	16	Lynx	9	11	67	22
3	15	Pygmy shrew Wolverine	13	8	23	69
7	26	Yuma myotis Long-eared myotis Fringed myotis Long-legged myotis	13	8	92	0
3	14	Hermit warbler	4	0	0	100
9	30	Ash-throated flycatcher Bushtit	10	0	20	80
NA	38	Black rosy finch Gray-crowned rosy finch	11	0	100	0
NA	39	Lewis' woodpecker (resident population)	1	0	100	0
NA	40	Brown-headed cowbird	13	0	0	100

NA = not applicable; these species not assigned to families.

[a] Groups are listed in descending order by percentage of ERUs with a trend category of -1 or -2.

[b] Trend categories were defined such that -2 = a decrease ≥60 percent; -1 = a decrease ≥20 percent and <60 percent; 0 = a decrease or increase of <20 percent; 1 = an increase ≥20 percent and <60 percent; and 2 = an increase ≥60 percent.

Figure 7—Relative change (RCH_G) in source habitats, from historical to current periods, for each of 40 groups of broad-scale species of focus, on all lands versus public lands and mixed-ownership lands at the scale of the basin.

of all between-group species pairings was near zero (*r* = 0.02), further suggesting that clustering of species into groups efficiently captured similar direction and magnitude of species-level trends within each multispecies group.

Range of coefficients between individual species within each group varied widely, however, with *r* values as high as 0.96, and as low as 0.12. Despite this wide range, only 5 of the 177 coefficients (<3 percent) calculated for the within-group species pairings were <0.20 (fig. 8): (1) pygmy shrew and wolverine (*r* = 0.12, group 15); (2) long-eared myotis and long-legged myotis (*r* = 0.15, group 26); (3) long-eared myotis and fringed myotis (*r* = 0.17, group 26); (4) Wyoming ground squirrel and longnose snake (*r* = 0.18, group 32); and (5) Wyoming ground squirrel and Mojave black-collared lizard (*r* = 0.18, group 32). In five other cases (<3 percent), correlation coefficients were >0.20 but <0.40: (1) California mountain kingsnake and black-chinned hummingbird (*r* = 0.33, group 24); (2) long-legged myotis and Yuma myotis (*r* = 0.33, group 26); (3) white-tailed antelope squirrel and Wyoming ground squirrel (*r* = 0.30, group 32); (4) white-tailed antelope squirrel and longnose snake (*r* = 0.35, group 32); and (5) white-tailed antelope squirrel and Mojave black-collared lizard (*r* = 0.36, group 32);

Notably, 9 of the 10 correlation coefficients <0.40 involved just five species—long-legged myotis, Wyoming ground squirrel, white-tailed antelope squirrel, longnose snake, and Mojave black-collared lizard —as a member of a species pairing. Also, the 10 coefficients <0.40 involved just 4 groups: 15, 24, 26, and 32. Finally, of the 11 species that were part of one or more pairings where *r* was <0.40, all 11 (100 percent) were associated with trend categories for source habitats that were neutral (table 7); this is especially

Figure 8—Mean and range of correlation coefficients for species pairings within each group of broad-scale species of focus for groups containing more than one species. Mean for each group was calculated from Kendall's Tau correlation coefficients that were computed for each pair of species in the group. Single values are for groups containing two species (one coefficient for the one pair). Range of values is shown for groups containing 3 or more species. Specific pairings are identified for any pair of species with a correlation coefficient less than 0.4, which is denoted by the upper dotted line. The lower dotted line denotes the mean correlation coefficient (0.02) for all species pairings between groups.

noteworthy considering that habitats for most of the species underwent strong or very strong declines (trend categories of relative change of -1 or -2, table 7).

Management Implications

The large number of groups of species whose source habitats declined strongly or very strongly at the basin scale (trend categories of relative change of -1 or -2, table 9), combined with the diverse composition and structure of the source habitats of these species (vol. 3, appendix 1, table 1), suggests that no particular species or habitats, or small set of species or habitats, are easily identified as needing priority management. Rather, the large number of species undergoing decline in source habitats, combined with the diversity of habitats associated with these species, suggests that aggregations of large numbers of species and a wide array of source habitats may need management attention. Consequently, our findings suggest that habitat analysis and management of groups of species may be more efficient than a species-by-species approach. This point is especially germane, considering the large number of species (91 species and 97 species seasonal entries) analyzed here, and the consistent pattern shown between trends in source habitats at the species level versus trends for the same species calculated as groups (for example, examine trends in table 7 versus table 9).

The high consistency in direction and magnitude of change in source habitats for each group between all lands and public-mixed ownership lands further suggests that the same habitat issues may be of interest to both public and private land managers. That is, both public and private land managers, or regulatory managers with potential jurisdiction related to both public and private lands, would be faced with the same or a similar direction and magnitude of habitat trends, regardless of land ownership. This finding, however, may not hold at finer scales within the basin—such as subwatershed and watershed scales—where large differences in direction and magnitude of habitat trends may exist between land ownerships.

The relatively high, positive correlation coefficients that we calculated for most within-group species pairings versus the relatively low or negative coefficients calculated for between-group species pairings, have the following implications for interpretation of our group-level habitat trends:

1. The strong, positive correlations in habitat trends among species within most of the groups indicate that group-level results accurately represent individual species trends; this is especially encouraging, considering that most groups having strong correlations in their species-level habitat trends also were the groups that contained species associated with strong or very strong declines in source habitats. In these cases, the group-level trends reflected the species-level trends. This implication is especially important, considering that most attention presumably will be given to species and groups whose source habitats have undergone the strongest declines. In these cases, our group-level results appear most reliable.

2. The few groups containing species with low coefficients—namely groups 15, 24, 26, and 32—may yield group-level trends that could be misleading for one or more species within the groups. Many of the species involved in pairings having low correlation coefficients, however, are localized in their distributions, and thus have little effect on group-level trends. Examples are white-tailed antelope squirrel, Wyoming ground squirrel, longnose snake, Mojave black-collared lizard, and California mountain kingsnake. In these cases, the species-level contribution to the group trend is minor because ranges of the problem species (vol. 2, fig. 96) are narrow and thus do not contribute to calculation of habitat trend for most areas of the basin in which group-level trends were calculated. (See "Methods" for details about calculation of group-level trends in source habitats).

3. Implementation procedures presumably will consider results of our correlation analysis and account for the handful of low correlations as part of local analysis. The species listed in figure 8 and their associated groups are candidates for more detailed analysis as part of implementation.

Because of the accuracy and efficiency with which group-level trends reflect species-level changes in source habitats, we have emphasized and provided detailed results and management implications based on indepth analyses for all 40 groups of species in volume 2. An especially noteworthy section of volume 2 is the comprehensive set of issues, strategies, and practices identified for effective management of each group of species, as well as the synthesis of supporting, pertinent empirical literature about environmental requirements and population status and trends of each species in each group.

Family-Level Results and Discussion

Habitat Relations Among Families

Placement of 37 of the groups into 12 families (fig. 5, table 6) by using a combination of cluster analysis and empirical knowledge of similarities of species in habitat requirements resulted in distinct differences among families in the number of terrestrial community types and source habitats used (table 11). Family 4 had the most restricted number of terrestrial community types and source habitats used by species of any family, with habitats restricted to early-seral forests (table 11). Species in family 1 also were restricted to a small number of terrestrial community types, and in this case, the types were composed of low-elevation, late-seral forests (table 11). By contrast, species in family 2 used a higher number and variety of terrestrial community types that encompassed all elevations of late-seral forests. Species in family 3 used an even greater variety of forested conditions; habitats encompassed the highest number and type of source habitats within the highest number of terrestrial community types of any family dependent on forested habitats.

Species dependent strictly on rangelands were placed in families 10, 11 and 12. Species in families 11 and 12 were restricted to a relatively small number of terrestrial community types, with family 11 primarily dependent on sagebrush, and family 12 dependent on grassland and open-canopy sagebrush habitats (table 11). Species in family 10 used a broader set of

Table 11 - Pages 63–64

terrestrial communities, consisting of a greater variety of grassland, shrubland, woodland, and related cover types than those used by families 11 and 12.

Species in families 5, 6, 7, 8, and 9 were associated with various terrestrial community types, but each family's set of source habitats was distinctly different from the others (table 11). Habitats for species in family 9 were restricted to relatively few source habitats within the upland woodland and upland shrubland types. By contrast, species in family 5 used habitats that encompassed nearly all terrestrial community types. Species in family 6 also used various terrestrial communities, with the types composed of forests, woodlands, and montane shrubs. Terrestrial community types used by family 7 were similar to those used by family 6, with the main difference being the use of sagebrush types instead of montane shrubs. Finally, habitats for family 8 spanned a fairly restrictive but unusual combination of terrestrial community types composed of both early- and late-seral forests, as well as woodland, shrubland, and grassland types (table 11).

These differences in terrestrial communities and source habitats among the families resulted in distinctly different habitat trends for each family. In the following sections, we present results for each family and an overview of results across families. Results are summarized in terms of key vegetative themes, trends, and issues presumably of most interest to managers of FS- and BLM-administered lands within the basin. Specifically, the family-level results provide (1) a description of source habitats and special habitat features for species in the family; (2) a summary of family-level trends in source habitats from historical to current periods; (3) identification of the primary causes for the observed habitat trends and the ecological processes associated with the causes; and (4) a synthesis of broad-scale strategies that would benefit species and their source habitats.

Overview of Family-Level Results

The 12 families exhibited wide variation in the percentage of ERUs that had declining versus increasing or neutral habitat trends (table 12). Family 1 had the largest percentage of ERUs (85 percent, 11 of 13 ERUs) with declining trends (see "Methods," "Evaluating Change in Source Habitats by Family," for analysis steps used to characterize ERU habitat trends by family). Other families for which most of the ERUs had declining habitat trends included family 8 (82 percent, 9 of 11 ERUs), family 10 (69 percent, 9 of 13 ERUs), and families 4 and 12 (each 62 percent, 8 of 13 ERUs). A substantial percentage of ERUs also had declining trends for family 2 (46 percent, 6 of 13) and family 11 (39 percent, 4 of 13). Smaller percentages of ERUs had declining trends for family 6 (31 percent, 4 of 13), family 5 (23 percent, 3 of 13), family 7 (15 percent, 2 of 13), family 9 (10 percent, 1 of 10), and family 3 (8 percent, 1 of 12).

As found for the species and groups, declining habitat trends for families were associated with several species whose source habitats encompassed a diversity of forest and rangeland environments. For example, families 1, 4, 8, 10, and 12, which had the highest percentage of ERUs with negative habitat trends, were associated with source habitats as diverse as low-elevation, old-forest (family 1), early-seral forest (family 4), a combination of rangeland and early- and late-seral forest (family 8), herbland, shrubland, and woodland (family 10), and grassland and open-canopy sagebrush (family 12). In addition, nearly all families (even those with a small number of ERUs with declining habitat trends) contained one or more groups of species whose source habitats declined strongly or very strongly from historical to current periods (based on trend categories of relative change (TC_G) at the basin scale [rank of -1 or -2, table 9]). Exceptions were families 3 and 9, neither of which included groups having a declining trend category at the basin scale (table 9).

Management implications—Family-level habitat trends suggest that no particular species or habitats, or small set of species or habitats, are easily identified as needing priority management. This is because (1) several families had predominantly negative habitat trends across ERUs (table 12), (2) nearly all families contained groups of species whose source habitats declined strongly or very strongly at the basin scale (table 9), and (3) declining source habitats were diverse in composition and structure (vol. 3, appendix 1, table 1). The large number of species, spanning multiple groups and families, that experienced declines in source habitats, combined with the diversity of habitats associated with these species, suggest that aggregations of large numbers of species and a wide array of source habitats may need management attention.

Table 12—Percentage of watersheds in 3 trend categories for each family, by ecological reporting unit (ERU)

Family	ERU	ERU name	Percentage of watersheds decreasing	Percentage of watersheds neutral	Percentage of watersheds increasing	Dominant trend[a]
1	1	Northern Cascades	69	24	7	Decreasing
1	2	Southern Cascades	56	22	22	Decreasing
1	3	Upper Klamath	33	19	48	Neutral
1	4	Northern Great Basin	47	8	45	Neutral
1	5	Columbia Plateau	51	19	31	Decreasing
1	6	Blue Mountains	67	20	13	Decreasing
1	7	Northern Glaciated Mountains	94	5	2	Decreasing
1	8	Lower Clark Fork	95	4	1	Decreasing
1	9	Upper Clark Fork	77	20	3	Decreasing
1	10	Owyhee Uplands	77	3	20	Decreasing
1	11	Upper Snake	81	0	19	Decreasing
1	12	Snake Headwaters	86	4	10	Decreasing
1	13	Central Idaho Mountains	57	33	9	Decreasing
2	1	Northern Cascades	74	13	13	Decreasing
2	2	Southern Cascades	37	15	47	Neutral
2	3	Upper Klamath	7	5	88	Increasing
2	4	Northern Great Basin	18	10	71	Increasing
2	5	Columbia Plateau	44	10	46	Neutral
2	6	Blue Mountains	47	17	36	Neutral
2	7	Northern Glaciated Mountains	92	5	3	Decreasing
2	8	Lower Clark Fork	89	8	3	Decreasing
2	9	Upper Clark Fork	97	2	1	Decreasing
2	10	Owyhee Uplands	41	18	41	Neutral
2	11	Upper Snake	52	23	25	Decreasing
2	12	Snake Headwaters	75	17	8	Decreasing
2	13	Central Idaho Mountains	43	22	35	Neutral
3	1	Northern Cascades	17	45	37	Neutral
3	2	Southern Cascades	0	20	80	Increasing
3	3	Upper Klamath	5	7	88	Increasing
3	4	Northern Great Basin	7	23	70	Increasing
3	5	Columbia Plateau	23	25	52	Increasing
3	6	Blue Mountains	7	15	78	Increasing
3	7	Northern Glaciated Mountains	22	37	41	Neutral
3	8	Lower Clark Fork	47	40	13	Neutral
3	9	Upper Clark Fork	71	17	13	Decreasing
3	10	Owyhee Uplands	35	15	50	Neutral
3	11	Upper Snake	10	19	71	Increasing
3	12	Snake Headwaters	14	41	45	Neutral
3	13	Central Idaho Mountains	21	48	31	Neutral
4	1	Northern Cascades	30	8	63	Increasing
4	2	Southern Cascades	45	13	42	Neutral
4	3	Upper Klamath	98	0	2	Decreasing
4	4	Northern Great Basin	69	0	31	Decreasing

Table 12—Percentage of watersheds in 3 trend categories for each family, by ecological reporting unit (ERU) (continued)

Family	ERU	ERU name	Percentage of watersheds decreasing	Percentage of watersheds neutral	Percentage of watersheds increasing	Dominant trend[a]
4	5	Columbia Plateau	83	2	15	Decreasing
4	6	Blue Mountains	53	4	42	Decreasing
4	7	Northern Glaciated Mountains	81	7	11	Decreasing
4	8	Lower Clark Fork	96	3	2	Decreasing
4	9	Upper Clark Fork	81	9	10	Decreasing
4	10	Owyhee Uplands	47	8	45	Neutral
4	11	Upper Snake	59	13	28	Decreasing
4	12	Snake Headwaters	21	7	72	Increasing
4	13	Central Idaho Mountains	35	13	52	Increasing
5	1	Northern Cascades	29	54	17	Neutral
5	2	Southern Cascades	17	44	39	Neutral
5	3	Upper Klamath	9	50	41	Neutral
5	4	Northern Great Basin	4	90	6	Neutral
5	5	Columbia Plateau	59	39	2	Decreasing
5	6	Blue Mountains	34	48	17	Neutral
5	7	Northern Glaciated Mountains	36	43	22	Neutral
5	8	Lower Clark Fork	48	43	9	Neutral
5	9	Upper Clark Fork	82	13	5	Decreasing
5	10	Owyhee Uplands	20	80	0	Neutral
5	11	Upper Snake	60	40	0	Decreasing
5	12	Snake Headwaters	43	38	19	Neutral
5	13	Central Idaho Mountains	18	52	30	Neutral
6	1	Northern Cascades	43	36	21	Neutral
6	2	Southern Cascades	39	39	22	Neutral
6	3	Upper Klamath	5	14	81	Increasing
6	4	Northern Great Basin	13	13	74	Increasing
6	5	Columbia Plateau	28	7	65	Increasing
6	6	Blue Mountains	54	15	31	Decreasing
6	7	Northern Glaciated Mountains	63	22	15	Decreasing
6	8	Lower Clark Fork	93	3	4	Decreasing
6	9	Upper Clark Fork	87	11	2	Decreasing
6	10	Owyhee Uplands	27	18	55	Increasing
6	11	Upper Snake	18	27	55	Increasing
6	12	Snake Headwaters	34	19	47	Neutral
6	13	Central Idaho Mountains	48	22	30	Neutral
7	1	Northern Cascades	36	52	13	Neutral
7	2	Southern Cascades	5	51	44	Neutral
7	3	Upper Klamath	9	29	62	Increasing
7	4	Northern Great Basin	7	86	7	Neutral
7	5	Columbia Plateau	47	29	24	Neutral
7	6	Blue Mountains	23	46	31	Neutral
7	7	Northern Glaciated Mountains	25	55	20	Neutral
7	8	Lower Clark Fork	55	37	8	Decreasing

Table 12—Percentage of watersheds in 3 trend categories for each family, by ecological reporting unit (ERU) (continued)

Family	ERU	ERU name	Percentage of watersheds decreasing	Percentage of watersheds neutral	Percentage of watersheds increasing	Dominant trend[a]
7	9	Upper Clark Fork	44	38	18	Neutral
7	10	Owyhee Uplands	24	71	5	Neutral
7	11	Upper Snake	63	29	8	Decreasing
7	12	Snake Headwaters	42	30	29	Neutral
7	13	Central Idaho Mountains	34	36	30	Neutral
8	1	Northern Cascades	81	6	13	Decreasing
8	2	Southern Cascades	76	15	8	Decreasing
8	3	Upper Klamath	64	24	12	Decreasing
8	4	Northern Great Basin	10	83	7	Neutral
8	5	Columbia Plateau	77	22	1	Decreasing
8	6	Blue Mountains	90	8	2	Decreasing
8	7	Northern Glaciated Mountains	88	3	9	Decreasing
8	8	Lower Clark Fork	99	0	1	Decreasing
8	9	Upper Clark Fork	89	5	6	Decreasing
8	10	Owyhee Uplands	24	76	0	Neutral
8	13	Central Idaho Mountains	79	15	6	Decreasing
9	1	Northern Cascades	60	0	40	Decreasing
9	2	Southern Cascades	38	29	32	Neutral
9	3	Upper Klamath	17	8	75	Increasing
9	4	Northern Great Basin	7	4	89	Increasing
9	5	Columbia Plateau	5	10	85	Increasing
9	6	Blue Mountains	34	7	59	Increasing
9	7	Northern Glaciated Mountains	0	0	100	Increasing
9	10	Owyhee Uplands	9	25	66	Increasing
9	11	Upper Snake	9	9	81	Increasing
9	12	Snake Headwaters	33	11	56	Increasing
10	1	Northern Cascades	50	10	40	Neutral
10	2	Southern Cascades	57	16	27	Decreasing
10	3	Upper Klamath	77	9	14	Decreasing
10	4	Northern Great Basin	10	88	1	Neutral
10	5	Columbia Plateau	71	28	1	Decreasing
10	6	Blue Mountains	70	19	11	Decreasing
10	7	Northern Glaciated Mountains	92	4	4	Decreasing
10	8	Lower Clark Fork	85	8	8	Decreasing
10	9	Upper Clark Fork	67	11	22	Decreasing
10	10	Owyhee Uplands	16	84	0	Neutral
10	11	Upper Snake	65	35	0	Decreasing
10	12	Snake Headwaters	68	10	22	Decreasing
10	13	Central Idaho Mountains	35	37	28	Neutral
11	1	Northern Cascades	47	45	8	Neutral
11	2	Southern Cascades	47	23	30	Neutral
11	3	Upper Klamath	20	59	20	Neutral
11	4	Northern Great Basin	11	86	2	Neutral

Table 12—Percentage of watersheds in 3 trend categories for each family, by ecological reporting unit (ERU) (continued)

Family	ERU	ERU name	Percentage of watersheds decreasing	Percentage of watersheds neutral	Percentage of watersheds increasing	Dominant trend[a]
11	5	Columbia Plateau	50	33	17	Neutral
11	6	Blue Mountains	43	41	16	Neutral
11	7	Northern Glaciated Mountains	54	31	15	Decreasing
11	8	Lower Clark Fork	100	0	0	Decreasing
11	9	Upper Clark Fork	58	30	13	Decreasing
11	10	Owyhee Uplands	24	72	4	Neutral
11	11	Upper Snake	76	22	2	Decreasing
11	12	Snake Headwaters	82	14	3	Decreasing
11	13	Central Idaho Mountains	42	30	27	Neutral
12	1	Northern Cascades	95	5	0	Decreasing
12	2	Southern Cascades	86	11	4	Decreasing
12	3	Upper Klamath	93	2	4	Decreasing
12	4	Northern Great Basin	11	88	1	Neutral
12	5	Columbia Plateau	80	19	1	Decreasing
12	6	Blue Mountains	78	11	12	Decreasing
12	7	Northern Glaciated Mountains	96	1	2	Decreasing
12	8	Lower Clark Fork	100	0	0	Decreasing
12	9	Upper Clark Fork	65	4	31	Decreasing
12	10	Owyhee Uplands	45	53	2	Neutral
12	11	Upper Snake	31	65	5	Neutral
12	12	Snake Headwaters	48	28	25	Neutral
12	13	Central Idaho Mountains	45	12	43	Neutral

[a] ERUs were classified as increasing or decreasing if >50 percent of the watersheds had positive or negative trends, respectively. ERUs not classified as increasing or decreasing were classified as neutral. See "Forming Families of Groups to Summarize Results Among Multiple Groups" in "Methods" section for details about assigning trends to watersheds.

Correlation of Habitat Trends Among Species Within Families

Relative change in source habitats was positively correlated ($P < 0.05$) for 520 (94 percent) of the 556 within-family species pairings for the 10 families that contained multiple species. Only 36 within-family species pairings were not correlated ($P > 0.05$), and only 3 (<1 percent) were negatively correlated ($P < 0.05$). Moreover, the grand mean of all correlation coefficients, calculated from the means of all species pairings within each family, was relatively high ($r = 0.52$). Mean coefficients for each family, however, varied from a low of 0.23 (family 3) to a high of 0.96 (family 9).

In general, the mean within-family coefficients were higher for families whose species were associated with a smaller, more specialized set of source habitats, and progressively lower for families whose species were associated with an increasingly larger, more diverse set of habitats. For example, mean within-family coefficients were 0.53 and 0.55 for the two families whose source habitats were restricted largely to old-forest stages (families 1 and 2). Mean within-family coefficients were similarly high (0.60 to 0.72) for the three families whose source habitats were wholly or largely restricted to rangelands (families 10, 11, and 12), and highest (0.96) for the family with the most restricted set of source habitats (family 9). By contrast, mean within-family coefficients ranged from 0.23 to 0.34 for the four families whose source habitats either spanned a broad range of forest structural stages (family 3) or encompassed diverse combinations of forest and rangeland habitats (families 5, 6, and 7).

Management implications—The correlation coefficients for species pairings within each family were less positive and more variable relative to the coefficients calculated for species pairings within each group (fig. 8). For example, <3 percent of the within-group species pairings had coefficients that were <0.20, but 6 percent of the within-family species pairings had coefficients <0.20. Moreover, the grand mean of all coefficients for the within-group species pairings was 0.66, whereas the grand mean of all within-family coefficients was 0.52.

The more variable and less positive coefficients of species pairings within familes versus those within groups is not surprising, given the more diverse set of habitats associated with species within each family versus group. These results have the following implications for any management strategy that relies on family-level habitat trends:

1. Use of the family-level habitat trends for habitat management is a coarse-filter approach. Coarse-filter management assumes that managing an appropriate amount and arrangement of all representative land areas and habitats will provide for the needs of all associated species (Hunter 1991) (see "Methods," "Designing a Hierarchical System of Single- and Multi-species Assessment"). Such an assumption, by using family-level habitat trends as the basis for a coarse-filter approach, would be tenuous when applied to individual subbasins, watersheds, or subwatersheds, given the family-level correlation results. A coarse-filter approach that relies on family-level habitat trends can likely be effective, however, in devising credible broad-scale ecosystem strategies across large geographic areas of the basin. Such family-level strategies will be more accurate and defensible when devised for areas as large as individual or multiple ERUs, or for large numbers of subbasins or watersheds. If subbasins or watersheds are used as the basis for devising family-level strategies, a minimum of 5 to 10 subbasins or 75 to 150 watersheds would be needed; such areas would provide sufficient accuracy (based on table 2) to detect the most dominant habitat trends common to most species and groups in a family, and provide sufficient geographic coverage to dampen some or much of the species variability in family-level habitat trends that occur on individual watersheds or small collections of watersheds.

2. Any coarse-filter approach based on family-level habitat trends should include an analysis of how well such an approach accommodates habitat needs for each group of species and for individual species that have been identified as having undergone strong, widespread declines in aerial extent of source habitats. Such an analysis would test how well the coarse-filter approach meets the needs of species or groups that likely require highest management attention, and allow for the coarse-filter approach to be "fine-tuned" to ensure its effectiveness for all species. For example, managers may not be compelled to devise a habitat strategy for the "Forest and Range Mosaic Family" (family 5) because most family-level trends were neutral or positive (table 12); closer examination of group-level trends within the family, however, shows that trends for group 22 (composed of California and Rocky Mountain bighorn sheep) were largely negative for most ERUs (table 10) and for the basin as a whole (table 9). Managers should check for and accommodate such results in their broad-scale ecosystem strategies.

Family 1—Low-Elevation, Old-Forest Family

Groups 1, 2, and 3 compose family 1 (table 6). These three groups include the white-headed woodpecker, white-breasted nuthatch, pygmy nuthatch, migratory population of Lewis' woodpecker, and western gray squirrel (table 6). Ranges of these species are shown in figures 3, 6, and 9 of volume 2.

Source habitats and special habitat features—All species in family 1 depend on late-seral multi- and single-storied lower montane forests as source habitats (table 11). Some family members also use old-forest cover types in the upper montane, riparian woodlands, and upland woodlands community groups (table 11; vol. 3, appendix 1, table 1). Source habitats for family 1 occur in all ERUs, but habitats were never common. Historically, these habitats typically composed less than 25 percent of the area in most watersheds (vol. 2, figs. 4a, 7a, and 10a). Today, source habitats for family 1 (vol. 2, figs. 4b, 7b, and 10b) still occur in all 13 ERUs but are particularly scarce within the Columbia

Figure 9—Trend in source habitats for family 1 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

Plateau, Upper Snake, Northern Great Basin, and Owyhee Uplands. In the remaining nine ERUs, source habitats are more common but still compose <25 percent of most watersheds.

All species in family 1 require large-diameter (>53 cm [21 in]) snags or trees with cavities for nesting, foraging, or both (vol. 3, appendix 1, table 2). The possible exception is the western gray squirrel, which uses cavities of snags and large hollow trees for nesting and resting, but these structures may not be a requirement (Ryan and Carey 1995). The Lewis' woodpecker is associated closely with recent burns and responds favorably to stand-replacing fires (see Tobalske 1997), whereas habitat for other species in family 1 is usually maintained by frequent, low-intensity burns that retain old-forest structure.

Broad-scale changes in source habitats—Source habitats declined in 70 percent of watersheds basin-wide between the historical and current periods (fig. 9). Thirteen percent of watersheds had increasing trends, and the remaining 17 percent were stable. Eleven ERUs exhibited declining trends in >50 percent of watersheds (table 12). The only ERUs with predominantly neutral trends were the Upper Klamath and Northern Great Basin ERUs, and of these, the Northern Great Basin ERU contained little habitat historically.

Declines in source habitats for family 1 are related largely to reductions in the old-forest lower montane community type. Declines in both late-seral single-layered and late-seral multi-layered lower montane occurred in all ERUs that had declining habitat trends, and these declines were considered ecologically significant except for the old-forest multi-layered stage in the Blue Mountains and Central Idaho Mountains (Hann and others 1997).

The importance of restoration for species in this family is highlighted by the magnitude of the declines. Basin-wide, the current extent of late-seral single-layered lower montane forests represents an 81-percent decline in the historical areal extent, and the extent of multi-layered forests represents a 35-percent decline (Hann and others 1997). These declines were particularly pronounced in the Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork, where nearly 100 percent of these community types have been lost (Hann and others 1997). Declines in source habitats for family 1 are among the most widespread and strongest of any declines observed for any set of species that we included in our analysis.

Primary causes for habitat trends and the associated ecological processes—Timber harvest and fire exclusion were the two primary causes for the widespread, strong decline in source habitats for family 1 (Hann and others 1997). Timber harvest has resulted in the replacement of late-seral, lower montane source habitats with mid-seral forests. Fire exclusion has resulted in a gradual shift in stand composition from shade-intolerant tree species such as ponderosa pine to shade-tolerant species such as Douglas-fir and grand fir. Additionally, human occupancy of and use of lands that historically supported lower montane forests have increased and presumably contributed to declines in source habitats.

The magnitude of decline in historical vegetation structure and composition has been greater for the lower montane community group than any other forest community groups (Hann and others 1997), partly because lower elevation forests were more accessible for logging and contained high-value, large-diameter timber. Moreover, lower elevation forests historically were subject to more frequent, light surface or underburn fire events; structures in these forests therefore were more susceptible to decreases in fire frequency than were forests at higher elevations. This combination of intensive harvest of the larger overstory trees and the exclusion of fire has created an environment favorable for the increase of shade-tolerant trees characteristic of the montane community group. The resulting forest structure and composition is not suitable for many species in family 1 because of greater density of small-diameter trees and logs, and changed species composition. For example, high stand density can make foraging difficult for the Lewis' woodpecker, an aerial insectivore, and can reduce vigor of oaks used by western gray squirrels for foraging. The loss of large-diameter trees and large snags can limit the abundance of nesting structures for the white-breasted nuthatch, pygmy nuthatch, white-headed woodpecker, and Lewis' woodpecker. A concurrent decline in large down logs has occurred, which may be of concern for other species associated with this group.

Source habitats for family 1 also shifted geographically across large areas of the basin since historical times (see fig. 9). Source habitats that underwent no change or an increase are now farther south (fig. 9) and represent a warmer average environment. Many of these environments with increasing amounts of habitat are only increasing because of fire exclusion in what would have been fire-maintained savannahs dominated by shrubs or herbs with scattered large trees. Environments with neutral changes in habitat have a complex combination of areas with (1) slow succession rates, such that change in response to fire exclusion has not affected broad-scale cover type and structural stage composition; or (2) a neutralizing mix of late-seral forest increases from fires exclusion in savannah types and decreases from timber harvest. The habitats where declines occurred are to the north with cooler average temperatures and higher habitat productivity.

Finally, extensive fragmentation of historical landscape patterns has occurred in lower elevation watersheds that support habitats of family 1 (Hann and others 1997, Hessburg and others 1999). Broad-scale departure as a result of fragmented ownership patterns, high road densities, and timber harvest occurred in 8 of the 13 ERUs.

Restoration of source habitats will be difficult for family 1 because the existing composition and structure of vegetation represents a substantial departure from historical conditions. The current vegetation is more susceptible to stand-replacing fires and increasingly vulnerable to insect- and disease-related tree mortality. These conditions may require active management to restore more desirable forest structure and composition.

Other factors affecting the family—Roads may facilitate a reduction in the density of large-diameter trees and snags as habitat for family 1, as suggested by the lower density of large-diameter trees, snags, and logs associated with roaded areas (Hann and others 1997). Roads also likely facilitate the legal and illegal shooting of western gray squirrels in association with increased human access provided by roads.

Issues and strategies for conservation—The following issues and strategies for family 1 relate to declines in source habitats and special habitat features.

Issues—

1. Basin-wide decline in late-seral interior and Pacific ponderosa pine and large (>53 cm [21 in]) overstory and emergent trees.
2. Basin-wide loss of large-diameter snags (>53 cm [21 in]).
3. Declines in old-forest aspen and cottonwood/willow.
4. Declines in shrub and herb understories of montane and lower montane forests in response to increased density of small trees and downed wood, litter, and duff.
5. Loss or decline of oak trees as a cover type and within other cover types.
6. Fragmentation of lower elevation landscape patterns.
7. Exclusion of light surface or underburn fires that occurred frequently and extensively.
8. Broad-scale shift of family 1 habitats to environments with warmer average temperatures.

Strategies—The following strategies could be considered to address issues related to species belonging to family 1:

1a. (To address issue no. 1) Retain stands of interior and Pacific ponderosa pine where old-forest conditions are present, and manage to promote their long-term sustainability through the use of prescribed burning and understory thinning.

1b. (To address issue no. 1) Primarily in the northern parts of the basin where old forests have transitioned to mid-seral stages, identify mid-seral stands that could be brought into old-forest conditions in the near future and use appropriate silvicultural activities to encourage this development.

2. (To address issue no. 2) As a short-term strategy retain all large-diameter (>53 cm [21 in] d.b.h.) ponderosa pine, cottonwood, Douglas-fir, and western larch snags within the basin, preferably in clumps, and provide opportunities for snag recruitment throughout the montane and lower montane

Figure 10—Trend in source habitats for family 2 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

communities. As a long-term strategy, conduct mid-scale assessment of species snag use and the dynamics of snags in landscapes and adjust the strategy or groups of subbasins.

3. (To address issue no. 3) Within all ERUs with cottonwood-willow stands, maintain existing old forests, and identify younger stands for eventual development of old-forest structural conditions. Return natural hydrologic regimes to large river systems, particularly in the Central Idaho Mountains, Upper Snake, and Snake Headwaters ERUs where large riparian cottonwood woodlands still remain.

4. (To address issue no. 4) Rejuvenate and enhance shrub and herb understory of lower montane community groups (old-forest ponderosa pine) in the Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and Blue Mountains ERUs. Throughout the range of the Lewis' woodpecker, allow some stand-replacing wildfires to burn in lower montane wilderness and other lands managed with a natural process emphasis (for example designated wilderness, research natural areas, and areas of critical environmental concern). Such opportunities can be found particularly in the Central Idaho Mountains, Blue Mountains, and Snake Headwaters ERUs, and in western Montana. Minimize mechanized harvest and site-preparation activities that increase susceptibility to exotic and noxious weed invasion, soil erosion, or high densities of tree regeneration.

5. (To address issue no. 5) Manage for the maintenance and restoration of oak woodlands, particularly along the eastern flank of the Cascade Range within and between existing populations of western gray squirrel.

6. (To address issue no. 6) Look for opportunities to acquire lands in lower elevation forest and forest-rangeland mosaics. Close and restore excess roads to reduce fragmentation of landscapes by roads. Use thinning to repattern landscapes to a more native condition. Where natural process areas occur, prioritize road closures and restoration in adjacent watershed to increase the interior core of habitats with native patterns.

7. (To address issue no. 7) Continue a strategy of wildfire suppression of stand-replacing fires except where such fires would benefit habitat for Lewis' woodpecker under the conditions specified in issue no. 4. Use prescribed fire, timber harvest, and thinning to change forest composition and structure to reduce risk of stand-replacing wildfires and shift to maintenance with prescribed underburn fires.

Family 2—Broad-Elevation, Old-Forest Family

Family 2 consists of 24 species of birds and mammals within groups 4 to 13 (table 6). Example species are marten, fisher, flammulated owl, northern goshawk, pileated woodpecker, boreal owl, northern flying squirrel, and black-backed woodpecker. Ranges of each species in family 2 are shown in figures 12, 15, 18, 21, 24, 27, 30, 33, 36, and 39, volume 2.

Source habitats and special habitat features—All species in family 2 use late-seral multi- and single-layered stages of the montane community as source habitats. Source habitats for some species also include late-seral stages of the subalpine community or the lower montane community, or both (table 11). In addition, source habitats for the northern flying squirrel include the understory reinitiation stage of most cover types within subalpine, montane, lower montane, and riparian woodland communities. Source habitats for family 2 overlap those of family 1 but encompass a broader array of cover types and elevations than habitats for family 1 (vol. 3, appendix 1, table 1). Species of family 1 are primarily restricted to lower elevation forests of interior Douglas-fir and ponderosa pine forests.

Fifteen species in family 2 depend on snags for nesting or foraging; four of these species also use down logs to meet life requisites; four species also use large, hollow trees (vol. 3, appendix 1, table 2). Downed logs, lichens, and fungi of late-seral forests provide habitat for many prey species of northern goshawk, flammulated owl, boreal owl, great gray owl, fisher, and marten (Gibilsco and others 1995, Hayward and Verner 1994, Reynolds and others 1992, Thompson and Colgan 1987). Stand-replacing, large burns and other beetle-infested stands provide high concentrations of prey (wood-boring beetles) for three-toed and black-backed woodpeckers (Koplin 1969). Juxtaposition of early- and late-seral stages is needed to meet all aspects of life functions for the silver-haired bat, hoary bat, and great gray owl, which are identified as contrast species (see "Glossary," vol. 3).

Late-seral source habitats used by marten, fisher, and boreal owl, however, may be negatively affected by increased fragmentation brought about by juxtaposing their need for late-seral habitats with early-seral habitats (Hargis 1996, Hayward and Verner 1994, Jones 1991). Thus, the negative response of marten, fisher, and boreal owl to juxtaposition of their source habitats with forest openings versus the positive response of silver-haired bat, hoary bat, and great gray owl to these same conditions must be considered when managing the spatial arrangement of early- versus late-seral habitats for species in family 2.

Broad-scale changes in source habitats—Source habitats for family 2 declined in most watersheds. Basin-wide, 59 percent of watersheds exhibited declining trends, 28 percent increased, and the remaining 13 percent were neutral (fig.10). Watersheds with declining trend were concentrated in the northern part of the basin and in the Snake River drainage; those with increasing trend were mostly in the south-central and southwestern areas of the basin (fig. 10). The Northern Cascades, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, Upper Snake, and Snake Headwaters ERUs had declining trends in more than 50 percent of their watersheds (table 12). The Blue Mountains, Central Idaho Mountains, and Columbia Plateau had predominantly neutral trends, but nevertheless, each of these ERUs had a substantial percentage of watersheds with declining trends: 47 percent in the Blue Mountains, 43 percent in the Central Idaho Mountains, and 44 percent in the Columbia Plateau (table 12). Watersheds with increasing trends were concentrated in the Upper Klamath and Northern Great Basin ERUs (table 12; fig. 10). Abundance of source habitats in the Northern Great Basin, however, was minor as there are few watersheds within this ERU that contain source habitats for family 2.

Although source habitats for family 2 declined in most watersheds, not all species-level trends for members of family 2 exhibited a declining trend. Exceptions were three-toed woodpecker, Vaux's swift, great gray owl, and woodland caribou (tables 7, 8). Source habitats for the three-toed woodpecker exhibited positive trends, and those of the woodland caribou and great gray owl were neutral primarily because their habitats do not include the lower elevation old forests of Sierra Nevada mixed-conifer, western white pine, or ponderosa pine (vol. 3, appendix 1, table 1), which generally declined more than upper elevational cover types (vol. 3, appendix 1, table 4). Source habitats of the Vaux's swift were neutral primarily because of its unique combination of source habitats and range distribution. That is, Vaux's swift uses only the montane terrestrial community, which had a mixture of declining and increasing trends in areal extent basin-wide (Hann and others 1997; vol. 3, appendix 1, table 4), and its range does not include the Snake Headwaters and Upper Snake ERUs (vol. 2, fig. 18), where significant declining trends were projected for family 2.

Primary causes for habitat trends and the associated ecological processes—Timber harvest techniques, exclusion of fire, and resulting changes in insect and disease infestation dynamics are the primary causes for trends in source habitats for family 2. Suppression of wildfires has resulted in a shift in stand composition from shade-intolerant to shade-tolerant species within lower montane, montane, and subalpine communities. Timber harvest activities have had a similar effect, favoring the removal of shade-intolerant tree species (such as western larch, western white pine, and ponderosa pine), and the retention and growth of shade-tolerant understories, which are more susceptible to fire, insect, and disease (such as grand fir, western redcedar, western hemlock, and Douglas-fir).

Declines in source habitats were particularly associated with late-seral lower montane single-layer forest, which was projected to have had an 80-percent decline in areal extent since the historical period and with late-seral subalpine multi-layer forest, which had a projected decline of 64 percent (Hann and others 1997). Although of less magnitude, declines also occurred in late-seral forests of the montane and subalpine terrestrial communities. There was an ecologically significant increase in the late-seral single-layer subalpine community, but this only affected a relatively small area. The areal extent of late-seral lower montane, montane, and subalpine forests were found to be below their historical minimum in 78, 59, and 63 percent of the subbasins, respectively (Hann and others 1997).

There was a substantial spatial shift from historical to current in the distribution of family 2 habitat that was somewhat similar to that of family 1 (see fig. 10). As with family 1, the areas with neutral or increasing trends were generally in the southern part of the basin, whereas the areas with decreasing trends were farther north. Patterns of family 2 are not, however, identical to those for family 1. Family 2 habitats often increased where family 1 habitats were neutral. This is generally because successional processes are more rapid in the montane and subalpine environments than they are in lower montane environments, so these habitats for family 2 responded more quickly to fire suppression than those for family 1.

As with family 1, the areas of greatest decline are to the north or in the high elevations of the Snake Headwaters where the combination of timber harvest, fire exclusion, and insect-disease mortality of stressed trees is causing a shift to mid-seral or early-seral forests. The area of greatest increase was in the Upper Klamath where there were vast increases in both single-layer and multi-layer montane old forests (Hann and others 1997). These late-seral forests in the Klamath, however, have been extensively affected by selective harvest and fire exclusion and may not have old-forest characteristics at the mid scale (Hessburg and others 1999).

Other factors affecting the family—Roads increase human access into source habitats and have the potential to negatively affect most species in family 2. Fourteen species in family 2 rely on snags for nesting and foraging, and snag densities are lower in roaded versus unroaded areas of the basin (Hann and others 1997). Survival of marten and fisher can decline because of fur harvesting if trapping is not regulated carefully (Fortin and Cantin 1994, Jones 1991, Quick 1956). Roads potentially increase trapping pressure on marten and fisher, resulting in significantly higher captures in roaded versus unroaded areas (Hodgman and others 1994) and in logged versus unlogged areas (Thompson 1994). Roads also increase mortality of woodland caribou. Fatal collisions with automobiles occur on open roads in woodland caribou habitat (Scott and Servheen 1985). A high percentage of the annual mortality in the 1980s was attributed to illegal harvest by hunters and poachers (Scott and Servheen 1985), and both legal and illegal take of other ungulate species have been facilitated by road access (for example, Cole and others 1997).

Patterns of road density also are associated with departures from the historical landscape patterns. Broad-scale landscape patterns were found to be highly fragmented in correlation with low to moderate elevation and proximity to moderate or higher road densities (Hann and others 1997). Fragmentation and substantial declines of the late-seral lower montane forests, simplification of the montane forest, and fragmentation of the subalpine forest resulted in broad-scale departures from historical landscape patterns for 8 of the 13 ERUs (Hann and others 1997).

Issues and strategies for conservation—The primary issues for family 2 relate to source habitats, special habitat features, and road-related human disturbances.

Issues—

1. Declines in late-seral forests of subalpine, montane, and lower montane communities and associated attributes such as large trees, large snag, large down logs, lichen, and fungi.

2. Tradeoffs between source habitats for species in family 2 and habitats for species in family 1.

3. Balancing the fragmentation of late-seral habitats for marten, fisher, and boreal owl versus juxtaposition of early- and late-seral habitats for silver-haired bat, hoary bat, and great gray owl.

4. Broad-scale departures from historical landscape patterns.

5. Negative effects of road-related human activities.

6. Reduction in the extent of frequent, light underburning and light surface fires.

Strategies—The following strategies could be considered to address issues related to species belonging to family 2. It is important that source habitats for both families 1 and 2 be considered together in the design of conservation strategies. For example, efforts to restore the composition and structure of lower montane forests may involve thinning or the use of fire in areas where shade-tolerant species now dominate. Such areas currently serve as source habitats for many species in family 2. Consequently, the maintenance of an appropriate network of these habitats would be essential for restoring lower montane forests in a manner that provides for both families.

The historical ranges of area covered by these habitats could be used as one guide to establishing this balance (Hann and others 1997). In addition, the disturbance processes that create and maintain these habitats could be considered in determining where habitats are to be maintained. Sites where shade-tolerant species are at lower risk of broad-scale loss because of insects, disease, and fire could be managed to maintain those habitats for family 2, whereas areas prone to frequent disturbance could be managed to simulate the disturbance processes that historically maintained the composition and structure of lower montane forests and thereby benefit family 1.

A similar strategy could be used in the ERUs where habitat has clearly increased for both families, such as parts of the Southern Cascades, Upper Klamath, and Northern Great Basin. Here, both families would benefit from retention of a network of old-forest habitats with management also aimed at increasing the extent of fire-maintained communities.

The following strategies have been developed to address issues related to the species in family 2, for application in a spatial context that also meets the needs of family 1, as described above.

1a. (To address issue no. 1) Retain stands of late-seral forests in the subalpine, montane, and lower montane communities; actively manage to promote their long-term sustainability; and manage young stands to develop late-seral characteristics. In the Southern Cascades and Upper Klamath ERUs, prescribed burns and understory thinning may be required to avoid loss of late-seral forests. In the Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, Upper Snake and Snake Headwaters ERUs, it may be necessary to identify mid-seral forests in the lower montane community that could be brought to late-seral conditions because late-seral lower montane forests that can be mapped at the broad scale have been eliminated in these areas.

1b. (To address issue no. 1) As a short-term strategy, retain all large-diameter (>53 cm [21 in] d.b.h.) snags and large trees in the subalpine, montane, and lower montane communities, preferably in clumps, and provide opportunities for snag recruitment. As a long-term strategy, conduct mid-scale assessment to determine biophysical snag dynamics at a watershed scale and adjust the strategy by subbasin or groups of subbasins.

1c. (To address issue no. 1) Include family 2 conservation within a larger, ecosystem context that addresses management of primary cavity nesters and the small-mammal prey base for species within family 2. This includes maintenance of old-forest attributes such as coarse woody debris, fungi, and lichens.

2. (To address issue no. 2) Integrate the short-term strategy for conservation of current family 2 habitat with conservation of current family 1 habitat through mid-scale step-down assessment. Concurrently, develop a long-term strategy to repattern watersheds basin-wide to a mosaic of sustainable levels of family 1 and family 2 habitats.

3. (To address issue no. 3) Increase connectivity of disjunct habitat patches and prevent further reduction of large blocks of contiguous habitat. For martens and fishers, provide large contiguous areas of forested habitat at the home range scale. Notably, these species are generally not affected by forest openings less than about 120 m (390 ft) wide (Hargis and McCullough 1984, Koehler and Hornocker 1977), so large contiguous areas with small forest openings would also benefit the species with contrasting habitat needs: silver-haired bat, hoary bat, and great gray owl. For boreal owls, evaluate the links among subpopulations and use that information to identify areas that are highest priority for retention and restoration of habitat. This is of particular concern in the Northern Glaciated Mountains, Upper Clark Fork, and Lower Clark Fork ERUs, where reduction in the extent of source habitats has increased the isolation of remaining habitat patches.

4. (To address issue no. 4) Integrate a long-term strategy to repattern forest and forest-range landscape mosaics at the watershed scale through mid-scale step-down assessment. Develop patterns that consider issue no. 3 (fragmentation) in context of historical patterns as well as the biophysical succession-disturbance regimes.

5. (To address issue no. 5) Minimize or avoid road construction within late-seral forests. Obliterate or restrict use of roads after timber harvests and other management activities. Give special consideration to obliteration of roads that would help reduce poaching of caribou.

6. (To address issue no. 6) Continue a strategy of wildfire suppression in most managed forests while allowing stand-replacing wildfires to burn in wilderness areas,

areas of critical environmental concern (ACECs), and other natural process areas. Stand-replacing wildfires in such natural process areas are of particular benefit to black-backed and three-toed woodpeckers. In managed areas, use prescribed fire, timber harvest and thinning to change forest composition and structure to reduce risk of stand-replacement wildfires and loss of large emergent trees and overstory trees to benefit other species in family 2. Shift fire regimes to mixed fire behavior (as defined by Hann and others 1997), underburns, and creeping-irregular disturbance events through use of prescribed fire.

Family 3—Forest Mosaic Family

Family 3 is composed of groups 14 through 17 and consists of 6 species: the hermit warbler, pygmy shrew, wolverine, lynx, blue grouse (summer habitat only), and mountain quail (summer) (table 6). Ranges of these species are shown in figs. 39, 42, 45, 48, and 51, volume 2.

Source habitats and special habitat features— Species within this family tend to be habitat generalists in montane forests; most species also use subalpine forests, lower montane forests, or riparian woodlands as source habitats (table 11). The blue grouse and mountain quail are the only species in this family that use upland shrublands, and during summer, the blue grouse also uses upland herblands. Source habitats generally include all structural stages.

Downed logs are a special habitat feature for the wolverine and lynx because they serve as potential resting and denning sites (vol. 3, appendix 1, table 2). Wolverines also have been found to use talus slopes as denning sites (Copeland 1996), and therefore talus is considered a special habitat feature for this species.

Special habitat features for the mountain quail are the shrub-herb understory in forest communities and shrub-herb riparian vegetation (vol. 3, appendix 1, table 2). Areas with abundant shrubs in the understory are used for cover as well as forage (Brennan and others 1987, Zwickel 1992). Riparian areas appear to be preferred, because mountain quail within the basin are primarily found within 100 to 200 m (328 to 656 ft) of a water source (Brennan 1989).

The blue grouse is considered a contrast species (vol. 3, appendix 1, table 2) because the species requires a juxtaposition of forest and nonforest vegetation structure to meet all aspects of its ecology (see "Glossary," vol. 3, for definition of contrast species and related terms). Blue grouse nest on the ground but use trees for roosting and flush into trees when disturbed. Breeding areas are generally on the forest/shrub interface (Zwickel 1992).

Broad-scale changes in source habitats—Trends in source habitat extent differ across the basin for family 3, with neutral trends predominating. Within all watersheds having source habitats, 22 percent exhibited declining trends, 32 percent had increasing trends, and 46 percent were neutral (fig. 11). Six ERUs had increasing trends in >50 percent of watersheds, six had neutral trends, and only the Upper Clark Fork ERU had predominantly decreasing trends (table 12). Increasing trends were mostly in the south and central ERUs: the Southern Cascades, Upper Klamath, Northern Great Basin, Columbia Plateau, Blue Mountains, and Upper Snake ERUs (fig. 11, table 12).

Although the overall extent of source habitats for family 3 changed little since the historical period, there were notable changes in the extent of terrestrial community types that compose source habitat. Within the lower montane community, ecologically significant declines were projected basin-wide for early- and late-seral stages, but these were partially offset by ecologically significant increases in mid-seral lower montane forests (Hann and others 1997). There also were contrasting trends among the various structural stages of the subalpine community: ecologically significant decreases in late-seral multi-layer forests, and ecologically significant increases in late-seral single-layer and early-seral forests. Within the montane community, mid-seral structures exhibited ecologically significant increases throughout the basin, whereas there were declines in both early- and late-seral structures (Hann and others 1997). The Upper Clark Fork had declining trends in 71 percent of watersheds (table 12) because seven of nine communities with significantly declining trends decreased by more than 80 percent, and these declines were only partially offset by increases in mid-seral forests (Hann and others 1997).

Figure 11—Trend in source habitats for family 3 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

Primary causes for habitat trends and the associated ecological processes—Although forest habitats as a whole for the forest mosaic family did not show significant broad-scale change from historical to current periods, there were substantial changes in community structure and spatial distribution. Early- and mid-seral montane forests were influenced by cycling disturbance regimes that moved mid-seral to early-seral condition while early-seral forest succeeded to mid-seral condition. Because of these transitions, much of the current early-seral forest lacks the historical structure, which included large snags and large emergent trees that survived crown fires, clumps of upland trees that survived because of mixed fire behavior, narrow stringers of old-forest structure in riparian, and large down logs (Hann and others 1997, Hessburg and others 1999). In essence, timber harvest practices substantially simplified the fine-scale attributes of early-seral patches. In addition, harvested early-seral areas have more disturbed soil and are more heavily infested by exotic plants such as Canada thistle and spotted knapweed instead of native understory herbs and shrubs.

Because much of this change in forest structure resulted from management activity, the change can be correlated with road density. Mid-seral patches in areas of moderate to high road densities declined in densities of large trees, large snags, and large down logs, but increased in small tree density, small down wood, and litter-duff depths (Hann and others 1997, Hessburg and others 1999). In contrast, mid-seral patches in areas of low road densities still retained the large emergent tree, large snag, and large down log components but had similar trends of increased small-tree density, small down wood, and litter-duff depth. These changes in fine-scale components of mid-seral patches in proximity to roads were attributed to a complex combination of timber harvest, woodcutting, fire exclusion, blister rust mortality of western white pine and whitebark pine, and increased insect-disease tree mortality that resulted from harvest-induced changes in tree composition to more susceptible species (Hessburg and others 1999). Changes in areas of low road densities or unroaded areas were attributed primarily to fire exclusion, effects of blister rust mortality, and increased insect-disease mortality because of competition-induced stress from high small-tree densities.

Another significant transition from the historical to current period was the shift of fire-maintained upland herbland to mid-seral lower montane forests (1.3 percent basin-wide) (Hann and others 1997). The analysis of Hessburg and others (1999) and Hann and others (1997) indicated that the fire-maintained upland herbland was typically a savannah with scattered large ponderosa pine and Douglas-fir trees and snags. The shift of this type to relatively dense, stressed mid-seral ponderosa pine and Douglas-fir was attributed primarily to fire exclusion and excessive livestock grazing, which decreased the competitive ability of the native grasses.

A substantial spatial shift also occurred from historical to current periods in the distribution of habitats for family 3 (fig. 11). Watersheds with decreasing trends generally occurred to the north and east in a mosaic with watersheds that showed no change. The increases generally occurred to the south and west. In the North Cascades and Northern Glaciated Mountains ERUs, some watersheds with increasing trends were scattered in a mosaic with watersheds with neutral trends.

Early-seral lower montane and montane departure classes with less than the historical range of variability (HRV) minimum occurred in 79 and 44 percent of subbasins, respectively, whereas early-seral subalpine forests occurred above the HRV maximum in 56 percent of subbasins (Hann and others 1997). Mid-seral lower montane, montane, and subalpine communities had levels of subbasin departure greater than the historical maximum for 58, 57, and 30 percent, respectively. Late-seral lower montane, montane, and subalpine had levels of subbasin departure lower than the historical minimum for 78, 59, and 63 percent, respectively.

Of particular pertinence to habitats for family 3 was the fact that departure of landscape mosaic pattern was high in 8 of the 13 ERUs for the current period compared to the historical period (Hann and others 1997). Broad-scale mosaic patterns were moderately fragmented in 5 of the 13 ERUs, whereas moderately simplified in 7 of 13 ERUs. The implication for family 3 forest habitat generalists is that fragmented landscapes could lack adequate connectivity, whereas simplified landscapes could lack important structural components. This trend is supported by the terrestrial community departures discussed earlier, which indicate that subbasins typically have less diversity and evenness (simplified) of communities than historically.

The patterns of some subbasins are more fragmented (more and smaller patches), whereas other subbasins are more homogeneous. The mid-scale assessment of Hessburg and others (1999) confirmed a similar trend at the watershed scale.

Family 3 may be one of the best families to use as an indication of context for forests of the basin. The lack of overall change in amount of forests could indicate that the general decline of some species in this family may not be habitat-related. Some members of the family may have declined, in part, because of hunting, trapping, or other negative interactions with humans. It is also plausible, however, that the population declines are partially a result of change in landscape pattern and simplification of several forest attributes that have occurred, and continue to occur across the basin, among subbasins, and within subbasins, watersheds, and terrestrial communities.

Other factors affecting the family—Trapping can be a significant source of mortality for wolverine (Banci 1994) and lynx (Bailey and others 1986, Carbyn and Patriquin 1983, Mech 1980, Nellis and others 1972, Parker and others 1983, Ward and Krebs 1985). Currently in the basin, wolverine can be trapped in Montana (Banci 1994). Increased roads have provided trappers greater access to lynx and wolverine populations.

Other forms of human disturbance such as heliskiing, snowmobiles, backcountry skiing, logging, hunting, and summer recreation have been suggested as having potentially negative effects on wolverines and lynx, but the effects are not well documented (Copeland 1996, Hornocker and Hash 1981, ICBEMP 1996c, Koehler and Brittell 1990). Most of these recreational activities occur, however, in high-elevation areas used as denning sites by wolverine, and production of young at denning sites is considered a primary factor limiting wolverine population growth (Copeland 1996, Magoun and Copeland 1998).

Low-elevation riparian shrub habitat is of primary importance to quail, especially during severe winters. Hydroelectric impoundments along the Columbia River and its tributaries have eliminated thousands of acres of habitat by flooding low-elevation, primarily winter, habitat for mountain quail (Brennan 1990).

Other factors that have negatively impacted riparian shrublands are historical livestock grazing practices, agriculture, excessive recreational use, encroachment of exotic plants, and road construction (Brennan 1990, Murray 1938, Vogel and Reese 1995). Basin-wide analysis of riparian vegetation found significant changes, including widespread declines in riparian shrublands (Lee and others 1997, Quigley and others 1996). Because of the scale of our analysis and the fine-scale nature of riparian shrubland habitats, the results of our analysis likely do not reveal the true loss in this important habitat component for mountain quail.

Issues and strategies for conservation—At the broad-scale, source habitats for family 3 have not declined to the extent observed for families 1 and 2 because the species in this family are capable of using a wider variety of cover types and structural stages than the species in the two old-forest families. Conservation strategies proposed for families 1 and 2 generally will benefit broad-scale habitats for family 3. There are additional issues and strategies relative to quality of habitat and effects of changes in landscape pattern and simplification of forests. The following issues and strategies are provided:

Issues—

1. Potentially negative impacts of human disturbance on wolverine and lynx populations.

2. Loss of riparian shrubland for mountain quail at finer scales than this broad-scale assessment.

3. Changes in landscape pattern and simplification of forests across subbasins, within subbasins and watersheds, and within terrestrial communities.

Strategies—The following strategies could be considered to address issues related to species belonging to family 3:

1a. (To address issue no. 1) Provide large areas with low road density and minimal human disturbance for wolverine and lynx, especially where populations are known to occur. Manage human activities and road access to minimize human disturbance in areas of known populations.

1b. (To address issue no. 1) Manage wolverine and lynx in a metapopulation context, and provide adequate links among existing populations. Areas supporting dispersal likely would not require the same habitat attributes needed to support self-sustaining populations (Banci 1994).

2. (To address issue no. 2) Maintain and restore riparian shrublands through restoration of historical hydrologic regimes where feasible, through control of livestock grazing, and through better management of roads and recreation.

3. (To address issue no. 3) Conduct mid-scale step-down assessment of current conditions relative to landscape departure patterns of succession-disturbance regimes. Focus short-term restoration of watersheds on those that depart greatly from succession-disturbance regimes, that do not contain susceptible populations of species of high conservation concern, and that are at high risk of loss of biophysical capability. In such watersheds, continue suppression of stand-replacing, high-severity wildfires, and initiate prescribed fire appropriate to the biophysical succession-disturbance regime and timed to protect biophysical capability.

Family 4—Early-Seral Montane and Lower Montane Family

This family has one member, the lazuli bunting (group 18). Its range is shown in vol. 2, figure 54.

Source habitat and special habitat features—The lazuli bunting was assigned a separate family because of its unique dependence on early-seral, shrub-dominated conditions in forested environments. Source habitats for the family were defined as the stand initiation stages of subalpine, montane, lower montane, and riparian woodland communities (table 11; vol. 3, appendix 1, table 1). Most cover types that serve as source habitat are in the montane community.

Broad-scale changes in source habitats—Source habitats declined in 60 percent of watersheds basin-wide between the historical and current periods (fig. 12). Seven percent of watersheds had neutral trends, and 33 percent had increasing trends. At least 50 percent of watersheds had decreasing trends in eight ERUs: Upper Klamath, Northern Great Basin, Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and Upper Snake (table 12). Habitats increased in at least 50 percent of watersheds in the Northern Cascades, Snake Headwaters, and Central Idaho Mountains. Trends were mixed in the Southern Cascades and Owyhee Uplands ERUs.

Ecologically significant increases occurred in early-seral subalpine forests in all three ERUs with positive trends, and early-seral montane forests increased in two of them (Hann and others 1997). Within the eight ERUs that showed overall declines in source habitats, early-seral lower montane forests underwent ecologically significant declines in all of them, and early-seral montane forests declined in five of them.

Primary causes for habitat trends and the associated ecological processes—Fire suppression and the frequency and rate of timber harvest are the main causes for the widespread, strong decline in early-seral source habitats for family 4. In particular, Hann and others (1997) found a substantial basin-wide decline of early-seral lower montane forests (-77 percent) and a slight decline in early-seral montane (-8 percent). In addition, Hann and others (1997) found high levels of HRV departure for early-seral habitats in lower montane and montane forests, reflecting a combination of intensive timber harvest, fire suppression, roading, and invasion of exotic plants. This high HRV departure in early-seral habitats was associated with a substantial reduction in patch size and habitat quality (Hessburg and others 1999).

Spatial trends in source habitats for lazuli bunting resulted from variable types and intensities of timber harvest concurrent with fire suppression across the basin. Recent timber harvest has increased areas of the stand initiation stage in some areas, whereas fire suppression has tended to decrease area of the stand initiation stage to a much larger extent (Hann and others 1997).

Trends for family 4 were spatially disjunct (fig. 12). Increases occurred in the Northern Cascades, Central Idaho Mountains, and Snake Headwaters in response to wildfires and some timber harvest. Decreases occurred throughout much of the rest of the basin in response to the overwhelming effects of fire exclusion, with few watersheds showing a neutral response.

Figure 12—Trend in source habitats for family 4 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

In general, declines occurred in the more mesic environments with milder temperatures and higher productivity. By contrast, increases occurred in environments with cooler average temperatures and lower productivity.

Of particular concern relative to the early-seral structure is the finding of Hann and others (1997) and Hessburg and others (1999) that current conditions do not resemble historical conditions at a patch scale. Early-seral communities historically were found to have scattered large tree emergents that survived stand-replacing and mixed-fire events as well as large- and medium-size snags. Current early-seral communities commonly are now devoid of large tree emergents and snags, have comparatively high levels of disturbed soil, and contain exotic weeds. In addition, the commonly used 5-year regeneration objective of accelerating the regeneration process by planting may have shortened the time that stands remain in the early-seral stage (Hann and others 1997). Planting in postfire habitats also shortens the duration of the stand-initiation stage. The practice of planting also reduces the abundance of herb, forb, and shrub structure from early-seral stands.

Other factors affecting the family—Hutto (1995) found that lazuli buntings demonstrated a strong positive response to early successional burned forests resulting from stand-replacing fires in western Montana and northern Wyoming. In addition, lazuli buntings are Neotropical migrants and thus are affected by factors outside of their breeding habitat within the basin.

Issues and strategies for conservation—The primary issues and strategies for family 4 relate to declines in source habitats.

Issues—

1. Reduction in early-seral terrestrial communities.

2. Altered frequency of stand-replacement fires.

3. Reduction of shrubs in early-seral vegetation types.

Strategies—The following strategies could be considered to address issues related to species belonging to family 4. Four broad-scale strategies would be effective in improving habitat for lazuli buntings and other postfire-dependant species:

1. (To address issues no. 1 and no. 2) Restore fire as an ecological process in the montane and lower montane community groups.

2. (To address issues no. 1 and no. 2) Implement silvicultural strategies and practices that result in composition and structure of vegetation that mimic effects of historical fire regimes.

3. (To address issue no. 3) Allow natural development of early-seral and postfire habitats to increase the representation of early-seral shrubs where appropriate for the biophysical environment. Change reforestation goals to allow for development and maintenance of postfire habitats that are dominated by shrubs and herbs.

Family 5—Forest and Range Mosaic Family

Family 5 consists of groups 19, 20, 21, and 22, which include the gray wolf, grizzly bear, mountain goat, long-eared owl, and two subspecies of bighorn sheep (table 6). Ranges of these species are shown in figures 57, 60, 63, and 66, volume 2.

Source habitats and special habitat features—Species in family 5 use a broad range of forest, woodlands, and rangelands as source habitats (table 11; vol. 3, appendix 1, table 1). Source habitats include all terrestrial community groups except for exotics and agriculture. The Rocky Mountain and California bighorn sheep differ from other family members in that they do not use habitats in the montane, lower montane, and upland woodland community groups. The long-eared owl also does not use alpine or subalpine community groups as source habitats.

The long-eared owl is considered a contrast species, requiring a juxtaposition of contrasting vegetation structures to meet all life history needs (vol. 3, appendix 1, table 2). Where forests are adjacent to open areas, trees are typically used for nest sites. Where forests are not present, nests are placed in tall shrubs (Holt 1997). Special habitat features for the mountain goat and both subspecies of bighorn sheep are cliffs, talus, and shrub/herb riparian vegetation (vol. 3, appendix 1, table 2). Cliffs provide important escape terrain, and shrub/herb riparian vegetation provides high-quality forage for these mountain-dwelling herbivores. No special habitat features were identified for

Figure 13—Trend in source habitats for family 5 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

the gray wolf or grizzly bear, although the grizzly bear also seeks talus areas and shrub/herb riparian vegetation for high-quality forage during summer.

Broad-scale changes in source habitats—Basin-wide, 51 percent of watersheds had stable trends in source habitats, 35 percent had decreasing trends, and 14 percent had increasing trends (fig. 13). The greatest declines were in the Lower Clark Fork ERU, where 82 percent of watersheds showed declines (table 12). Other ERUs with decreasing trends in >50 percent of watersheds were the northern half of the Columbia Plateau, Upper Clark Fork, and Upper Snake ERUs. Increasing trends for family 5 were mostly in the Upper Klamath and Central Idaho Mountains ERUs and in portions of the Northern Glaciated Mountains ERU (fig. 13), but the average trends in all three ERUs was neutral (table 12).

Primary causes for habitat trends and the associated ecological processes—Trends were spatially disjunct and correlated with human-caused effects. Declines occurred in correlation with invasion of exotic plants and agriculture and urban development in environments with generally longer growing seasons and more productive soils. Neutral areas occurred primarily in the rangelands, dry forest, or cold forest where productivity is lower and thus where less agricultural and urban development occur. Cover type and structural stage transitions in the montane and lower montane community groups resulted in no net change in source habitats for family 5. For example, extensive declines in old-forest structural stages of all forest cover types have occurred (Hann and others 1997), but these losses have been offset by increases in mid-seral stages that also serve as source habitats for the gray wolf, grizzly bear, long-eared owl, and to a lesser extent, the mountain goat. Bighorn sheep do not use most structural stages of forest cover types, so the structural transitions that occurred did not affect their source habitat. Ecologically significant losses of all structural stages occurred in western white pine, whitebark pine, western larch, and limber pine (Hann and others 1997).

Within nonforest terrestrial communities, upland herbland and upland shrubland have strongly declined, whereas three new terrestrial communities, urban, agriculture, and exotic herbland, have emerged since the historical period (Hann and others 1997), none of which serves as source habitat for this family. Source habitat declines in the Columbia Plateau and Upper Snake ERUs were attributed primarily to the conversion of upland shrubland and upland herbland to agriculture (Hann and others 1997). Currently, 42 percent of the Columbia Plateau and 36 percent of the Upper Snake ERU are now in agriculture. Similar transitions occurred in the Lower and Upper Clark Fork ERUs, although the areal extent of the transitions was less.

Of particular relevance to habitats for family 5 is the fact that forest and range landscape patterns have changed extensively across the basin (Hann and others 1997, Hessburg and others 1999). The spatial redistribution of forest and range terrestrial communities has resulted in 80 percent of all subbasins being below the minimum for HRV for one or more forest or range terrestrial communities. Only 2 percent of landscape patterns were projected to have patterns consistent with the biophysical succession-disturbance regime across all ownership and 5 percent on FS- and BLM-administered public lands. Forest landscape patterns have highly fragmented mosaics but simplified patch composition and structure in roaded areas, whereas unroaded areas were more simplified in both mosaic and patch composition and structure. Rangelands were more simplified in both mosaics (except in areas of exotic plant invasion) and patch composition and structure. Forest-rangeland landscapes responded somewhat similar to forest landscapes but with higher diversity of types. These changes in landscape patterns may have substantially changed foraging and other life functions for species in family 5, which may have contributed to the substantial range contractions that have occurred for all species in this family (vol. 2, figs. 57, 60, and 66) with the exception of the long-eared owl (fig. 63, vol. 2).

Other factors affecting the family—Human disturbance is a primary factor affecting most species in family 5. Most mortalities of the gray wolf and grizzly bear are due to humans. About 84 percent of all known mortalities of wolves on the Montana-British Columbia-Alberta border were human-caused (Pletscher and others 1997), and in the northern Rockies, 85 to 94 percent of all deaths (1974-96) of marked grizzly bears >1 year old were due to humans (Mattson and others 1996b). Additionally, human activities result in the displacement of wolves and grizzly bears from otherwise high-quality habitat (Mace and others 1996, Mladenoff and others 1995), and human developments cause habitat fragmentation (Noss and others 1996).

Mountain goats and bighorn sheep are not subjected to the same negative attitudes as wolves and grizzly bears, but they are nevertheless highly susceptible to hunting, both legal and illegal (Johnson 1983, Matthews and Coggins 1994). Also, human activities such as recreational hiking, road construction, timber harvesting, and mining can cause physiological stress and displacement from habitats (Chadwick 1972, Hamilton and others 1982, Hicks and Elder 1979, Johnson 1983, Joslin 1986, MacArthur and others 1982). Of all species in family 5, the long-eared owl seems to be the least affected by direct human disturbances.

All species in family 5 except for the long-eared owl are considered road-sensitive because the negative impacts from human activities often are increased where roads are present. A disproportionate number of human-caused mortalities occur near roads, both for wolves (Mech 1970) and grizzly bears (Mattson and others 1996b). Roads, particularly highways, have been documented as a source of mortality for mountain goats through vehicle collisions (Singer 1978). Also, roads increase hunter access for both mountain goat and bighorn sheep herds (Johnson 1983).

The condition of habitats for bighorns and mountain goats has been altered over the last century because of changes in historical fire regimes. Fire suppression has resulted in an increase in the density of trees in formerly open stands, reducing forage quantity, forage quality, and openness, all of which make such stands largely unsuitable for bighorn sheep and mountain goat. For the Rocky Mountain bighorn, fire-suppressed stands have created barriers between historical winter and summer range, thereby preventing occupancy of the total range even though each isolated range is currently suitable (Wakelyn 1987).

Riparian vegetation has declined in extent basin-wide, because of disruption of hydrologic regimes from dams, water diversions, road construction, grazing, and increased recreational use along stream courses (Lee and others 1997, USDA Forest Service 1996). Loss of riparian vegetation has degraded important foraging areas for bighorn sheep, mountain goats, and grizzly bears and potential nesting habitat for the long-eared owl.

Bighorn sheep are highly susceptible to pneumonia after exposure to bacteria (*Pasteurella* spp.), viruses (*Parainfluenza* type-3), lungworm, and stress agents (Foreyt 1994, Wishart 1978). Major reductions or total extirpation of bighorn herds from pneumonia outbreaks are well-documented (Cassirer and others 1996, Coggins 1988, Onderka and Wishart 1984, Spraker and others 1984). Abundant circumstantial evidence (Coggins 1988, Foreyt and Jessup 1982, Martin and others 1996) and recent direct evidence exist (Foreyt 1994; Rudolph and others, in prep.) that domestic and exotic sheep are the source of nonendemic bacteria and viruses predisposing bighorn sheep to pneumonia. Disease transmission from domestic animals is not a major threat to other species in family 5. It is mentioned here, however, because it is currently the most significant factor affecting bighorn sheep conservation.

Issues and strategies for conservation—The primary issues for family 5 relate to direct and indirect human impacts on populations and habitat quality. These issues areas are as follows:

Issues—

1. Habitat fragmentation (poor juxtaposition of seasonal ranges as well as isolation of small populations) because of agricultural, industrial, and recreational development.

2. Displacement from suitable habitats because of human activities and the facilitation of human activities by roads.

3. Degradation and loss of native upland shrublands, upland grasslands, riparian shrublands, and riparian woodlands.

4. Changes in landscape patterns of source habitats and reduction in forage quantity and quality for mountain goats and bighorn sheep because of changes in fire regimes.

5. Disease transmission potential between domestic sheep and bighorn sheep.

6. Excessive bear and wolf mortality from conflicts with humans.

7. Habitat fragmentation or simplification across the basin among subbasins, watershed scale landscape mosaics, and at patch composition and structure.

Strategies—The following strategies could be considered to address issues related to species belonging to family 5. These strategies are appropriate for all areas of the basin with current populations of one or more of the species in family 5, or with suitable, unoccupied habitat where recovery of these species has been identified as a management goal.

1. (To address issue no. 1) Seek opportunities to increase habitat links between isolated populations and seasonal foraging areas caused by human land uses. For wolves and grizzly bears, design interregional habitat connectivity across all ERUs where populations are currently present (Northern Cascades, Northern Glaciated Mountains, Upper Clark Fork, Lower Clark Fork, Central Idaho Mountains, and Snake Headwaters).

2a. (To address issue no. 2) Reduce human activities near important seasonal foraging areas of any species in family 5 and around known wolf dens and lambing and kidding areas of bighorn sheep and mountain goats.

2b. (To address issue no. 2) Develop a policy for road construction, maintenance, and obliteration on public lands to reduce human access to specific areas considered key to the conservation of species in family 5.

3a. (To address issue no. 3) Increase quality and amount of riparian shrublands and woodlands through restoration of hydrologic flows, vegetation restoration, road management, and control of grazing and recreational activities.

3b. (To address issue no. 3) Maintain and restore native upland shrublands and upland grasslands, particularly in the northern half of the Columbia Plateau, Lower Clark Fork, Upper Snake, and Snake Headwaters ERUs.

4. (To address issue no. 4) For mountain goats and bighorn sheep, restore habitat links between summer and winter range and access to escape cover that have been lost because of changes in historical fire regimes. Restore quality and quantity of forage where succession has caused substantial reductions. Implement use of prescribed fire to reestablish inherent fire regime-vegetation patterns.

5. (To address issue no. 5) Actively control the potential for disease transmission between bighorns and domestic livestock.

6. (To address issue no. 6) Reduce the prevalence of conflict situations and human-caused mortalities of bears and wolves.

7. (To address issue no. 7) Conduct mid-scale assessment as part of multiscale step-down implementation to identify risks and opportunities for restoration among subbasins, repattern priority watersheds based on the biophysical succession-disturbance patterns, and conserve or restore patch composition and structure to mimic that appropriate to the succession-disturbance regime.

Family 6—Forest, Woodland, and Montane Shrub Family

This family consists of groups 23, 24, and 25 (table 6). Species in these groups are the sharptail snake, California mountain kingsnake, northern goshawk (winter), rufous hummingbird, broad-tailed hummingbird, and black-chinned hummingbird. The ranges of these species are generally widespread throughout the basin except for the two snake species, which have small, isolated ranges (vol. 2, figs. 69, 72, 75).

Source habitats and special habitat features— Source habitats of the six species in this family consist of montane and lower montane forests, riparian and upland woodlands, chokecherry-serviceberry-rose, mountain mahogany, and riparian shrublands (table 11; vol. 3, appendix 1, table 1). Source habitats for family 6 occur in all 13 ERUs. However, habitat for most species was never common in the Northern Great Basin, Columbia Plateau, Owyhee Uplands, or Upper Snake (vol. 2, figs. 70, 73, and 76).

Special habitat features include nectar-producing flowers for the hummingbird species and logs and talus for the snake species (vol. 3, appendix 1, table 2).

Broad-scale changes in source habitats—Changes in source habitats were variable across the basin. Source habitats had decreasing trends in 45 percent of the

Figure 14—Trend in source habitats for family 6 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

watersheds in the basin and increasing trends in 37 percent (fig. 14). The Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork had an overall decreasing trend, whereas the Snake Headwaters and Central Idaho Mountains had overall neutral trends (table 12). The four primarily nonforested ERUs—Northern Great Basin, Columbia Plateau, Owyhee Uplands, and Upper Snake—as well as the Upper Klamath have little habitat overall but showed increasing trends.

Reasons for increases and decreases were variable, but declines were associated with reductions in late-seral and early-seral lower montane and montane forests, riparian woodlands, and riparian shrublands (Hann and others 1997). Increases were associated with transitions to mid-seral coniferous forest (primarily managed young forests) and to increases in the upland woodland community group. Large increases in juniper/sagebrush in all or parts of the Upper Klamath, Northern Great Basin, Columbia Plateau, Blue Mountains, Upper Snake, and Snake Headwaters ERUs contributed to much of the increases shown in figure 14.

Primary causes for habitat trends and the associated ecological processes—Fire exclusion, heavy livestock grazing, intensive timber harvest, and road-building have contributed to changes in areal extent and quality of source habitats for family 6. Trends in conditions of shrubs, logs, talus and flowers are not available at the broad scale, and these special habitat features are particularly important to the life histories of many species in family 6. Activities that may negatively affect these special habitat features include fire exclusion, timber harvest, road construction and maintenance, livestock grazing, and mining. Fire exclusion may impact flower abundance by increasing forest canopy closure, thereby reducing the amount of herbaceous understory and an associated decline in fire-adapted forbs. Heavy grazing also has reduced the density of understory plants used as a food source (nectar) by hummingbirds (Saab and Rich 1997).

At a broad scale, an ecologically significant decline occurred in early-seral (-77 percent) and late-seral single-layer lower montane (-80 percent), and a slight decline in early-seral montane (-8 percent), which would be the major shrub-, flowering forb-, and grass-producing forest stages of family 6 source habitats. Almost all subbasins of the basin currently are less than the HRV minimum for these stages. These habitats transitioned primarily to mid-seral lower montane and montane stages. Even in the historical condition, the mid-seral stages have higher density of tree overstory and thus have less shrub and herb understory diversity than the early-seral or late-seral single-layer stages. In the current condition, however, the areas in mid-seral were found to have even less shrub and understory diversity than historically because of fire exclusion. Consequently, fine-scale attributes for species in family 6, such as shrubs, forbs, and down logs, likely have been reduced further in abundance in mid-seral habitats compared to historical conditions.

In addition, an ecologically significant decline occurred in the upland shrubland terrestrial community (-31 percent) from historical to current periods. Most of the upland shrub that declined was of the sagebrush-steppe type, which for nonpublic lands was converted primarily to agriculture. On public lands, about a third of the decline transitioned to upland woodland (juniper/sagebrush); this was considered an increase in source habitat for family 6 but would be of lower habitat quality for those species associated with herbaceous shrubs than the mountain shrubs of the lower montane and montane forests.

Most species in family 6 seem to be adapted to forest openings, down logs, shrubs, and flowering forbs. This type of condition would be found in correlation with frequent underburn or mixed-fire events. Current shrub and herbaceous (forb and graminoid) diversity and productivity have declined considerably as a result of fire exclusion, increased tree density, and excessive livestock grazing. In addition, a basin-wide decline occurred in mid-scale detectable riparian shrubland correlated with excessive livestock grazing (Lee and others 1997). Large down logs have declined in areas accessible to roads as a result of woodcutting and timber harvest of large trees, which are the recruitment source for logs.

Of additional pertinence to source habitats for family 6 is the fact that landscape patterns at subbasins and watershed scales changed substantially from historical to current, with only 2 percent estimated to retain their native pattern according to Hann and others (1997) and Hessburg and others (1999). These authors found that most ERUs exhibited high levels of departure from the historical biophysical succession-disturbance regimes and simplification of many of fine-scale attributes important to species in family 6.

Trends of watershed change for the forest, woodland, and montane shrub habitats for family 6 were spatially disjunct (fig. 14). Decreases generally occurred in the northern and eastern portions of the basin, whereas increases and neutral changes were in a mosaic in the central and southern portions of the basin. These patterns resulted from the combination of fire exclusion across all forests and rangelands of the basin, and the timber harvest practices that occurred in the northern portion of the basin. In general, the increases have occurred in environments that are warmer, drier, and less productive, and declines have occurred in more mesic habitats.

Other factors affecting the family—Humans have had a direct effect on all species of snakes through collection, harassment, accidental mortalities, as well as intentional killing because of fear and hate (Brown and others 1995). Also of particular concern with these snake species is population isolation: both the California mountain kingsnake and sharptail snake have small, isolated distributions in the basin (vol. 2, fig 72).

Little is known about the population dynamics of the goshawk (Squires and Reynolds 1997). Several studies, however, have documented a positive relation between prey abundance and nest success (Doyle and Smith 1994, Linden and Wikman 1983, Ward and Kennedy 1996), which presumably also exists between prey abundance and goshawk survival during winter (recall that source habitats for goshawk in family 6 are winter habitat only). Habitat components associated with high prey abundance for goshawk—such as snags, down logs, herbaceous understories, and interspersion of different structural stages—may have been negatively affected by past management activities.

The three hummingbird species are Neotropical migrants. The availability of habitats used during migration, as well as their winter habitat, are critical components, and information on the abundance of or trends in these habitats is lacking.

Issues and strategies for conservation—Because species in family 6 use various cover types and structural stages, issues and strategies for the species are directed at maintaining diversity of vegetation conditions, with emphasis on restoration of habitats and vegetative components that have declined.

Issues—

1. Decline in the abundance of late- and early-seral forests.

2. Likely loss of forest openings with herbaceous understories that provide for small-mammal prey base (for goshawk), and flowers (for hummingbird species).

3. Overall loss of riparian woodlands and herbaceous shrublands, including loss of herbaceous shrubs within these communities.

4. Loss of habitat connectivity particularly for the sharptail snake and California mountain kingsnake.

5. Negative effects of human disturbance to the sharptail snake and California mountain kingsnake.

6. Decline in snags and logs and other important structural components used by sharptail snake, California mountain kingsnake, and the prey of goshawk.

7. Broad-scale changes in landscape patterns in combination with cumulative effects of simplification of fine-scale environmental factors at the ERU, subbasin, watershed, and patch scales (based on results of Hann and others [1997] and Hessburg and others [1999]).

Strategies—The following strategies could be considered to address issues related to species belonging to family 6:

1. (In support of issues no. 1 and no. 2) Enhance landscape diversity by increasing the mix of early- and late-seral stages, particularly in ponderosa pine, western larch, and western white pine types. Increase late-seral forests in the Southern Cascades, Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs, where declines have been strongest. Increase early-seral forests in the Columbia Plateau, Northern Glaciated Mountains, and Lower Clark Fork ERUs in response to strong declines.

2. (In support of issues no. 1 and no. 2) Use prescribed fire and understory thinning to increase vegetative diversity. Several of the species in this

family depend on forest openings and understory shrubs, both of which were maintained historically through natural fire regimes.

3. (In support of issue no. 3) Seek opportunities to improve connectivity among isolated populations of the sharptail snake and California mountain kingsnake.

4. (In support of issue no. 7) Conduct mid-scale step-down assessment of current conditions relative to landscape patterns of succession-disturbance regimes. Focus short-term restoration of watersheds on those that are in high departure, do not contain susceptible populations of species of high conservation concern and are at high risk of loss of biophysical capability. Continue suppression of stand-replacing, high-severity wildfires, and initiate prescribed fire appropriate to the biophysical succession-disturbance regime and timed to protect biophysical capability.

Family 7—Forest, Woodland, and Sagebrush Family

Groups 26, 27, and 28 compose family 7. These three groups include the pine siskin and eight species of bats (table 10). Ranges of these species are shown in figures 78, 81, and 84, volume 2.

Source habitats and special habitat features— Family 7 members use a complex pattern of forest, woodlands, and sagebrush cover types (table 11; vol. 3, appendix 1, table 1). Although the species in family 7 use a broad range of cover types and structural stages as source habitats, all but the pine siskin have special requirements for nesting or roosting (vol. 3, appendix 1, table 2). The bat species use cliffs, caves, mines, and buildings for day roosts and hibernacula (Manning and Knox-Jones 1989, Nagorsen and Brigham 1993). The pallid bat, long-eared myotis, fringed myotis, and long-legged myotis also use large-diameter (>53 cm [21 in]) trees and snags with exfoliating bark for maternity roosts and day roosts (Nagorsen and Brigham 1993, Ormsbee and McComb 1998, Rabe and others 1998).

Suitable roosting structures often limit bat distribution and population size (Humphrey 1975, Nagorsen and Brigham 1993, Perkins and Peterson 1997). For example, the distribution of Townsend's big-eared bat is closely tied to the presence of caves and cavelike structures because they roost in large colonies and require a ceilinglike substrate for hanging (Idaho State Conservation Effort 1995, Nagorsen and Brigham 1993). The spotted bat also appears limited in roost site selection, with all roosts reported in crevices of high cliffs (Nagorsen and Brigham 1993, Sarell and McGuinness 1993, Wai-Ping and Fenton 1989). Snag-roosting bats require specific conditions usually provided by exfoliating bark or large cavities, and must shift their use to other snags when snag decomposition changes these conditions. Rabe and others (1998) suggest that snag-roosting bats may require higher densities of snags than cavity-nesting birds, because the stage at which snags are suitable for bat roosts is extremely short-lived, requiring the use of several snags over the course of a lifetime of a bat.

Shrub/herb riparian areas are a special habitat feature for two members of family 7, the Yuma myotis and long-eared myotis. The Yuma myotis specializes in foraging over water, where it eats midges and emergent aquatic insects (Whitaker and others 1977). The long-eared myotis concentrates most of its foraging in riparian areas, where it is a hover-gleaner (Barclay 1991, Nagorsen and Brigham 1993). Although shrub/herb riparian areas are not considered a requirement for the other bat species in this family, all use riparian areas for foraging because of high insect density.

Broad-scale changes in source habitats—Trends in source habitats were mixed: 47 percent of the watersheds basin-wide had neutral trends; 21 percent had increasing trends, and 32 percent had declining trends (fig. 15). Watersheds with declining trends were concentrated in the Lower Clark Fork and Upper Snake ERUs, and in the northern half of the Columbia Plateau ERU (fig. 15, table 12). The only ERU with increasing trends in more than 50 percent of its watersheds was the Upper Klamath.

Primary causes for habitat trends and the associated ecological processes—Stable trends in broad-scale source habitats throughout much of the basin reflect the wide range of cover types and nearly all structural stages of forests used as source habitats by species in family 7. The basin has experienced dramatic declines in old-forest structural stages of all forest cover types (Hann and others 1997), but for family 7, these losses have been offset by increases in mid-seral stages that

VIEW

MAP

Figure 15—Trend in source habitats for family 7 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

also serve as source habitats. Populations of this family, however, likely could be in decline across their range because of basin-wide changes in landscape patterns and simplification of patch composition and structure (per results of Hann and others [1997] and Hessburg and others [1997]).

Declines in source habitats in the Lower Clark Fork were associated with the broad-scale transition of upland woodland to upland herbland (Hann and others 1997), the latter being a terrestrial community group that does not provide source habitat for family 7. In both the Upper Snake and Columbia Plateau ERUs, source habitat declines were attributed primarily to the conversion of upland shrubland to agriculture. Currently, 36 percent of the Upper Snake ERU and 23 percent of the Columbia Plateau are now in agriculture. Not all species in family 7 are affected by these declines because some of these species either do not occur in these ERUs or do not use upland shrubland as source habitats. The species most affected are long-eared myotis, Yuma myotis, small-footed myotis, Townsend's big-eared bat, and spotted bat.

Increasing trends in most watersheds within the Upper Klamath ERU were primarily due to the transition of upland herbland to several forest community groups that serve as source habitats. These include both mid- and late-seral lower montane and upland woodland terrestrial community types (Hann and others 1997). The transition of upland herbland to lower montane was also responsible for increasing trends in other ERUs, particularly in the central and southeastern areas of the basin.

In contrast to most other families, the mosaic of increasing, decreasing, and neutral trends was not highly disjunct spatially (see fig. 15). There was slight correlation of neutral trends with range landscape patterns and dry forest. Decreasing trends were somewhat correlated with the northerly and eastern portions of the basin, whereas increasing trends were scattered.

Other factors affecting the family—The bat species in family 7 are sensitive to human disturbance of roost sites and loss of roost sites. The most straightforward source of impact is destruction of the structure, i.e., loss of snags through timber harvests, and removal of old buildings and bridges or closure of mines and caves for safety reasons (Perlmeter 1995, Pierson and others 1991). The second source of impact is disturbance of roosting bats, primarily by recreational activities in or near caves, but also from mining, road construction, and any other activities near roosts (Pierson and others 1991). During winter, the transition from torpor requires a large caloric output, and repeated disturbances can drain the energy reserves of bats and lead to starvation (Nagorsen and Brigham 1993). The third source of impacts at roost sites is purposeful killing of roosting bats.

Roads indirectly affect bat species by increasing human access to roost sites. Caves have become more accessible, increasing the amount of human visitation and potential harassment of bats. The presence of roads also increases the likelihood that snags will be cut for fuelwood (Hann and others 1997).

Riparian vegetation has declined in extent basin-wide, because of disruption of hydrologic regimes from dams and water diversions, road construction, grazing, and increased recreational use along stream courses (Lee and others 1997, USDA Forest Service 1996). Loss and degradation of riparian vegetation likely has reduced the diversity of insect prey for bats. Moreover, the loss of riparian woodlands has reduced the availability of sites for day and nursery roosts. Perkins and Peterson (1997) attributed the low detection of bats in the Owyhee Mountains to the lack of suitable roosts, particularly in riparian areas.

Pine siskin foraging behavior, geographic location, and population levels are highly influenced by the combination of current population level and food availability: an abundance of seeds will cause the population to expand, and if the next year's crop is unable to support the expanded population, the birds will move elsewhere (Bock and Lepthien 1976).

Issues and strategies for conservation—Because the species in family 7 are habitat generalists, changes that have occurred in terrestrial community groups since the historical period have resulted in few substantial changes in the extent of source habitats. The primary issues for family 7 relate to human impacts on populations and on special habitat features needed for roosting and foraging. These issues include the following:

Issues—

1. Loss of potential roost sites because of mine closures, destruction of abandoned buildings, snag removal, deliberate fumigation of buildings, and levels of human activity that cause roost abandonment.

2. Excessive disturbance of roosting bats because of human activities and roads as a facilitator of such activities.

3. Degradation and loss of native riparian vegetation.

Strategies—The following strategies could be considered to address issues related to the bat species in family 7. These strategies are appropriate for all areas of the basin. Strategies for pine siskin populations have not been formulated because the causes for apparent population declines at the continental scale are unknown.

1. (To address issue no. 1) Protect all known roost sites (nurseries, day roosts, and hibernacula) and restore useability of historical roosts where feasible. Actively manage for the retention and recruitment of large-diameter (>53 cm [21 in] snags in all forest cover types and structural stages.

2. (To address issue no. 2) Reduce levels of human activities around known bat roosts through road management, signs, public education, and bat gates.

3. (To address issue no. 3) Maintain and improve the condition of riparian vegetation for bat foraging areas.

Family 8—Rangeland and Early- and Late-Seral Forest Family

The western bluebird (group 29) is the sole member of this family. This species was placed in its own family because its source habitats are a unique combination of woodlands, shrublands, grasslands, and early- and late-seral forests. Range of the western bluebird is displayed in figure 87, volume 2.

Source habitats and special habitat features— Source habitats for family 8 are early-seral and late-seral single-storied montane and lower montane forests, riparian and upland woodlands, and upland shrub and herblands (table 11). Additionally, burned pine forests likely function as source habitats. Juxtaposition of forested and open areas is a necessary characteristic of source habitats. Snags are a special habitat feature for nesting, although the snags may be relatively small (<53 cm [21 in]) (vol. 3, appendix 1, table 2).

Broad-scale changes in source habitats—Basin-wide, source habitats for the western bluebird declined in 72 percent of watersheds and increased in only 5 percent (fig. 16). These declines are stronger than those observed for most species included in this assessment (table 12). Source habitats have declined in at least 50 percent of watersheds in 9 of the 11 ERUs in which this species occurs (tables 8 and 12). Only the Northern Great Basin and the Owyhee Uplands showed a neutral trend (table 12).

Primary causes for habitat trends and the associated ecological processes—Declines in source habitat resulted from ecologically significant basin-wide declines in early-seral lower montane forest, late-seral lower montane, single-layer forest, upland shrublands, and upland herblands (Hann and others 1997). Of the terrestrial communities providing source habitats for bluebirds, only upland woodlands showed a basin-wide increase from historical to current conditions. There were ecologically significant decreases in upland herblands in all nine ERUs where source habitats declined for bluebirds, and decreases in early- and late-seral single-storied lower montane forests in eight of these nine ERUs. See discussions in families 1, 2, 6, and 10 for additional information on causes for habitat trends and the associated ecological processes.

Our evaluation at the broad-scale did not assess the distribution of foraging habitat in relation to nesting habitat. Additional analysis of the juxtaposition of foraging with nesting habitat is needed at a finer scale of resolution. Results for source habitats shown here for both the current and historical periods are likely overestimates as they do not take into account the need for juxtaposition of habitats.

Other factors that affect the family—Some western bluebirds that breed in the basin migrate to California and Baja California in winter and could be affected by conditions on those wintering grounds.

Figure 16—Trend in source habitats for family 8 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

Issues and strategies for conservation—The primary issues and strategies for family 8 relate to declines in source habitats.

Issues—

1. Reductions in early- and late-seral montane and lower montane forests.

2. Possibly unsustainable conditions in late-seral montane and lower montane forests where large transitions have occurred from shade-intolerant to shade-tolerant species.

3. Reductions and degradation of upland shrublands and herblands.

Strategies—The following strategies could be considered to address issues related to family 8.

1. (To address issue no. 1) Maintain and restore early- and late-seral montane and lower montane forests where these cover types have declined. Both the extent and pattern of these habitats are of concern because source habitats for western bluebirds are found in edge areas.

2. (To address issue no. 2) Restore succession-disturbance regimes to patterns consistent with biophysical variation in those ERUs and portions of ERUs where substantial habitat remains, such as the Northern Great Basin, Owyhee Uplands, or southern portion of Columbia Plateau.

3. (To address issue no. 3) Restore upland shrub and herbland cover types, and manage these areas to maintain plant composition and structure similar to that consistent with the biophysical succession-disturbance regimes. Reduce risk of exotic plant invasion and restore invaded areas to more closely represent native composition and structure.

Family 9—Woodland Family

This family is composed of the two species in group 30, the ash-throated flycatcher and bushtit (table 6). Range maps for these species are shown in figure 90, volume 2.

Source habitats and special habitat features— Source habitats consist primarily of the upland woodland and upland shrubland community groups, including juniper woodlands, mixed-conifer woodlands, juniper/sagebrush woodlands, Oregon white oak, and mountain mahogany (table 11; vol. 3, appendix 1, table 1). The ash-throated flycatcher also uses old-forest cottonwood-willow. Snags are a special habitat feature for ash-throated flycatchers (vol. 3, appendix 1, table 2).

Broad-scale changes in source habitats—Source habitats for family 9 increased strongly within the basin (fig. 17); specifically, source habitats increased in 70 percent of watersheds and decreased in only 18 percent. Fifty percent or more of the watersheds in 8 of the 10 ERUs containing source habitats had increasing trends: Upper Klamath, Northern Great Basin, Columbia Plateau, Blue Mountains, Northern Great Basin, Owyhee Uplands, Upper Snake, and Snake Headwaters (table 12). Source habitats in the Northern Great Basin represent <1 percent of the ERU. Only the Northern Cascades had a greater number of watersheds with decreasing rather than increasing amount of source habitats. The Southern Cascades generally had a neutral trend.

Primary causes for habitat trends and the associated ecological processes—Increasing trends in source habitats were due to increases in the juniper/sagebrush cover type. The extent of juniper/sagebrush woodlands has more than doubled in the basin, primarily because of the combined effects of livestock grazing and fire suppression (Hann and others 1997). The upland woodland community group, which includes juniper/sagebrush, underwent ecologically significant increases in five of the eight ERUs that had an increasing habitat trend for family 9. Broad-scale trends in the other source habitat types, especially old-forest cottonwood-willow, Oregon white oak, and mountain mahogany, are difficult to determine at the scale of this analysis (Hann and others 1997).

The increase in woodland extent has produced communities of lower habitat quality then occurred historically. Historical woodland types were typically on soils with scattered clumps of surface rock that protected juniper and other woodland tree species from

Figure 17—Trend in source habitats for family 9 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

fire. The fire regime maintained a somewhat open shrub/herb understory that was high-quality habitat for family 9 species. Historical excessive grazing and fire exclusion has resulted in much higher density of woodland trees and loss of the shrub/herb understory in these native woodland types (Hann and others 1997). Also, as a result of fire exclusion, some of the sagebrush zones have transitioned to dense woodlands of one size class that lacks the structural diversity and snags of native woodlands.

Other factors affecting the family—Insects are the primary prey for these species. Understory shrubs and grasses provide habitat for insects, and excessive grazing can degrade these habitats.

Issues and strategies for conservation—Results of our analysis suggest no cause for broad-scale concern about source habitats for family 9. However, strategies that play a part in overall ecosystem management, and that ensure long-term availability of source habitats for this family, are suggested below.

Issues—

1. Identification and retention of woodlands that are present under inherent succession and disturbance regimes versus identification and reduction of woodlands that exist primarily because of fire exclusion and other land uses.

2. For ash-throated flycatchers, loss of trees with natural cavities or trees suitable for excavation by other species because of juniper removal.

3. Degradation and loss of native understory shrubs and grasses that provide substrates for arthropod prey.

Strategies—The following strategies could be considered to address issues related to species belonging to family 9:

1a. (To address issues no. 1 and no. 2) Plan the conversion of juniper to other, more desirable native shrubs and grasses such that blocks of old-growth juniper are retained within and juxtaposed to the restored areas over space and time. Retention of large or deformed trees and older stands of juniper would benefit species in this family as well as families 6, 7, and 10. Value of older stands of juniper would be highest if stands are retained that have a preponderance of older trees that are hollow or that contain cavities; such trees are used as nest sites by ash-throated flycatchers, especially when located in or near areas dominated by native understory shrubs and grasses. Assure that the retention of woodlands is consistent with the biophysical succession-disturbance regimes.

1b. (To address issues no. 1 and no. 2) Retain representative stands of old-growth western junipers especially in areas containing old junipers with cavities and hollow centers for potential nest sites of ash-throated flycatchers.

2. (To address issue no. 3) Protect and restore native understory shrubs and grasses in source habitats. Reduce risk of exotic plant invasion and restore invaded areas to more closely represent native composition and structure.

Family 10—Range Mosaic Family

Family 10 consists of 17 species of birds, mammals, and reptiles within groups 31 and 32 (table 6). The ranges of the species in this family primarily cover the rangeland ERUs, and several of the species have restricted ranges within only one or two ERUs (vol. 2, figs. 93 and 96).

Source habitats and special habitat features—This family is characterized by species that primarily use various shrublands, herblands, and woodlands. All species in family 10 use several cover types in the upland shrubland and upland herbland community groups as source habitats (table 11). All species except the short-eared owl, pronghorn, Preble's shrew, white-tailed antelope squirrel, and Uinta ground squirrel also use upland woodlands as source habitats. Exotic herbland is an additional source habitat for the ferruginous hawk, burrowing owl, short-eared owl, and lark sparrow. The short-eared owl is the only species in the family that uses riparian herbland.

Figure 18—Trend in source habitats for family 10 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

Several special habitat features have been identified for family 10 (vol. 3, appendix 1, table 2). The burrowing owl requires burrows excavated by other species or natural cavities in lava flows or rocky areas for nest sites; the Preble's shrew uses down logs; the pronghorn antelope is associated with shrub/herb riparian areas for parts of the year; the striped whipsnake and longnose snake use talus areas, and the striped whipsnake also uses cliffs. Many species in this family prefer open cover types with a high percentage of grass and forbs in the understory, either for foraging or nesting.

Broad-scale changes in source habitats—Trends in source habitats were predominantly declining for family 10 (fig. 18). Basin-wide, 52 percent of watersheds exhibited declining trends, whereas 10 percent were projected to have increased. Neutral trends were projected for the remaining area. Watersheds with declining trends were concentrated in the northern half of the basin and in the Snake River drainage, whereas watersheds with neutral trends were mostly in the south-central portions of the basin (fig. 18). Nine ERUs had declining trends in >50 percent of watersheds, and the remaining four had neutral trends in >50 percent of watersheds (table 12). There were no ERUs with predominantly increasing trends.

Individually, all species in family 10 had declining or strongly declining trends in source habitats except for the long-nosed leopard lizard, Mojave black-collared lizard, longnose snake, Wyoming ground squirrel, and white-tailed antelope squirrel, all of which have fairly small and disjunct ranges within the basin (vol. 2, figs 93 and 96). Source habitats for these species were projected to be neutral (table 7).

Primary causes for habitat trends and the associated ecological processes—Upland shrubland and herbland terrestrial communities both had ecologically significant declines (-67 and -31 percent, respectively), whereas upland woodland increased (50 percent) (Hann and others 1997). About 70 percent of the upland shrubland decline transitioned to agriculture on private lands, whereas the decline on public lands was a transition somewhat evenly split among exotic herbland, upland herbland, and upland woodland. About 66 percent of the upland herbland decline transitioned to agriculture on private lands, whereas the decline on public lands was a transition of 13 and 21 percent, respectively, to mid-seral lower montane forest and upland shrubland. Upland woodland was above the historical maximum across 40 percent of subbasins but below for 34 percent. Dominant transitions for upland woodland increase came from upland shrubland, whereas decreases went to upland herbland. Declines in woodland came primarily from the loss of aspen and cottonwood woodland types through excessive livestock grazing and lack of fire in the northeastern and eastern portions of the basin, whereas increases came from increased juniper woodland types in the south-central and western portions of the basin.

In general, patch habitat quality for family 10, the herbland, shrubland, and woodland source habitats, declined from historical to current periods because of conversion to agriculture, successional transitions caused by fire exclusion, and excessive livestock grazing. Current upland shrubland and upland herbland patches were found to have higher canopy closure of shrubs, less species and layer diversity of understory shrubs and herbs, and less herbaceous productivity (Hann and others 1997). Almost two thirds of upland shrubland patches were estimated to contain some component of exotic plant species, and at least one third was estimated to have an understory dominated by exotic plant species. Current upland herbland patches were found to have lower canopy closure of grasses and less diversity of species and layers, with lower productivity of herbs, as compared to historical conditions. The communities with transitions to and from upland woodland may be the ecosystems most at risk. Dense upland woodlands created through transition from upland shrubland because of fire exclusion and excessive livestock grazing were found to often have nutrient-limited soils that limit the ability of understory herbaceous species to regenerate and provide soil cover. This lack of understory plant cover may be exacerbating erosion of surface soils in steep terrain, thereby reducing site capability. Limited nutrients also may be tied up in the juniper foliage and lost when intense summer wildfires occur.

Trends of watershed change for family 10 source habitats were highly spatially disjunct (fig. 18). Decreases occurred extensively across the western, northern, central, and eastern portions of the basin. Neutral trends occurred in a concentrated area of the south-central portion of the basin, and increases were minor. These changes occurred in response to extensive fire exclusion, agricultural development, exotic invasions, and excessive livestock grazing across the more productive portions of the basin. The watersheds exhibiting neutral or positive trends would, if investigated at a finer scale, likely show a decrease in source habitat because of extensive decline in fine-scale habitat quality. Because of the invasion of exotics, the historical effects of excessive livestock grazing, the permanent loss of many habitats to agricultural conversion, and a 95-percent change in frequency and severity of fire, we conclude there is little that is similar to historical conditions for this terrestrial family.

Other factors affecting the family—Losses of native perennial grass and forb understories within the upland shrublands, because of excessive livestock grazing combined with cheatgrass and other exotic plant invasions, are microhabitat features that cannot be evaluated directly with the broad-scale analysis. Because species in family 10 favor grass or shrub-grass types for nesting, foraging, or hiding, we know that the grass component of historical shrublands was important. Wiens and Rotenberry (1981) found significant correlations between the coverage of grass and the densities of both western meadowlarks ($r = 0.62$, $P < 0.001$) and lark sparrows ($r = 0.37$, $P < 0.05$). Forbs comprise most of pronghorn diets during spring and summer, and livestock grazing decreases the abundance of forbs (Yoakum 1980). Removal of grass cover by livestock potentially has detrimental effects on the short-eared owl (Marti and Marks 1989). Areas dominated by dense stands of cheatgrass or other exotic plants may preclude use by longnose leopard lizards (Stebbins 1985), longnose snakes (Beck and Peterson 1995), and collared lizards.

Microbiotic, or cryptogamic crust, is projected to have been widely distributed throughout the source habitats for this group, particularly in the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs but also scattered in the Columbia Plateau ERU (Hann and others 1997, map 3.59). Evidence indicates that microbiotic crusts improve soil stability, productivity, and moisture retention; moderate extreme temperatures at the soil surface; and enhance seedling establishment of vascular plants (Belnap and Gardner 1993, Harper and Pendleton 1993, Johansen and others 1993, St. Clair and others 1993). The BLM in Idaho has recognized the potential importance of microbiotic crusts by proposing standards for rangeland health that include the maintenance of these crusts to ensure proper functioning and productivity of native plant communities (USDI Bureau of Land Management 1997). These crusts were widely destroyed by trampling during the excessive livestock grazing of the late 1800s and early 1900s (Daubenmire 1970, MacCracken and others 1983, Mack and Thompson 1982, Poulton 1955). Currently, high-intensity grazing and altered fire regimes modify shrub-steppe plant communities and threaten the maintenance and recovery of microbiotic crusts (Belnap 1995, Kaltenecker 1997, St. Clair and Johansen 1993).

Soil compaction caused by livestock grazing could negatively affect both the longnose snake and ground snake. These burrowers benefit from loose, sandy, and friable soils (Beck and Peterson 1995, Nussbaum and others 1983).

Human activities associated with roads are known to impact ferruginous hawks, short-eared owls, burrowing owls (Bechard and Schmutz 1995, Green and Anthony 1989, Lokemoen and Duebbert 1976, Olendorff and Stoddart 1974, Ramakka and Woyewodzic 1993, Schmutz 1984, White and Thurow 1985) and western meadowlarks (Lanyon 1994). Harassment of pronghorn by snowmachines and all-terrain vehicles stresses animals at all times of the year (Autenrieth 1978). Accidental and deliberate mortality of snakes potentially increase in direct proportion to roading and traffic in the basin. Although the three species of snakes in this family may not be as frequently killed by vehicles as are some more common species (such as gopher snake and western rattlesnake), increasing human access to source habitats likely will result in more deliberate killing of snakes. Because reptiles are increasingly popular as pets, all reptile species in this group, particularly the lizards, likely are impacted by collecting (Lehmkuhl and others 1997). Road access intensifies the pressure on reptile populations by increasing the ease with which reptiles can be collected.

Poisoning and other eradication potentially affect populations of all four species of ground squirrels. Ground squirrels also are popular targets for recreational shooting. The typically small size of Washington ground squirrel colony populations makes them particularly vulnerable to extirpation (Tomich 1982). Recreational shooting of marmots and ground squirrels impacts burrowing owls because the owls are accidentally or deliberately shot (Marti and Marks 1989). Pesticide use leads to direct mortality in burrowing owls, short-eared owls (Marti and Marks 1989), and western meadowlarks (Griffin 1959). Pesticides may also reduce populations of burrowing owls through a reduction in the populations of burrowing mammals.

Pronghorn movement is restricted or completely impeded by net-wire and other fences that prevent them from crossing beneath the lower strand (Helms 1978, Oakley and Riddle 1974, Yoakum 1980). Roads are readily crossed by pronghorn, but snow accumulating in roadside ditches also may present barriers to movement during winter (Bruns 1977).

Issues and strategies for conservation—The primary issues for family 10 relate to source habitats, special habitat features, and road-related human disturbances.

Issues—

1. Permanent and continued loss of large acreage of upland shrublands and upland herblands because of conversion to agriculture, brush control, cheatgrass invasion, and excessive livestock grazing.

2. Loss of native perennial grass and forb understories within the upland shrublands.

3. Soil compaction and loss of the microbiotic crust.

4. Adverse effects of human disturbance and roads as a facilitator of these effects.

Strategies—The following strategies could be considered to address issues related to species belonging to family 10:

1a. (To address issue no. 1) Identify and conserve large areas of remaining native upland shrublands and upland herblands where ecological integrity is still relatively high, and manage to promote their long-term sustainability. Large contiguous blocks of public land in the Northern Great Basin and Owyhee Uplands could be considered, as well as native vegetation that currently exists on military lands in Washington (Rickard and Poole 1989, Schuler and others 1993, Smith 1994).

1b. (To address issue no. 1) Conduct mid-scale step-down assessment of current conditions relative to landscape patterns of succession-disturbance regimes. Focus short-term restoration of watersheds on those that are in high departure, do not contain susceptible populations of species of high conservation concern and that are at high risk of loss of biophysical capability.

2. (To address issue no. 2) Restore the native grass and forb components of the upland woodland, shrubland, and grassland community groups to historical levels throughout the basin. Restoration measures include seedings and plantings in combination with effective methods of site preparation, effective management of grazing by domestic and wild ungulates, and control of human activities such as offroad vehicle usage and other ground-disturbing factors.

3. (To address issue no. 3) Reduce causes of soil compaction, particularly within source habitats of the longnose snake and ground snake. This factor may be important in the Owyhee Uplands ERU in particular. Restore the microbiotic crust in ERUs with potential for redevelopment, specifically the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs and, to a lesser extent, the Columbia Plateau ERU.

4. (To address issue no. 4) Reduce the negative effects of factors associated with roads. These include the indiscriminate poisoning and recreational shooting of ground squirrels, accidental and deliberate killing of snakes and lizards, the capture of reptiles as pets, and the poaching and disturbance of pronghorn populations.

5. (To address issue no. 4) To the extent possible, encourage activities that reduce mortality and stress on species in family 10. For example, modify existing fences and construct new fences in pronghorn range to allow passage by pronghorns (Yoakum 1980); modify agricultural practices to minimize direct mortality of nesting birds (Clark 1975); and reduce use of pesticides when feasible.

Family 11—Sagebrush Family

This family consists of groups 33, 34, and 35. The included species are listed in table 6; example species are sage grouse, loggerhead shrike, pygmy rabbit, and kit fox. The species ranges within this family are generally located throughout the primarily rangeland type communities across the basin (vol. 2, figs. 99, 102, and 105).

Source habitats and special habitat features—
Species in family 11 group together based on their nearly common use of open and closed low-medium

Figure 19—Trend in source habitats for family 11 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

shrub stages of big sagebrush, low sage, and mountain big sagebrush (table 11; vol. 3, appendix 1, table 1). Other important source habitats include salt desert shrub, antelope bitterbrush-bluebunch wheatgrass, and herbaceous wetlands. Four species (sage thrasher, brewer's sparrow, sage sparrow, and loggerhead shrike) also use upland woodlands. Special habitat features include riparian meadows (sage grouse), and burrows (kit fox).

Broad-scale changes in source habitats—Source habitats are limited in the Northern Cascades, Southern Cascades, Northern Glaciated Mountains, and Lower Clark Fork ERUs, with few watersheds containing habitats for few species within this family (vol. 2, figs. 100, 103, and 106). Overall, 42 percent of the watersheds in the basin had declining trends, and 45 percent had neutral trends (fig. 19). Of the eight ERUs that contained a substantial number of watersheds with source habitats, five showed overall neutral trends (Upper Klamath, Northern Great Basin, Columbia Plateau, Blue Mountains, Owyhee Uplands, and Central Idaho Mountains), and three showed declining trends (Upper Clark Fork, Upper Snake, and Snake Headwaters) (table 12). Fifty percent of the watersheds in the Columbia Plateau showed a declining trend.

Habitat loss on an absolute scale ranged from -9 percent for the loggerhead shrike (group 35) to -15 percent for group 33, which contains the sage grouse, sage thrasher, and pygmy rabbit among others (table 9). All of the species in this group except the kit fox showed relative declines $\geq$ 20 percent across the basin (table 7). Wet meadows and riparian vegetation, cover types used for brood-rearing by sage grouse, have declined substantially since historical times (Lee and others 1997, Quigley and others 1996).

No information is available to determine whether changes in availability of burrows for kit fox dens, or in soil conditions needed for burrow excavation, have occurred in the basin. A lack of suitable loose-textured soil for burrow construction may be a natural limiting factor for kit fox in southeastern Oregon (Keister and Immell 1994). Two other species in this family, pygmy rabbit and sagebrush vole, construct their own burrows, and any factors that may negatively affect soil texture or quality may negatively affect these species as well. Voles seldom use compacted or rocky soil (Maser and others 1974) and may be absent from areas that have suffered soil erosion because of heavy livestock grazing (Maser and Strickland 1978).

Primary causes for habitat trends and the associated ecological processes—Trends of these habitats can be taken in similar context as family 10. That is, the same patterns of broad-scale redistribution of habitats, broad-scale reduction, and fragmentation and simplification of habitats at multiple spatial scales (as described by Hann and others 1997) were associated with family 11 habitats in a similar manner as those associated with family 10.

The major cause for change in source habitats for groups in family 11 has been a significant loss of upland shrubland habitat, which showed the largest decline (-11 percent) of any terrestrial community basin-wide (Hann and others 1997). The single largest loss in cover types within the basin was the decline in big sagebrush (-8 percent), which is considered source habitat for all species within this family. The large-scale loss of upland shrubland habitat was attributed to several factors, including the increase in agriculture and the conversion of lands to other exotic forbs and annual grasses. The largest transition of any terrestrial community was from upland shrubland to agriculture (+9 percent) (Hann and others 1997). The ERUs with the biggest changes were the Columbia Plateau and Upper Snake. The former is now nearly half agricultural lands, whereas the latter is nearly one-third. Agriculture also now occupies over a tenth of the Owyhee Uplands.

The abundance of upland woodlands, primarily the juniper/sagebrush cover type, increased significantly (from less than 1 percent to about 2 percent) basin-wide (Hann and others 1997), which in some cases may have offset the relative losses shown in the upland shrublands.

Much of the area that at the broad scale is mapped as source habitat currently may, in fact, at a finer scale be unsuitable because of changes in soil or understory vegetation. Altered fire regimes and livestock grazing in many areas have removed much of the native herbaceous understories, which are important habitat features for several members of this group. In some areas, native herbaceous understories also have been replaced by unsuitable exotic vegetation.

Habitat condition for family 11 can be described by the composite ecological integrity ratings (Quigley and others 1996) that show most of the habitat to have a "low" rating. Many of the subbasins that have a "low" rating include lands used for agricultural and grazing uses. Primary risks to the ecological integrity over most of the area with source habitats for this family include overgrazing, exotic grass and forb invasion, and continued declines in herbland and shrubland habitats (Quigley and others 1996).

Other factors affecting this family—Grazing and altered fire regimes have been linked to continued losses of microbiotic crusts (Belnap 1995, Kaltenecker 1997, St. Clair and Johansen 1993). There is increasing evidence that microbiotic crusts improve soil productivity and moisture retention, moderate extreme temperatures at soil surfaces, and enhance seeding establishment of vascular plants (Belnap and Gardner 1993, Harper and Pendleton 1993, Johansen and others 1993, St. Clair and others 1993). The effects of past losses and continued threats to microbiotic crusts across the basin may affect restoration efforts of upland herbland and shrubland environments.

Little information is available on effects of landscape patterns on species in this family. Research by Knick and Rotenberry (1995) indicates that both the sage thrasher and sage sparrow are more likely to be found in areas with larger patches of habitat as compared to the Brewer's sparrow, which is known to occupy small patches of suitable habitat within a matrix of unsuitable vegetation.

Several species in this family are known to be negatively affected by human disturbance from various causes. Kit fox are vulnerable to poisoned baits placed to destroy coyotes (Orloff and others 1986). Vehicular collisions may be an important source of mortality of loggerhead shrikes because shrikes often forage and nest along roads (Blumton 1989, Craig 1978, Flickinger 1995, Yosef 1996). Lastly, roads and associated human disturbance can be especially harmful to grouse during the lekking and wintering periods (Marks and Saab 1987, Saab and Marks 1992).

The sage sparrow, Brewer's sparrow, and lark bunting are infrequently parasitized by brown-headed cowbirds (Ehrlich and others 1988). The sage thrasher also is parasitized but rejects cowbird eggs (Rich and Rothstein 1985).

Issues and strategies for conservation—

1. Loss of and degradation of sagebrush habitats because of conversion to agriculture, altered fire regimes, and livestock grazing. A change in fire regimes and livestock grazing has left much of the area susceptible to invasion of cheatgrass and other nonnative vegetation. Altered fire regimes and livestock grazing also may have played a role in the loss of microbiotic crusts.

2. Adverse effects of human disturbance.

3. Redistribution, fragmentation, and simplification of habitats outside of the HRV (per Hann and others [1997]).

Strategies—The following strategies could be considered to address issues related to species belonging to family 11. Primary strategies for improvement of source habitats for family 11, outlined below, are similar to many strategies identified for family 10:

1. (In support of issue no. 1) Identify and conserve remaining core areas of shrub-steppe and other source habitats where ecological integrity is still high (Quigley and others 1996); examples are the Northern Great Basin and Owyhee Uplands ERUs that contain large blocks of public land. Conservation measures include control of cheatgrass and other exotic plants, proper management of grazing by domestic and wild ungulates, and maintenance of the Conservation Reserve Program on private lands. Conservation of large core areas will provide long-term habitat stability; such areas will function as anchor points for restoration, corridor connections, and for other key functions of landscape management.

2. (In support of issue no. 1) Restore the native grass, forb, and shrub composition within the sagebrush cover types, and in other shrubsteppe cover types used by species in family 10. Restore selected areas of cheatgrass monocultures, by using seedings and other manipulations, for areas that would provide key spatial links for populations in family 10.

3. (In support of issue no. 1) Retard the spread of nonnative vegetation. Use fire prevention and suppression, planting of fire-resistant vegetation, and explore the use of "green-stripping" techniques to

Figure 20—Trend in source habitats for family 12 within each of 2,562 watersheds in the interior Columbia basin. Trend for each watershed is shown as one of three categories: increasing, decreasing, or no change. A watershed was classified as increasing if >50 percent of the groups in a family increased in source habitats >20 percent, considering only those groups that occurred in the watershed. A watershed was classified as decreasing if >50 percent of the groups in a family decreased in source habitats >20 percent, considering only those groups that occurred in the watershed. Watersheds not classified as increasing or decreasing were classified as no change.

control the spread of cheatgrass in areas that are susceptible to cheatgrass invasion and that are currently dominated by native shrubsteppe vegetation.

4. (In support of issue no. 1) Restore the microbiotic crust in ERUs with potential for redevelopment (that is areas near propagule sources, and with suitable soil, vegetation, and climatic characteristics [see Belnap 1993, Belnap 1995, Kaltenecker 1997, Kaltenecker and Wicklow-Howard 1994]); specifically focus on the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs and, to a lesser extent, the Columbia Plateau ERU (Hann and others 1997, map 3.59).

5. (In support of issue no. 1) Maintain or restore riparian vegetation and associated water tables to benefit microhabitats for sage grouse through rangeland management (for example, grazing management of domestic and wild ungulates).

6. (In support of issue no. 2) Minimize adverse effects of human disturbance. Implement road closures or other management that reduces human activities and presence in source habitats.

7. (In support of issue no. 3) Conduct midscale step-down assessment of current conditions relative to landscape patterns of succession-disturbance regimes. Focus short-term restoration of watersheds on those that are in high departure, do not contain susceptible populations of species of high conservation concern, and are at high risk of loss of biophysical capability.

Family 12—Grassland and Open-Canopy Sagebrush Family

Family 12 consists of the four species in groups 36 and 37: Columbian sharp-tailed grouse (summer), clay-colored sparrow, grasshopper sparrow, and Idaho ground squirrel (table 6). The sharp-tailed grouse and Idaho ground squirrel are year-round residents, whereas the grasshopper sparrow and clay-colored sparrows breed only in the basin. Most species in this family have limited or reduced distributions, or both (vol. 2, figs. 108 and 111).

Source habitats and special habitat features— Membership in family 12 was based on their close associations with upland herblands, primarily fescue-bunchgrass but, additionally, all species except the clay-colored sparrow use open-canopied sagebrush communities (table 11; vol. 3, appendix 1, table 1). Additional cover types used by one or more species are chokecherry-serviceberry-rose, wheatgrass bunchgrass, native forbs, and herbaceous wetlands.

Although no special habitat features were identified for species in family 12, microhabitat characteristics probably limit these species' distributions within the source habitats identified above. Sharp-tailed grouse use areas in more mesic (>30 cm [12 in] of annual precipitation) shrublands and grasslands (Meints and others 1992) and where the topography is rolling (Saab and Marks 1992). Winter habitat for sharp-tailed grouse, primarily mountain and riparian shrubs, was not modeled here because of the fine-scale nature of those specific cover types. The clay-colored sparrow may be attracted to sites that have dense shrubs in a matrix of more open grasslandlike vegetation (Janes 1983). Lastly, the Idaho ground squirrel inhabits meadows dominated by shallow soils and small intrusions of deeper soil for nest burrows (USDA Forest Service and USDI Fish and Wildlife Service 1996); such meadows are typically surrounded by ponderosa pine forests.

Broad-scale changes in source habitats—Restricted ranges and reductions in ranges of most species in family 12 increase the susceptibility of these populations to habitat declines, which occurred consistently and strongly across most or all ERUs (table 12) and associated watersheds (fig. 20). Source habitats declined in 60 percent of the watersheds throughout the basin. Specifically, source habitats declined in eight ERUs (Northern Cascades, Southern Cascades, Upper Klamath, Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork) (table 12). Greater than 45 percent of the watersheds in the Owyhee Uplands, Snake Headwaters, and Central Idaho Mountains also had declining trends, whereas >65 percent of the watersheds in the Northern Great Basin and Upper Snake had neutral trends.

Primary causes for habitat trends and the associated ecological processes—Trends of source habitats for family 12 can be taken in similar context as for families 10 and 11. That is, the same patterns of broad-scale redistribution of habitats, and of broad-scale reduction, fragmentation, and simplification of habitats at multiple spatial scales (as described by Hann and others 1997) were associated with family 12.

Declines in source habitats for family 12 resulted from basin-wide declines that occurred primarily in upland shrubland and upland herblands (Hann and others 1997). The largest declines of terrestrial communities basin-wide were upland shrublands (-11 percent) and upland herblands (-10 percent) (Hann and others 1997). The two largest decreases in cover types across the basin were big sagebrush (-8 percent) and fescue-bunchgrass (-5 percent).

The open-canopy low-medium structural stage of mountain big sagebrush and big sagebrush experienced some of the greatest absolute declines on an ERU basis. The combined absolute decline for the open-canopy low-medium structural stage of these two sagebrush types declined in the Upper Snake (-40 percent), Owyhee Uplands (-20 percent), Columbia Plateau (-13 percent), Snake Headwaters (-7 percent), and Northern Great Basin (-2 percent) (vol. 3, appendix 1, table 4). In these open-canopied cover types, shrubs and trees eventually invade much of the area that was occupied by grasses and forbs when fire is absent. Woody species tie up nitrogen and other trace nutrients causing a decline in site productivity. Subsequently, foliage cover, basal cover, and litter from the grasses and forbs decline, causing exposure of the surface soil, which leads to erosion. Erosion potentials in these areas can be aggravated by excessive livestock grazing (as well as excessive grazing by wild ungulates in concentrated areas, typically only on winter range). Once the surface soil becomes eroded and the subsoil is exposed, the environment becomes more conducive to other woody species that better compete for subsoil moisture.

Bunchgrasses, critical habitat components for family 12, were substantially impacted by high-intensity grazing in the late 1800s and early 1900s (USDA Forest Service 1996). For the Idaho ground squirrel, meadow habitats of sagebrush and herbaceous vegetation surrounded by pine forest are decreasing because of forest encroachment (USDA Forest Service and USDI Fish and Wildlife Sevice 1996).

Fire can either enhance or degrade habitats for species in this family depending on cover type, timing, frequency, intensity, size of burn, soils, and precipitation. It is likely that all species in family 12 avoid burns immediately after the fire because of loss of grass or shrub cover, and return to burned sites after grasses are restored. Most species of sagebrush do not resprout and may not regenerate for 5 to 15 years after fires. In contrast, many species of deciduous shrubs (for example chokecherry-serviceberry-rose) usually resprout immediately after fire. Also, exotic vegetation can invade after fire, depending on the soils and precipitation.

Mountain shrubs (chokecherry-serviceberry-rose), shrub-wetlands, and herbaceous wetlands, other source habitats that are key components of sharp-tailed grouse habitat during late summer, fall, and winter, naturally occur in small patches and were difficult to map at the scale of this analysis. Accurate information, therefore, was not available on habitat trends in mountain shrub and wetland cover types

Other factors affecting the family—Grazing can negatively affect grasshopper sparrows (Bock and Webb 1984, Saab and others 1995), and sharp-tailed grouse (Marks and Saab Marks 1987, Saab and Marks 1992). High-intensity grazing negatively affects the other species of this group (clay-colored sparrows and Idaho ground squirrels) because of losses of native perennial grasses and forbs, which are essential habitat components for these species.

Grazing and altered fire regimes have been linked to continued losses of microbiotic crusts (Belnap 1995, Kaltenecker 1997, St. Clair and Johansen 1993). Increasing evidence shows that microbiotic crusts improve soil productivity and moisture retention, moderate extreme temperatures at soil surfaces, and enhance seeding establishment of vascular plants (Belnap and Gardner 1993, Harper and Pendleton 1993, Johansen and others 1993, St. Clair and others 1993). The effects of past losses and continued threats to microbiotic crusts across the basin may affect restoration efforts of upland herbland and shrubland environments.

Where hayfields and similar agricultural lands have replaced native source habitats or are now located adjacent to such habitats, substantial mortality can be associated with annual tillage, particularly for grasshopper sparrow. Early season mowing of hayfields causes major nest failures in grassland-nesting species (Knapton 1994, Smith 1963).

Human disturbances related to the expansion of residential developments, increases in road densities, and associated recreational activities may exacerbate losses of suitable habitat within the historical range of Columbian sharp-tailed grouse (Giesen and Connelly 1993, Tirhi 1995). Idaho ground squirrel populations are susceptible to sport shooting (Moroz and others 1995) as well as loss of habitat from human developments (USDA Forest Service and USDI Fish and Wildlife Service 1996). The clay-colored sparrow and grasshopper sparrow also are susceptible to continued loss in habitat because of continued expansion of residential developments.

Issues and strategies for conservation—The magnitude and consistency of declines in source habitats for family 12 were as strong as or stronger than those experienced for any other family, with the possible exception of family 1. Such declines are reinforced by the strength and consistency of habitat declines that we observed at a species level for members of this family (tables 7 and 8). Declines in source habitats for the Idaho ground squirrel, grasshopper sparrow, and clay-colored sparrow were second, third, and fifth highest among all species in the basin (table 7). Moreover, declines in source habitats for the Columbian sharp-tailed grouse were in the upper 20 percent of all species-level declines (table 7).

Issues—

1. Loss of upland herbland and upland shrubland vegetation basin-wide.

2. Degradation of upland herbland and upland shrublands habitats because of invasions of exotic forbs and grasses, excessive livestock grazing, altered fire regimes, and herbicide and pesticide use.

3. Human disturbance and human encroachment, and roads as a facilitator of these negative effects.

4. Isolated and disjunct populations for Columbian sharp-tailed grouse and Idaho ground squirrels.

5. Redistribution, fragmentation, and simplification at basin, ERU, subbasin, watershed, and patch scales compared to HRV (per findings of Hann and others [1997]).

Strategies—The following strategies could be considered to address issues related to species belonging to family 12. The large and widespread declines in source habitats for species in family 12 are notable and compelling from a management perspective. Strategies to improve source habitats for this family partially overlap with strategies for families 10 and 11:

1. (In support of issue no. 1) Identify and conserve remaining large areas of open-canopied big sagebrush, fescue-bunchgrass, mountain big sagebrush, wheatgrass bunchgrass, native forbs and other source habitats where source habitats have not declined strongly, such as in the Northern Great Basin, Upper Snake, and Snake Headwaters ERUs. Conservation measures include control of cheatgrass and other exotic plants; reductions in grazing by domestic and wild ungulates; and maintenance of or increased participation in the Conservation Reserve Program on private lands. Conservation of large areas will provide long-term habitat stability; such areas will function as anchor points for restoration, corridor connections, and for other key functions of landscape management.

2. (In support of issue no. 2) Restore the historical composition of native grasses, forbs, and shrubs within the big, mountain big, and low sagebrush, fescue- and wheatgrass bunchgrass, native forb, and chokecherry-serviceberry-rose cover types used by species in family 12, in all ERUs that have undergone strong declines in source habitats. Restoration measures include seedings and plantings in combination with effective methods of site preparation, reductions in grazing pressure by domestic and wild ungulates, control of invading exotic plants, reductions in human activities such as offroad vehicle usage, control of road access and associated motorized traffic, and control of other ground-disturbing factors not part of site preparation.

3. (In support of issue no. 2) Restore the microbiotic crust in ERUs with potential for redevelopment (i.e., areas near propagule sources, and with suitable soil, vegetation, and climatic characteristics [see Belnap 1993, Belnap 1995, Kaltenecker 1997, Kaltenecker and Wicklow-Howard 1994]): Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs and, to a lesser extent, the Columbia Plateau ERU (Hann and others 1997, map 3.59).

4. (In support of issue no. 3) Reduce the negative effects of factors associated with roads on species in family 12 (tables 13 and 14). Negative effects associated with roads include human disturbance of sharp-tailed grouse leks and recreational shooting of Idaho ground squirrels. Example mitigations include seasonal road closures during the grouse lekking period and restrictions on recreational shooting of ground squirrels.

5. (In support of issue no. 4) Restore historical, native composition of meadow vegetation within the range of the Idaho ground squirrel; augment restoration with possible measures to control populations of Columbian ground squirrels, which may have a competitive advantage with the Idaho ground squirrel in areas of sympatry.

6. (In support of issue no. 4) Hasten recovery of populations of sharp-tailed grouse through the use of translocations in areas where habitats have undergone restoration or are deemed to be of sufficient quality and size to support the species' long-term persistence. Use land transactions to consolidate areas containing suitable habitats, or that could be restored to suitability, as part of translocation strategies.

7. (In support of issue no. 5) Conduct midscale step-down assessment of current conditions relative to landscape patterns of succession-disturbance regimes. Focus short-term restoration of watersheds on those that are in high departure, do not contain susceptible populations of species of high conservation concern, and are at high risk of loss of biophysical capability.

Species Negatively Affected by Factors Associated With Roads

Species-Road Relations

Various road-associated factors can negatively affect habitats and populations of terrestrial vertebrates (Bennett 1991, Forman and Hersperger 1996, Forman and others 1997, Mader 1984, Trombulak and Frissell 2000). We identified 13 factors that were consistently associated with roads in a manner deleterious to terrestrial vertebrates (table 13), based on results from a plethora of studies conducted in Europe, North America, and Australia (with examples of this literature cited in table 13). Effects of road-associated factors can be direct, such as habitat loss and fragmentation (Miller and others 1996, Reed and others 1996) or indirect, such as population displacement or avoidance in areas near roads in relation to motorized traffic and associated human activities (Mader 1984). Indirect effects can be subtle, such as the negative effects of all-terrain vehicles (Busack and Bury 1974, Lukenbach 1978) that can and do travel over a myriad of off-road and on-road conditions, and whose movements are facilitated by road access.

Based on the factors listed in table 13, >70 percent of the 91 broad-scale species of focus were found to be negatively affected by one or more factors associated with roads (table 14). Negative factors associated with roads, and their specific effects on habitats and populations, are diverse and not always easily recognized. These factors go beyond the obvious, direct effects of habitat loss from road construction and maintenance, which affects all species. Despite the diversity of factors and effects, several generalizations are obvious from the summaries in table 13 and from the literature cited in table 14:

1. Road construction converts large areas of habitat to nonhabitat (Forman 2000, Hann and others 1997, Reed and others 1996); the resulting motorized traffic facilitates the spread of exotic plants and animals, further reducing quality of habitat for native flora and fauna (Bennett 1991, Hann and others 1997). Roads also create habitat edge (Mader 1984, Reed and others 1996); increased edge changes habitat in favor of species that use edges, and to the detriment of species that avoid edges or experience increased mortality near or along edges (Marcot and others 1994).

2. Species that depend on large trees, snags, or down logs, particularly cavity-using birds and mammals, are vulnerable to increased harvest of these structures along roads (Hann and others 1997). Motorized access facilitates firewood cutting, as well as commercial harvest, of these structures.

3. Several large mammals are vulnerable to poaching, such as caribou, pronghorn, mountain goat, bighorn sheep, wolf, and grizzly bear (e.g., Dood and others 1985, 1986; Knight and others 1988; McLellan and

Table 13—Road-associated factors that negatively affect habitats or populations of terrestrial vertebrates, a generalized description of each factor's effect in relation to roads, and example citations linking roads as a facilitator of the factors and effects

Road-associated factor	Effect of factor in relation to roads	Example citations
Snag reduction	Reduction in density of snags due to their removal near roads, as facilitated by road access	Hann and others (1997), Quigley and others (1996)
Down log reduction	Reduction in density of large logs due to their removal near roads, as facilitated by road access	Hann and others (1997), Quigley and others (1996)
Habitat loss and fragmentation	Loss and resulting fragmentation of habitat due to establishment and maintenance of road and road right-of-way	Forman and others (1997), Reed and others (1996)
Negative edge effects	Specific case of fragmentation for species that respond negatively to openings or linear edges created by roads (such as habitat-interior species [Marcot and others 1994])	Forman and others (1997), Mader (1984), Reed and others (1996)
Over-hunting	Nonsustainable or nondesired legal harvest by hunting, as facilitated by road access	Christensen and others (1991), Unsworth and others (1993)
Over-trapping	Nonsustainable or nondesired legal harvest by trapping, as facilitated by road access	Bailey and others (1986), Hodgman and others (1994)
Poaching	Increased illegal take (shooting or trapping) of animals, as facilitated by road access	Cole and others (1997), McLellan and Shackleton (1988)
Collection	Collection of live animals for human uses (e.g., amphibians and reptiles collected for use as pets), as facilitated by the physical characteristics of roads or by road access	Nussbaum and others (1983)
Harassment or disturbance at specific use sites	Direct interference of life functions at specific use sites due to human or motorized activities, as facilitated by road access (e.g., increased disturbance of nest sites, breeding leks, or communal roost sites)	Forman (1995), White (1974)
Collisions	Death or injury resulting from a motorized vehicle running over or hitting an animal on a road	Blumton (1989), Boarman and Sazaki (1996), Vestjens (1973)
Movement barrier	Preclusion of dispersal, migration, or other movements as posed by a road itself or by human activities on or near a road or road network	Bennett (1991), Mader (1984)
Displacement or avoidance	Spatial shifts in populations or individual animals away from a road or road network in relation to human activities on or near a road or road network	Forman and Hersperger (1996), Mech and others (1988)
Chronic, negative interactions with humans	Increased mortality of animals (e.g., euthanasia or shooting of gray wolves or grizzly bears) due to increased contact with humans, as facilitated by road access	Mace and others (1996), Thiel (1985)

Table 14—Effects of road-associated factors on habitats and populations of broad-scale species of focus[a]

Group	Species	Snag reduction	Down log reduction	Negative edge effects	Over-hunting	Over-trapping	Poaching	Collection	Harass-ment	Collisions	Movement barrier	Displace-ment	Chronic, negative interactions
1	Pygmy nuthatch	A		B									
1	White-breasted nuthatch	A		B									
1	White-headed woodpecker	A		B									
2	Lewis' woodpecker (migrant)	A							3				
3	Western gray squirrel			B			60					60	
4	Blue grouse (winter)												
5	Fisher	A	A	B		10, 11				12*		10*	
5	Flammulated owl	A		B									
5	N. goshawk (summer)			B									
5	American marten	A	A	B		4*, 5, 6							
6	Brown creeper	A		B									
6	Chestnut-backed chickadee	A		B									

Table 14—Effects of road-associated factors on habitats and populations of broad-scale species of focus[a] (continued)

Group	Species	Snag reduction	Down log reduction	Negative edge effects	Over-hunting	Over-trapping	Poaching	Collection	Harass-ment	Collisions	Movement barrier	Displace-ment	Chronic, negative interactions
6	Golden-crowned kinglet			B									
6	Hammond's flycatcher			B									
6	Hoary bat	A											
6	Pileated woodpecker	A	A	B									
6	Silver-haired bat	A											
6	Vaux's swift	A		B									
6	Varied thrush												
6	Williamson's sapsucker	A		B									
6	Winter wren	A	A	B									
7	Boreal owl	A		B									
8	Great gray owl	A											
9	Black-backed woodpecker	A		B									
10	Olive-sided flycatcher												
11	Three-toed woodpecker	A		B									

Table 14—Effects of road-associated factors on habitats and populations of broad-scale species of focus[a] (continued)

Group	Species	Snag reduction	Down log reduction	Negative edge effects	Over-hunting	Over-trapping	Poaching	Collection	Harass-ment	Collisions	Movement barrier	Displace-ment	Chronic, negative interactions
11	White-winged crossbill									69*			
12	Woodland caribou			B			13*			13*			
13	N. flying squirrel	A	A	B									
14	Hermit warbler			B									
15	Pygmy shrew										C		
15	Wolverine		A			47, 55			68				
16	Lynx		A			31*, 32			33				
17	Blue grouse (summer)								34				
17	Mountain quail (summer)								34				
18	Lazuli bunting												
19	Gray wolf			18*			16*, 17, 20, 21, 22, 72			17*, 20*, 21, 22*, 72		15*, 18*, 19*, 23*, 72	16, 19*, 17, 20, 22, 70* 72
19	Grizzly bear						24, 25, 26, 14*, 72			24*, 26*, 72		14*, 25*, 28*, 29*, 30*, 72	24, 14*, 30*, 72
20	Mountain goat				40		C		39, 41	38*		39*	
21	Long-eared owl												
22	California bighorn sheep				C		C		66			C	

Table 14—Effects of road-associated factors on habitats and populations of broad-scale species of focus[a] (continued)

Group	Species	Snag reduction	Down log reduction	Negative edge effects	Over-hunting	Over-trapping	Poaching	Collection	Harass-ment	Collisions	Movement barrier	Displace-ment	Chronic, negative interactions
22	Rocky Mt. bighorn sheep				65		65		44			42, 43, 57	
23	Broad-tailed hummingbird												
23	Rufous hummingbird												
24	Black-chinned hummingbird												
24	California mountain kingsnake							35, 67		36*			
24	Sharptail snake		A					67		36*			
25	N. goshawk (winter)												
26	Fringed myotis	A					7, 71		7, 71				
26	Long-eared myotis	A					7, 71		7, 71				
26	Long-legged myotis	A					7, 71		7, 71				
26	Yuma myotis						7, 71		7, 71				
27	Townsend's big-eared bat						7, 8, 71		7, 8, 71			7, 8	

Table 14—Effects of road-associated factors on habitats and populations of broad-scale species of focus[a] (continued)

Group	Species	Snag reduction	Down log reduction	Negative edge effects	Over-hunting	Over-trapping	Poaching	Collection	Harass-ment	Collisions	Movement barrier	Displace-ment	Chronic, negative interactions
27	Pine siskin									69*			
28	Pallid bat	A											
28	Spotted bat												
28	Western small-footed myotis						7, 71		7, 71				
29	W. bluebird	A											
30	Ash-throated flycatcher	A											
30	Bushtit												
31	Burrowing owl						37		48				
31	Ferruginous hawk						73, 74, 75		49, 50	73			
31	Lark sparrow												
31	Pronghorn			27	46		27		9*, 46		45*, 27		
31	Short-eared owl							37, 76, 77		51	76, 77		
31	Vesper sparrow												
31	W. meadowlark												52

Table 14—Effects of road-associated factors on habitats and populations of broad-scale species of focus[a] (continued)

Group	Species	Snag reduction	Down log reduction	Negative edge effects	Over-hunting	Over-trapping	Poaching	Collection	Harass-ment	Collisions	Movement barrier	Displace-ment	Chronic, negative interactions
32	Ground snake							56, 67		C			
32	Longnose leopard lizard							56, 67		C			
32	Longnose snake							56, 67		C			
32	Mojave black-collared lizard							56, 67		C			
32	Preble's shrew		A								C		
32	Striped whipsnake							56, 67		C			
32	Uinta ground squirrel				C		C						
32	Washington ground squirrel				C		C						
32	White-tailed antelope squirrel				C		C						
32	Wyoming ground squirrel				C		C						
33	Brewer's sparrow												
33	Lark bunting												
33	Pygmy rabbit												
33	Sagebrush vole										C		
33	Sage sparrow												

Table 14—Effects of road-associated factors on habitats and populations of broad-scale species of focus[a] (continued)

Group	Species	Snag reduction	Down log reduction	Negative edge effects	Over-hunting	Over-trapping	Poaching	Collection	Harass-ment	Collisions	Movement barrier	Displace-ment	Chronic, negative interactions
33	Sage grouse (summer)								C				
33	Sage grouse (winter)												
33	Sage thrasher												
34	Black-throated sparrow												
34	Kit fox						63						
35	Loggerhead shrike									61, 62, 64*			
36	Col. sharp-tailed grouse (summer)						53, 54		53, 54				
37	Clay-colored sparrow												
37	Grasshopper sparrow												
37	Idaho ground squirrel						58, 59						
38	Black rosy finch												
38	Gray-crowned rosy finch												

Table 14—Effects of road-associated factors on habitats and populations of broad-scale species of focus[a] (continued)

Group	Species	Snag reduction	Down log reduction	Negative edge effects	Over-hunting	Over-trapping	Poaching	Collection	Harass-ment	Collisions	Movement barrier	Displace-ment	Chronic, negative interactions
39	Lewis' woodpecker (resident)	1, 2											
40	Brown-headed cowbird												

* = Cited reference makes a direct link with roads as a facilitator of the factor's effect. Cited references not marked by an asterisk establish the factor as a problem for the species but do not address whether roads facilitate the factor's effect.

[a] Factors and effects listed here are defined in table 13. Factors and effects were documented from empirical literature and literature summaries, with each number listed below denoting a footnoted study. Presumed effects are denoted by a letter corresponding to a footnote that describes each presumed effect and cites the supporting literature related to other species of the taxa. A factor not marked with a number or letter (blank cells) indicates that we could find no research results on the factor in relation to the species or related taxa. Blank cells in this table therefore indicate no studies found rather than no effect of the factor.

A = Species depends on snags, down logs, or both structures to meet life requisites (Thomas and others 1979; volume 3, appendix 1, table 2); consequently, the species presumably is affected by a reduction in density of these structures and the documented links of this effect with roads (Hann and others 1997, Quigley and others 1996).

B = Species presumably responds negatively to openings or linear edges created by roads based on its dependence on closed-canopy habitats and lack of dependence on disturbed or contrasting habitats of openings and closed-canopy forests (such as "habitat-interior" species [Marcot and others 1994]); additional research is needed, however, to validate the presumption.

C = Factor is presumed to have a negative effect on the species based on documented effects of the factor on species of similar life history or taxa. For poaching or over-hunting of large mammals, documented effects include Cole and others (1997), Dood and others (1986), Knight and others (1988), McLellan and Shackleton (1988), Mech (1970), Scott and Servheen (1985), Stelfox (1971), and Yoakum (1978). For over-harvest and poaching of ground squirrels ("plinking"), effects are described by Ingles (1965). For collisions of reptiles with vehicles, documented effects are summarized by Vestjens (1973) and Bennett (1991). For roads as barriers to movements of small mammals, documented effects are described by Mader (1984), Swihart and Slade (1984), and Merriam (1989). For displacement of all taxa, documented effects are summarized by Bennett (1991). For any other effects on taxa marked with a "C" but not explicitly identified here, documented effects are summarized by Bennett (1991). Presumed effects of factors marked with a "C" require additional research to validate the presumption.

References:

1. Hann and others 1997; 2. Quigley and others 1996; 3. Bock 1970; 4. Hodgman and others 1994; 5. Fortin and Cantin 1994; 6. Thompson 1994; 7. Nagorsen and Brigham 1993; 8. Idaho State Conservation Effort 1995; 9. Autenrieth 1978; 10. Coulter 1966; 11. Jones 1991; 12. Paragi and others 1994; 13. Scott and Servheen 1985; 14. Mace and others 1996; 15. Thurber and others 1994; 16. Mech 1970; 17. Van Ballenberghe and others 1975; 18. Mladenoff and others 1995; 19. Thiel 1985; 20. Fritts and others 1985; 21. Pletscher and others 1997; 22. Bangs and Fritts 1996; 23. Singer 1979; 24. Knight and others 1988; 25. McLellan and Shackleton 1988; 26. Dood and others 1986; 27. Yoakum 1978; 28. Kasworm and Manley 1990; 29. Mattson and others 1987; 30. Mattson and others 1992; 31. Bailey and others 1986; 32. Parker and others 1983; 33. Koehler and Brittell 1990; 34. ICBEMP 1996d; 35. ICBEMP 1996a; 36. Brown and others 1995; 37. Marti and Marks 1989; 38. Singer 1978; 39. Chadwick 1972; 40. Johnson 1983; 41. Joslin 1986; 42. Hamilton and others 1982; 43. Hicks and Elder 1979; 44. MacArthur and others 1979; 45. Bruns 1977; 46. Helms 1978; 47. Hornocker and Hash 1981; 48. Green and Anthony 1989; 49. Bechard and Schmutz 1995; 50. Lokemoen and Duebbert 1976; 51. Olendorff and Stoddart 1974; 52. Lanyon 1994; 53. Giesen and Connelly 1993; 54. Tirhi 1995; 55. Banci 1994; 56. Lehmkuhl and others 1997; 57. Geist 1971; 58. Moroz 1995; 59. USDA Forest Service and USDI Fish and Wildlife Service 1996; 60. Washington Department of Wildlife 1993b; 61. Yosef 1996; 62. Flickinger 1995; 63. DeStefano 1990; 64. Blumton 1989; 65. Stelfox 1971; 66. Taylor and others 1993; 67. Nussbaum and others 1983; 68. Copeland 1996; 69. Ehrlich and others 1988; 70. Mech 1973; 71. Tuttle 1988; 72. Frederick 1991; 73. Howard 1975; 74. Harmata 1981; 75. Gilmer and others 1985; 76. Clark 1975; 77. Holt 1992.

Shackleton 1988; Mech and others 1970; Scott and Servheen 1985; Stelfox 1971; Yoakum 1978). Roads facilitate poaching (Cole and others 1997).

4. Wolves and grizzly bears experience chronic, negative interactions with humans, and roads are a key facilitator of such interactions (Mace and others 1996, Mattson and others 1992, Thiel 1985). Repeated, negative interactions of these two species with humans increase mortality of both species and often cause high-quality habitats near roads to function as population sinks (Mattson and others 1996a, 1996b; Mech 1973).

5. Carnivorous mammals such as marten, fisher, lynx, and wolverine are vulnerable to over-trapping (Bailey and others 1986, Banci 1994, Coulter 1966, Fortin and Cantin 1994, Hodgman and others 1994, Hornocker and Hash 1981, Jones 1991, Parker and others 1983, Thompson 1994, Witmer and others 1998), and over-trapping can be facilitated by road access (Bailey and others 1986, Hodgman and others 1994, Terra-Berns and others 1997, Witmer and others 1998). Movement and dispersal of some of these species also are believed to be inhibited by high rates of traffic on highways (Ruediger 1996) but this belief has not been validated. Carnivorous mammals such as lynx also are vulnerable to increased mortality from highway accidents with motorized vehicles (as summarized by Terra-Berns and others 1997).

6. Reptiles seek roads for thermal cooling and heating, and in doing so, these species experience significant, chronic mortality from motorized vehicles (Vestjens 1973). Highways and other roads with moderate to high rates of motorized traffic may function as population sinks for many species of reptiles, thereby resulting in reduced population size and increased isolation of populations (Bennett 1991). For example, in Australia, 5 million reptiles and frogs are estimated to be killed annually by motorized vehicles on roads (Ehmann and Cogger 1985, as cited by Bennett 1991). Roads also facilitate human access into habitats for collection and killing of reptiles.

7. Many species are sensitive to harassment or human presence during particular seasons, which is often facilitated by road access; potential reductions in productivity, increases in energy expenditures, or displacements in population distribution or habitat use can occur (Bennett 1991, Mader 1984 Trombulak and Frissell 2000). Examples are human disturbance of leks (sage grouse and sharp-tailed grouse), of nests (raptors such as ferruginous hawk), and of dens (kit fox). Another example is elk avoidance of large areas near roads open to traffic (Lyon 1983, Rowland and others 2000), with the magnitude of elk avoidance increasing with rate of traffic (Wisdom and others 1999, Johnson and others 2000).

8. Bats are vulnerable to disturbance and displacement caused by human activities in caves, mines, and on rock faces (Hill and Smith 1984, Nagorsen and Brigham 1993). Cave or mine exploration and rock-climbing are examples of recreation that potentially reduce population fitness of bats that roost in these sites (Nagorsen and Brigham 1993, Tuttle 1988). Such activities may be facilitated by human developments and road access (Hill and Smith 1984).

9. Ground squirrels often are targets of recreational shooting ("plinking"), which is facilitated by human developments and road access (Ingles 1965). Most species of ground squirrels included in our analysis are local endemics; consequently, these small, isolated populations may be especially vulnerable to recreational shooting, potentially resulting in severe reductions or local extirpations of populations.

10. Roads often restrict the movements of small mammals (Mader 1984, Merriam and others 1988, Swihart and Slade 1984). Consequently, roads can function as barriers to population dispersal and movement of some species of small mammals (Oxley and Fenton 1974).

11. Many granivorous birds are attracted to grains and seeds along roadsides, thereby resulting in high mortality from vehicle collisions (Vestjens 1973). For example, pine siskins and white-winged crossbills are attracted to road salt, which can result in mortality from vehicle collisions (Ehrlich and others 1988).

12. Terrestrial vertebrates inhabiting areas near roads accumulate lead and other toxins that originate from motorized vehicles, with potentially lethal but largely undocumented effects (Bennett 1991).

In summary, no terrestrial vertebrate taxa appear immune to the myriad of road-associated factors that degrade habitat or that increase mortality. These multifaceted effects have strong management implications for landscapes characterized by moderate to high densities of roads, which is the typical pattern across large areas of the basin (figs. 21, 22). That is, about 51 percent of the basin supports road densities estimated as moderate, high, or extremely high (Quigley and others 1996). Specific implications of this pattern for species affected negatively by roads are as follows:

1. Source habitats likely are underused for many of the species listed in table 14 when such habitats exist in areas that contain moderate to high road density. In some cases, the presence of moderate or high densities of roads may index areas that function as population sinks and that would otherwise function as source environments if road density was low or zero.

2. Species listed in table 14 whose source habitats have undergone strong declines across the basin (see "Species-Level Results", and "Group-Level Results," this volume) may be affected in a synergistic manner by the combination of scarce or declining habitats and negative factors associated with roads. If this is true, our analysis of trends in source habitats underestimates the presumed effects of change in environmental conditions on such species and groups.

3. Mitigating the negative effects stemming from road-associated factors on the species listed in table 14 will be as challenging, or perhaps more challenging, than that of maintaining or restoring vegetation used as source habitats by these species. Mitigation will require effective control of human access and roads in relation to management of livestock, timber, recreation, hunting, trapping, and mineral development. Mitigation will require intensive investments of money and resources that are sustained over long periods. Setting priorities for mitigation and implementing effective mitigative measures likely will require extensive, new research about species-road relations. Such research could be designed and conducted as joint management experiments between managers and researchers.

Mapping Road Density in Relation to Abundance of Source Habitats for Terrestrial Carnivores

Composite carnivore map of habitat abundance and road density—Subbasins having both zero to low road density and moderate to high abundance of source habitats for any of the four species of terrestrial carnivores (grizzly bear, gray wolf, wolverine, or lynx), considering current habitat abundance within each of the historical range of the species, were concentrated in seven distinct areas (fig. 23), identified as follows: area 1—the Greater Yellowstone Area, defined as subbasins within the eastern portion of the Snake Headwaters ERU; area 2—the Northern Continental Divide Area, centered within and adjacent to Glacier National Park and composed of subbasins within the extreme eastern portion of the Northern Glaciated Mountains ERU; area 3—the North Cascades Area, defined as the segment of North Cascades National Park that overlays one subbasin of the Northern Cascades ERU; area 4—the Bitterroot-Central Idaho Area whose subbasins overlap the Selway-Bitterroot Wilderness and the Frank Church River of No Return Wilderness within the Central Idaho Mountains ERU; area 5—the Eagle Cap Wilderness-Hells Canyon Area, composed of subbasins within the extreme eastern portion of the Blue Mountains ERU; area 6—the Owyhee Area, defined as subbasins within the southern half of the Owyhee Uplands ERU and southeast portion of the Northern Great Basin ERU; and area 7—the Crater Lake Area, composed of the portion of Crater Lake National Park that overlays one subbasin in the Upper Klamath ERU (fig. 23). Estimated habitat abundance for each carnivore species in relation to road density is summarized in the following sections and compared to the composite carnivore habitat-roads map of figure 23.

Grizzly bear—Subbasins having both zero to low road density and moderate to high abundance of source habitats for grizzly bear, considering current conditions within the historical range of the species (fig. 24), were concentrated in all seven areas that were identified on the composite carnivore habitat-roads map (compare fig. 24 with fig. 23). Interestingly, four of the seven areas—Greater Yellowstone, Continental Divide, North Cascades, and Bitterroot-Central Idaho—are within areas currently occupied by grizzly bear, or are within areas that have had occasional sightings or potential occurrences since 1970

Figure 21—Pixel-based predictions of road density classes within the basin (from Quigley and others 1996).

Figure 22—Generalized classes of road density estimated to dominate each subbasin. See “Methods”, “Summarizing Knowledge About Species-Road Relations,” for description of the steps used to estimate the dominant road class.

Figure 23—Seven areas composed of one or more subbasins that are dominated by zero to low road density and that also are dominated by moderate to high abundance of source habitats for either grizzly bear, gray wolf, wolverine, or lynx. Area number, name, and location are: area 1—Greater Yellowstone Area, defined as subbasins within the eastern portion of the Snake Headwaters ERU; area 2—Northern Continental Divide Area, centered within and adjacent to Glacier National Park and composed of subbasins within the extreme eastern portion of the Northern Glaciated Mountains ERU; area 3—North Cascades Area, defined as the segment of North Cascades National Park that overlays one subbasin of the Northern Cascades ERU; area 4—Bitterroot-Central Idaho Area whose subbasins overlap the Selway-Bitterroot Wilderness and the Frank Church River of No Return Wilderness within the Central Idaho Mountains ERU; area 5—Eagle Cap Wilderness-Hells Canyon Area, composed of subbasins within the extreme eastern portion of the Blue Mountains ERU; area 6—Owyhee Area, defined as subbasins within the southern half of the Owyhee Uplands ERU and southeast portion of the Northern Great Basin ERU; and area 7— Crater Lake Area, composed of the portion of Crater Lake National Park that overlays one subbasin in the Upper Klamath ERU.

(Mattson and others 1995). The other three areas—Eagle Cap Wilderness-Hells Canyon, Owyhee, and Crater Lake—have had no verified grizzly bear occurrences since early European settlement (late 1800s to early 1900s, Mattson and others 1995), although use of lower elevations within the Owyhee Area was probably incidental or infrequent.[4]

Also of interest is the fact that two other areas currently occupied by grizzly bear—the Selkirk and Cabinet-Yaak Ecosystems (Mattson and others 1995), each located within the portion of the Northern Glaciated Mountains ERU in northern Idaho and northwestern Montana—contain no subbasins having both moderate to high abundance of source habitats and zero to low road density (fig. 24). Consequently, these areas were not detected by our mapping criteria. That is, all subbasins within the Selkirk and Cabinet-Yaak Ecosystems have low abundance of source habitats, moderate to high road density, or both (fig. 24). Although our mapping criteria did not detect these two areas, it is noteworthy that the Selkirk and Cabinet-Yaak ecosystems are believed to contain less than 20 grizzly bears (Knick and Kasworm 1989, Wielgus and Bunnell 1995). The relatively small number of bears present in these ecosystems suggests that environmental conditions may not be as conducive to maintenance of self-sustaining bear populations as would other areas of the basin that we identified with our mapping criteria.

Gray wolf—Subbasins having both zero to low road density and moderate to high abundance of source habitats for gray wolf, considering current conditions within the historical range of the species, were concentrated in five areas: Greater Yellowstone, Continental Divide, Bitterroot-Central Idaho, Owyhee, and Eagle Cap Wilderness-Hells Canyon (compare fig. 25 with fig. 23). Three of these same areas used by grizzly bear—Greater Yellowstone, Continental Divide, and Bitterroot-Central Idaho—also are currently occupied by wolf (USDI Fish and Wildlife Service 1997). The other two areas—Eagle Cap Wilderness-Hells Canyon and Owyhee—have had no verified wolf occurrences since early European settlement (USDI Fish and Wildlife Service 1987) and, in contrast to the other three areas, have not benefitted from translocation programs or from immigration of wolves from areas outside the basin (USDI Fish and Wildlife Service 1997). However, recent sightings of radio-collared wolves (from Idaho) in the Blue Mountains ERU suggest that the Eagle Cap Wilderness-Hells Canyon Area may already be used by some wolves at least seasonally.

Wolverine—Subbasins having both zero to low road density and moderate to high current abundance of source habitats for wolverine, considering all areas within the historical range of the species, were concentrated in the Greater Yellowstone, Northern Continental Divide, North Cascades, Bitterroot-Central Idaho, Eagle Cap Wilderness-Hells Canyon, and Crater Lake Areas (compare fig. 26 with fig. 23). Interestingly, all six of these areas have had verified occurrences of wolverine since 1961, based on mapped observations by Maj and Garton (1994). The largest concentration of these occurrences appears to be within the Bitterroot-Central Idaho Area, based on an overlay of fig. 26 with Maj and Garton's (1994) 1961-93 maps of wolverine observations (Wisdom 2000).

Also of interest is the fact that >90 percent of the wolverine observations compiled by Maj and Garton (1994) for 1961-93 encompass subbasins containing moderate to high abundance of the source habitats of this species (Wisdom 2000). Moreover, <10 percent of these verified wolverine observations were located in subbasins containing low abundance of source habitats. This high concentration of wolverine observations in relation to subbasins having moderate to high abundance of wolverine source habitats also is congruent with areas of the basin that likely have higher potential to support reproductive den sites (per descriptions of Copeland [1996] and Magoun and Copeland [1998]).

Lynx—The map for lynx (fig. 27) was similar to that for wolverine (fig. 26). That is, the same five areas—Greater Yellowstone, Northern Continental Divide, North Cascades, Bitterroot-Central Idaho, and Eagle Cap Wilderness-Hells Canyon—contained the only subbasins having both moderate to high habitat abundance and zero to low road densities (compare fig. 27 with fig. 23). The sixth area identified for wolverine—Crater Lake—was assumed to be outside the geographic

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[4] Personal communication. 1998. David Mattson, U.S.Geological Survey, Forest and Rangeland Ecosystem Science Center and Department of Fish and Wildlife Resources, University of Idaho, Moscow, ID 83844-1136.

VIEW

MAP

Figure 24—Low, moderate, and high abundance of source habitats for grizzly bear in relation to zero and low road densities for each of 164 subbasins in the interior Columbia basin.

VIEW

MAP

Figure 25—Low, moderate, and high abundance of source habitats for gray wolf in relation to zero and low road densities for each of 164 subbasins in the interior Columbia basin.

Figure 26—Low, moderate, and high abundance of source habitats for wolverine in relation to zero and low road densities for each of 164 subbasins in the interior Columbia basin.

Figure 27—Low, moderate, and high abundance of source habitats for lynx in relation to zero and low road densities for each of 164 subbasins in the interior Columbia basin.

range of the lynx (Marcot and others, in prep.). A more recent summary of occurrence data (McKelvey and others 1999), suggests, however, that lynx occur in portions of the southern Cascades of Oregon outside the range map of Marcot and others (in prep.).

In contrast to wolverine, most verified lynx locations, based on combined data from Maj and Garton (1994) and Lewis and Wenger (1998), corresponded to subbasins having a high abundance of lynx source habitats, regardless of road density (Wisdom 2000). That is, lynx locations verified by Maj and Garton (1994) from 1961 to 1993 and by Lewis and Wenger (1998) from 1977 to 1998 corresponded closely to subbasins of high abundance of source habitats rather than to subbasins having both zero to low road density and moderate to high habitat abundance. Similar results were found when lynx locations of McKelvey and others (1999) were overlaid in relation to our subbasin maps of lynx habitat abundance and road density (Wisdom 2000).

Management implications—Several interesting patterns emerged from the overlays of road density with current habitat abundance for grizzly bear, wolf, wolverine, and lynx, especially when current or recent occurrence data for all four species was considered. First, most of the subbasins having both moderate to high abundance of source habitats and zero to low road density occurred within or adjacent to National Parks or Wilderness Areas. Second, most of these subbasins occurred within areas of high elevation. Third, most of these subbasins were identified within areas currently occupied by most or all of the four species. Two other areas, however, currently occupied by grizzly bear—the Selkirk and Cabinet-Yaak Ecosystems (Mattson and others 1995)—were not identified by our mapping exercise because subbasins within these areas had low abundance of source habitats, moderate to high road density, or both (fig. 24). And finally, the pattern of lynx observations corresponded more closely to subbasins of high habitat abundance rather than to subbasins identified by our mapping criteria.

Although these patterns are interesting and often agreed in general terms with knowledge of habitat requirements and known occurrences of all four species, our maps are strictly qualitative and not validated through formal research. As such, our maps should be considered working hypotheses that must be tested as part of large-scale studies that evaluate a range of environmental conditions in relation to rigorous surveys of the presence and absence of each species. Such an evaluation has been proposed for lynx (Ruggiero and others 1999) and similar evaluations have occurred for wolf and grizzly bear in parts of the basin (e.g., Merrill and others [1999] for grizzly bear and evaluations described by Bangs and Fritts [1996] for gray wolf). Notably missing are any large-scale evaluations for wolverine or more comprehensive evaluations for wolf or grizzly bear that encompass the entire basin and adjacent ecosystems. Such evaluations are needed to corroborate the patterns displayed in our maps and to elucidate more fine-scale relations between environmental conditions and the likelihood of population occurrence for all four species.

Given these limitations, our maps could be useful to managers when considered in tandem with other large-scale data on wolf, grizzly bear, wolverine, and lynx. The mapping pattern shown here illustrates an especially important point for all four species: that large areas of the basin composed of moderate or high abundance of source habitats may not be used, or may be underused, by many or all of the four species, presumably because of negative interactions with humans that are facilitated by roads and human developments. For gray wolf and grizzly bear, researchers have verified a strong, negative relation between road density and population fitness (e.g., Mace and others 1996, Mattson and others 1996b, Mech and others 1988, Thiel 1985). Similar relations have been hypothesized for wolverine and lynx within the basin (ICBEMP 1996b, 1996c), and limited research on lynx (Bailey and others 1986 and as summarized by Terra-Berns and others 1997) outside the basin supports the hypothesis that population fitness is lower in areas characterized by increased road access. Because of these observed or suspected effects on population fitness, our maps identified a handful of large areas of abundant source habitats that have low road density. Presumably these areas have higher potential to support populations that could persist without additive mortality that may be caused by road-associated factors. Thus, managers interested in conserving the few large blocks of remaining habitats that are relatively secure from human disturbances for terrestrial carnivores would want to focus on maintenance and improvement of the seven areas identified in our analysis (fig. 23), particularly the Greater Yellowstone, Continental Divide, North Cascades, and Bitterroot-Central Idaho Areas. These areas could be effective "building blocks" from which an overall network of habitat and human activity strategies could be devised to ensure a high probability of well-distributed, persistent populations of all four species in the basin.

Validating Agreement Between Change in Source Habitats and Expert Opinion-Based Habitat Outcomes

Direction of change (historical to current) in source habitats agreed 81 to 84 percent of the time with a like direction of change in historical to current habitat or cumulative effects outcomes (Lehmkuhl outcomes) for 68 of our broad-scale species of focus that also were evaluated by Lehmkuhl and others (1997). The consistency of agreement between our trends in source habitats and the Lehmkuhl outcomes reflected strong, underlying congruity; this was true for habitat trends in relation to the habitat outcomes, as well as to the cumulative effects outcomes, for both the Eastside EIS and the Upper Columbia River EIS areas.

Thirteen species, however, had trends in source habitats that differed in direction from either the habitat or the cumulative effects outcomes (table 15). Trends in source habitats versus the Lehmkuhl outcomes generally differed for one of two reasons: (1) the expert panels for Lehmkuhl and others (1997) considered fine-scale characteristics of habitat, such as snag abundance, riparian features, or habitat patchiness, that we could not address with the large pixel size (100 ha [247 ac]) used for our source habitat analysis; or (2) the expert panels for Lehmkuhl and others (1997) considered effects of roads or other nonvegetative factors that we did not consider in our source habitat analysis. These two differences in evaluation criteria potentially account for contradictions in direction in trends of source habitats versus outcomes for 10 of the 13 species listed in table 15. For example, the expert panels for Lehmkuhl and others (1997) cited fine-scale habitat features as the primary basis for evaluating 8 of the 13 species, and cited roads or other nonvegetative features, as the primary basis for evaluating 2 other species. When these 10 species are removed from the analysis, the direction of change in source habitats versus the direction of change in the Lehmkuhl outcomes agreed 95 to 97 percent of the time.

Although such high agreement between source habitat trends and the Lehmkuhl outcomes is compelling, it is not unexpected for at least two reasons. One is the overlap (at least 25 percent) that existed between experts who served on the panels of Lehmkuhl and others (1997) and the experts who served on our panels that identified source habitats; experts serving on both panels would be expected to identify source habitats in the same manner in which they based their outcome projections. A second reason is that most species experts tend to agree on the habitat factors and effects that contribute to population persistence, and all of these experts draw from the same set of empirical knowledge, regardless of overlap in experts serving on both panels.

Nonetheless, the congruity between trends in source habitats and those found in Lehmkuhl and others (1997), although strictly correlative, indicates that direction of change in source habitats reflects a like direction of change in projected, long-term population persistence for any given species. That is, species whose source habitats underwent a strong decline from historical to current periods also should be expected to have an estimated lower likelihood of population persistence currently than historically. Moreover, a strong decline in source habitats presumably contributes largely or wholly to the reduced likelihood of population persistence, based on empirical knowledge conveyed by the experts. These final points are important to Federal managers who must demonstrate compliance with viability requirements of ESA, NFMA, and related laws. Given the congruity of results presented here, it seems that our methods of analyzing trends in source habitats may be useful in analyzing future habitat scenarios for EIS alternatives in terms of compliance with Federal viability requirements.

Major Findings and Implications

1. Source habitats for most species declined strongly from historical to current periods across large areas of the basin. Strongest declines were for species dependent on low-elevation, old-forest habitats (family 1), for species dependent on combinations of rangelands or early-seral forests with late-seral forests (family 8), and for species dependent on

Table 15—Species for which trends in source habitats differed from habitat outcomes of Lehmkuhl and others (1997)

Common name	Relative change in source habitats	Change in habitat outcome	Reasons for habitat outcome (from panel notes)	Most likely reasons for difference
Vaux's swift	-7.99	Increase	Increase in habitat due to fire suppression and subsequent increase in grand-fir, which provides source habitat for this species	Although grand fir did increase in some areas, when considering all source habitats for Vaux's swift, habitat declined slightly basin-wide.
Fringed myotis	17.36	Decrease	Loss of large snags and increased human disturbance	We did not evaluate change in snag abundance or the effects of human disturbance.
Long-legged myotis	17.16	Decrease	Loss of large snags	We did not evaluate changes in snag abundance.
Three-toed woodpecker	22.44	Decrease	Loss of snags	We did not evaluate changes in snag abundance.
Mountain quail	16.09	Decrease	Reduction in riparian shrub cover and species composition due to grazing	We did not analyze the fine-scale attributes of riparian habitats.
Black-chinned hummingbird	14.37	Decrease	Fire suppression has reduced amount of openings, and there has been an increase in fragmentation of riparian areas	We did not evaluate patchiness of habitats or fine-scale riparian attributes.
Olive-sided flycatcher	17.55	Decrease	Fire suppression has reduced patchiness of late- and early-seral habitat, and important pine habitat	We did not evaluate patchiness of habitats.
Lynx	14.49	Decrease	Overtrapping and negative effects of logging on prey habitat juxtaposition	Our evaluation did not include effects of trapping or patchiness of habitats.
Wolverine	14.41	Decrease	Roads and human disturbance	Our evaluation did not explicitly measure road effects or other nonvegetative factors.

Table 15—Species for which trends in source habitats differed from habitat outcomes of Lehmkuhl and others (1997) (continued)

Common name	Relative change in source habitats	Change in habitat outcome	Reasons for habitat outcome (from panel notes)	Most likely reasons for difference
Striped whipsnake	-20.59	No change on BLM/FS Eastside lands	Population has not declined on Eastside BLM and FS lands because these lands have not undergone the increase in agricultural development and dam construction as have the private lands or Upper Columbia River Basin BLM and FS lands	Basin-wide, the habitait outcome score of a negative change matches the decline in source habitat.
Sharptail snake	55.23	Decrease	Always patchy distribution, but situation has declined due to agriculture and urban development, and perhaps climate change	Our analysis did not measure changes in overall population distribution from historical that the panelists estimated.
Mojave black-collared lizard	-3.14	No change in Upper Columbia River Basin CumEff	Habitat has become more fragmented, and has declined due to agriculture, non-native vegetation, invasion of exotics, and reservoir development	Most of species range is on BLM-administered lands, which did show a decline in habitait outcome. Although there was no change in the weighted mean score, the distribution of habitat outcome scores was lower in the historical period.
White-winged crossbill	-46.41	No change	Nomadic species associated with spruce, higher elevation forests. Species not negatively affected by the increased fragmentation caused by relatively small amounts of logging of that habitat.	Unknown, though source habitats include both upper and lower montane late-seral forests, which did decline basin-wide.

native grassland and open-canopy sagebrush habitats (family 12). Widespread but less severe declines also occurred for most species dependent on old-forest habitats present in several elevation zones (family 2); for species dependent on early-seral forests (family 4); for species dependent on native herbland, shrubland, and woodland habitats (family 10); and for species dependent on native sagebrush habitats (family 11). Source habitats for all of the above-named families have become increasingly fragmented, simplified in structure, and infringed on or dominated by exotic plants.

2. Primary causes for decline in old-forest habitats (families 1 and 2) are intensive timber harvest and large-scale fire exclusion (Hann and others 1997). Additional causes for decline in low-elevation, old-forest habitats are conversion of land to agriculture and to residential or urban development (Hann and others 1997). These same causes—intensive timber harvest and large-scale fire exclusion—also are primarily responsible for the large decline in early-seral habitats (family 4).

3. Primary causes for decline in native herbland, woodland, grassland, and sagebrush habitats (families 10, 11, and 12) are excessive livestock grazing, invasion of exotic plants, and conversion of land to agriculture and residential and urban development (Hann and others 1997). Altered fire regimes also are responsible for decline in native grassland and shrubland habitats.

4. Various road-associated factors negatively affect habitats or populations of most species analyzed here. Effects of road-associated factors can be direct, such as habitat loss and fragmentation because of road construction and maintenance. Effects also can be indirect, such as displacement or increased mortality of populations in areas near roads in relation to motorized traffic and associated human activities. Because of the high density of roads present across large areas of the basin, effects from road-associated factors must be considered additive to that of habitat loss. Moreover, it is likely that many habitats are underused by several species because of the effects of roads and associated factors; this may be especially true for species of carnivorous mammals, particularly gray wolf and grizzly bear.

5. Implications of our results for managing old-forest structural stages include the potential to conserve old-forest habitats in subbasins and watersheds where decline has been strongest; manipulate mid-seral forests to accelerate development of late-seral stages where such manipulations can be done without further reduction in early- or late-seral forests; and restore fire and other disturbance regimes in all forested structural stages to hasten development and improvement in the amount, quality, and distribution of old-forest stages. Many of the practices designed to restore old-forest habitats also can be designed to restore early-seral habitats. For example, long-term restoration of more natural fire regimes will hasten development of both early- and late-seral structural conditions, and minimize area of mid-seral habitats, which few if any species depend on as source habitat.

6. Implications of our results for managing rangelands include the potential to conserve native grasslands and shrublands that have not undergone large-scale reduction in composition of native plants; control or eradicate exotic plants on native grasslands and shrublands where invasion potential or spread of exotics is highest; and restore native plant communities, by using intensive range practices, where potential for restoration is highest. Restoration includes the potential to manipulate livestock grazing systems and stocking rates where existing or past grazing practices have contributed to the decline in native grasslands and shrublands.

7. Implications of our summary of road-associated effects include the potential to mitigate a diverse set of negative factors associated with roads. Comprehensive mitigation of road-associated factors would require a substantial reduction in the density of existing roads as well as effective control of road access in relation to management of livestock, timber, recreation, hunting, trapping, mineral development, and other human activities. Efforts to restore habitats without simultaneous efforts to reduce road density and control human disturbances will curtail the effectiveness of habitat restoration, or even contribute to its failure; this is because the large number of species that are simultaneously affected by decline in habitat as well as by road-associated factors.

8. Implications of all our results, when considered at multiple spatial scales ranging from the basin, ERU, subbasin, and watershed, provide spatially explicit opportunities for conservation and restoration of source habitats across various land ownerships and jurisdictions. Moreover, our results provide temporally explicit opportunities for design of long-term efforts to restore source habitats that have undergone strong, widespread decline, with simultaneous design of efforts to conserve these same habitats where they exist currently. Use of our findings to conduct effective spatial and temporal prioritization of restoration and conservation efforts for terrestrial species and habitats represents a major opportunity for resources managers in the basin.

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Abbreviations

Centimeter	(cm)
Hectare	(ha)
Inch	(in)
Kilometer	(km)
Meter	(m)
Mile	(mi)
Year	(yr)

References

Anon. 1996. Sierra Nevada Ecosystem Project. Davis, CA: University of California-Davis; final report to Congress; Centers for Water and Wildland Resources report 38. [Irregular pagination].

Alverson, William S.; Kuhlmann, Walter; Waller, Donald M. 1994. Wild forests: conservation biology and public policy. Washington, DC: Island Press. 300 p.

Autenrieth, R. 1978. Guidelines for the management of pronghorn antelope. In: Proceedings of the 8th biennial pronghorn antelope workshop; 1978 May 2-4; Jasper, AB. Jasper, AB: Alberta Recreation, Parks and Wildlife, Fish and Wildlife Division: 472-525.

Bailey, T.N.; Bangs, E.E.; Portner, M.F. [and others]. 1986. An apparent overexploited lynx population on the Kenai Peninsula, Alaska. Journal of Wildlife Management. 50: 279-290.

Banci, Vivian. 1994. Wolverine. In: Ruggiero, Leonard F.; Aubry, Keith B.; Buskirk, Steven W. [and others], tech. eds. 1994. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the Western United States. Gen. Tech. Rep. RM-254. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 7-37. Chapter 2.

Bangs, Edward E.; Fritts, Steven H. 1996. Reintroducing the gray wolf to central Idaho and Yellowstone National Park. Wildlife Society Bulletin. 24(3): 402-413.

Barclay, R.M.R. 1991. Population structure of temperate zone insectivorous bats in relation to foraging behavior and energy demand. Journal of Animal Ecology. 60: 165-178.

Bashore, T.L.; Tzilkowski, W.M.; Bellis, E.D. 1985. Analysis of deer-vehicle collision sites in Pennsylvania. Journal of Wildlife Management. 49: 769-774.

Bechard, M.J.; Schmutz, J.K. 1995. Ferruginous hawk (*Buteo regalis*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 172. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union. 20 p.

Beck, Jonathan M.; Peterson, Charles R. 1995. Movements and habitat selection of the longnose snake (*Rhinocheilus lecontei*) in southwestern Idaho. Tech. Bull. 95-18. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office. 30 p.

Belnap, J. 1995. Surface disturbances: their role in accelerating desertification. Environmental Monitoring and Assessment. 37: 39-57.

Belnap, Jayne. 1993. Recovery rates of cryptobiotic crusts: inoculant use and assessment methods. Great Basin Naturalist. 53: 89-95.

Belnap, Jayne; Gardner, John S. 1993. Soil microstructure in soils of the Colorado Plateau: the role of the cyanobacterium *Microcoleus vaginatus*. Great Basin Naturalist. 53: 40-47.

Bennett, A.F. 1991. Roads, roadsides and wildlife conservation: a review. In: Saunders, D.A.; Hobbs, R.J., eds. Nature conservation 2: the role of corridors. Victoria, Australia: Surrey Beatty and Sons: 99-118.

Blumton, A.K. 1989. Factors affecting loggerhead shrike mortality in Virginia. Blacksburg, VA: Virginia Polytechnical Institute and State University. 85 p. M.S. thesis.

Boarman, William I.; Sazaki, Marc. 1996. Highway mortality in desert tortoises and small vertebrates: success of barrier fences and culverts. In: Evink, G.; Garrett, P.; Berry, J., eds. Proceedings, transportation and wildlife: reducing wildlife mortality and improving wildlife passageways across transportation corridors; 1996, April 3-May 2; Orlando, FL: Department of Transportation, Federal Highway Administration: 169-173.

Bock, C.E.; Webb, B. 1984. Birds as grazing indicator species in southeastern Arizona. Journal of Wildlife Management. 48(3): 1045-1049.

Bock, Carl E.; Lepthien, L.W. 1976. Synchronous eruptions of boreal seed-eating birds. American Naturalist. 110: 559-579.

Brennan, L.A. 1990. What happened to the mountain quail in Idaho? Quail Unlimited. 9: 42-43, 69.

Brennan, Leonard A. 1989. Report on a survey of mountain quail in eastern Oregon, southeastern Washington, western Idaho, and northern Nevada. Boise, ID: The Chukar Foundation. 58 p. 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.

Brennan, L.A.; Block, W.M.; Gutierrez, R.J. 1987. Habitat use by mountain quail in northern California. Condor. 89: 66-74.

Brown, Herbert A.; Bury, R. Bruce; Darda, David M. [and others]. 1995. Reptiles of Washington and Oregon. Seattle, WA: Seattle Audubon Society. 176 p.

Bruns, E.H. 1977. Winter behavior of pronghorns in relation to habitat. Journal of Wildlife Management. 41(3): 560-571.

Busack, Stephen D.; Bury, R. Bruce. 1974. Some effects of off-road vehicles and sheep grazing on lizard populations in the Mojave Desert. Biological Conservation. 6: 179-183.

Carbyn, L.N.; Patriquin, D. 1983. Observations on home range sizes, movements and social organization of lynx (*Lynx canadensis*) in Riding Mountain National Park, Manitoba. Canadian Field-Naturalist. 97: 262-267.

Cassirer, E. Frances; Oldenburg, Lloyd E.; Coggins, Victor L. [and others]. 1996. Overview and preliminary analysis of a bighorn sheep dieoff, Hells Canyon 1995-96. In: Proceedings of the 10th biennial symposium. Northern wild sheep and goat council; 1996 April 29-May 3; Silverthorne, CO. Thermopolis, WY: Northern Wild Sheep and Goat Council. 10: 78-86.

Chadwick, Douglas H. 1972. Mountain goat ecology—logging relationships in the Bunker Creek drainage of western Montana. State of Montana; Project No. W-120-R3, No. 91.1; big game research, final job report. 228 p. On file with: U.S. Department of Agriculture; Agricultural Research Service, National Agricultural Library, 10301 Baltimore Blvd., Beltsville, MD 20705-2351. Record no. 20722.

Christensen, A.G.; Lyon, L.J.; Lonner, T.N., comps. 1991. Proceedings of a symposium on elk vulnerability; 1991 April 10-12; Bozeman, MT. Bozeman, MT: Montana Department of Fish, Wildlife, and Parks, Montana State University. 330 p.

Clark, Richard J. 1975. A field study of the short-eared owl, *Asio flammeus* (Pontoppidan), in North America. Wildlife Monographs. 47: 1-67.

Coggins, V.L. 1988. The Lostine Mountain bighorn sheep die-off and domestic sheep. Proceedings of the biennial symposium of the Northern Wild Sheep and Goat Council. 6: 66-76.

Cole, E.K.; Pope, M.D.; Anthony, R.G. 1997. Effects of road management on movement and survival of Roosevelt elk. Journal of Wildlife Management. 61: 1115-1126.

Conroy, Michael J.; Cohen, Yosef; Frances, James C. [and others]. 1995. Parameter estimation, reliability, and model improvement for spatially explicit models of animal populations. Ecological Applications. 5(1): 17-19.

Conroy, Michael J.; Noon, Barry R. 1996. Mapping of species richness for conservation of biological diversity: conceptual and methodological issues. Ecological Applications. 6(3): 763-773.

Copeland, Jeffrey P. 1996. Biology of the wolverine in central Idaho. Moscow, ID: University of Idaho. 138 p. M.S. thesis.

Coulter, M.W. 1966. Ecology and management of fishers in Maine. Syracuse, NY: Syracuse University, College of Forestry. 183 p. Ph.D. dissertation.

Craig, R.B. 1978. An analysis of the predatory behavior of the loggerhead shrike. Auk. 95(2): 221-234.

Daubenmire, R. 1970. Steppe vegetation of Washington. Bull. 62. Pullman, WA: State of Washington, Agricultural Experiment Station. 131 p.

DeStefano, Stephen. 1990. Investigation of the status of kit foxes in southeastern Oregon. Tech. Rep. 90-5-01. Portland, OR: Oregon Department of Fish and Wildlife, Wildlife Diversity Program. 60 p.

Dood, Arnold R.; Brannon, Robert D.; Mace, Richard D. 1986. Final programmatic environmental impact statement: the grizzly bear northwestern Montana. Helena, MT: Montana Department of Fish, Wildlife, and Parks. 287 p.

Dood, Arnold R.; Brannon, Robert D.; Mace, Richard D. 1985. Management of grizzly bears in the northern continental divide ecosystems, Montana. Transactions of the 51st North American Wildlife and Natural Resources Conference. 51: 162-177.

Doyle, F.I.; Smith, J.M.N. 1994. Population responses of northern goshawks to the 10-year cycle in numbers of snowshoe hares. Studies in Avian Biology. 16: 122-129.

Easter-Pilcher, Andrea. 1996. Implementing the Endangered Species Act: assessing the listing of species as endangered or threatened. BioScience. 46(5): 355-363.

Ehmann, H.; Coggar, H. 1985. Australia’s endangered herpetofauna: a review of criteria and politics. In: Green, G.; Shine, R.; Ehmann, H., eds. Biology of Australian frogs and reptiles. Sydney, Australia: Surrey Beatty and Sons and Royal Zoological Society of New South Wales: 435-447.

Ehrlich, Paul R.; Dobkin, David S.; Wheye, Darryl. 1988. The birder's handbook. New York: Simon and Schuster. 785 p.

Eidenshink, J. 1992. The conterminous U.S. AVHRR data set. Photogrammetric Engineering and Remote Sensing. 58(6): 809-813.

Flather, C.H.; Joyce, L.A.; Bloomgarden, C.A. 1994. Species endangerment patterns in the United States. Gen. Tech. Rep. RM-GTR-241. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 42 p.

Flather, Curtis H.; Knowles, Michael S.; Kendall, Iris A. 1998. Threatened and endangered species geography: characteristics of species hotspots in the conterminous United States. BioScience. 48(5): 365-376.

Flickinger, E.L. 1995. Loggerhead fatalities on a highway in Texas. In: Yosef, R.; Lohrer, F.E., eds. Shrikes (Laniidae) of the world: biology and conservation: 1st international shrike symposium; 1993 January 11-15; Lake Placid, FL. In: Proceedings of the Western Foundation of Vertebrate Zoology; 6(1): 67-69.

Foreyt, William J. 1994. Effects of controlled contact exposure between healthy bighorn sheep and llamas, domestic goats, mountain goats, cattle, domestic sheep, or mouflon sheep. In: Proceeding of the 9th biennial symposium. Northern Wild Sheep and Goat Council; 1994 May 2-6; Cranbrook, BC. Thermopolis, WY: Northern Wild Sheep and Goat Council. 9: 7-14.

Foreyt, William J.; Jessup, David A. 1982. Fatal pneumonia of bighorn sheep following association with domestic sheep. Journal of Wildlife Diseases. 18: 163-168.

Forman, Richard T.T. 1995. Land mosaics: the ecology of landscapes and regions. Cambridge, England: Cambridge University Press. 325 p.

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

Forman, Richard T.T.; Hersperger, Anna M. 1996. Road ecology and road density in different landscapes, with international planning and mitigation solutions. In: Evink, G.; Garrett, P.; Berry, J., eds. Transportation and wildlife: reducing wildlife mortality and improving wildlife passageways across transportation corridors. Proceedings of a symposium; 1996 April 3-May 2; Orlando, FL: Department of Transportation, Federal Highway Administration: 1-23.

Forman, Richard T.T.; Friedman, Debras S.; Fitzhenry, David [and others]. 1997. Ecological effects of roads: toward summary indices and an overview of North America. In: Canters, Kees, ed. Proceedings, international conference on habitat fragmentation, infrastructure, and the role of ecological engineering: 1995 September 12-17; Maastricht and The Hague, The Netherlands. [The Netherlands]: Ministry of Transport, Public Works, and Water Management: 40-54.

Fortin, C.; Cantin, M. 1994. The effects of trapping on a newly exploited American marten population. 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: 179-191.

Fraser, D. 1979. Sightings of moose, deer, and bears on roads in northern Ontario. Wildlife Society Bulletin. 7: 181-184.

Frederick, Glenn P. 1991. Effects of forest roads on grizzly bears, elk, and gray wolves: a literature review. Publ. R1-91-73. Libby, MT: Kootenai National Forest. 53 p.

Fritts, Steven H.; Paul, William J.; Mech, L. David. 1985. Can relocated wolves survive? Wildlife Society Bulletin. 13: 459-463.

Gibilisco, Chuck; Heinemeyer, K.S.; Jones, Jeff; Mack, Curt. 1995. Conservation strategy for fisher *(Martes pennanti*) and marten (*Martes americana*) in Idaho. In: Idaho Department of Fish and Game, Nez Perce Tribe, and Sawtooth National Forest. Forest carnivores in Idaho. Habitat Idaho State Conservation Effort; Idaho Department of Fish and Game, Idaho Department of Parks and Recreation; U.S. Department of the Interior, Bureau of Land Management; U.S. Department of Agriculture, Forest Service, Northern and Intermountain Regions; U.S. Department of the Interior, Fish and Wildlife Service: 43-70. Draft report dated April 21, 1995. 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, USDA Forest Service, 304 N. 8th Street, Boise, ID 83702.

Giesen, K.M.; Connelly, J.W. 1993. Guidelines for management of Columbian sharp-tailed grouse habitats. Wildlife Society Bulletin. 21(3): 325-333.

Geist, Valerius. 1971. Mountain sheep: a study in behavior and evolution. Chicago: The University of Chicago Press. 383 p.

Gilmer, D.S.; Evans, D.L.; Konrad, P.M.; Stewart, R.E. 1985. Recoveries of ferruginous hawks banded in south-central North Dakota. Journal of Field Ornithology. 56: 184-187.

Gravenmier, Rebecca A.; Wilson, Andrew; Steffenson, John. 1997. Information system development and documentation. In: Quigley, Thomas M.; Arbelbide, S.J., tech. eds. An assessment of ecosystem components in the interior Columbia basin including portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 2012-2066. Vol. 4. Chapter 8. (Quigley, Thomas M., tech. ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Green, G.A.; Anthony, R.G. 1989. Nesting success and habitat relationships of burrowing owls in the Columbia basin, Oregon. Condor. 91(2): 347-354.

Griffin, Daude N. 1959. The poisoning of meadowlarks with insecticides. Wilson Bulletin. 71(2): 193.

Hamilton, K.S.; Holl, S.K.; Douglas, C.L. 1982. An evaluation of the effects of recreational activity on bighorn sheep in the San Gabriel Mountains, California. Transactions of the Annual Meeting of the Desert Bighorn Council. 26: 50-55.

Hann, Wendel J.; Jones, Jeffrey L.; Karl, Michael G. [and others]. 1997. Landscape dynamics of the basin. In: Quigley, Thomas M.; Arbelbide, S.J., tech. eds. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 337-1055. Vol. 2. Chapter 3. (Quigley, Thomas M., tech. ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Hann, Wendel J.; Jones, Jeffrey L.; Keane, Robert E. [and others]. 1998. Landscape dynamics. Journal of Forestry. 96(10): 10-15.

Hansen, A.J.; Garmon, S.L.; Marks, B. 1993. An approach for managing vertebrate diversity across multiple-use landscapes. Ecological Applications. 3: 481-496.

Hardy, Colin C.; Burgan, Robert E.; Ottmar, Roger D.; Deeming, John E. 1996. A database for spatial assessments of fire characteristics, fuel profiles, and PM10 emissions. In: Keane, Robert E.; Jones, Jeffrey L.; Riley, Laurienne S.; Hann, Wendel J., tech. eds. Compilation of administrative reports: multi-scale landscape dynamics in the basin and portions of the Klamath and Great basins. [Irregular pagination]. 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.

Hargis, C.D. 1996. The influence of forest fragmentation and landscape pattern on American martens and their prey. Logan, UT: Utah State University. 154 p. Ph.D. dissertation.

Hargis, Christina D.; McCullough, Dale R. 1984. Winter diet and habitat selection of marten in Yosemite National Park. Journal of Wildlife Management. 48(1): 140-146.

Harmata, A.R. 1981. Recoveries of ferruginous hawks banded in Colorado. North American Bird Bander 6:144-147.

Harper, Kimball T.; Pendleton, Rosemary L. 1993. Cyanobacteria and cyanolichens: can they enhance availability of essential minerals for higher plants? Great Basin Naturalist. 53: 59-72.

Haynes, Richard W.; Graham, Russell T.; Quigley, Thomas M. 1998. A framework for ecosystem management in the interior Columbia basin. Journal of Forestry. 96(10): 4-9.

Hayward, G.D.; Verner, J., eds. 1994. Flammulated, boreal, and great gray owls in the United States: a technical conservation assessment. Gen. Tech. Rep. RM-253. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 213 p.

Helms, B. 1978. Antelope and energy development. In: Proceedings of the 8th biennial pronghorn antelope workshop; 1978 May 2-4; Jasper, AB. Jasper, AB: Alberta Recreation, Parks and Wildlife, Fish and Wildlife Division: 206-215.

Hessburg, Paul F.; Smith, Bradley G.; Kreiter, Scott D. [and others]. 1999. Historical and current forest and range landscapes in the interior Columbia River basin and portions of the Klamath and Great Basins. Part I: 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. (Quigley, Thomas M., ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Hicks, Lorin L.; Elder, James M. 1979. Human disturbance of Sierra Nevada bighorn sheep. Journal of Wildlife Management. 43(4): 909-915.

Hill, J.D.; Smith, J.D. 1984. Bats: a natural history. Austin, TX: University of Texas Press. 243 p.

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(4): 593-600.

Holt, Denver W. 1997. The long-eared owl (*Asio otus*) and forest management: a review of the literature. Journal of Raptor Research. 32: 175-186.

Holt, D.W. 1992. Notes on short-eared owl (*Asio flammeus*) nest sites, reproduction and territory sizes in coastal Massachusetts. Canadian Field-Naturalist. 106: 352-356.

Hornocker, M.G.; Hash, H.S. 1981. Ecology of the wolverine in northwestern Montana. Canadian Journal of Zoology. 59(7): 1286-1301.

Howard, R.P. 1975. Breeding ecology of the ferruginous hawk in northern Utah and southern Idaho. Logan, UT: Utah State University. [Page numbers unknown]. M.A. thesis.

Humphrey, S.R. 1975. Nursery roosts and community diversity of nearctic bats. Journal of Mammalogy. 56(2): 321-346.

Hunter, M.L., Jr. 1991. Coping with ignorance: the coarse-filter strategy for maintaining biodiversity. In: Kohm, K.A., ed. Balancing on the brink of extinction. Covelo, CA: Island Press: 266-281.

Hunter, Malcolm L. 1990. Wildlife, forests, and forestry: principles of managing forests. Englewood Cliffs, NJ: Prentice-Hall. 370 p.

Hutto, R.L. 1995. Composition of bird communities following stand-replacement fires in northern Rocky Mountain conifer forests. Conservation Biology. 9(5): 1041-1058.

Idaho State Conservation Effort. 1995. Habitat conservation assessment and conservation strategy for the Townsend’s big-eared bat. Boise, ID. 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.

Ingles, L.G. 1965. Mammals of the Pacific States. Stanford, CA: Stanford University Press. 506 p.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996a. Expert panel notes for amphibians and reptiles terrestrial habitat assessment, fall 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996b. Expert panel notes for carnivore terrestrial habitat assessment, fall 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996c. Expert panel notes for carnivore viability assessment, spring 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996d. Expert panel notes for raptor and upland game bird viability assessment, spring 1996, for the interior Columbia basin. Unpublished notes. 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.

Janes, S.W. 1983. Status, distribution, and habitat selection of the grasshopper sparrow in Morrow County, Oregon. Murrelet. 64: 51-54.

Jensen, Mark; Goodman, Iris; Brewer, Ken [and others]. 1997. Biophysical environments of the basin. In: Quigley, Thomas M.; Arbelbide, Sylvia J., tech. eds. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins: volume 1. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 99-307. (Quigley, Thomas M., tech. ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Johansen, Jeffrey R.; Ashley, John; Rayburn, William R. 1993. Effects of rangefire on soil algal crusts in semiarid shrub-steppe of the lower Columbia basin and their subsequent recovery. Great Basin Naturalist. 53: 73-88.

Johnson, Bruce K.; Kern, John W.; Wisdom, Michael J. [and others]. 2000. Resource selection and spatial partitioning of mule deer and elk during spring. Journal of Wildlife Management. 64(3): 684-697.

Johnson, Rolf L. 1983. Mountain goats and mountain sheep of Washington. Biology Bulletin No. 18. Olympia, WA: Washington State Game Department. 196 p.

Jones, Jeffrey L. 1991. Habitat use of fisher in north central Idaho. Moscow, ID: University of Idaho. 147 p. M.S. thesis.

Joslin, G. 1986. Mountain goat population changes in relation to energy exploration along Montana’s Rocky Mountain front. Biennial Symposium Northern Wild Sheep and Goat Council. 5: 253-271.

Kaltenecker, J.H. 1997. The recovery of microbiotic crusts following post-fire rehabilitation on rangelands of the western Snake River plain. Boise, ID: Boise State University. 99 p. M.S. thesis.

Kaltenecker, J.; Wicklow-Howard, M. 1994. Microbiotic soil crusts in sagebrush habitats of southern Idaho. 48 p. (plus an appendix). Prepared for Eastside Ecosystem Management Project. 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.

Kasworm, Wayne F.; Manley, T.L. 1990. Road and trail influences on grizzly bears and black bears in northwest Montana. International Conference on Bear Research and Management. 8: 79-84.

Keane, Robert E.; Long, Donald G.; Menakis, James P. [and others]. 1996. Simulating coarse-scale vegetation dynamics using the Columbia River basin succession model-CRBSUM. Gen. Tech. Rep. INT-GTR-340. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 50 p.

Keister, George P.; Immell, David. 1994. Continued investigations of kit fox in southeastern Oregon and evaluation of status. Tech. Rep. 94-5-01. Portland, OR: Oregon Department of Fish and Wildlife, Wildlife Diversity Program. 33 p.

Kiester, A. Ross; Scott, Michael J.; Csuti, Blair [and others]. 1996. Conservation prioritization using GAP data. Conservation Biology. 10(5): 1332-1342.

Knapton, Richard W. 1994. Clay-colored sparrow (*Spizella pallida*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 120. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists’ Union. 16 p.

Knick, S.T.; Rotenberry, J.T. 1995. Landscape characteristics of fragmented shrubsteppe habitats and breeding passerine birds. Conservation Biology. 9(5): 1059-1071.

Knick, Steven T.; Kasworm, Wayne. 1980. Shooting mortality in small populations of grizzly bears. Wildlife Society Bulletin. 17(1): 11-15.

Knight, R.R.; Blanchard, B.M.; Eberhardt, L.L. 1988. Mortality patterns and population sinks for Yellowstone grizzly bears, 1973-1985. Wildlife Society Bulletin. 16: 121-125.

Koehler, G.M.; Brittell, J.D. 1990. Managing spruce fir habitat for lynx and snowshoe hares. Journal of Forestry. 88: 10-14.

Koehler, Gary M.; Hornocker, Maurice G. 1977. Fire effects on marten habitat in the Selway-Bitterroot Wilderness. Journal of Wildlife Management. 41(3): 500-505.

Koehler, Gary M.; Aubry, Keith B. 1994. Lynx. In: Ruggiero, L.; Aubry, K.; Buskirk, S. [and others], tech. eds. 1994. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the Western United States. Gen. Tech. Rep. RM-254. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 74-98. Chapter 4.

Koplin, James R. 1969. The numerical response of woodpeckers to insect prey in a subalpine forest in Colorado. Condor. 71(4): 436-438.

Krebs, Charles J. 1989. Ecological methodology. New York: Harper-Collins. 654 p.

Landres, Peter B.; Verner, Jared; Thomas, Jack Ward. 1988. Ecological uses of vertebrate indicator species: a critique. Conservation Biology. 2(4): 316-328.

Lanyon, W.E. 1994. Western meadowlark (*Sturnella neglecta*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 104. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union. 20 p.

Lee, Danny C.; Sedell, James R.; Rieman, Bruce E. [and others]. 1997. Broad-scale assessment of aquatic species and habitats. In: Quigley, Thomas M.; Arbelbide, Sylvia J., tech. eds. An assessment of ecosystem components in the interior Columbia Basin and portions of the Klamath and Great basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 1057-1496. Vol. 3. (Quigley, Thomas M., tech. ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Lee, Danny C.; Sedell, James R.; Rieman, Bruce E. [and others]. 1998. Aquatic species and habitats. Journal of Forestry. 96(10): 16-21.

Lehmkuhl, John F.; Raphael, Martin G.; Holthausen, Richard S. [and others]. 1997. Historical and current status of terrestrial species and the effects of proposed alternatives. In: Quigley, Thomas M.; Lee, Kristine M.; Arbelbide, Sylvia J., tech. 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. Vol. 2.

Lewis, Lyle; Wenger, C. Richard. 1998. Idaho's Canada lynx: pieces of the puzzle. Tech. Bull. 98-11. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office. 19 p.

Linden, H.; Wikman, M. 1983. Goshawk predation on tetraonids: availability of prey and diet of the predator in the breeding season. Journal of Animal Ecology. 52: 953-968.

Lokemoen, J.T.; Duebbert, H.F. 1976. Ferruginous hawk nesting ecology and raptor populations in northern South Dakota. Condor. 78(4): 464-470.

Losensky, B. John. 1995. A description and photographic record of the historical vegetation of the Columbia River basin. Walla Walla, WA: Interior Columbia Basin Ecosystem Management Project; completion report; contract INT-94951-RJVA. [Irregular pagination]. On file with: U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Forestry Science Laboratory, 800 Block E. Beckwith, P.O. Box 8089, Missoula, MT 59807.

Losensky, B. John. 1994. Historical vegetation types of the interior Columbia River basin. Walla Walla, WA: Interior Columbia Basin Ecosystem Management Project; completion report; contract INT-94951-RJVA. [Irregular pagination]. On file with: U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Forestry Science Laboratory, 800 Block E. Beckwith, P.O. Box 8089, Missoula, MT 59807.

Loveland, T.R.; Merchant, J.M.; Ohlen, D.O.; Brown, J.F. 1991. Development of a land-cover characteristics database for the conterminous U.S. Photogrammetric Engineering and Remote Sensing. 57(11): 1453-1463.

Luckenbach, Roger A. 1978. An analysis of off-road vehicle use on desert avifaunas. Transactions of the North American Wildlife and Natural Resources Conference. 43: 157-162.

Ludwig, John A.; Reynolds, James F. 1988. Statistical ecology. New York: John Wiley and Sons. 337 p.

Lyon, L. Jack. 1983. Road density models describing habitat effectiveness for elk. Journal of Forestry 81: 592-595.

MacArthur, R.A.; Geist, V.; Johnston, R.H. 1982. Cardiac and behavioral responses of mountain sheep to human disturbance. Journal of Wildlife Management. 46: 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.

MacCracken, J.G.; Alexander, L.E.; Uresk, D.W. 1983. An important lichen of southeastern Montana rangelands. Journal of Range Management. 36: 35-37.

Mace, Richard D.; Waller, John S.; Manley, Timothy L. [and others]. 1996. Relationships among grizzly bears, roads and habitat in the Swan Mountains, Montana. Journal of Applied Ecology. 33: 1395-1404.

Mack, R.N.; Thompson, J.N. 1982. Evolution in steppe with few large, hooved mammals. American Naturalist. 119: 757-773.

Mader, H.J. 1984. Animal habitat isolation by roads and agricultural fields. Biological Conservation. 29: 81-96.

Magoun, Audrey J.; Copeland, Jeffrey P. 1998. Characteristics of wolverine reproductive den sites. Journal of Wildlife Management. 62: 1313-1320.

Maj, M.; Garton, E.O. 1994. Fisher, lynx, and wolverine summary of distribution information. In: Ruggiero, L.; Aubry, K.; Buskirk, S. [and others], tech eds. 1994. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the Western United States. Gen. Tech. Rep. RM-254. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 169-175. (appendix B).

Mannan, R.W.; Morrison, Michael L.; Meslow, E. Charles. 1984. Comment: the use of guilds in forest bird management. Wildlife Society Bulletin. 12: 426-430.

Manning, R.W.; Knox-Jones, J. 1989. *Myotis evotis*. Mammalian Species. No. 329. Baltimore, MD: American Society of Mammalogists. 5 p.

Marcot, Bruce G.; Castellano, Michael A.; Christy, John A. [and others]. 1997. Terrestrial ecology of the basin. In: Quigley, Thomas M.; Arbelbide, S.J., tech. eds. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 1497-1713. Vol. 3. (Quigley, Thomas M., tech. ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Marcot, Bruce G.; Wales, Barbara C.; Murphy, Wally. [In prep.]. Range maps for terrestrial vertebrates in the interior Columbia basin. On file with: Bruce Marcot, U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 1221 S.W. Yamhill Street, Suite 200, Portland, OR 97208.

Marcot, Bruce G.; Wisdom, Michael J.; Li, Hiram W.; Castillo, Gonzalo C. 1994. Managing for featured, threatened, endangered, and sensitive species and unique habitats for ecosystem sustainability. Gen. Tech. Rep. PNW-GTR-329. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 39 p.

Marks, J.S.; Saab Marks, V. 1987. Influence of radio collars on survival of sharp-tailed grouse. Journal of Wildlife Management. 51(2): 468-471.

Marti, C.D.; Marks, J.S. 1989. Medium-sized owls. Proceedings of the western raptor management symposium and workshop; 1987 October 26-28; Boise, ID. National Wildlife Federation Science and Technical Series. 12: 124-133.

Martin, Kevin D.; Schommer, Tim; Coggins, Victor. 1996. Literature review regarding the compatibility between bighorn and domestic sheep. Biennial Symposium of the Northern Wild Sheep and Goat Council. 10: 72-77.

Maser, C.; Hammer, E.W.; Brown, C. [and others]. 1974. The sage vole, *Lagurus curtatus*, in the Crooked River National Grassland, Jefferson County, Oregon: a contribution to its life history and ecology. Saegetierkundliche Mitteilungen. 22: 193-222.

Maser, C.; Strickland, G.S. 1978. The sage vole, *Lagurus curtatus*, as an inhabitant of subalpine sheep fescue *Festuca ovina* communities on Steens Mountain: an observation and interpretation. Northwest Science. 52: 276-284.

Matthews, Patrick E.; Coggins, Victor L. 1994. Status and history of mountain goats in Oregon. Biennial Symposium of the Northern Wild Sheep and Goat Council. 9: 69-74.

Mattson, David J.; Blanchard, Bonnie M.; Knight, Richard R. 1992. Yellowstone grizzly bear mortality, human habituation, and whitebark pine seed crops. Journal of Wildlife Management. 56(3): 432-442.

Mattson, David J.; Herrero, Stephen; Wright, Gerald; Pease, Craig M. 1996a. Designing and managing protected areas for grizzly bears: how much is enough? In: Wright, R.G., ed. National parks and protected areas: their role in environmental protection. Cambridge, MA: Blackwell Science: 133-164.

Mattson, David J.; Herrero, Stephen; Wright, Gerald; Pease, Craig M. 1996b. Science and management of Rocky Mountain grizzly bears. Conservation Biology. 10(4): 1013-1025.

Mattson, David J.; Knight, Richard R.; Blanchard, Bonnie M. 1987. The effects of developments and primary roads on grizzly bear habitat use in Yellowstone National Park, Wyoming. International Conference on Bear Research and Management. 7: 259-273.

Mattson, D.J.; Wright, G.R.; Kendall, K.C.; Martinka, C.J. 1995. Grizzly bears. In: LaRoe, E.T.; Farris, G.S.; Puckett, C.E. [and others], eds. Our living resources. Washington, DC: U.S. Department of the Interior, National Biological Service: 103-105.

McKelvey, Kevin S.; Aubry, Keith B.; Ortega, Yvette K. 1999. History and distribution of lynx in the contiguous United States. In: Ruggiero, Leonard F.; Aubry, Keith B.; Buskirk, Steven W. [and others], eds. Ecology and conservation of lynx in the United States. Niwot, CO: University Press of Colorado, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 207-264. Chapter 8.

McLellan, B.N.; Shackleton, D.M. 1988. Grizzly bears and resource-extraction industries: effects of roads on behaviour, habitat use, and demography. Journal of Applied Ecology. 25: 451-460.

Mech, L.D. 1970. Implications of wolf ecology to management. In: Jorgensen, S.E.; Faulkner, L.E.; Mech, L.D., eds. Proceedings of a symposium on wolf management in selected areas of North America. [Meeting dates unknown]; [Meeting location unknown]. [Place of publication unknown]: U.S. Department of the Interior, Fish and Wildlife Service: 39-44.

Mech, L.D. 1973. Wolf numbers in the Superior National Forest of Minnesota. Res. Pap. NC-97. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Research Experiment Station. 10 p.

Mech, L.D. 1980. Age, sex, reproduction, and spatial organization of lynxes colonizing northeastern Minnesota. Journal of Mammology. 61: 261-267.

Mech, L.D.; Fritts, S.H.; Radde, G.L.; Paul, W.J. 1988. Wolf distribution and road density in Minnesota. Wildlife Society Bulletin. 16: 85-87.

Meints, D.R.; Connelly, J.W.; Reese, K.P. [and others]. 1992. Habitat suitability index procedure for Columbian sharp-tailed grouse. Idaho Forest Wildlife and Range Experiment Station Bulletin. Moscow, ID: University of Idaho, College of Forestry and Range Sciences. (vi): 1-27.

Menakis, James P.; Long, Donald G.; Keane, Robert E. 1996. The development of key broad-scale layers and characterization files. In: Keane, Robert E.; Jones, Jeffrey L.; Riley, Laurienne S.; Hann, Wendel J., tech. eds. Compilation of administrative reports: multi-scale landscape dynamics in the basin and portions of the Klamath and Great basins. [Irregular pagination]. 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.

Merriam, G.; Kozakiewicz, M.; Tsuchiya, E.; Hawley, K. 1988. Barriers as boundaries for metapopulations and demes of *Peromyscus leucopus* in farm landscapes. Landscape Ecology. 2: 227-235.

Merrill, Troy; Mattson, David J.; Wright, R. Gerald; Quigley, Howard B. 1999. Defining landscapes suitable for restoration of grizzly bears *Ursus arctos* in Idaho. Biological Conservation. 87(2): 231-248.

Miller, J.R.; Joyce, L.A.; Knight, R.L.; King, R.M. 1996. Forest roads and landscape structure in the southern Rocky Mountains. Landscape Ecology. 11: 115-127.

Mladenoff, David J.; Sickley, Theodore A.; Haight, Robert G.; Wydeven, Adrian P. 1995. A regional landscape analysis and prediction of favorable gray wolf habitat in the northern Great Lakes region. Conservation Biology. 9(2): 279-294.

Moroz, Paul. 1995. Idaho ground squirrel, *Spermophilus brunneus brunneus*, habitat conservation assessment (HCA), conservation strategy (CS). Idaho Interagency Conservation/Prelisting Effort. Idaho Department of Fish and Game; Idaho Department of Parks and Recreation; U.S. Department of the Interior, Fish and Wildlife Service and Bureau of Land Management; U.S. Department of Agriculture, Forest Service. 26 p. 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.

Morrison, M.L.; Marcot, B.G.; Mannan, R.W. 1992. Wildlife-habitat relationships: concepts and applications. Madison, WI: University of Wisconsin Press. 343 p.

Murray, T.B. 1938. Upland game birds in Idaho and their future. University of Idaho Bulletin 33: 55-60.

Nagorsen, D.W.; Brigham, R.M. 1993. Bats of British Columbia. Vancouver, BC: UBC Press. 164 p.

Nellis, Carl H.; Wetmore, Stephen P.; Keith, Lloyd B. 1972. Lynx-prey interactions in central Alberta. Journal of Wildlife Management. 36(2): 320-329.

Noss, R.F. 1987. From plant communities to landscapes in conservation inventories: a look at The Nature Conservancy (USA). Biological Conservation. 41: 11-37.

Noss, R.F.; LaRoe, E.T., III; Scott, J.M. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Biological report 28. Washington, DC: National Biological Service.

Noss, Reed F.; Quigley, Howard B.; Hornocker, Maurice G. [and others]. 1996. Conservation biology and carnivore conservation in the Rocky Mountains. Conservation Biology. 10(4): 949-963.

Nussbaum, Ronald A.; Brodie, Edmund D., Jr.; Storm, Robert M. 1983. Amphibians and reptiles of the Pacific Northwest. Moscow, ID: University of Idaho Press. 332 p.

Oakley, C.; Riddle, P. 1974. The impact of a severe winter and fences on antelope mortality in south central Wyoming. Proceedings of the 6th antelope states workshop; 1974 February 19 -21; Salt Lake City, UT. [Place of publication unknown]: [Publisher unknown]: 155-171.

Olendorff, R.R.; Stoddart, J.W., Jr. 1974. The potential for management of raptor populations in western grassland. In: Hamerstrom, F.N.; Harrell, B.N.; Olendorff, R.R., eds. Management of raptors: Proceedings of the conference on raptor conservation, part 4; 1973 March 22-24; Fort Collins, CO. Vermillion, SD: Raptor Research Foundation. Raptor Research Reports. 2: 44-48.

Onderka, D.K.; Wishart, W.D. 1984. A major die-off from pneumonia in southern Alberta. Proceedings of the biennial symposium of the Northern Wild Sheep and Goat Council. Helena, MT: Idaho Department of Fish and Game. 4: 356-363.

Orloff, S.; Hall, F.; Spiegel, L. 1986. Distribution and habitat requirements of the San Joaquin kit fox in the northern extreme of their range. Transactions of the Western Section of the Wildlife Society: 2260-2270.

Ormsbee, Patricia C.; McComb, William C. 1998. Selection of day roosts by female long-legged myotis in the central Oregon Cascade Range. Journal of Wildlife Management. 62(2): 596-602.

Ottmar, Roger D.; Alvarado, Ernesto; Hessburg, Paul F. [and others]. 1996. Historical and current forest and range landscapes in the interior Columbia River basin and portions of the Klamath and Great basins. Part II: Linking vegetation patterns to potential smoke production and fire behavior. Draft 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.

Oxley, D.J.; Fenton, M.B. 1974. The effects of roads on populations of small mammals. Journal of Applied Ecology. 25: 1073-1087.

Paragi, T.F.; Arthur, S.M.; Krohn, W.B. 1994. Seasonal and circadian activity patterns of female fishers, *Martes pennanti*, with kits. Canadian Field Naturalist. 108(1): 52-57.

Parker, G.R.; Maxwell, J.W.; Morton, L.D. [and others]. 1983. The ecology of lynx (*Lynx canadensis*) on Cape Brenton Island. Canadian Journal of Zoology. 61: 770-786.

Perkins, J. Mark; Peterson, Joshua R. 1997. Bat distribution in the juniper woodlands of the Idaho Owyhee Mountains, summer 1996. Tech. Bull. 97-4. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office. 12 p.

Perlmeter, S.I. 1995. Bats and bridges: a field study of the thermal conditions and social organization of night roosts in the Willamette National Forest. North York, ON: York University. 98 p. M.S. thesis.

Pierson, Elizabeth D.; Rainey, William E.; Koontz, Dolora M. 1991. Bats and mines: experimental mitigation for Townsend's big-eared bat at the McLaughlin mine in California. In: Corner, R.D., Davis, P.R., Foster, S.Q. [and others], eds. Proceedings V: issues and technology in the management of impacted wildlife; 1991 April 8-10; Snowmass Resort, CO. Boulder, CO: Thorne Ecological Institute: 31-42.

Pletscher, Daniel H.; Ream, Robert R.; Boyd, Diane K. [and others]. 1997. Population dynamics of a recolonizing wolf population. Journal of Wildlife Management. 61(2): 459-465.

Poulton, C.E. 1955. Ecology of the non-forested vegetation in Umatilla and Morrow Counties, Oregon. Pullman, WA: Washington State University. 166 p. Ph.D. dissertation.

Pulliam, H.R. 1988. Sources, sinks, and population regulation. American Naturalist. 132(5): 652-661.

Pulliam, H.R.; Danielson, B.J. 1991. Sources, sinks, and habitat selection: a landscape perspective on population dynamics. American Naturalist. 137: 55-66.

Quick, H.F. 1956. Effects of exploitation on a marten population. Journal of Wildlife Management. 20(3): 267-274.

Quigley, Thomas M.; Haynes, Richard W.; Graham, Russell T., tech. eds. 1996. Integrated scientific assessment for ecosystem management in the interior Columbia basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-382. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 303 p. (Quigley, Thomas M., ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Rabe, Michael J.; Morrell, Thomas E.; Green, Heather [and others]. 1998. Characteristics of ponderosa pine snag roosts used by reproductive bats in northern Arizona. Journal of Wildlife Management. 62(2): 612-621.

Ramakka, J.M.; Woyewodzic, R.T. 1993. Nesting ecology of ferruginous hawk in northwestern New Mexico. Journal of Raptor Research. 27(2): 97-101.

Raphael, Martin G.; Marcot, Bruce G.; Holthausen, Richard S.; Wisdom, Michael J. 1998. Terrestrial species and habitats. Journal of Forestry. 96(10): 22-27.

Reed, Rebecca A.; Johnson-Barnard, Julia; Baker, William L. 1996. Contribution of roads to forest fragmentation in the Rocky Mountains. Conservation Biology. 10: 1098-1106.

Reynolds, R.T.; Graham, R.T.; Reiser, M.H. [and others]. 1992. Management recommendations for the northern goshawk in the Southwestern United States. Gen. Tech. Rep. RM-217. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 184 p.

Rich, T.D.; Rothstein, S.I. 1985. Sage thrashers reject cowbird eggs. Condor. 87: 561-562.

Rickard, William H.; Poole, Linda D. 1989. Terrestrial wildlife of the Hanford Site: past and future. Northwest Science. 63(4): 183-192.

Romesburg, H.C. 1984. Cluster analysis for researchers. Belmont, CA: Lifetime Learning Publications. 334 p.

Rowland, Mary M.; Wisdom, Michael J.; Johnson, Bruce K. [and others]. 2000. Elk distribution and modeling in relation to roads. Journal of Wildlife Management. 64(3): 672-684.

Rudolph, Karen M.; Cowan, Lisa M.; Hunter, David L. [and others]. [In prep.]. Sharing of *Pasteurella* spp. among free-ranging bighorn sheep, domestic sheep, and feral goats.

Ruediger, Bill. 1996. The relationship between rare carnivores and highways. In: Evink, G.; Garrett, P.; Berry, J., eds. Transportation and wildlife: reducing wildlife mortality and improving wildlife passageways across transportation corridors. Proceedings of a symposium; 1996 April 3-May 2; Orlando, FL. Orlando, FL: Department of Transportation, Federal Highway Administration: 24-38.

Ruggiero, Leonard F.; Aubry, Keith B.; Buskirk, Steven W. [and others]. 1999. Ecology and conservation of lynx in the United States. Niwot, CO: University Press of Colorado and U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 480 p.

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 Agriculture, Forest Service, Pacific Northwest Research Station. 36 p.

Saab, V.A.; Bock, C.E.; Rich, T.D.; Dobkin, D.S. 1995. Livestock grazing effects in western North America. In: Martin, Thomas E.; Finch, Deborah M., eds. Ecology and management of neotropical migratory birds. New York: Oxford University Press: 311-353.

Saab, V.A.; Marks, J.S. 1992. Summer habitat use by Columbian sharp-tailed grouse in western Idaho. Great Basin Naturalist. 52(2): 166-173.

Saab, V.A.; Rich, T.D. 1997. Large-scale conservation assessment for neotropical migratory land birds in the interior Columbia River basin. Gen. Tech. Rep. PNW-GTR-399. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 57 p. (Quigley, Thomas M., ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Sarell, M.J.; McGuinness, K.P. 1993. Rare bats of the shrub-steppe ecosystem of eastern Washington. Gen. Tech. Rep. Olympia, WA: Washington Department of Wildlife.

SAS Institute, Inc. 1989a. Cluster procedure. In: SAS/STAT user’s guide, version 6. 4th ed. SAS Institute: 519-614. Vol. 1. Available from: SAS Institute, Inc., SAS Campus Drive, Cary, NC 27513.

SAS Institute, Inc. 1989b. Corr procedure. SAS procedures guide, version 6. 3d ed. SAS Institute: 209-235. Available from: SAS Institute, Inc., SAS Campus Drive, Cary, NC 27513.

SAS Institute, Inc. 1989c. Tree procedure. SAS/STAT user's guide, version 6. 4th ed. SAS Institute: 1613-1632. Vol. 2. Available from: SAS Institute, Inc., SAS Campus Drive, Cary, NC 27513.

Sauer, J.R.; Peterjohn, B.J.; Schwartz, S.; Hines, J.E. 1996a. The North American breeding bird survey home page; version 95.1. Laurel, MD: Patuxent Wildlife Research Center. URL http://www.mbr-pwrc.usgs.gov/bbs/bbsold.html.

Sauer, John R.; Pendleton, Grey W.; Peterjohn, Bruce G. 1996b. Evaluating causes of population change in North American insectivorous songbirds. Conservation Biology. 10(2): 465-478.

Schmutz, J.K. 1984. Ferruginous and Swainson’s hawk abundance and distribution in relation to land use in southeastern Alberta. Journal of Wildlife Management. 48(4): 1180-1187.

Schuler, Carol A.; Rickard, William H.; Sargeant, Glen A. 1993. Conservation of habitats for shrub-steppe birds. Environmental Conservation. 20(1): 57-64.

Scott, M.D.; Servheen, G. 1985. Caribou ecology. Boise, ID: Idaho Department of Fish and Game; job comp. report, proj. no. W-160-R-11. 136 p.

Scott, Michael J.; Davis, Frank; Csuti, Blair [and others]. 1993. GAP analysis: a geographic approach to protection of biological diversity. Wildlife Monograph no. 123. 41 p.

Singer, F.J. 1978. Behavior of mountain goats in relation to U.S. Highway 2, Glacier National Park, Montana. Journal of Wildlife Management. 42(3): 591-597.

Smith, Michael Roger. 1994. Evaluating the conservation of avian diversity in eastern Washington: a geographic analysis of upland breeding birds. Seattle: University of Washington. 89 p. M.S. thesis.

Smith, R.L. 1963. Some ecological notes on the grasshopper sparrow. Wilson Bulletin. 75(2): 159-165.

Southern Appalachian Man and the Biosphere Cooperative (SAMAB). 1996. The Southern Appalachian assessment summary report. Rep. 1. Atlanta, GA: U.S. Department of Agriculture, Forest Service, Southern Region. 118 p.

Spraker, T.R.; Hibbler, C.P.; Schoonveld, G.G.; Adbetm, W.S. 1984. Pathological changes and microorganisms found during a stress related die-off. Journal of Wildlife Diseases. 20: 319-327.

Squires, J.R.; Reynolds, R.T. 1997. Northern goshawk (*Accipiter gentilis*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 298. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists’ Union. 32 p.

St. Clair, L.L.; Johansen, J.R. 1993. Introduction to the symposium on soil crust communities. Great Basin Naturalist. 53(1): 1-4.

St. Clair, Larry L.; Johansen, Jeffrey R.; Rushforth, Sammuel R. 1993. Lichens of soil crust communities in the Intermountain area of the Western United States. Great Basin Naturalist. 53: 5-12.

Stebbins, Robert C. 1985. Peterson field guides: western reptiles and amphibians. Boston, MA: Houghton Mifflin Co. 336 p.

Stelfox, John G. 1971. Bighorn sheep in the Canadian Rockies: a history 1800-1970. The Canadian Field-Naturalist. 85: 101-122.

Swihart, R.K.; Slade, N.A. 1984. Road crossing in *Sigmodon hispidus* and *Microtus ochrogaster*. Journal of Mammalogy. 65: 357-360.

Szaro, Robert C. 1986. Guild management: an evaluation of avian guilds as a predictive tool. Environmental Management. 10(5): 681-688.

Taylor, Elroy; McCoy, Matthew; Bodie, Walt. 1993. California bighorn sheep ecology. Study II: habitat selection. Jobs 1 and 3. Boise, ID: Idaho Department of Fish and Game; job completion rep.; proj. no. W-160-R-20. 34 p.

Terra-Berns, Mary; Call, Paula; Harris, Chuck [and others]. 1997. Canada lynx in Idaho: past, present, and future. Boise, ID: Idaho Department of Fish and Game. 79 p.

Thiel, R.P. 1985. Relationship between road densities and wolf habitat suitability in Wisconsin. American Midland Naturalist. 113: 404-407.

Thomas, Jack Ward; Anderson, Ralph G.; Maser, Chris; Bull, Evelyn L. 1979. Snags. In: Thomas, Jack Ward, tech. ed. Wildlife habitats in managed forests, the Blue Mountains of Oregon and Washington. Agric. Handb. No. 553. Washington, DC: U.S. Department of Agriculture: 60-77.

Thompson, I.D. 1994. Marten populations in uncut and logged boreal forests in Ontario. Journal of Wildlife Management. 58(2): 272-280.

Thompson, I.D.; Colgan, P.W. 1987. Numerical responses of martens to a food shortage in north central Ontario. Journal of Wildlife Management. 51(4): 824-835.

Thurber, Joanne M.; Peterson, Rolf O.; Drummer, Thomas D.; Thomasma, Scott A. 1994. Gray wolf response to refuge boundaries and roads in Alaska. Wildlife Society Bulletin. 22(1): 61-68.

Tirhi, M.J. 1995. Washington state management plan for sharp-tailed grouse. Olympia, WA: Washington Department of Fish and Wildlife, Wildlife Management Program. 99 p.

Tobalske, B.W. 1997. Lewis' woodpecker (*Melanerpes lewis*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 284: Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union. 28 p.

Tomich, P. Quentin. 1982. Ground squirrels (*Spermophilus beecheyi* and allies). In: Chapman, Joseph A.; Feldhamer, George A., eds. Wild mammals of North America: biology, management, and economics. Baltimore, MD: Johns Hopkins University Press: 192-208.

Trombulak, S.C.; Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology. 14: 18-30.

Tuttle, M.D. 1988. America's neighborhood bats. Austin, TX: University of Texas Press. 96 p.

Unsworth, J.M.; Kuck, L.; Scott, M.D.; Garton, E.O. 1993. Elk mortality in the Clearwater Drainage of northcentral Idaho. Journal of Wildlife Management. 57: 495-502.

U.S. Department of Agriculture, Forest Service. 1996. Status of the interior Columbia basin: summary of scientific findings. Gen. Tech. Rep. PNW-GTR-385. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station; U.S. Department of the Interior, Bureau of Land Management. 144 p.

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management. 1994. Record of decision for amendments to Forest Service and Bureau of Land Management planning document within the range of the northern spotted owl. [Place of publication unknown]. 74 p. [plus Attachment A: standards and guidelines].

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management. 1997a. Eastside draft environmental impact statement. Boise, ID: Interior Columbia Basin Ecosystem Management Project. [Irregular pagination]. 2 vol.

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management. 1997b. Upper Columbia River basin draft environmental impact statement. Boise, ID: Interior Columbia Basin Ecosystem Management Project. [Irregular pagination]. 2 vol.

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Fish and Wildlife Service. 1996. Conservation agreement: *Spermophilus brunneus brunneus*, northern Idaho ground squirrel. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest; Boise, ID: U.S. Department of the Interior, Fish and Wildlife Service. 14 p.

U.S. Department of the Interior, Bureau of Land Management. 1997. Proposed standards for rangeland health and guidelines for livestock grazing management. BLM/ID/PT-97/002+4120. Boise ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office.

U.S. Department of the Interior, Fish and Wildlife Service. 1987. Northern Rocky Mountain wolf recovery plan. Denver, CO: U.S. Department of the Interior, Fish and Wildlife Service. 119 p.

Van Ballenberghe, Victor; Erickson, Albert W.; Byman, David. 1975. Ecology of the timber wolf in northeastern Minnesota. Wildlife Monographs. No. 43. 43 p.

Verner, Jared. 1984. The guild concept applied to management of bird populations. Environmental Management. 8(1): 1-14.

Vestjens, W.J.M. 1973. Wildlife mortality on a road in New South Wales. Emu. 73: 107-112.

Vogel, C.A.; Reese, K.P. 1995. Mountain quail status report: a document preliminary to a habitat conservation assessment for mountain quail. 33 p. 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.

Wai-Ping, V.; Fenton, M.B. 1989. Ecology of the spotted bat (*Euderma maculatum*) roosting and foraging behavior. Journal of Mammalogy. 70(3): 617-622.

Wakelyn, Leslie A. 1987. Changing habitat conditions on bighorn sheep ranges in Colorado. Journal of Wildlife Management. 51(4): 904-912.

Ward, J.M.; Kennedy, P.L. 1996. Effects of supplemental food on size and survival of juvenile northern goshawks. Auk. 113: 200-208.

Ward, Richard M.P.; Krebs, Charles J. 1985. Behavioural responses of lynx to declining snowshoe hare abundance. Canadian Journal of Zoology. 63(12): 2817-2824.

Western, David; Pearl, Mary C., eds. 1989. Conservation for the twenty-first century. New York: Oxford University Press. 365 p.

Whitaker, John O., Jr.; Maser, Chris; Keller, Laurel E. 1977. Food habits of bats of western Oregon. Northwest Science. 51(1): 46-55.

White, C.E. 1974. Current problems and techniques in raptor management and conservation. Transactions of the North American Wildlife Conference. 39: 301-312.

White, C.M.; Thurow, T.L. 1985. Reproduction of ferruginous hawks exposed to controlled disturbance. Condor. 87(1): 14-22.

Wielgus, Robert B.; Bunnell, Fred, L. 1995. Tests of hypotheses for sexual segregation in grizzly bears. Journal of Wildlife Management. 59(3): 552-560.

Wiens, John A.; Rotenberry, John T. 1981. Habitat associations and community structure of birds in shrub-steppe environments. Ecological Monographs. 5(1): 21-41.

Wisdom, Michael J.; Cimon, Norman J.; Johnson, Bruce K. [and others]. 1999. Distribution and spatial partitioning of mule deer and elk in relation to traffic. Unpublished report. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, 1401 Gekeler Lane, La Grande, OR 97850.

Wisdom, Michael J. 2000. Overlays and correspondence of lynx and wolverine occurrence data with habitat abundance and road density estimates in the Interior Columbia Basin. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Forestry and Range Sciences Laboratory, 1401 Gekeler Lane, La Grande, OR 97850.

Wishart, William. 1978. Bighorn sheep. In: Schmidt, John L.; Gilbert, Douglas L., eds. Big game of North America: ecology and management. Harrisburg, PA: Stackpole Books: 161-171.

Witmer, Gary W.; Martin, Sandra K.; Sayler, Rodney D. 1998. Forest carnivore conservation and management in the interior Columbia basin: issues and environmental correlates. Gen. Tech. Rep. PNW-GTR-420. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 51 p. (Quigley, Thomas M., ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Yoakum, J. 1980. Habitat management guides for the American pronghorn antelope. Tech. Note 347. Denver, CO: U.S. Department of the Interior, Bureau of Land Management. 77 p.

Yoakum, James D. 1978. Pronghorn. In: Schmidt, John L.; Gilbert, Douglas L., eds. Big game of North America. Harrisburg, PA: Stackpole Books: 103-121.

Yosef, R. 1996. Loggerhead shrike (*Lanius ludovicianus*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 231. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union. 28 p.

Zwickel, Fred C. 1992. Blue grouse, *Dendragapus obscurus*. In: Poole, Alan F.; Stettenheim, Peter; Gill, Frank B., eds. The birds of North America. No. 15. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union. 28 p.

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May 2000

Source Habitats for Terrestrial Vertebrates of Focus in the Interior Columbia Basin: Broad-Scale Trends and Management Implications

Volume 2– Group Level Results

**Michael J. Wisdom, Richard S. Holthausen,
Barbara C. Wales, Christina D. Hargis,
Victoria A. Saab, Danny C. Lee, Wendel J. Hann,
Terrell D. Rich, Mary M. Rowland, Wally J. Murphy,
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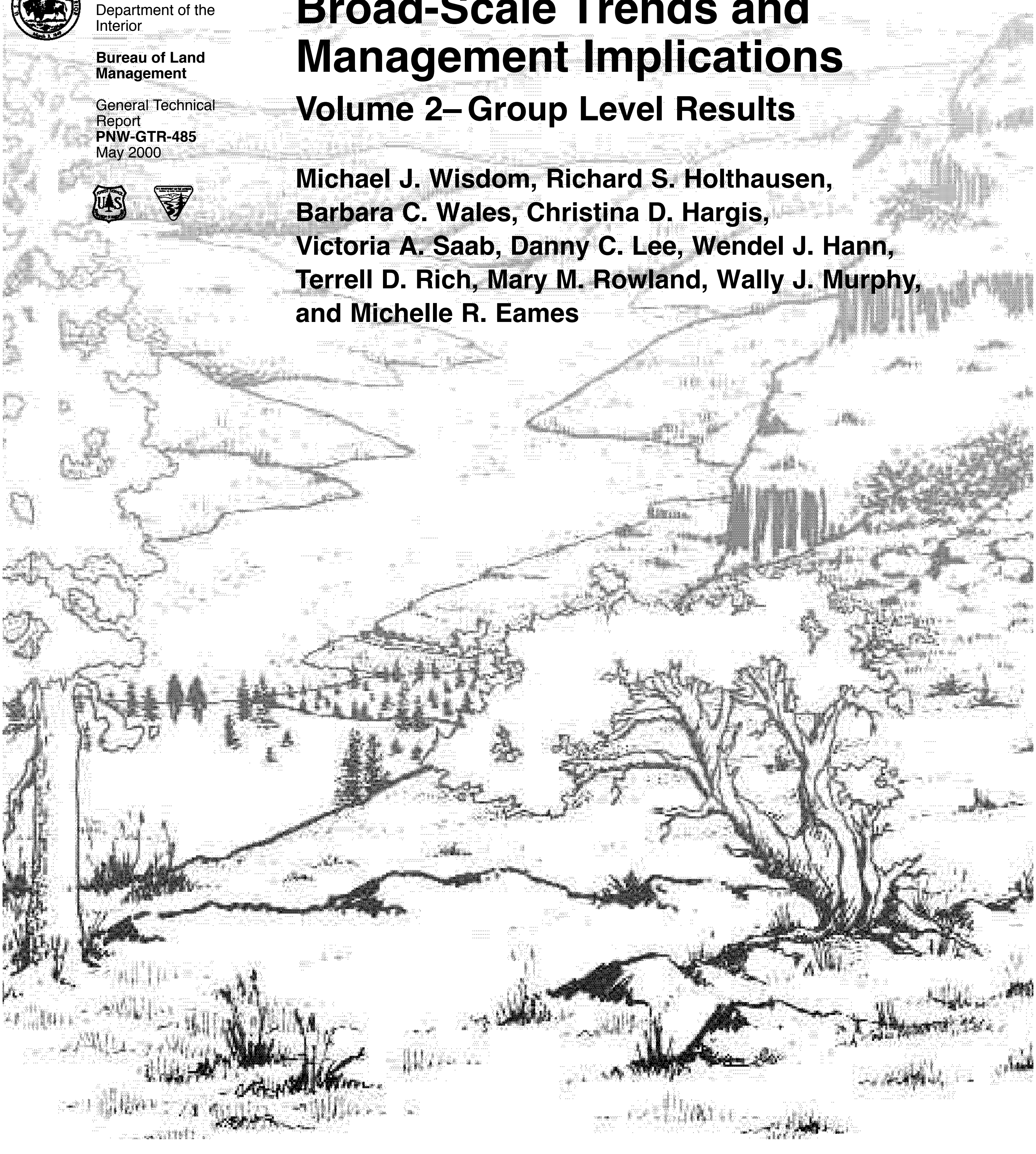

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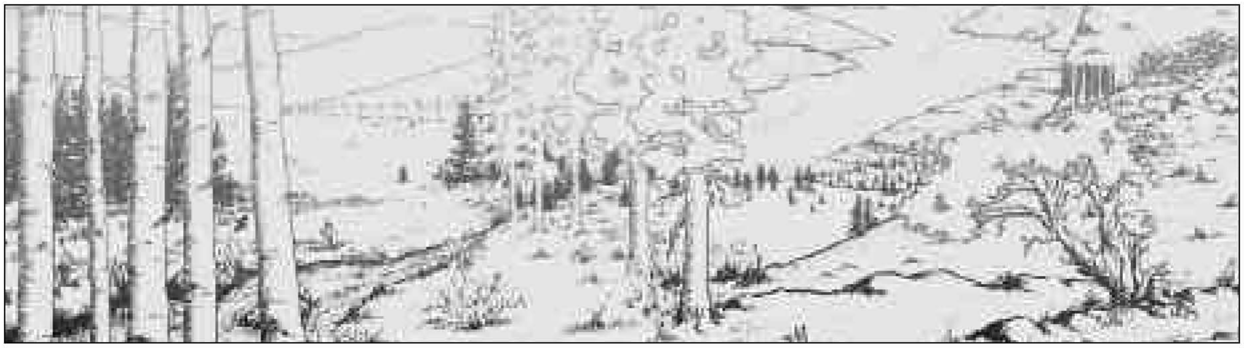

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Interior Columbia Basin Ecosystem Management Project: Scientific Assessment

Thomas M. Quigley, Editor

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Abstract

Wisdom, Michael J.; Holthausen, Richard S.; Wales, Barbara C.; Hargis, Christina D.; Saab, Victoria A.; Lee, Danny C.; Hann, Wendel J.; Rich, Terrell D.; Rowland, Mary M.; Murphy, Wally J.; Eames, Michelle R. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia basin: broad-scale trends and management implications. Volume 2—Group level results. Gen. Tech. Rep. PNW-GTR-485. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 3 vol. (Quigley, Thomas M., tech. ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

We defined habitat requirements (source habitats) and assessed trends in these habitats for 91 species of terrestrial vertebrates on 58 million ha (145 million acres) of public and private lands within the interior Columbia basin (hereafter referred to as the basin). We also summarized knowledge about species-road relations for each species and mapped source habitats in relation to road densities for four species of terrestrial carnivores. Our assessment was conducted as part of the Interior Columbia Basin Ecosystem Management Project (ICBEMP), a multiresource, multidisciplinary effort by the USDAForest Service (FS) and the USDI Bureau of Land Management (BLM) to develop an ecosystem-based strategy for managing FS and BLM lands within the basin. Our assessment was designed to provide technical support for the ICBEMPand was done in five steps. First, we identified species of terrestrial vertebrates for which there was ongoing concern about population or habitat status (species of focus), and for which habitats could be estimated reliably by using a large mapping unit (pixel size) of 100 ha (247 acres) and broad-scale methods of spatial analysis. Second, we evaluated change in source habitats from early European settlement (historical, circa 1850 to 1890) to current (circa 1985 to 1995) conditions for each species and for hierarchically nested groups of species and families of groups at the spatial scales of the watershed (5th hydrologic unit code [HUC]), subbasin (4th HUC), ecological reporting unit, and basin. Third, we summarized the effects of roads and road-associated factors on populations and habitats for each of the 91 species and described the results in relation to broad-scale patterns of road density. Fourth, we mapped classes of the current abundance of source habitats for four species of terrestrial carnivores in relation to classes of road density across the 164 subbasins and used the maps to identify areas having high potential to support persistent populations. And fifth, we used our results, along with results from other studies, to describe broad-scale implications for managing habitats deemed to have undergone long-term decline and for managing species negatively affected by roads or road-associated factors.

Our results indicated that habitats for species, groups, and families associated with old-forest structural stages, with native grasslands, or with native shrublands have undergone strong, widespread decline. Implications of these results for managing old-forest structural stages include consideration of (1) conservation of habitats in subbasins and watersheds where decline in old forests has been strongest; (2) silvicultural manipulations of mid-seral forests to accelerate development of late-seral stages; and (3) long-term silvicultural manipulations and long-term accommodation of fire and other disturbance regimes in all forested structural stages to hasten development and improvement in the amount, quality, and distribution of old-forest stages. Implications of our results for managing rangelands include the potential to (1) conserve native grasslands and shrublands that have not undergone large-scale reduction in composition of native plants; (2) control or eradicate exotic plants on native grasslands and shrublands where invasion potential or spread of exotics is highest; and (3) restore native plant communities by using intensive range practices where potential for restoration is highest.

Our analysis also indicated that >70 percent of the 91 species are affected negatively by one or more factors associated with roads. Moreover, maps of the abundance of source habitats in relation to classes of road density suggested that road-associated factors hypothetically may reduce the potential to support persistent populations of terrestrial carnivores in many subbasins. Management implications of our summarized road effects include the potential to mitigate a diverse set of negative factors associated with roads. Comprehensive mitigation of road-associated factors would require a substantial reduction in the density of existing roads as well as effective control of road access in relation to management of livestock, timber, recreation, hunting, trapping, mineral development, and other human activities.

A major assumption of our work was that validation research will be conducted by agency scientists and other researchers to corroborate our findings. As a preliminary step in the process of validation, we found high agreement between trends in source habitats and prior trends in habitat outcomes that were estimated as part of the habitat outcome analysis for terrestrial species within the basin. Results of our assessment also were assumed to lead to finer scale evaluations of habitats for some species, groups, or families as part of implementation procedures. Implementation procedures are necessary to relate our findings to local conditions; this would enable managers to effectively apply local conservation and restoration practices to support broad-scale conservation and restoration strategies that may evolve from our findings.

Keywords: Cluster analysis, conservation, forest management, habitat, habitat condition, habitat management, habitat trend, interior Columbia basin, Interior Columbia Basin Ecosystem Management Project, landscape ecology, landscape analysis, population viability, rangeland management, terrestrial vertebrates, spatial analysis, species of focus, sink, sink environment, source, source environment, source habitat, source habitats, restoration, species groups, monitoring, validation research, viability, wildlife, wildlife-habitat relations.

Foreword

This publication consists of three volumes so that our findings—which consist of hundreds of tables, figures, pages of text, and supporting citations—could be presented in a manner most usable to resource managers, biologists, and the public. Volume 1 is designed as an overview of objectives, methods, key results, and management implications. Volumes 2 and 3 contain increasingly detailed results that support and complement results in volume 1. We believe that resource managers may find sufficient detail in the generalized results and implications presented in volume 1, but that management biologists and other users of the results and supporting data will want to refer to all three volumes. Results, management implications, and supporting citations provided in volume 2 are especially important to consider as part of step-down implementation procedures and related management conducted by field units within the interior Columbia basin. By contrast, information in volume 1 may be particularly useful in serving broad-scale planning issues, objectives, and strategies for the interior Columbia basin as a whole. Regardless of application, all three volumes are intended to function together as a comprehensive assessment of habitat trends and a summary of other environmental factors affecting terrestrial vertebrates whose population or habitat status is of ongoing concern to resource managers. Data underlying most tables presented in the three volumes also are available at the web site for the ICBEMP: http://www.icbemp.gov/spatial/metadata/databases.

Preface

The Interior Columbia Basin Ecosystem Management Project was initiated by the Forest Service and the Bureau of Land Management to respond to several critical issues including, but not limited to, forest and rangeland health, anadromous fish concerns, terrestrial species viability concerns, and the recent decline in traditional commodity flows. The charter given to the project was to develop a scientifically sound, ecosystem-based strategy for managing the lands of the interior Columbia River basin administered by the Forest Service and the Bureau of Land Management. The Science Integration Team was organized to develop a framework for ecosystem management, an assessment of the socioeconomic and biophysical systems in the basin, and an evaluation of alternative management strategies. This paper is one in a series of papers developed as background material for the framework, assessment, or evaluation of alternatives. It provides more detail than was possible to disclose directly in the primary documents.

The Science Integration Team, although organized functionally, worked hard at integrating the approaches, analyses, and conclusions. It is the collective effort of team members that provides depth and understanding to the work of the project. The Science Integration Team leadership included deputy team leaders Russell Graham and Sylvia Arbelbide; landscape ecology—Wendel Hann, Paul Hessburg, and Mark Jensen; aquatic—Jim Sedell, Kris Lee, Danny Lee, Jack Williams, and Lynn Decker; economic— Richard Haynes, Amy Horne, and Nick Reyna; social science—Jim Burchfield, Steve McCool, Jon Bumstead, and Stewart Allen; terrestrial—Bruce Marcot, Kurt Nelson, John Lehmkuhl, Richard Holthausen, Randy Hickenbottom, Marty Raphael, and Michael Wisdom; spatial analysis—Becky Gravenmier, John Steffenson, and Andy Wilson.

Thomas M. Quigley
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Contents

Introduction

This volume is the second in a three-volume publication that defines and assesses trends in source habitats for 91 terrestrial vertebrate species within the interior Columbia River basin (hereafter referred to as "basin") (See "Glossary," vol. 3, for terms used in this paper). This assessment was conducted as part of the Interior Columbia Basin Ecosystem Management Project (ICBEMP), a multiresource, multidisciplinary effort by the USDA Forest Service (FS) and the USDI Bureau of Land Management (BLM) to develop an ecosystem-based strategy for managing lands within the basin administered by the FS and BLM. The assessment area extends over 58 million ha[1] (145 million acres) in eastern Washington, eastern Oregon, Idaho, western Montana, and small portions of Nevada, California, Wyoming, and Utah (figs. 1 and 2). The purpose of this publication is to provide technical support to the ICBEMP regarding trends in the areal extent of wildlife habitats in the basin, as well as management implications regarding those trends. Additionally, it can be used to provide a broad-scale view of how wildlife habitats have changed in the basin since early European settlement and factors that have contributed to those changes.

This publication focuses on source habitats rather than all habitats in which a species is known to occur. Source habitats are those characteristics of macrovegetation that contribute to stationary or positive population growth for a species in a specified area and time. Source habitats contribute to source environments (Pulliam 1988, Pulliam and Danielson 1991), which represent the composite of all environmental conditions that results in stationary or positive population growth for a species in a specified area and time. The distinction between source habitats and source environments is important for understanding our evaluation and its limitations. For example, source habitats for a bird species during the breeding season would include those characteristics of vegetation that contribute to successful nesting and rearing of young, but would not include nonvegetative factors, such as the effects of pesticides on thinning of eggshells, which also affect production of young. Consequently, we have tried to identify all factors that affect population performance of each species as a complement to our explicit analysis of source habitats. For our analysis, we relied on published literature and guidance from species experts to identify source habitats and additional factors that presumably affect population performance.

The 91 species in our analysis are organized into 40 groups, 37 of which are then organized into 12 families. Groups are composed of one or more species that share common source habitats, as defined by vegetation cover types and structural stages. Similar groups also are clustered into families whose source habitats generally fall into similar terrestrial community groups, a broader classification that includes several cover types. Group size ranges from 1 to 17 species, and family size ranges from one to nine groups.

Volume 1 describes methods used to select species for analysis, place them in groups and families, estimate source habitats, and analyze habitat trends. It also includes general analyses of source habitat trends at all three levels—species, group, and family—including a correlation analysis that evaluates how well species-level trends in source habitats are reflected in the higher level group- and family-level trends. Volume 1 also identifies causes for the observed trends and ecological processes important for maintaining source habitats as part of the family-level results. Additionally, volume 1 provides a special section on species and groups that are negatively affected by road-related human activities. In volume 2, we present more detailed results on the analysis of source habitat trends at the group level in support of the more generalized results presented in volume 1. The appendices in volume 3 provide further data and results in support of both volumes 1 and 2.

For each of the 40 groups discussed in volume 2, we specifically present results on source habitat trends, interpret those results, and discuss management implications. In the results section, we list the species included in each group, display range maps for each of the species, and describe source habitats and special habitat features for each species. Source habitats

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[1] See "Abbreviations," p. 396, for definitions of abbreviated units of measure.

Figure 1—Assessment boundaries of the Interior Columbia Basin Ecosystem Management Project and the 13 ecological reporting units.

Figure 2—Land ownership within the Interior Columbia Basin Ecosystem Management Project science assessment area.

and special habitat features for each species in each group and family are listed in volume 3, appendix 1, tables 1 and 2.

In the results section of volume 2, we specifically display maps that compare the historical and current distribution of source habitats within the basin for each group, and describe changes in areal extent that were projected to have occurred since the historical period. These changes are analyzed at the watershed level, a unit of land whose mean size is about 22 500 ha (56,000 acres). The watershed results are summarized by ecological reporting units (ERUs), which represent 13 broad geographical regions within the basin (fig. 1) that differ significantly in biophysical characteristics (Hann and others 1997).

The section on interpretation of results in volume 2 consists of four components. First, we provide a description of the vegetation changes that underlie source habitat changes. Ecological processes and management actions that caused the vegetation changes are described in volume 1, and more thoroughly in Hann and others (1997). Second, changes from historical to current in the condition of special habitat features are disclosed for those features for which information is available. Third, factors other than habitat that significantly affect species in the group are discussed, with emphasis on the effects of specific management activities and other human disturbances. Finally, any available data on population status and trends for any species in the group are presented. We have not performed any correlations or added discussion of anecdotal similarity between habitat trends and population trends because our habitat analysis addresses different time frames and different geographic areas than do population trend data available for most species.

The final section of volume 2 discusses management implications based on both the findings of this analysis and published literature for each group of species. Management implications are presented in three parts. First, issues relevant to species in the group are discussed. These include issues related to broad-scale source habitats, special habitat features, and other factors that significantly influence the group. Broad strategies that could be used to resolve these issues are presented, and geographic priorities for the strategies are offered where appropriate. The third part of the management implications section consists of specific on-the-ground management practices that could be used in the implementation of the strategies. In all cases, the discussion of strategies and practices is intended to be addressed within the context of broader ecosystem-based objectives. Implementation of the strategies and practices for any single group without consideration of other ecosystem elements would not be appropriate.

The list of strategies and practices outlined for each group of species in volume 2 should be considered a menu of possible approaches that could be adopted by managers to help achieve their objectives for conservation and restoration of habitats. Before any of these approaches are adopted, they should be analyzed to determine their effectiveness, their compatibility with overall ecosystem management objectives, and their applicability to specific situations. Testing and validation should continue through all the geographic scales of implementation.

In summary, the strategies presented at the family level in volume 1 represent a synthesis of similar group strategies developed in volume 2. Volume 1 therefore provides a broader, more generalized perspective of source habitat trends in the basin, whereas volume 2 offers a more specific, indepth coverage of the same analysis. Thus, users of our publication can refer to volume 1 for an overview of results and implications, refer to volume 2 for detailed results that support the overview, and refer to volume 3 for the most specific results and information in support of both volumes 1 and 2.

Group 1—Pygmy Nuthatch, White-Breasted Nuthatch, and White-Headed Woodpecker

Results

Species ranges, source habitats, and special habitat features—Group 1 consists of the pygmy nuthatch, white-breasted nuthatch, and white-headed woodpecker, all of which are year-round residents within the basin.[2] The pygmy nuthatch is widespread except for the Columbia Plateau and southern portions of the basin, and the white-breasted nuthatch occurs throughout most of the basin (fig. 3). The white-headed woodpecker has the most restricted range, occurring in the eastern slope of the Cascade Range, the Blue Mountains, the Okanogan Mountains, and mountains of Idaho. Source habitats for group 1 are found in old forests of Sierra Nevada mixed-conifer and ponderosa pine cover types. The white-breasted nuthatch also breeds in old forests of aspen and cottonwood-willow, in Oregon white oak, and in unmanaged young forests of interior ponderosa pine (vol. 3, appendix 1, table 1).

A special habitat feature for group 1 is large-diameter snags for nesting and foraging (vol. 3, appendix 1, table 2). Both nuthatches are secondary cavity nesters and can use various nesting structures (McEllin 1979), whereas the white-headed woodpecker is a primary cavity excavator of soft snags and is therefore more limited by the degree of wood decay suitable for nest hole excavation (Garrett and others 1996). White-headed woodpeckers typically nest in snags and leaning logs, and occasionally nest in the dead tops of live trees (Garrett and others 1996, Milne and Hejl 1989). White-breasted nuthatches nest in natural cavities of live ponderosa pine more often than in snags (Brawn and Balda 1988, McEllin 1979). Suitable nest sites for all three species usually are found within the upper diameter classes of trees and snags. Average diameters reported for nest trees are 57.93 ± 3.65 cm (22.80 ± 1.43 in [$\bar{x}$ ± SE]) for pygmy nuthatch (McEllin 1979), 53.77 ± 1.56 cm (21.16 ± 0.61 in [$\bar{x}$ ± SE]) for white-breasted nuthatch (McEllin 1979), and 80 ± 65 cm (31 ± 25 in [$\bar{x}$ ± SE]) for white-headed woodpecker (Garrett and others 1996, Milne and Hejl 1989).

All three species forage primarily on live trees. White-breasted nuthatches glean insects from tree trunks and were observed in Colorado to spend nearly 75 percent of foraging time on ponderosa pine trunks (Bock 1969). In the same study, pygmy nuthatches foraged more generally in live ponderosa pine, dividing their foraging time fairly equally among needles, branches, and trunks. In Oregon, 80 percent of white-headed woodpecker foraging time was on live trees, and a preference was shown for trees with diameters >25 cm (10 in) (Bull and others 1986a).

Broad-scale changes in source habitats—Source habitats for group 1 likely occurred throughout the mountainous areas of the basin historically, and were most extensive throughout the Cascade Range, the Okanogan Mountains, and in central Oregon (fig. 4A). Currently, source habitats cover roughly the same geographical extent, but habitat patches appear more disjunct (fig. 4B). The Upper Klamath ERU continues to provide extensive source habitats, but elsewhere, <25 percent of most watersheds within the distribution of these species currently contains source habitats.

Basin-wide, >50 percent of watersheds had strong negative declines in the availability of source habitats (fig. 5). This basin-wide trend was mirrored within six ERUs that also had strong negative declines in more than 50 percent of the watersheds within the individual ERU boundaries: the Northern Cascades, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, Upper Snake, and Snake Headwaters ERUs (fig. 5). Source habitats in the Upper Snake and Snake Headwaters ERUs were less than 2 percent of either ERU, both historically and currently (vol. 3, appendix 1, table 3). The extent of coverage in the Northern Cascades, Northern Glaciated Mountains, and Lower Clark Fork, however, was substantial historically, accounting for 19 to 24 percent of the total area of these ERUs (vol. 3, appendix 1, table 3). In general, areas predominated by declining trends were in the northern basin, whereas the central and southwestern parts of the basin had mixed trends (fig. 4C).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats—Most projected declines in source habitats were due to losses, particularly in the northern part of the basin, of late-seral forests that today are in early- and mid-seral stages

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[2] See table 1, volume 1, for common and scientific names of the vertebrate species of focus, and appendix 3, volume 3, for scientific and common names of plants and animals not addressed as terrestrial vertebrate species of focus.

Figure 3—Ranges of species in group 1 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 4—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage area of source habitats from historical to current periods (C), for group 1 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

VIEW FIGURE

Figure 5—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 1, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

(vol. 3, appendix 1, table 4). Throughout the basin, mid-seral shade-tolerant forests seem to be at nearly twice their historical levels (Hann and others 1997). A widespread change has been the transition of Pacific and interior ponderosa pine old forests to mid-seral stands of interior Douglas-fir and grand fir-white fir.

Managed young-forest structural stages of ponderosa pine, used as source habitats for the white-breasted nuthatch, generally had strongly increasing trends corresponding to the decline in old-forest structural stages. In contrast, unmanaged young forests, characterized by higher snag densities than managed forests, experienced strong declines throughout the range of group 1 (vol. 3, appendix 1, table 4).

Within the cottonwood-willow cover type, old forests had strongly declining trends throughout the basin (vol. 3, appendix 1, table 4) and generally remain only in stands smaller than the 1-km^2 (0.4-mi^2) mapping unit used in this analysis. These losses occurred from changes in historical hydrologic regimes. Flooding by reservoirs eliminated many cottonwood-willow stands, and reservoirs also reduced periodic flooding, a disturbance that is frequently needed for cottonwood seed establishment (Merigliano 1996, Rood and Heinze-Milne 1989). The declines in riparian woodlands and old-forest ponderosa pine documented for the basin are part of a larger picture of similar declines throughout the Western United States (Noss and others 1995).

Condition of special habitat features—Large-diameter ponderosa pine snags are a special habitat feature for group 1. In roaded areas with a history of timber sales, large-diameter snags $\geq$53 cm (21 in) have been reduced basin-wide (Hann and others 1997, Hessburg and others 1999, Quigley and others 1996). Nesting and foraging substrates for group 1 have therefore been reduced.

Other factors affecting the group—Roads indirectly affect group 1 because roaded areas in the basin have fewer snags than unroaded areas (Hann and others 1997). Roads enable snags to be cut, either in conjunction with timber sales, or by individuals seeking firewood. The additional loss of snags in areas where snags are already in low density could limit populations of species in group 1.

Population status and trends—Population trends were estimated for all three species by using Breeding Bird Survey (BBS) route data from 1966 to 1995 (Sauer and others 1996). These data have not been summarized for the basin, but summaries for various states, USDI Fish and Wildlife Service regions, and BBS physiographic regions are available. Pygmy nuthatch numbers were stable within all summary geographic areas of relevance to the basin, which were physiographic region 64 (Central Rocky Mountains), USDI Fish and Wildlife Service Region 1 (5 western states), and the Western United States (11 western states) (Sauer and others 1996). White-breasted nuthatch numbers were stable in physiographic region 64 but increased 3.6 percent annually (n = 149, $P < 0.01$) in USDI Fish and Wildlife Service Region 1 and about the same throughout the Western United States. White-headed woodpecker numbers were not summarized for physiographic region 64 but increased 3.3 percent annually (n = 45, $P < 0.10$) in USDI Fish and Wildlife Service Region 1 and similarly throughout the 11 Western states (Sauer and others 1996).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integration of potential resource objectives for group 1 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The results of our habitat trend analysis suggest the following issues are of high priority for group 1:

1. Basin-wide decline in late-seral interior and Pacific ponderosa pine.

2. Basin-wide loss of large-diameter snags (>53 cm [21 in]).

3. High risk of additional loss of ponderosa pine habitat through stand-replacing fires.

4. Decline in old forests of aspen and cottonwood-willow.

Potential strategies—The following strategies could be used to reverse broad-scale declines in source habitats:

1. (To address issue no. 1) Retain stands of interior and Pacific ponderosa pine where old-forest conditions are present, and actively manage to promote their long-term sustainability. The white-headed woodpecker has the most restricted distribution of all group members, and therefore the retention of existing old forests is particularly important within the range of this species where declines in old forests have been most pronounced: watersheds within the Northern Cascades, Northern Glaciated Mountains, Upper Clark Fork, Lower Clark Fork, and Blue Mountains ERUs.

2. (To address issue no. 1) Restore dominance of ponderosa pine to sites where transition to other cover types has occurred.

3. (To address issues no. 1 and no. 2) Accelerate development of late-seral conditions, including snag recruitment, within stands that are currently in mid-seral stages. Areas for emphasis are the same as those listed for strategy no. 1.

4. (To address issue no. 2) Include provisions for snag retention and snag recruitment where needed in all management plans involving forests used as source habitats for group 1.

5. (To address issue no. 3) Reduce risk of stand-replacing fires in late-seral ponderosa pine.

6. (To address issue no. 4) Within all ERUs with cottonwood-willow stands, maintain existing old forests and identify younger stands for eventual development of old-forest structural conditions. Return natural hydrologic regimes to large river systems, particularly in the Central Idaho Mountains, Upper Snake, and Snake Headwaters ERUs where large riparian cottonwood woodlands still remain.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategies nos. 1-4) Use understory thinning and prescribed burns to enhance development of ponderosa pine old forests and to reduce fuel loads. Refer to Blair and others (1995) for specific recommendations about live tree densities for the old-forest structural stage.

2. (In support of strategy no. 4) Retain existing snags, particularly if >53 cm (21 in), and provide measures for snag replacement. Review existing or develop new snag guidelines that reflect local ecological conditions and that address snag numbers, diameter, height, decay class, species, and distribution.

3. (In support of strategy no. 4) Reduce road densities in managed forests where ponderosa pine snags are currently in low abundance. Close roads after timber harvests and other management activities, and minimize the period when such roads are open, to minimize removal of snags along roads. In addition, or as an alternative to road management, actively enforce fuel wood regulations to minimize removal of large snags.

4. (In support of strategy no. 4) Restrict fuel wood permits to disallow snag cutting where ponderosa pine snags are in low abundance, and particularly where existing roads cannot be closed. Blair and others (1995) recommend that public fuel wood harvest should be limited to trees <38 cm (15 in) diameter at breast height (d.b.h.).

Group 2—Lewis' Woodpecker (Migrant Population)

Results

Species ranges, source habitats, and special habitat features—Group 2 consists of populations of Lewis' woodpecker that breed but do not overwinter in the basin. Breeding occurs in portions of all ERUs except the Upper Klamath and Northern Great Basin (fig. 6).

Source habitats of Lewis'woodpecker include old-forest, single-storied structural stages of ponderosa pine and multi-storied stages of Douglas-fir, western larch, and riparian cottonwood woodlands (vol. 3, appendix 1, table 1). Unlike most woodpecker species, the Lewis'woodpecker is an aerial insectivore and requires openings for foraging maneuvers. Their breeding distribution is strongly associated with the distribution of ponderosa pine in western North

Figure 6—Ranges of species in group 2 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

America (see Diem and Zeveloff 1980). This species often is classified as a specialist in burned pine forest habitat, although suitability of burned areas as habitat may differ with postfire age, size and intensity of burn, and geographic region (Block and Brennan 1987, Bock 1970, Linder 1994, Raphael and White 1984, Saab and Dudley 1998). Burned ponderosa pine forests created by stand-replacing fires seem to be highly productive source habitats compared to unburned pine or cottonwood riparian forest (see Tobalske 1997). Burned versus unburned stand condition was not included in the analysis of source habitat extent but is addressed in regards to source habitat quality.

Among nine cavity-nesting species, Lewis' woodpecker was a highly successful nester and the most abundant species nesting in a large (100 000 ha [250,000 acres]), recently burned pine forest in western Idaho (Saab and Dudley 1998). Openings in partially logged, burned forests likely provide greater opportunities for aerial foraging. Within the large burned forests in western Idaho, Lewis'woodpecker nested (1) almost exclusively in salvage-logged units (1.1 nests per km [1.7 per mi] surveyed), compared to unlogged units (0.05 nests per km [0.08 per mi] surveyed); (2) in sites where snags were distributed in clumps; (3) in areas with densities of snags >23 cm (9 in) d.b.h. averaging 59.3 snags per ha (24 snags per acre); and (4) in areas with snag densities for trees >53 cm d.b.h. (21 in) averaging 15.6 snags per ha (6.3 snags per acre) (Saab and Dudley 1998). Nest sites generally are associated with an abundance of flying insects, open-canopy forest or tree clumps, snags, and dense ground cover in the form of shrubs, downed material, and grasses (Bock 1970, Saab and Dudley 1998, Tashiro-Vierling 1994, Tobalske 1997, Vierling 1997). In burned habitats in Wyoming (Linder 1994) and California (Block and Brennan 1987), the percentage of shrub canopy in breeding areas was 13 to 16 percent.

Snags are a special habitat feature for this species (vol. 3, appendix 1, table 2). Lewis'woodpeckers require large snags in an advanced state of decay or trees with soft sapwood for ease of cavity excavation

(Bock 1970, Raphael and White 1984, Saab and Dudley 1995). Additionally, Lewis' woodpeckers usurp occupied cavities (Saab and Dudley 1995), reuse old cavities created by strong excavators (for example, hairy woodpecker, black-backed woodpecker, or northern flicker), or nest in natural cavities of trees (Bock 1970, Saab and Dudley 1995, Tashiro-Vierling 1994, Vierling 1997). Reuse of old nests and excavation of highly decayed wood probably are associated with their weak excavation morphology compared to that of other woodpeckers (see Tobalske 1997). Nest tree species are typically ponderosa pine and cottonwood, and less commonly aspen, lodgepole pine, juniper, willow species, and paper birch (Tobalske 1997). Snags and trees used for nesting are generally larger in diameter and more heavily decayed than that expected based on availability of such snags. In burned ponderosa pine forests of western Idaho, nest trees were large ($\bar{x}$ ± SD = 44.5 ± 1.8 cm d.b.h. [17.5 ± 0.7 in]) and were of heavier decay than were trees measured at random ($n = 206$ nests; Saab and Dudley 1998). In Colorado, cottonwood nest trees had a larger d.b.h. (112.6 ± 38.8 cm [44.3 ± 15.3 in]) than random trees ($n = 47$ nests; Tashiro-Vierling 1994, Vierling 1997). In burned pine-fir forests of the Sierras, nest height averaged 7.3 m (24.0 ft), tree height 11.4 m (37.4 ft), tree d.b.h. 66.5 cm (26.2 in), and tree diameter at cavity 52.2 cm (20.6 in) ($n = 37$ nests; Raphael and White 1984).

Broad-scale changes in source habitats— Historically, the greatest concentrations of Lewis' woodpecker source habitats (excluding burned coniferous forest and riparian habitat that were not considered at the scale of this analysis) were in the Northern Glaciated Mountains, Lower Clark Fork, and Blue Mountains ERUs (fig. 7A). Up to 50 percent of several watersheds within these ERUs are thought to have provided source habitats, whereas lesser amounts of source habitats likely occurred in most watersheds of the Columbia Plateau, Southern Cascades, Upper Clark Fork, Central Idaho Mountains, and Snake Headwaters ERUs (fig. 7A).

The current amount of source habitat is significantly reduced from historical levels in all 11 ERUs that provide source habitat (fig. 7B). The Central Idaho Mountains currently provide the most contiguous habitats, yet these comprise <25 percent of most watersheds (fig. 7B).

Dramatic declines in source habitats seem widespread, based on strong negative trends in 85 percent of the watersheds throughout the basin (figs. 7C and 8). Strong negative trends were particularly evident in the northern watersheds of the basin (Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs), where more than 95 percent of the watersheds experienced declines (fig. 8). Relative change in extent of source habitats for the Lewis'woodpecker was the greatest (that is, most negative) of any species analyzed in this report (vol. 1, table 7).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Declines in areal extent of source habitats were due primarily to a basin-wide alteration of old-forest ponderosa pine to mid-seral structural stages (Hann and others 1997). The current extent of mid-seral dry forest types is nearly twice the historical level (Hann and others 1997). In the northern and central ERUs, less than 10 percent of the historical extent of interior ponderosa pine in the old-forest single-story structural stage remains (vol. 3, appendix 1, table 4). Late-seral western larch also underwent immense declines and is nearly absent at the broad scale in all ERUs in which it historically occurred (vol. 3, appendix 1, table 4).

Within the cottonwood-willow cover type, old forests have strongly declining trends throughout the basin (see vol. 3, appendix 1, table 4) and generally remain only in stands smaller than the 1-km^2 (0.4-mi^2) mapping unit used in this analysis. These losses occurred from changes in historical hydrologic regimes. Flooding by reservoirs eliminated many cottonwood-willow stands, and reservoirs also reduced periodic flooding, a disturbance that is frequently needed for cottonwood seed establishment (Merigliano 1996, Rood and Heinze-Milne 1989). The declines in riparian woodlands, old-forest ponderosa pine, and western larch documented for the basin are part of a larger picture of similar declines throughout the Western United States (Noss and others 1995).

Condition of special habitat features— Abundance of large (>53 cm [21 in]), heavily decayed snags for nesting has been reduced basin-wide because of changes in vegetation structure from old-forest single stratum to mid-seral structures as well as snag removal by woodcutters (Hann and others 1997, Hessburg and others 1999, Quigley and others 1996). Reductions in

Figure 7—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 2 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 8—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 2, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

the amount of old-forest single stratum and stand initiation structures have reduced forest patch openings that allow foraging maneuvers. In the central and southern regions of the basin, increases in closed-canopy, multi-storied forests have reduced understory shrubs and presumably reduced the abundance of associated arthropods on which Lewis'woodpecker feed.

Other factors affecting the group— Road densities have significantly increased throughout the basin (Hann and others 1997), thereby allowing greater human access into forested regions and greater potential for snag removal along roads. Prolonged human presence at or near nest sites may cause abandonment (Bock 1970), although stable populations coexist with park development and heavy tourist use during the breeding season in British Columbia (Siddle and Davidson 1991). Chlorinated hydrocarbons (such as DDT, a pesticide formerly used in fruit orchards and gardens) could have potential negative effects on Lewis'woodpeckers (Tobalske 1997) because they sometimes nest in agricultural settings (Sorensen 1986, Tashiro-Vierling 1994). Elevated energetic costs and stress may be associated with high rates of territorial encounters with European starlings, which could reduce reproductive success even if Lewis'woodpecker dominates the interaction (Siddle and Davidson 1991).

Population status and trends— Breeding Bird Surveys indicate that population trends have been stable within the basin from 1968 to 1994 (Saab and Rich 1997). Saab and Rich (1997), however, included the Lewis' woodpecker as one of 15 Neotropical migrants in the basin that are of high concern to management under all future management themes for the basin, because of the close association of the species with old forest stages of ponderosa pine. Populations may have declined by about 60 percent within the Western United States since the 1960s, on the basis of BBS data (1966 to 1995, -4.0 percent per yr, $n = 61$, $P < 0.01$; Sauer and others 1996). Also, Christmas Bird Counts (CBC) showed a decline in Lewis' woodpecker observations across the entire range of the species, from an average of 10 birds per 1,000 observation hours in 1960 to about four birds per 1,000 observation hours in 1989 ($n = 20$, $P < 0.05$; Tashiro-Vierling 1994).

Trend data generated by the BBS and CBC may not be adequate for monitoring populations of Lewis' woodpecker (Saab and Rich 1997, Tobalske 1997) because of their sporadic distribution (Bock 1970) and relatively uncommon status (DeSante and Pyle 1986). Dramatic cycles of abundance may be related to local changes in habitat (Bock 1970) and to nomadic behavior of Lewis'woodpeckers in search of burned forests for nesting habitat.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 2 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues— The following issues were identified from results of our analysis in combination with relevant vegetation dynamics documented by Hann and others (1997):

1. Declines in shrub understories of montane and lower montane forests.

2. Basin-wide decline in old forests of interior and Pacific ponderosa pine and interior western larch.

3. Basin-wide decline in old forests of cottonwood woodlands.

4. Decline in availability of large snags and trees for foraging and nesting.

5. Potential for negative impacts from agricultural pesticides.

Potential strategies— The issues identified above suggest the following broad-scale strategies for the long-term persistence of Lewis'woodpecker.

1. (To address issue no. 1) Rejuvenate and enhance shrub understory of lower montane community groups (old-forest ponderosa pine) and montane community groups that include interior Douglas-fir and western larch in the Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and Blue Mountains ERUs.

2. (To address issue no. 2) Restore degraded stands and maintain high-quality existing stands of old-forest interior and Pacific ponderosa pine, interior

Douglas-fir, western larch, and cottonwood-willow. Protection and restoration of existing old forests is especially important within the range of this species where declines in old forests have been most pronounced. Areas of emphasis include Blue Mountains, Northern Glaciated Mountains, Upper Clark Fork, Lower Clark Fork, and Central Idaho Mountains ERUs. Within these same ERUs, accelerate development of old forests within stands that are currently mid-seral structural stages.

3. (To address issue no. 3) Within all ERUs with cottonwood-willow stands, maintain existing old forests, and identify younger stands for eventual development of old-forest structural conditions. Return natural hydrologic regimes to large river systems, particularly in the Central Idaho Mountains, Upper Snake, and Snake Headwaters ERUs where large cottonwood riparian woodlands still remain.

4. (To address issue no. 4) Retain all large-diameter (>53 cm d.b.h. [21 in]) ponderosa pine, cottonwood, Douglas-fir, and western larch snags within the basin, preferably in clumps, and provide opportunities for snag recruitment.

5. (To address issue no. 5) Reduce exposure to pesticides during nesting season. Avoid use of toxic chlorinated hydrocarbons and organophosphorus insecticides near Lewis'woodpecker nesting sites.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategies no. 1 and no. 2) Use prescribed burns and understory thinning of small-diameter trees (<25 cm d.b.h. [10 in]) to maintain existing old-forest ponderosa pine stands and to accelerate development of midsuccessional stages to old-forest conditions. These practices also can be used to enhance and develop shrub understories (>13 percent shrub canopy) to attract arthropod prey.

2. (In support of strategies no. 1 and no. 2) Allow stand-replacing wildfires to burn in lower montane wilderness and other lands managed with a reserve emphasis (for example, designated wilderness, research natural areas, and areas of critical environmental concern). Such opportunities can be found particularly in the Central Idaho Mountains, Blue Mountains, and Snake Headwaters ERUs, and in western Montana.

3. (In support of strategy no. 4) Develop measures for snag recruitment in unburned forests. Management for snag recruitment (particularly broken-topped snags) in unburned forests with high risks of stand-replacing fires will provide nest trees during the first few years after wildfire when other trees are not easily excavated.

4. (In support of strategy no. 4) In salvage-logged, postfire ponderosa pine forests, retain snags in clumps rather than evenly spaced, leaving both hard and soft decay classes to lengthen the time that those stands are suitable for nesting by Lewis' woodpeckers. Snag densities should approximate 59 snags per ha (24 snags per acre) of d.b.h. size >23 cm [9 in], and of these, about 15 snags per ha (6 snags per acre) should be large snags (>53 cm d.b.h. [21 in]) (Saab and Dudley 1998).

5. (In support of strategy no. 4) Minimize the density of roads open to motorized vehicles. Close roads after timber harvests and other management activities, and maintain short periods during which such roads are open to minimize removal of snags along roads. In addition or as an alternative to road management, actively enforce fuel wood regulations to minimize removal of large snags.

6. (In support of strategy no. 4) Restrict fuel wood permits to disallow snag cutting where ponderosa pine snags are in low abundance, and particularly where existing roads cannot be closed. Blair and others (1995) recommend for Idaho that public fuel wood harvest should be limited to trees <38 cm (15 in) d.b.h.

7. (In support of strategy no. 5) Avoid use of toxic agricultural insecticides near Lewis'woodpecker nest sites.

Group 3—Western Gray Squirrel

Results

Species ranges, source habitats, and special habitat features—Group 3 is composed of the western gray

VIEW

MAP

Figure 9—Ranges of species in group 3 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

squirrel, a year-round resident of the basin. The western gray squirrel is distributed within the western portion of the basin. Its range includes the Southern Cascades, most of the Northern Cascades and Upper Klamath, and portions of the Northern Glaciated Mountains, Columbia Plateau, and Northern Great Basin ERUs (fig. 9). Currently, however, only small, disjunct areas within this range are occupied by squirrel populations (Ryan and Carey 1995).

Source habitats for the western gray squirrel include interior ponderosa pine and Oregon white oak woodlands (vol. 3, appendix 1, table 1). Structural stages of interior ponderosa pine that provide source habitat are old-forest single-story, old-forest multi-story, and both managed and unmanaged young forest.

Mast-producing trees are an important component of western gray squirrel habitat. Species of mast-producing trees differ throughout the range of the squirrel and include both the native Oregon white oak and introduced English and black walnuts (Barnum 1975). The western gray squirrel uses tree cavities and stick nests as winter dens and for rearing young (Ryan and Carey 1995). The presence of a contiguous tree canopy that allows for arboreal travel around nest sites is also an important habitat feature (ICBEMP 1996c).

Broad-scale changes in source habitats—The trend in broad-scale source habitats for western gray squirrels from historical to current periods was mixed (fig. 10). Moderate or strong decreases were projected in about 30 percent of the watersheds basin-wide, with moderate to strong increases in nearly an equal number (fig. 11). In the Northern Cascades, there were negative and strongly negative trends in about 65 percent of the watersheds (fig. 11). More than half the watersheds in the Northern Great Basin had declining or strongly declining trends. In the Columbia Plateau, there were increasing or strongly increasing trends in about 65 percent of watersheds (fig. 11). Other ERUs either showed mixed trends in source habitats (Southern Cascades, Upper Klamath) or had few watersheds that fell within the range of the squirrel (Blue Mountains, Northern Glaciated Mountains).

Figure 10—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 3 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 11—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 3, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Declines in source habitats in the Northern Cascades were due to large decreases in old-forest single-story, old-forest multistory, and unmanaged young-forest structural stages of interior ponderosa pine (vol. 3, appendix 1, table 4). In the Northern Great Basin, most of the decline resulted from decreases in old-forest single-storied interior ponderosa pine (vol. 3, appendix 1, table 4). Increasing trends in the Columbia Plateau were mostly due to increases in the managed young-forest stage of interior ponderosa pine.

Although oak woodlands were listed as an important source habitat, there was not a measurable vegetation change in this cover type in the ERUs within the range of the species (vol. 3, appendix 1, table 4). In many cases, oak woodlands do not occur in large patches in the basin and may not have been adequately sampled by the 1-km^2 (0.4-mi^2) pixel size used to interpret vegetation.

Condition of special habitats features— Mast-producing trees, such as oak, likely have declined primarily because of increasing human developments (Washington Department of Wildlife 1993c). In roaded areas with a history of timber harvests, densities of large-diameter trees (>53 cm [21 in] d.b.h.) have declined from historical conditions (Hann and others 1997, Hessburg and others 1999, Quigley and others 1996), thus reducing the availability of cavities.

Other factors affecting the group— Introduced eastern fox squirrels and gray squirrels (eastern) are potential competitors in parts of the range of the western gray squirrel (Ryan and Carey 1995). Humans often shoot western gray squirrels both legally and illegally. In Washington, the western gray squirrel is protected from hunting; in Oregon, however, the western gray squirrel is a game species and is regarded as a pest in nut orchards (Ryan and Carey 1995).

Local extirpations caused by mange infestations have seriously affected populations of western gray squirrels. Recovery of populations from disease outbreaks may be difficult when populations are small and widely dispersed (Ryan and Carey 1995).

Population status and trends— Although there is no specific evidence of a reduction in range of western gray squirrels from historical conditions, there is evidence that populations within the range are sparser and more scattered (Washington Department of Wildlife 1993c). This suggests a declining population trend, but there are no direct population data available to confirm the trend.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 3 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Our results, combined with literature and other empirical information, suggest that the following issues are important for the western gray squirrel:

1. Loss of habitat because of increased human development, timber harvest, and other management activities.

2. Loss or decline of oak trees as a cover type and within other cover types.

3. Isolation of squirrel populations because of loss of habitat.

4. Interspecific competition with nonnative squirrels.

5. Direct mortality because of hunting and illegal shooting.

Potential strategies—Issues for the squirrel suggest that the following strategies may help land managers effectively address declines in habitats or populations within the range of the squirrel in the basin:

1. (To address issues no. 1 and no. 2) Across the current range of the squirrel, provide source habitats composed of young- and old-forest interior ponderosa pine stands that include an oak component.

2. (To address issue no. 2) Manage for the maintenance and restoration of oak woodlands.

3. (To address issue no. 3) Provide connectivity among current squirrel populations (Ryan and

Carey 1995) by increasing the areal extent of habitats where these have declined, particularly in watersheds within the Northern Cascades, Southern Cascades, and Upper Klamath ERUs.

4. (To address issues no. 4 and no. 5) Coordinate with other agencies and parties on cooperative efforts to ensure that habitats and populations are maintained.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Where mixed-coniferous/deciduous forest stands have the potential to support a significant oak component, manage them to provide a mixed tree species composition by (1) killing overtopping conifers to allow oaks to grow to an open form; (2) thinning dense pure oak and conifer-oak stands to reduce crowding and water stress and allow remaining oaks to become larger, more vigorous, more productive, and more fire-resistant; (3) removing smaller conifer trees under the oak canopy that are competing with oaks for water and that will eventually overtop the oaks (Ryan and Carey 1995); and (4) retaining old and large conifers within oaks stands where these trees are widely spaced and have an open crown that intercepts little sunlight while providing good year-round shelter for wildlife and their nests (Ryan and Carey 1995).

2. (In support of strategy no. 2) Manage oak woodlands to achieve the following attributes: (1) large, live, open-form oaks; (2) nearby water; (3) adjacent intergrading stands of ponderosa pine; (4) associated deciduous trees and shrubs; (5) a second age class of closed-form oaks to replace aging oaks; (6) natural prairie plant associations to provide an open to patchy understory; and (7) corridors linking habitat fragments (Ryan and Carey 1995). Minimum size of oak stands should be 2 ha (5 acres), with a desired size of 4 ha (10 acres) (Ryan and Carey 1995).

3. (In support of strategies no. 2 and no. 3) Identify and emphasize the location of mature oak stands in relevant management plans, particularly where such stands could potentially link existing populations. Include oak preservation in planning criteria (Ryan and Carey 1995). Increase public awareness of Oregon white oak and western gray squirrels (Ryan and Carey 1995).

4. (In support of strategy no. 4) Improve coordination among state agencies to design hunting seasons to target only areas of crop depredations and to avoid introductions of competitive species.

Group 4—Blue Grouse (Winter)

Results

Species ranges and source habitats—This group consists of winter habitat for blue grouse. Blue grouse are widely distributed across the basin, occurring along the crest of the Cascade Range, in the Blue Mountains, and throughout Idaho and western Montana (fig. 12). Spring and summer habitat for blue grouse occurs at lower elevation than winter habitat, and is discussed in group 17. Specific winter source habitats for blue grouse are old-forest single-story, old-forest multi-story, and understory reinitiation stages of interior Douglas-fir, western larch, Sierra Nevada mixed conifer, Pacific ponderosa pine, and interior ponderosa pine; and mixed-conifer woodlands (vol. 3, appendix 1, table 1).

Broad-scale changes in source habitats—Significant areas of blue grouse winter range occur in 9 of the 13 ERUs (fig. 12). Within the winter range of the blue grouse, there has been an overall decline in its winter habitat with about 70 percent of watersheds showing a moderate or strong decline (figs. 13 and 14). Moderate or strong declines occurred in source habitat in at least 50 percent of watersheds within eight ERUs that included the Northern Cascades, Southern Cascades, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, Snake Headwaters, and Central Idaho Mountains (figs. 13 and 14). Moderate or strong habitat increases were projected in over 50 percent of watersheds only in the Upper Klamath. The Northern Great Basin, Columbia Plateau, Owyhee Uplands, and Upper Snake ERUs contain only small areas of blue grouse winter habitat (fig. 13).

Figure 12—Ranges of species in group 4 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Many of the cover types and structural stage combinations estimated to provide source habitats for wintering blue grouse have decreased in area from historical to current periods (Hann and others 1997; vol. 3, appendix 1, table 4). Interior ponderosa pine old-forest single-story stage was the major contributor to declines in habitat in seven of the eight ERUs, with moderate or strong declines (vol. 3, appendix 1, table 4). Other habitats that declined within these ERUs were interior ponderosa pine understory reinitiation and old-forest multi-storied stages, interior Douglas-fir old-forest single- and multi-storied stages, western larch old-forest multi-storied stage, and mixed-conifer woodland (vol. 3, appendix 1, table 4). In the Upper Klamath, the only ERU for which a moderate or strong increase was projected, the largest increases were projected for interior ponderosa pine old-forest multi-storied stage and interior Douglas-fir old-forest single- and multi-storied stages (vol. 3, appendix 1, table 4).

Other factors affecting the group— Blue grouse are sedentary during winter, moving only 69 m (226 ft) per day on average (Cade and Hoffman 1993, Hines 1986). Their sedentary nature makes them vulnerable to various predators such as lynx, red fox, weasels, American marten, merlin, prairie falcon, northern goshawk, and Cooper's hawk (Zwickel 1992). There are, however, no reports of predation seriously depressing blue grouse populations.

Population status and trends—Although blue grouse still occupy most of their original range (fig. 12), accounts suggest higher historical densities in parts of their range (Zwickel 1992). There are, however, no empirical data on population trend for blue grouse within the basin.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 4 with broader,

Figure 13—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 4 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 14—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 4, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Our analysis indicates winter habitats for blue grouse have declined in the basin; the following issue could be addressed for this species within overall ecosystem-based strategies:

1. Reduction in the amount of montane and lower montane old forests.

Potential strategies—Blue grouse winter habitat could be improved by strategies that focus on the following:

1. (To address issue no. 1) Retain existing interior ponderosa pine, interior Douglas-fir, and western larch old forests, with highest priority for retaining watersheds that still support substantial blue grouse winter habitat within ERUs that have shown large decreases in habitat.

2. (To address issue no. 1) Manage early- and mid-seral montane and lower montane forests to accelerate restoration of late-seral conditions of interior ponderosa pine, interior Douglas-fir, and western larch.

Practices that support strategies— The following practice would be effective in implementing the strategies listed above:

1. (In support of strategies no. 1 and no. 2) Retain remnant, large trees (Pekins and others 1991) in all seral stages of montane forests. In a Colorado study, Cade and Hoffman (1990) found wintering blue grouse in late-seral Douglas-fir stands as small as 1 ha (2.5 acres). Remington and Hoffman (1996) recommended selective logging that would retain clumps of trees of that size.

Group 5—Northern Goshawk (Summer), Flammulated Owl, American Marten, and Fisher

Results

Species ranges, source habitats, and special habitat features—Group 5 consists of the northern goshawk, flammulated owl, American marten, and fisher. Only summer habitat for northern goshawks is included in this group. Goshawk winter habitat is analyzed separately as group 25 because it includes juniper habitats not used by other members of this group. Flammulated owls migrate out of the basin in winter, so only their breeding habitat is represented in this group. Goshawks occur throughout forested areas of the basin (fig. 15). Flammulated owls are broadly distributed throughout the Northern Cascades, Northern Glaciated Mountains, Upper and Lower Clark Fork, Blue Mountains, Central Idaho Mountains, and Upper Klamath ERUs. The range of the American marten includes parts of the western, central eastern, and northeastern portions of the basin (fig. 15). Currently the fisher occurs in the western portion of the basin and in central and northern Idaho and western Montana (fig. 15); historically its range included more areas in the northern, central, and eastern portions of the basin (fig. 15).

Source habitats common to all four species are late-seral stages of the montane community group; unmanaged young forests also are source habitats because this structural stage, like late-seral stages, contains sufficient large-diameter snags and logs needed for various life functions of species in the group (vol. 3, appendix 1, table 1). Managed young-forest stages do not provide source habitat because of the lack of remnant large trees and snags. Source habitats for martens extend up into these same stages of subalpine forests, whereas habitats for goshawks and flammulated owls extend down into the same stages of lower montane forests. For goshawks, flammulated owls, and martens, source habitat also is provided by the old-forest multi-storied and unmanaged young-forest stages of aspen, whereas goshawks, flammulated owls, and fishers find source habitat in these same stages of cottonwood-willow. In addition, flammulated owls use limber pine (McCallum and Gehlbach 1988) and mixed-conifer woodlands as source habitats, and goshawks use chokecherry-serviceberry-rose as source habitats.

Goshawks nest in various forest structural conditions, from open, parklike stands of aspen (Younk and Bechard 1994) to multi-storied old forests (Reynolds 1983). Nest stands are generally characterized by large trees and the densest canopy cover available within the area (Reynolds and others 1992) but are occasionally located in small-diameter trees (Hayward and Escano 1989, Squires and Ruggiero 1996). Foraging occurs in various cover types and structural stages, and the juxtaposition of several habitats may

Figure 15—Ranges of species in group 5 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 15—Ranges of species in group 5 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

enhance the quality of foraging habitat around nest sites (Hargis and others 1994). Home range for a nesting pair is estimated at >2400 ha (5,930 acres) (Hargis and others 1994, Kennedy and others 1994, Reynolds and others 1992).

Martens seem more sensitive to patch size than are other group members and usually avoid clearcuts dominated by grasses, forbs, and saplings, especially in winter. These areas do not provide access to the subnivean zone or offer protection from predation, and they have more severe microclimatic conditions than areas with forest cover (Buskirk and Powell 1994). At the broad scale, the presence of multiple clearcuts may render the entire landscape unsuitable. In Utah, martens were rarely found in areas with >25 percent of the landscape in a combination of natural openings and clearcuts (Hargis 1996). In Maine, no adult female territories were found in landscapes with >31 percent of mature forest cover removed (Chapin 1995).

Although fishers will cross openings to access forested areas (Arthur and others 1989), a negative association with clearcuts has been documented. Fisher occurrence in California was positively associated with large stands of mature forest and distance from clearcuts (Rosenburg and Raphael 1986); fishers in Idaho avoided stands with <40 percent canopy cover (Jones 1991, Jones and Garton 1994).

Old forests consisting of ponderosa pine and Douglas-fir seem to be a key component of flammulated owl home ranges (Reynolds and Linkhart 1992). Home ranges composed of at least 75 percent old ponderosa pine/Douglas-fir forest were occupied more continuously than home ranges consisting of less than 75 percent in this forest type (Reynolds and Linkhart 1990). Variability in the structure of these old stands seems important to support life functions of flammulated owls. Roosting occurs in fairly dense stands. Goggans (1986) showed that tree densities immediately surrounding roost trees average 2016 per ha (816 per acre), whereas overall home ranges average 589 trees per ha (238 per acre). In contrast, relatively open stands seem to be selected for foraging (Linkhart 1984), and open, mature stands are selected for nest sites (McCallum 1994). In two Oregon studies, mean d.b.h. of nest trees was 56.3 cm (22.2 in) (Goggans 1986) and 72.0 cm (28.4 in) (Bull and others 1990).

Several special habitat features have been identified for this group (see vol. 3, appendix 1, table 2). Fishers and American martens use down logs. Downed woody material is likely the key component of foraging areas for marten (Coffin and others 1997), providing habitat for many of their prey, particularly southern red-backed voles, and subnivean access to prey during winter (Corn and Raphael 1992). Fishers and martens depend on down logs for resting and denning (Buskirk and Powell 1994, Raphael and Jones 1997). Snags are a special habitat feature for flammulated owls, fishers, and martens. Flammulated owls nest in cavities in both snags and large live trees (Bull and others 1990, McCallum and Gehlbach 1988). Snags provide rest sites and den sites for fishers and martens.

Broad-scale changes in source habitats— Historically, source habitats likely occurred throughout the forested portions of the basin, with some of the greatest concentrations in the western, central, and northern portions of the basin (fig. 16A). Currently, the largest extent of source habitats is in the south-central and southwestern portions of the basin (fig. 16B). The primary change from historical to current times has been a broad shift in the geographic distribution of source habitats away from the north and towards the southwestern portion of the basin (fig. 16C).

Basin-wide, there were moderately or strongly declining habitat trends in nearly 70 percent of watersheds within the range of species in group 5, and neutral or increasing trends in about 30 percent of watersheds (fig. 17).

Positive changes in source habitat occurred in more than 50 percent of watersheds in the Upper Klamath and Northern Great Basin ERUs; mixed trends in the Southern Cascades and Upper Snake ERUs; and negative trends in more than 50 percent of watersheds in all remaining ERUs (figs. 16 and 17). The most strongly negative trends were projected across the northern portion of the basin in the Northern Cascades, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs (figs. 16 and 17).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Interior ponderosa pine old-forest single-story stage declined in all but one of the ERUs in which source habitat declined in more than 50 percent of watersheds (vol. 3, appendix 1, table 4). Interior ponderosa pine old-forest multi-story stage declined in nearly half of these ERUs. Less consistent declines were projected for the old-forest single-story stage of interior Douglas-fir; the old-forest multi-story stages of interior Douglas-fir, lodgepole pine, grand fir-white fir, Engelmann spruce-subalpine fir, western larch, and western white pine; the unmanaged young forest stages of whitebark pine, Engelmann spruce-subalpine fir, western larch, and lodgepole pine; and mixed-conifer woodland (vol. 3, appendix 1, table 4). In the ERUs with the most strongly negative trends, the Northern Cascades, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork, negative trends were projected for up to nine of these habitat types (vol. 3, appendix 1, table 4). In the Upper Klamath, the only ERU with a significant amount of source habitat for the group and a positive trend in more than 50 percent of watersheds, the increasing trend was associated with increases in the old-forest multi-story stages of interior ponderosa pine, interior Douglas-fir, lodgepole pine, and grand fir-white fir; and the old-forest single-story stage of interior Douglas-fir. In addition, riparian woodland (including aspen and cottonwood-willow) declined basin-wide, and also underwent a shift from early- and late-seral stages to mid-seral stages (Hann and others 1997).

Condition of special habitat features— Densities of large-diameter snags (>53 cm [21 in] d.b.h.) declined basin-wide from historical to current levels (Hann and others 1997, Hessburg and others 1999, Quigley and others 1996). Trends in snag abundance ultimately affect the availability of large down logs and cavities.

Other factors affecting the group— Populations of martens and fishers can be impacted by fur harvesting if trapping is not carefully regulated (Fortin and Cantin 1994, Jones 1991, Quick 1956). Trapping also affects populations by altering the sex and age structure through the disproportionate capture of juveniles and males (Hodgman and others 1994, Quick 1956). Historically, both martens and fishers were heavily

Figure 16—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 5 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 17—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 5, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of >20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

trapped in the basin. Currently, martens are still trapped in all states in the basin, but fishers are only trapped in Montana (Heinemeyer 1995).

Secondary roads in forested areas increase trapping pressures for martens and fishers, resulting in significantly higher captures in roaded versus unroaded areas (Hodgman and others 1994) and in logged versus unlogged areas, in which the difference was again attributed to higher road densities in logged stands (Thompson 1994). Secondary roads also might increase the likelihood that snags and logs will be removed for fuel wood. This could impact fishers, martens and flammulated owls, and also could have a negative effect on the prey base for goshawks (Reynolds and others 1992).

Studies have shown that fisher, marten, and goshawk populations respond to food limitation. Fisher populations can undergo fluctuations related to prey abundance (Powell and Zielinski 1994). Marten populations also have been observed to decline after a decline in principal prey species (Thompson and Colgan 1987, Weckwerth and Hawley 1962). Some of the decline is due to lower reproductive rates in females, but evidence of starvation also has been observed (Hodgman and others 1994, Weckwerth and Hawley 1962). Several studies suggest that goshawk populations are frequently food-limited. In Alaska and the Yukon where snowshoe hare is a dominant prey item, goshawk numbers fluctuate with snowshoe hare cycles (Doyle and Smith 1994). A review of several studies by Widen (1989) suggests correlations between goshawk numbers and other prey. Maj and others (1995) suggest that heavy levels of grazing in ponderosa pine communities may degrade insect habitat and reduce prey populations for flammulated owls.

Changes in forest structure related to fire suppression seem to increase the extent of some of the cover types and structural stages judged to be source habitats for goshawks. However, such stands, which are characterized by closed canopies and dense conifer understory, may not be as valuable to goshawks as the more open habitats, which they replaced. A high density of small-diameter understory trees may be detrimental to foraging and nesting aspects of goshawk ecology in at least three ways: (1) by obstructing flight corridors used by goshawks to obtain forest-associated prey; (2) by suppressing tree growth needed to produce large-diameter trees for nest sites; and (3) by reducing the growth of an herbaceous understory that supports potential prey species (Reynolds and others 1992). Therefore, although fire suppression may have increased the extent of multi-storied closed forests within the basin, the inherent value of these stands may be less than that of more open stands maintained by fire. This supposition warrants further investigation.

Conversely, the harvest of large-diameter overstory trees can create forest structures that are more open than normally used by goshawks. A secondary effect is increased competition with raptors adapted to more open habitats (Moore and Henny 1983). Goshawk nest sites are more frequently used by red-tailed hawks, great horned owls, or long-eared owls in harvested areas than in unharvested sites (Crocker-Bedford 1990, Patla 1990).

Flammulated owls are Neotropical migrants, so their population status may be affected by conditions of their winter habitat. Their winter range is suspected to be in southern Mexico and northern Central America (McCallum 1994).

Population status and trends—Fishers may be close to extirpation in Washington (Aubry and Houston 1992, cited in Powell and Zielinski 1994), and sightings are rare in Oregon. The last reliable reports of native fishers in Idaho and Montana were during the 1920s (Dodge 1977, Weckwerth and Wright 1968, cited in Powell and Zielinski 1994). Fisher populations were reintroduced to Idaho in the 1960s and to Montana in the 1950s and 1980s (Powell and Zielinski 1994). Projected declines in source habitats may have contributed to historical extirpations, coupled with the effects of trapping and the fragmented nature of remaining habitats.

The distribution of marten within the basin has been fairly stable since historical times, but population changes are not known, other than through trapping records, which fluctuate widely with fur prices and may not reflect actual population trends.

The BBS data for the goshawk were insufficient to determine population trends for the basin (Saab and Rich 1997) or for any state or physiographic region within the basin (Sauer and others 1996) because of low detection of goshawks under the BBS survey method. Sufficient data were available, however, for western North America to indicate a stable trend in numbers between the years 1966 and 1995 (Sauer and others 1996).

A separate trend estimate was derived from fall migration counts conducted by Hawkwatch International at four locations in Utah and New Mexico. These data indicated an average rate of decline in migrating goshawks of about 4 percent annually between 1977 and 1991 (Hoffman and others 1992). The extent to which the migration data represented local declines near the survey stations was not determined.

No population trend data were found for flammulated owls. The BBS survey method is not adequate for surveying flammulated owls because of low numbers and nocturnal behavior. Specialized monitoring would be required to determine the population trend of owls (Saab and Rich 1997).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 5 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The following issues were identified from the results of our analysis and published research:

1. Reduction in the amount of old-forests and associated structures (snags, logs, and cavities), particularly within the montane and lower montane community groups.
2. Fragmentation of habitat.
3. Low population numbers of fisher.
4. Negative effects resulting from higher road densities in source habitats. For marten throughout the basin and fishers in Montana, there is increased trapping pressure associated with roads. For all species in the group, loss of snags and logs associated with firewood collection may be higher along open roads.
5. Declines in overall extent of aspen and cottonwood-willow, and shifts from early- and late-seral to mid-seral stages of these cover types (vol. 3, appendix 1, table 4).
6. Possibly unsustainable conditions of old forests where there have been large transitions from shade-intolerant to shade-tolerant tree species. This last issue stems from the exclusion of fire from many forested communities, which has resulted in increased susceptibility to stand-replacing fires (USDAForest Service 1996).
7. Decline in suitable foraging areas around goshawk nest sites. On Federal lands, the immediate areas around active nests generally are protected from timber harvests, but the larger foraging areas surrounding nests frequently are managed without explicit consideration of goshawk foraging. Goshawks typically use a nest stand and nearby alternative nest stands for many years, and therefore, the long-term maintenance of suitable foraging areas is as important for successful reproduction as protection of the immediate nest stand.

Potential strategies—The following strategies could be used to reverse broad-scale declines in source habitats and populations:

1. (To address issue no. 1) Increase the representation of late-seral forests in all cover types used as source habitats, particularly in the northern half of the basin (Northern Cascades, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs).
2. (To address issue no. 2) Increase connectivity of disjunct habitat patches and prevent further reduction of large blocks of contiguous habitat.
3. (To address issues no. 3 and no. 4) Identify potential species strongholds for long-term management of marten and fisher (see practice no. 6 for criteria).
4. (To address issue no. 4) Reduce human disturbances in source habitats.
5. (To address issue no. 5) Restore aspen and cottonwood-willow forests, particularly the unmanaged young-forest and late-seral stages.

6. (To address issue no. 6) Reduce the risk of loss of habitat by focusing old-forest retention and restoration efforts on areas where fire regimes are either nonlethal or mixed (USDA Forest Service 1996). In ERUs where old-forest habitat has remained stable or increased from historical conditions, efforts could be focused on retaining existing habitat in areas with lower fire and insect risk while managing other areas to reduce risks of catastrophic loss of habitat.

7. (To address issue no. 7) Maintain stands with active goshawk nests in old-forest condition.

8. (To address issue no. 7) Embed the conservation of old forests within a larger, ecosystem context that considers historical fire regimes and landscape patterns and the habitat needs of species that are prey of the members of this group. For goshawks, Reynolds and others (1992) gave specific recommendations for promoting various cover types and structural stages in 2430 ha (6,005 acres) of potential home range around each active nest.

Practices that support strategies— The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) In the northern basin, identify representative stands of old forests for retention and mid-successional stages for development into old-forest conditions. Priority should be given to large blocks having high interior-to-edge ratios and few large openings.

2. (In support of strategy no. 1) Actively recruit snags and logs from green trees to increase the representation of old-forest structures (snags and logs) in mid-seral stands and in old forests where snags and logs are in low density or absent.

3. (In support of strategy no. 1) Retain slash piles and decks of cull logs to substitute for down logs over the short term. Raphael and Jones (1997) recommend retaining a minimum of 1.3 slash piles per ha (0.5 per acre) on a site that has been extensively harvested.

4. (In support of strategy no. 2) Where possible, use selection harvest rather than clearcutting. If clearcuts are used, aggregate cuts so that large blocks of unharvested forest are retained.

5. (In support of strategy no. 2) Adjust activities, including timber harvests, to provide links among currently isolated patches of source habitats.

6. (In support of strategy no. 3) Identify existing areas with the following desired conditions, or manage selected areas to create the following desired conditions for strongholds: existing populations of marten or fisher, or both; large, contiguous blocks of forest cover with a high percentage of late-seral stages, abundant snags and large logs, low road densities and overall low human disturbance, and potential connectivity to currently unoccupied source habitats.

7. (In support of strategy no. 4) Minimize new construction of secondary roads and close unneeded roads after timber harvest.

8. (In support of strategy no. 5) Use clearcutting to regenerate aspen. Where aspen regeneration is inhibited by domestic or wild ungulate browsing, use exclosures to protect regenerating stands or modify management to reduce browsing pressure.

9. (In support of strategy no. 5) Survey and map existing old forests of cottonwoods and reference their locations in land management planning documents. Monitor conditions of cottonwood stands to ensure that sufficient seedling or vegetative regeneration, or both, is occurring. Identify factors limiting regeneration so that appropriate corrective measures can be taken. For example, return natural hydrologic regimes to portions of large river systems that support cottonwood riparian woodlands.

10. (In support of strategy no. 6) Manage risks of catastrophic loss by using prescribed fire and thinning to reduce fuel loading and to encourage the development of forest openings, shrub openings, and shade-intolerant and fire-, insect-, and disease-resistant tree species.

11. (In support of strategy no. 7) Identify an area around each active goshawk nest site to be maintained in old-forest condition, and identify possible replacement stands. The Northern Goshawk Scientific Committee for the FS recommends three 12-ha (30-acre) nest stands per breeding pair and

three additional 12-ha (30-acre) replacement stands be located within a 2430-ha (6,000-acre) area that functions as a potential home range (Reynolds and others 1992).

12.(In support of strategies no. 6 and no. 8) Use silvicultural prescriptions in conjunction with restoration of fire regimes to create a desired mix of cover types and structural stages within the potential home range of each active goshawk nest. The Northern Goshawk Scientific Committee for the FS (Reynolds and others 1992) has identified two larger habitat use areas that extend beyond the nest site: a postfledgling-family area, encompassing about 170 ha (420 acres) around the nest and used by a nesting pair and offspring from the time the young leave the nest until they are independent, and a foraging area of about 2190 ha (5,411 acres) that provides the food resource during and after the breeding period (Reynolds and others 1992). For forests in the Southwestern United States, they recommended that four-fifths of each postfledgling family area and each foraging area be equally divided among four seral stages: young, mid aged, mature, and old forests, and the remaining one-fifth be equally divided between the seedling-sapling stage and grass-forb stage. These recommendations should be reviewed in light of different ecological conditions within the basin.

Group 6—Vaux's Swift, Williamson's Sapsucker, Pileated Woodpecker, Hammond's Flycatcher, Chestnut-Backed Chickadee, Brown Creeper, Winter Wren, Golden-Crowned Kinglet, Varied Thrush, Silver-Haired Bat, and Hoary Bat

Results

Species ranges, source habitats, and special habitat features—Group 6 consists of migratory breeding habitat for brown creepers, Hammond's flycatchers, Vaux's swifts, and Williamson's sapsuckers; resident summer habitat for varied thrushes, winter wrens, silver-haired bats, and hoary bats; and year-round habitat for chestnut-backed chickadees, golden-crowned kinglets, and pileated woodpeckers. Ranges within the basin for the 11 species in this group (fig. 18) tend to fit one of four broad patterns. Silver-haired bats and hoary bats occur throughout the basin in forested areas or woodlands. Brown creepers, Hammond's flycatchers, winter wrens, and golden-crowned kinglets generally occur throughout the forested areas of the basin. The range of Williamson's sapsucker differs from these four species as it does not extend all the way to the crest of the Cascade Range or to the southern extremes of the Central Idaho Mountains or Upper Klamath ERUs. Pileated woodpeckers, varied thrushes, chestnut-backed chickadees, and Vaux's swifts are distributed across forested areas in the western half of the basin, but their ranges do not extend to the southeastern portion of the Central Idaho Mountains below the Salmon River, or into the Snake Headwaters or Upper Snake ERUs.

Source habitats for the 11 species in group 6 are generally late-seral stages of the subalpine, montane, lower montane, and riparian woodland community groups (vol. 3, appendix 1, table 1). Source habitats shared in common by more than one-half of the species are the old-forest single- and multi-strata stages of grand fir-white fir, interior Douglas-fir, western larch, western white pine, western redcedar-western hemlock, Sierra Nevada mixed conifer, and mountain hemlock; and the old-forest multi-strata stage of Engelmann spruce-subalpine fir, Pacific silver fir-mountain hemlock, and red fir (vol. 3, appendix 1, table 1). Source habitats used by less than one-half the species include old-forest Pacific and interior ponderosa pine (used by brown creepers, Hammond's flycatchers, Williamson's sapsuckers, hoary bats, and silver-haired bats); old-forest whitebark pine and alpine larch (used by golden-crowned kinglets); old-forest lodgepole pine (used by golden-crowned kinglets, Hammond's flycatchers, hoary bats, and silver-haired bats); old-forest aspen (used by Williamson's sapsuckers, chestnut-backed chickadees, Hammond's flycatchers, hoary bats, and silver-haired bats); and old-forest cottonwood-willow (used by Williamson's sapsuckers, hoary bats, and silver-haired bats) (vol. 3, appendix 1, table 1). Hoary bats also use the stand initiation stage of all montane and lower montane forest types and of aspen and cottonwood-willow for foraging (vol. 3, appendix 1, table 1).

Figure 18—Ranges of species in group 6 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 18–Ranges of species in group 6 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 18—Ranges of species in group 6 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Several special habitat features exist for species in this group (vol. 3, appendix 1, table 2). Six of the bird species (brown creepers, chestnut-backed chickadees, pileated woodpeckers, Vaux's swifts, Williamson's sapsuckers, and winter wrens) depend on snags for nesting or roosting, or both (Bull and Hohmann 1993; Bull and others 1986a, 1992; Raphael and White 1984). Brown creepers, pileated woodpeckers, Vaux's swifts, and Williamson's sapsuckers use large (>53 cm [21 in] d.b.h.) snags (Bull and others 1986a, 1992; Bull and Hohmann 1993, Raphael and White 1984). Winter wrens and chestnut-backed chickadees use smaller diameter snags (Thomas and others 1979). Pileated woodpeckers forage on large snags and logs (Bull and Holthausen 1993, Mannan 1984), and winter wrens forage around and under logs (Van Horne and Bader 1990). Pileated woodpeckers and Vaux's swifts depend on large, hollow live or dead trees for roosting (Bull 1991, Bull and others 1992).

Special habitat features for both bat species include shrub/herbaceous wetland/riparian areas (vol. 3, appendix 1, table 2). Both species use contrasting habitats—forested areas for roosting and open areas for foraging. Snags are a special habitat feature for silver-haired bats. They roost in trees, snags, mines, caves, crevices, and buildings (Christy and West 1993). Day roost trees are usually characterized by being large (>53 cm [21 in] d.b.h.), dead or live with some defect, with loose bark and cracks. In an Oregon study, Betts (1996) found silver-haired bats roosting in live western larch and ponderosa pine, and in grand fir and ponderosa pine snags. The average diameter of these roost trees was 59.6 cm (23.5 in), and they were generally located on relatively densely forested slopes. The hoary bat is an edge-associated species, often roosting in deciduous trees or conifers at the edge of clearings (Perkins and Cross 1988, Shump and Shump 1982). Hoary bats are foliage roosters, with males, nonbreeding females, and breeding females located in different levels in the canopy (Christy and West 1993).

Broad-scale changes in source habitats— Source habitats for species in this group occur in all 13 ERUs (fig. 19), but amounts of habitat are relatively small in the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs. Basin-wide, source habitats for species in this group have declined moderately or strongly in more than 50 percent of watersheds containing appropriate habitat types (fig. 20). The pattern of habitat change, however, was highly variable across the basin with the northern part of the basin marked by generally strong declines and the southern part by strong increases (fig. 19). Moderate or strong declines in habitat from historical to current were projected in more than 50 percent of the watersheds in six ERUs: the Northern Cascades, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, Upper Snake, and Snake Headwaters (fig. 20). The declines were particularly strong across the northern basin in the Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs. Moderate or strong increases were projected in more than 50 percent of watersheds in the Southern Cascades, Upper Klamath, Northern Great Basin, and Columbia Plateau (fig. 20). More balanced mixes of increases and decreases were projected for the remaining three ERUs: Blue Mountains, Owyhee Uplands, and Central Idaho Mountains (fig. 20).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— The projected decline in source habitats reflected basin-wide declines in late-seral forest conditions (USDAForest Service 1996). Changes in late-seral forests, however, have differed among ERUs (tables 3.141 to 3.165 in Hann and others 1997). Late-seral lower montane multi-layer forests and late-seral subalpine multi-layer forests declined significantly in all six ERUs in which source habitats declined in more than 50 percent of watersheds; late-seral montane multi-layer forests declined in five of them; and late-seral lower montane single-layer forests declined in four of them (Hann and others 1997).

Late-seral montane multi-layer and single-layer forests each increased significantly in three of the four ERUs (Southern Cascades, Upper Klamath, Northern Great Basin, and Columbia Plateau) in which source habitats increased in more than 50 percent of watersheds. Much of this change was due to shifts from shade-intolerant, late-seral lower montane forest types to shade-tolerant, late-seral montane forest types. The increase in the fourth ERU, the Columbia Plateau, appears to be somewhat anomalous. It was likely the result of a moderate increase in the open canopy stem-exclusion stage of interior ponderosa pine (vol. 3, appendix 1, table 4), which serves as source habitat only for hoary bats (primarily foraging habitat).

VIEW

MAP

Figure 19—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 6 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 20—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 6, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Condition of special habitat features— Snags are a special habitat feature for seven of the species in this group, and large hollow trees for two species. Densities of large-diameter (>53 cm [21 in] d.b.h.) snags likely have declined basin-wide from historical to current levels (Hann and others 1997, Hessburg and others 1999, Quigley and others 1996). Historical to current trends in smaller diameter snags were variable, with no clear basin-wide trend emerging (Hann and others 1997).

The number of caves available for roosts across the basin likely has stayed the same, and mines may now provide additional roost or hibernacula areas. Cave and mine suitability, however, can be affected by recreational use, such as cave exploration, which increases with higher road densities near caves. Historical road densities were lower than current densities. Road densities are high in intensively managed forest lands of both public and private ownership, and the highest densities typically occur in developed urban-rural areas (USDA Forest Service 1996, p. 85).

Across the basin, there were widespread declines in shrublands in riparian zones (USDA Forest Service 1996, p. 101). Forest conversion and streamside disturbances have degraded and fragmented riparian vegetation. This may have negatively impacted the shrub/herbaceous wetland/riparian foraging areas for the hoary and silver-haired bats.

Other factors affecting the group— Four of the species in this group (brown creepers, Hammond's flycatchers, Vaux's swifts, and Williamson's sapsuckers) are Neotropical migrants and may be affected by habitat conditions on their wintering grounds. The bat species also are thought to winter outside the basin, although exact migration routes and winter ranges are not clear (Christy and West 1993).

Hoary bats eat moths, beetles, and mosquitos (Barclay 1985, 1986; Rolseth and others 1994; Shump and Shump 1982; Whitaker and others 1977). The silver-haired bat is an opportunistic feeder and eats moths, flies, beetles, and various other insects (Whitaker and others 1981). Management activities such as the use of pesticides that cause declines of insect species may negatively affect these bats. Also, direct contact with pesticides can cause illness or death in bats. Although most organochlorine pesticides that cause accumulation of chemicals up the food chain have been banned or highly restricted in the United States, the relatively short-lived organophospates can provide high risks during application (Clark 1988). For example, a large die-off of bats observed in Arizona after the application of methyl parathion, was believed to be linked to direct contact with the chemical (Clark 1988).

Grazing can have an adverse impact on the insect prey of bats (Clark 1988, Nagorsen and Brigham 1993, Perlmeter 1995, Ports and Bradley 1996). Roads also may facilitate harvest of snags for firewood and so may indirectly affect habitat for the species that use snags.

Population status and trends— Saab and Rich (1997) reported stable population trends, based on data from BBS routes within the basin, for Williamson's sapsuckers, Vaux's swifts, Hammond's flycatchers, brown creepers, and golden-crowned kinglets. Breeding Bird Survey data analyzed within other geographic boundaries (Sauer and others 1996), however, indicate a significant decline from 1966 to 1994 for brown creepers in eastern Oregon and Washington (-7.4 percent per year, $n = 15$, $P < 0.01$). Breeding Bird Survey data also indicate a significant increase in pileated woodpeckers in northwestern Montana (6.1 percent per year, $n = 41$, $P < 0.01$, 1966 to 1994; Sauer and others 1996) but a significant decrease in eastern Oregon and Washington (-7.8 percent per year, $n = 8$, $P < 0.05$, 1966 to 1979; Sauer and others 1996). A significant increase is shown for winter wrens in eastern Oregon and Washington (7.8 percent per year, $n = 9$, $P < 0.05$, 1966 to 1979). Population data are not available for the bat species.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 6 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues— The following issues were identified from our analysis of source habitat trends:

1. Reductions in the extent of late-seral lower montane, montane, and subalpine forest (Hann and others 1997), particularly in the Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs.

2. Reductions in large snags and logs in landscapes that have been managed under traditional silvicultural practices (Hann and others 1997).

3. Possibly unsustainable conditions in late-seral stage montane forests where there have been large transitions from shade-intolerant to shade-tolerant species.

4. Degradation and loss of riparian habitat.

5. Abandonment of bat roosts because of human disturbance.

6. Reductions in the insect prey base for bats because of both land management activities and the use of pesticides.

7. Negative effects of pesticide and insecticide spraying.

Potential strategies—The following strategies would benefit species in group 6:

1. (To address issues no. 1 and no. 2) Accelerate development of late-seral conditions in lower montane, montane, and subalpine forest types and retain large snags and logs in all forest seral stages. Habitat restoration efforts would be most beneficial if concentrated in the northern portions of the basin.

2. (To address issues nos. 1-3) In the southern portion of the basin, retain sufficient habitat to support species in this group while restoring forest conditions that are more resistant to catastrophic fire, insect, and disease problems. This could require management activities, including prescribed fire, that reduce the dominance of shade-tolerant tree species and increase the presence of shade-intolerant species (i.e., those most resistant to catastrophic fire and insect and disease problems).

3. (To address issue no. 4) Across the basin, maintain or improve riparian shrubland and riparian woodland communities.

4. (To address issues no. 2 and no. 5) Protect known and potential bat roosts across the basin. Specifically, maintain caves, mines, snags, and other such features for use as roosting areas and potential nurseries across the basin. Minimize human disturbance in these areas.

5. (To address issues no. 6 and no. 7) Minimize direct physiological effects on bats, as well as indirect effects on their insect prey, stemming from use of insecticides and pesticides.

6. (To address issues no. 6 and no. 7) Modify management practices as appropriate to enhance the insect prey base for bats.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Various silvicultural practices including thinning, burning, and uneven-age management could be used to help accelerate the development of old-forest conditions.

2. (In support of strategies no. 1 and no. 2) Both the retention and creation of snags are important for retention and development of old-forest characteristics. Techniques for snag management are well studied (Bull and others 1980, Bull and Partridge 1986) and have been extensively applied on National Forests (Bull and others 1986b). Retain existing snags, particularly if >53 cm (21 in), and provide measures for snag replacement. Review existing snag guidelines or develop guidelines that reflect local ecological conditions and address snag numbers, diameter, height, decay class, species, and distribution. Consider closing roads in areas that are deficient of snags and where cutting of snags or remnant trees for firewood contributes to the low snag densities. In addition, or as an alternative to road management, actively enforce fuel wood regulations to minimize removal of large snags.

3. (In support of strategy no. 2) To continue meeting habitat needs of species in this group, habitat retention efforts should be designed to maintain an appropriate network of old-forest habitats. Bull and Holthausen (1993) suggested managing areas of 1000 ha (2,471 acres) to meet needs of multiple

Figure 21—Ranges of species in group 7 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

pairs of pileated woodpeckers. Features of these areas were a substantial old forest and unlogged component, at least 8 snags per ha (3 snags per acre) with at least 20 percent of these >51 cm (20 in) d.b.h., and at least 100 logs per ha (40 logs per acre) with a preference for logs 38 cm (15 in) in diameter and larger. Such strategies could be coordinated with needs for ecosystem health by focusing old-forest retention areas in geographic locations where fire, insect, and disease risks are lowest.

4. (In support of strategy no. 3) Maintain or restore riparian vegetation around permanent and seasonal water sources.

5. (In support of strategy no. 4) Protect building roost sites. If possible, stabilize old structures that are important roosts.

6. (In support of strategy no. 6) Modify grazing practices to improve condition of degraded riparian areas for bat foraging.

Group 7—Boreal Owl

Results

Species ranges, source habitats, and special habitat features—Group 7 consists of the boreal owl. Within the basin, this species occurs in forested portions of eastern Washington, northern and central Idaho, western Montana, and the Blue Mountains and Cascade Range of Oregon (fig. 21). The boreal owl is a year-round resident of the basin.

Source habitats for boreal owls include old-forest and unmanaged young-forest stages of subalpine and montane forests and riparian woodlands (vol. 3, appendix 1, table 1). Specific cover types and structural stages that provide source habitat are the old-forest multi-story stages of Engelmann spruce-subalpine fir, Pacific silver fir-mountain hemlock, and aspen; and the old forest single- and multi-forest stages of interior Douglas-fir, western larch, and lodgepole pine. Unmanaged young-forest stages of all these cover types and of grand fir-white fir also serve as source habitats if suitable large-diameter snags are present. Source habitats typically support abundant lichens and fungal sporocarps, which provide important foods for southern red-backed voles, the principal prey of boreal owls (Hayward 1994c). These lichens and fungi are associated with coarse woody debris.

Boreal owls require snags or large trees with either natural cavities or cavities excavated by other species (vol. 3, appendix 1, table 2). Cavities excavated by pileated woodpeckers and northern flickers are the most common nest sites (Hayward 1994c). Tree and snag diameters used for nesting are generally large. For example, in Idaho, diameters of nest trees ranged from 26 to 61 cm (10 to 24 in) with an average of 41 cm (16 in). Of 19 nests, 10 were in snags whereas the remainder were in live trees (Hayward and others 1993).

At the home range scale, boreal owls are adapted to patchy landscapes and use several cover types and structural stages to meet different life history requirements (Hayward and others 1993). Landscapes that contain various old-forest cover types may support the greatest abundance of boreals (Hayward and others 1993). In portions of their range, boreal owls may occur in a patchy geographic pattern resulting in a metapopulation structure, with the long-term persistence of each population determined in part by its relation to other populations (Hayward 1994a).

Broad-scale changes in source habitats— Historically, the most concentrated areas of source habitat for boreal owls were in the Northern Cascades, Northern Glaciated Mountains, and Snake Headwaters ERUs (fig. 22A). Other ERUs that historically supported significant source habitat were the Southern Cascades, Lower Clark Fork, Upper Clark Fork, and Central Idaho Mountains ERUs.

Overall, source habitats were projected to have declined moderately or strongly in nearly 80 percent of the watersheds in the basin (fig. 23). Moderate or strong declines were projected for over 50 percent of watersheds in the Northern Cascades, Northern Glaciated Mountains, Upper Clark Fork, Lower Clark Fork, Snake Headwaters, and Central Idaho Mountains ERUs (fig. 23). Moderate or strong declines in over 50 percent of watersheds also were projected for the Columbia Plateau and Upper Snake, but these ERUs are peripheral to the range of boreal owls. Source habitats were projected to have increased moderately or strongly in over 50 percent of watersheds in the Southern Cascades, and there was a mixed pattern of change in the Blue Mountains ERU (fig. 23).

These trends have resulted in a broad shift in the geographic distribution of source habitats away from the northern ERUs and towards the central portions of the basin. Habitat losses have outweighed the gains, and current habitat distribution is substantially more disjunct than historically in the northern part of the basin (fig. 22).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Across the northern portion of the basin, the trend in forest structure has been an increase in mid-seral stages at the expense of both early- and late-seral stages (Hann and others 1997). Ecologically significant declines (Hann and others 1997) were projected for late-seral montane multi-story and single-story forests for the Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs. Late-seral subalpine multi-story forests also were projected to have declined significantly in two of these ERUs (Hann and others 1997). Specific habitat types for which there was greatest decline in areal extent within the three northern ERUs were western larch, interior Douglas-fir, and Engelmann spruce-subalpine fir old forests (vol. 3, appendix 1, table 4).

In the Southern Cascades, the source habitats that increased most strongly were single-storied old-forest Douglas-fir and multi-storied old-forest lodgepole pine (vol. 3, appendix 1, table 4). Increases in source habitats in portions of the Blue Mountains were associated largely with increases in multi-storied old-forests of Douglas-fir. In the Central Idaho Mountains ERU, the source habitats that decreased most in areal extent were old-forest single- and multi-storied Douglas-fir (vol. 3, appendix 1, table 4).

Condition of special habitat features—Densities of large-diameter snags and trees (>53 cm [21 in] d.b.h.) declined basin-wide from historical to current levels (Hann and others 1997, Hessburg and others 1999, Quigley and others 1996). Historical trends in smaller diameter snags were extremely variable (Hann and others 1997), so the overall basin-wide trend is unclear.

Figure 22—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 7 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 23—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 7, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Other factors affecting species within the group— Cavity availability is dependent on the presence of primary excavators, most notably the pileated woodpecker and northern flicker (Hayward 1994c). Changes in population levels of these and other cavity excavators could affect boreal owl nesting opportunities.

Changes in forest structure could alter habitat suitability for voles and other important prey species and affect population levels of these species. In particular, changes in the abundance of coarse woody debris, snags, lichens, and fungi could significantly alter habitat suitability for many species found in older structural stages. This could affect the food resource for boreal owls and have a direct bearing on reproductive success.

Population status and trends— No reliable estimates of boreal owl population densities or trends in North America are available (Hayward 1994c).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integration of potential resource objectives for group 7 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues— The following issues have been identified as potentially influencing boreal owl conservation:

1. Declines in late-seral subalpine and montane forests, particularly in the Northern Cascades, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and Snake Headwaters ERUs.

2. Declines in large aspen trees and forests primarily because of fire suppression. Hayward and others (1993) found a relatively high use of aspen for nesting compared to available habitats.

3. Increasingly disjunct distribution of source habitats that may affect population structure (Hayward 1994a, 1997) and persistence of boreal owls.

4. Loss of large-diameter snags (>45 cm [18 in] d.b.h. recommended by Hayward [1994a]).

5. Loss of microenvironments for small-mammal prey. Changes in forest structure and composition (such as loss of snags and logs) could alter habitat for primary prey species (Hayward 1994a).

Potential strategies— The following strategies can be used to address the issues listed above:

1. (To address issue no. 1) Maintain existing habitats and accelerate development of subalpine and montane old-forest conditions within stands that are currently in mid-seral structural stages, particularly in the Northern Glaciated Mountains, Upper Clark Fork, and Lower Clark Fork ERUs.

2. (To address issue no. 2) Restore aspen forests throughout the basin where they have been reduced. This is particularly important in areas where aspen provides most of the nesting habitat for boreal owls (Hayward 1997).

3. (To address issue no. 3) Provide adequate links among subpopulations. Evaluate the links among subpopulations and use that information to identify areas that are highest priority for retention and restoration of habitat. This is of particular concern in the Northern Glaciated Mountains, Upper Clark Fork, and Lower Clark Fork ERUs, where reduction in the extent of source habitats has increased the isolation of remaining habitat patches.

4. (To address issues no. 4 and no. 5) Retain large-diameter snags in all source habitats and provide for snag replacement over time.

5. (To address issue no. 5) Include boreal owl conservation within a larger, ecosystem context that addresses management of primary cavity nesters, small mammals, and forest structural components (Hayward 1994a).

Practices that support strategies— The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Adjust management activities to maintain and restore source habitats, particularly in the northern ERUs. Avoid extensive use of clearcuts, which may reduce habitat quality for 100 to 200 years (Hayward 1997). Small patch

cuts implemented on long rotations may be compatible with maintenance of habitat quality for boreal owls (Hayward 1997). Thinning from below may provide for development of nest structures.

2. (In support of strategy no. 2) Use clearcutting to regenerate aspen, focusing on the maintenance, at a landscape scale, of large aspen that provide nesting habitat for boreal owls (Hayward 1997). Where aspen regeneration is inhibited by domestic or wild ungulate browsing, use exclosures to protect regenerating stands and modify management to reduce browsing pressure.

3. (In support of strategy no. 4) Determine potential snag densities for each cover type used as source habitats by conducting surveys within remote areas, reserves, and natural areas. Use these baseline data to determine whether snags are below potential in other areas. Provide measures for snag protection and recruitment in all timber harvest plans.

Group 8—Great Gray Owl

Results

Species ranges, source habitats, and special habitat features—Group 8 consists of breeding habitat for the great gray owl, a year-round resident of the basin. Great gray owls are distributed holarctically across the boreal forests of North America and Eurasia; they also inhabit other forests types at the southern extent of their range within the United States (Duncan and Hayward 1994). Within the basin, the great gray owl is widely distributed, although at low population levels, across most forested areas (fig. 24).

Within the basin, source habitats for great gray owls are old-forest, unmanaged young forest, and stand-initiation stages of montane forests, Engelmann spruce-subalpine fir, and riparian woodlands (vol. 3, appendix 1, table 1). Shrub or herb-tree regeneration also provide source habitats (vol. 3, appendix 1, table 1). Source habitats in the stand-initiation stage and herb-tree regeneration are used primarily for foraging. Old and unmanaged young forests are used for nesting and roosting, and more open stands (11 to 59 percent canopy cover [Bull and Henjum 1990]) are used for foraging. Great gray owls are a contrast species, requiring the juxtaposition of habitats used for foraging and for nesting and roosting (vol. 3, appendix 1, table 2).

Snags are a special habitat feature for great gray owls (vol. 3, appendix 1, table 2). They do not build their own nests but rely on existing platforms such as stick nests originally created by other birds or formed by dwarf mistletoe brooms, depressions in broken-topped dead trees, stumps, or artificial platforms (Bull and Henjum 1990, Duncan 1992, Mikkola 1983, Nero 1980). In one study in northeastern Oregon (Bull and Henjum 1990), 51 percent of the nests were stick platforms, 29 percent were on artificial platforms, and 20 percent were in natural depressions on broken-topped dead trees (n = 49). Of the stick nests, 68 percent were made by northern goshawks, 12 percent made by red-tailed hawks, and 20 percent were natural platforms formed by dwarf mistletoe brooms. Large branches are needed to support large stick-nests averaging 74 cm (29 in) long, 65 cm (26 in) wide, and 27 cm (11 in) high (Bull and Henjum 1990), and nests in broken-topped trees must be wide enough to accommodate a family of owls. Such trees range from 46 to 94 cm (18 to 37 in) in d.b.h. (Bull and Henjum 1990).

Broad-scale change in source habitats—Historically, source habitats for the great gray owl presumably were broadly distributed throughout forested portions of the basin (fig. 25A). The greatest concentrations of habitat were in the northern portion of the basin in the Northern Cascades, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and Snake Headwaters ERUs (vol. 3, appendix 1, table 3). Source habitat is projected to have declined moderately or strongly in 50 percent of watersheds basin-wide, and to have increased moderately or strongly in nearly 40 percent of watersheds (fig. 26). Although the overall change in source habitat has not been great, there has been a significant shift in its geographic distribution with habitat becoming more extensive in the western and central portions of the basin and less abundant in the northeastern part (fig. 25C). Of the ERUs that support substantial source habitat, moderate or strong increases in more than 50 percent of watersheds were projected for the Southern Cascades, Upper Klamath, Blue Mountains, and Central Idaho Mountains. Decreases in more than 50 percent of watersheds were projected for the Columbia Plateau, Northern Glaciated

Figure 24—Ranges of species in group 8 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Mountains, Lower Clark Fork, Upper Clark Fork, and Snake Headwaters (fig. 26). Mixed trends were projected for the Northern Cascades ERU.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— The increase in habitat in the Southern Cascades, Upper Klamath, and Blue Mountains was primarily attributed to an increase in late-seral montane forests (Hann and others 1997). In the Blue Mountains, an increase in the stand-initiation structural stage also contributed to the increase in source habitats. In the Northern Cascades, increases in source habitats primarily were due to an increase in early-seral montane forests. Habitat also has increased in the Central Idaho Mountains where the increasing trend is primarily the result of an increase in late-seral multi-layer and early-seral montane forests.

In the ERUs where habitat for this species has declined (primarily the northern and eastern parts of the basin), habitat loss can be attributed primarily to the substantial reduction in late-seral montane and subalpine forests and early-seral montane forests (Hann and others 1997). The only exception is the Columbia Plateau, where source habitats declined primarily because of the reduction in abundance of shrub or herb-tree regeneration habitat (vol. 3, appendix 1, table 4). In all of the ERUs where source habitats are projected to have declined, there has been a significant increase in managed mid-seral montane forests since the historical period (Hann and others 1997).

Our evaluation at the broad-scale did not assess the distribution of foraging habitat in relation to that for nesting habitat. Further analysis of the juxtaposition of foraging with nesting habitats is needed at a finer scale of resolution. Average breeding home range size of individual adult great gray owls has been calculated as 4.5 km^2 (1.7 mi^2) (Bull and Henjum 1990) and 2.6 km^2 (1.0 mi^2) (Craighead and Craighead 1956), and

Figure 25—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 8 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 26—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 8, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

the ranges of adults are overlapping (Bull and Henjum 1990). Within each home range, a mixture of foraging and nesting habitat is needed. Analyses completed for the basin do not reveal landscape patterns at the scale of individual home ranges. Results for source habitats shown here for both the current and historical time periods are likely overestimates as they do not take into account the need for juxtaposition of habitats.

Condition of special habitat features— According to the landscape assessment (Hann and others 1997), the forests of the current period are more homogeneous than historical forests. Old-forest structures, remnant large trees, and the presence of medium to large trees in all forest structural classes have been reduced (Hann and others 1997). Densities of large-diameter snags (>53 cm [21 in] d.b.h.) likely declined basin-wide from historical to current levels (Quigley and others 1996, USDA Forest Service 1996). Presumably, the overall loss in large and medium trees and snag structures has reduced the availability of nest sites for great gray owls.

Other factors affecting the group— An additional factor may be the use of poisons to control pocket gopher populations. Such programs likely reduce the prey base for great gray owls (Hayward 1994b).

Population status and trends— No long-term, rigorous, or standardized surveys have been done of great gray owl populations within the basin (Duncan and Hayward 1994).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integration of potential resource objectives for group 8 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues— The following issues were derived from the analysis of source habitats and from published literature.

1. Decline of late- and early-seral stages of montane and subalpine forests, particularly in the northern and eastern parts of the basin.

2. Decline in availability of large trees and snags in all seral stages of montane and subalpine forests.

3. Encroachment of conifers into natural meadow systems, eliminating potential foraging habitat.

4. Reduced duration of early-seral stages because of intensive planting and thinning.

5. Decline in prey resulting from use of poisons to control pocket gophers.

Potential strategies— Habitat for great gray owls would benefit from the following strategies that address the issues listed above:

1. (To address issue no. 1) Conserve existing older forest that is considered source habitat for this species, particularly in the Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork. The older forests that are source habitats for great gray owls have greater likelihood of being used for nesting if such stands are near open or early forests, which are used for foraging.

2. (To address issue no. 1) Accelerate the development of old-forest conditions in existing mid-seral stands.

3. (To address issue no. 2) Maintain and recruit large (>50 cm [20 in] d.b.h.) (Bull and Henjum 1990) live trees and snags for potential nesting strata.

4. (To address issue no. 3) Maintain and restore natural meadow systems that are adjacent to or near areas of old forest and have nesting platforms for great gray owls.

5. (To address issues no. 1 and no. 4) Maintain a spatial and temporal mix of nesting (late-seral) and foraging (early-seral) habitats. Continuity of foraging habitat must be maintained through prudent long-term planning of timber harvest and other forest management activities.

6. (To address issues no. 1 and no. 2) In evaluating and managing for long-term habitat quality, consider factors that influence populations of nest-building species (goshawk, red-tailed hawks, and ravens) and tree pathogen-insect interactions that can influence branch development (dwarf mistletoe brooms).

Figure 27—Ranges of species in group 9 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

7. (To address issue no. 5) Avoid the use of poisons to control pocket-gopher populations near nesting habitat for great gray owls.

Practices that support strategies— The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Focus retention efforts for late-seral montane and subalpine forests on sites where risks of catastrophic loss are relatively low.

2. (In support of strategy no. 2) Use prescribed burning and precommercial thinning to accelerate the development of old-forest conditions in mid-seral stands.

3. (In support of strategy no. 3) Maintain and restore natural meadow systems with the use of prescribed burning and removal of encroaching conifers.

4. (In support of strategy no. 3) Close roads to minimize removal of snags where such removals are reducing habitat quality for great gray owls. In addition or as an alternative to road management, actively enforce fuel wood regulations to minimize removal of large snags.

Group 9—Black-Backed Woodpecker

Results

Species ranges, source habitats, and special habitat features—The black-backed woodpecker is a year-round resident that occurs in various forest types throughout the basin, except in southern Idaho ERUs (fig. 27). Source habitats of the black-backed woodpecker include old-forest stages of subalpine, montane, and lower montane forests and riparian woodlands (vol. 3, appendix 1, table 1). Both managed and unmanaged young-forest stages of lodgepole pine also provide source habitat (vol. 3, appendix 1, table 1).

Burned conifer forests (Caton 1996, Hoffman 1997, Hutto 1995, Marshall 1992, Saab and Dudley 1998) and other insect-infested forests (Goggans and others 1988) provide key conditions necessary for both nesting and foraging. Habitat requirements for nesting include mature and old trees infested with disease or heart rot, or in early stages of decay (Goggans and others 1988). This species forages almost exclusively on the larvae of bark beetles (Scolytidae) and wood-boring beetles (Cerambycidae and Buprestridae) (Marshall 1992), which are obtained from tree trunks by scaling or flaking bark (Bull and others 1986a) and by excavating logs and the base of large-diameter tree trunks (Villard 1994). Thus, black-backed woodpeckers require conditions that produce bark and wood-boring beetle sources, including fire-, wind- or insect-killed mature or old pines, and other trees that have flaky bark (Dixon and Saab, in prep.; Marshall 1992). Both live and dead trees are used for foraging. Once trees have dried out 2 to 3 yr after mortality, bark beetles decline, and use by this woodpecker also declines (Bull 1980). Populations are irruptive in response to bark beetle outbreaks in recently fire-killed forest stands or where trees become susceptible to bark beetle attacks through maturity (Baldwin 1968, Blackford 1955, Lester 1980). In the northern Rockies, early postfire conditions (1 to 5 yr after fire) are critical for supporting populations (Hutto 1995). Black-backed woodpecker abundance was not correlated to burn size but best correlated to the number of small snags remaining after fire in the northern Rockies (Hutto 1995). Summer home ranges for single birds differ in size from 72 to 328 ha (178 to 810 acres), depending on the quality of habitat (Goggans and others 1988). Goggans and others (1988) estimated that a single black-backed woodpecker requires an area of 193 ha (477 acres) of which 59 percent should be mature to old-forest conditions. They also suggested that a minimum management area for a nesting pair in lodgepole forests should be 387 ha (956 acres) of mature or old-forest conditions.

Snags are a special habitat feature for black-backed woodpeckers (vol. 3, appendix 1, table 2). Nest cavities are excavated in live trees with heart rot or recently killed trees (dead < 5 yr). This species nests in ponderosa pine, lodgepole pine, and western larch trees in the Blue Mountains (Bull and others 1986a). In central Oregon, they nested in mixed-coniferous and lodgepole forests that were undergoing a mountain pine beetle outbreak (Goggans and others 1988). Selection for mature and old stands was reported in central Oregon based on nest, foraging, and roost sites (Goggans and others 1988). Nesting birds favor unlogged compared to salvage logged stands of burned forests in western Idaho (Saab and Dudley 1998) and western Montana (Caton 1996). Black-backed woodpeckers generally select relatively small-diameter trees for nesting compared with other cavity nesters of similar size. In the Blue Mountains, mean d.b.h. of nest trees was 37 cm (14.6 in) ($n = 15$), and trees were generally tall (>15 m [49 ft]) and recently dead (<5 yr) (Bull and others 1986a). The mean d.b.h. of nest trees in central Oregon was 28 cm (11 in) ($n = 35$) (Goggans and others 1988). In burned ponderosa pine forests of western Idaho, nest tree d.b.h. averaged 32 cm (12.6 in) ($n = 17$), nest trees had relatively light decay, nest sites were located in tree clumps, and tree (>23 cm [9 in] d.b.h.) densities surrounding nests averaged 125 per ha (51 per acre) (104 per ha [42 per acre] in logged and 151 per ha [61 per acre] in unlogged units [Saab and Dudley 1998]).

In an Oregon forest with a bark beetle epidemic, overall nesting success averaged 68.5 percent ($n = 19$ nests) (Goggans and others 1988). In contrast, nest success was 100 percent for nests monitored in burned forests of western Idaho ($n = 27$) (Saab and Dudley 1998) and northwestern Wyoming ($n = 14$) (Hoffman 1997). Nest losses in Oregon were attributed to predation by flying squirrels and Douglas squirrels (Goggans and others 1988). Few mammalian nest predators were observed recolonizing the large-scale burns of western Idaho or the burns in northwestern Wyoming during the first 3 yr after fire (Dixon and Saab, in prep.). This suggests that large burned forests during early postfire years are potentially important source habitats for black-backed woodpecker.

Broad-scale changes in source habitats—The following analysis does not account for recently burned habitats that are likely important as source habitats for black-backed woodpeckers. Such areas are generally at too fine a scale, and too ephemeral, to have been reliably estimated in the landscape analysis.

Historically, source habitats for black-backed woodpeckers were broadly distributed throughout the range of the species within the basin (fig. 28A). The most concentrated areas of habitat occurred in portions of the Blue Mountains, Columbia Plateau, Upper Klamath, Southern Cascades, Northern Cascades, and Central Idaho Mountains ERUs (fig. 28A).

Figure 28—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 9 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 29—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 9, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

The current distribution of source habitats is more concentrated in the southern half of the basin and diminished in the northern half. The Upper Klamath, Southern Cascades, Blue Mountains, southern watersheds of the Columbia Plateau, and the Central Idaho Mountains currently support the greatest concentrations of habitat (fig. 28B). In contrast, source habitats in the northern portion of the basin are scarcer and less well distributed than historically (fig. 28B).

Moderate or strong declines in source habitats were projected in nearly 70 percent of watersheds throughout the basin, with moderate or strong increases in 23 percent of watersheds (fig. 29). The most widespread declines were in the northern and far eastern parts of the basin (fig. 28). Moderate or strong declines were projected in over 90 percent of watersheds within the Northern Glaciated Mountains, Lower and Upper Clark Forks, and Snake Headwaters ERUs (fig. 29). Moderate or strongly declining trends also were projected for over 50 percent of watersheds in the Northern Cascades, Columbia Plateau, and Blue Mountains ERUs. Moderately or strongly increasing trends were projected for the Upper Klamath ERU. More mixed trends were projected for remaining ERUs.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Source habitat declined in more than 50 percent of watersheds in seven ERUs—the Northern Cascades, Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and Snake Headwaters. In all but one of these (Snake Headwaters), ecologically significant declines occurred in late-seral lower montane forests (Hann and others 1997). In addition, there were also significant declines in late-seral montane forests in the three ERUs in the north end of the basin where source habitats declined most dramatically (Northern Glaciated Mountains, Upper Clark Fork, and Lower Clark Fork) (Hann and others 1997). The declines in the Snake Headwaters resulted from declines in both montane and subalpine late-seral forests (Hann and others 1997). Increases in the Upper Klamath ERU were due to increases in both lower montane and montane late-seral forest (Hann and others 1997).

Condition of special habitat features— Basin-wide declines from historical to current conditions were estimated for late-seral forest stands and for large snags (USDA Forest Service 1996) as well as for medium and large trees in all forest structural classes (Hann and others 1997). Based on these declines a decline in medium to large snags (23 to 53 cm d.b.h. [9 to 21 in]) is a reasonable assumption (see Quigley and others 1996 and USDA Forest Service 1996).

Other factors affecting the group— The natural pattern of beetle outbreaks has been altered through silvicultural practices and fire management policies. Silvicultural practices directed at maximizing wood production by harvesting trees before they are susceptible to bark beetle attacks, and salvage logging of beetle-infested, fire-killed, and wind-killed trees reduced the occurrence of beetles in some areas. Elsewhere, fire management policies have lengthened natural fire regimes and allowed more frequent occurrences of beetles.

Road densities have increased significantly throughout the basin (Hann and others 1997), thereby allowing greater human access into forested regions and subsequent increases in snag removal for firewood.

Usurpation of nest cavities by hairy woodpeckers (Goggans and others 1988) and by Lewis'woodpeckers (Saab and Dudley 1995) negatively affects black-backed woodpeckers. Stress and elevated energetic costs associated with territorial encounters with hairy and Lewis'woodpeckers potentially reduce reproductive success of black-backed woodpeckers.

Population status and change— Breeding Bird Surveys indicate that population trends from 1966 to 1995 have been stable within western North America (n = 16 routes) (Sauer and others 1996). Trend data generated by the BBS, however, may be inadequate for monitoring populations of black-backed woodpeckers because of their relatively uncommon status and because the species is often difficult to detect (Goggans and others 1988, Marshall 1992).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 9 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The following issues were developed from our analysis of source habitat trends and findings from other studies:

1. Decline of old forests, particularly in the northern portion of the basin.

2. Decline in availability of medium to large (23 to 53 cm [9 to 21 in]) trees and snags infected with bark beetles, disease, or heart rot, or in the early stages of decay.

3. Decline in availability of large (>387 ha [956 acre]) forest stands with bark beetle outbreaks because of salvage logging, particularly in the northern basin.

4. Altered frequency of stand-replacing fires.

Potential strategies—The issues identified above suggest the following broad-scale strategies would be effective in facilitating the long-term persistence of the black-backed woodpecker.

1. (To address issue no. 1) Maintain existing old forests that include interior ponderosa pine, interior Douglas-fir, western larch, lodgepole pine, grand fir-white fir, Engelmann spruce-subalpine fir, aspen, and red fir cover types over the short term. Accelerate development of old-forest conditions in stands that are currently in mid- or early-seral stages. Maintenance and restoration of old forests is especially important within the range of this species where declines in old forests have been most pronounced. Areas of emphasis include Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs.

2. (To address issue no. 2) Where suitable nesting and foraging trees and snags are limited, retain mature and old trees and snags susceptible to bark beetle infestations, disease, and heart rot, or in the early stages of decay.

3. (To address issue no. 3) Throughout the ranges of the species, manage watersheds to maintain foraging and nesting habitat, with the location of that habitat shifting through time. Maintain stands that have experienced beetle outbreaks and stand-replacing burns.

4. (To address issue no. 4) Restore fire as an ecological process in montane and lower montane forests.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Use silvicultural treatments of prescribed underburning and thinning only of small-diameter trees (<25 cm [10 in] d.b.h.) to accelerate development of mid-successional stages to old forests, particularly in cover types of ponderosa pine, Douglas-fir, and western larch.

2. (In support of strategy no. 2) Develop guidelines for retention of existing snags (>25 cm [10 in] d.b.h.) in all forests, especially those with recent stand-replacement fire, insects, and disease to lengthen the time that those stands are suitable for nesting by black-backed woodpecker. Close roads, particularly after postfire salvage, to minimize removal of snags for firewood. In addition, or as an alternative to road management, actively enforce fuel wood regulations to minimize removal of large snags.

3. (In support of strategy no. 2) Develop measures for snag recruitment in unburned forests. Snag recruitment in unburned forests, with high risks of stand-replacing fires, will provide nest trees during the first few years after wildfire.

4. (In support of strategy no. 3) Maintain some large (>387 ha [956 acre]) forest stands with bark beetle outbreaks for 5 yr, when beetle occupancy diminishes.

5. (In support of strategy no. 3) Avoid postfire salvage logging in portions of large burned forests to maintain contiguous burned stands of at least 387 ha [956 acres].

Figure 30—Ranges of species in group 10 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

6. (In support of strategy no. 3) Where postfire salvage logging is planned in burned, lower montane forests, retain snags in clumps rather than evenly spaced distributions and retain at least 104 snags per ha (42 per acre), of d.b.h. >23 cm (9 in).

7. (In support of strategies no. 3 and no. 4) Allow wildfires to burn in some forests with high fire risk to produce stand-replacing conditions, and avoid postfire salvage logging in portions of large burned forests for about 5 yr postfire.

Group 10—Olive-Sided Flycatcher

Results

Species range, source habitats, and special habitat features—Group 10 consists of migratory breeding habitat for olive-sided flycatchers. Their range within the basin extends throughout forested areas (fig. 30). Winter range for olive-sided flycatchers includes the Central American highlands, the Andes, and the Amazon (Willis and others 1993a).

Olive-sided flycatchers are a contrast species using coniferous old forests for nesting and either openings or gaps in old forests for foraging (vol. 3, appendix 1, table 2; Sharp 1992). Their source habitats are old-forest single- and multi-storied and stand-initiation stages of subalpine, montane, and lower montane forests. Specific cover types that serve as source habitat are Engelmann spruce-subalpine fir, interior Douglas-fir, red fir, grand fir-white fir, Sierra Nevada mixed conifer, and Pacific ponderosa pine. Olive-sided flycatchers are positively associated with recent burns (Hejl 1994).

Broad-scale changes in source habitats—The extent of source habitat for olive-sided flycatchers is substantial in nine ERUs: the Northern Cascades, Southern Cascades, Upper Klamath, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, Snake Headwaters, and Central Idaho Mountains (fig. 31B). Basin-wide, the trend in source habitat for

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Figure 31—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 10 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 32—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 10, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

olive-sided flycatchers is nearly neutral, with source habitats increasing and decreasing in almost equal numbers of watersheds (fig. 32). Trends differed geographically with habitat decreasing moderately or strongly in more than 50 percent of watersheds in three ERUs in the northern basin (Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork) and increasing moderately or strongly in more than 50 percent of watersheds in three ERUs in the southern basin (Southern Cascades, Upper Klamath, and Blue Mountains) (fig. 32). Trends were more mixed in the remaining three ERUs with significant source habitat (fig. 32).

Interpreting Results

Composition and structure associated with changes in source habitats—Increases in late-seral montane forests (Hann and others 1997) were consistent across the three ERUs (Southern Cascades, Upper Klamath, and Blue Mountains), with increasing trends in more than 50 percent of watersheds. The greatest contributors to the increases were old-forest single-storied interior Douglas-fir and grand fir-white fir in the Southern Cascades; old-forest single- and multi-storied interior Douglas-fir in the Upper Klamath; and old-forest multi-storied interior Douglas-fir and grand fir-white fir in the Blue Mountains (vol. 3, appendix 1, table 4). For the three ERUs with decreasing trends in more than 50 percent of watersheds (Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork), consistent decreases occurred in early seral lower montane and montane forests; late-seral lower montane and montane multi-layered and single-layered forests; and late-seral subalpine multi-layered forests (Hann and others 1997).

Condition of special habitat features—Changes in fire regimes (Hann and others 1997) likely have resulted in poorer habitat conditions for olive-sided flycatchers, but the magnitude of the change is unknown. Where altered fire regimes result in fewer but larger fires, it seems likely that the juxtaposition of the early- and late-seral habitats used by olive-sided flycatchers becomes less favorable. Likewise, decreases in both early- and late-seral forests in the Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork likely have resulted in a strong decrease in areas of contrasting habitat condition used by olive-sided flycatchers. Our evaluation at the broad scale, however, did not assess the distribution of foraging habitat in relation to that for nesting habitat. Further analysis of the juxtaposition of foraging with nesting habitats is needed at a finer scale of resolution.

Other factors affecting the group—Marshall (1988) suggests that changes in winter habitats have negatively affected olive-sided flycatchers.

Population status and trends—Breeding Bird Survey data indicate a significant decline from 1966 to 1994 for olive-sided flycatchers in eastern Oregon and Washington (-2.5 percent per yr, $n = 25$, $P < 0.01$) (Sauer and others 1996). Saab and Rich (1997) reported significant 10-yr and 26-yr declines (4.2 percent per year and 2.9 percent per year, respectively) for flycatchers on BBS routes within the basin. They included the olive-sided flycatcher as one of 15 Neotropical migrants in the basin that are of high concern under all future management themes.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 10 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The following issues were identified from our analysis of source habitat trends:

1. Reductions in early- and late-seral subalpine, montane, and lower montane forests, particularly in the Northern Glaciated Mountains and Upper and Lower Clark Forks.

2. Changes in fire regimes that result in fewer, larger, and more destructive fires, thereby reducing the areas of juxtaposed early- and late-seral forests.

Potential strategies—The following strategies would benefit species in group 10:

1. (To address issue no. 1) Accelerate development of late-seral conditions in lower montane, montane, and subalpine forests, particularly in the Northern Glaciated Mountains and the Upper and Lower Clark Fork.

2. (To address issues no. 1 and no. 2) Increase the amounts of early-seral lower montane and montane forests, focusing on early-seral conditions that result from fire. Such restoration efforts would be most beneficial if concentrated in the northern portions of the basin.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategies no. 1 and no. 2) Various silvicultural practices including thinning from below, burning, and uneven-age management could be used to help accelerate the development of old-forest conditions and the juxtaposition of early- and late-seral habitats used by olive-sided flycatchers.

Group 11—Three-Toed Woodpecker and White-Winged Crossbill

Results

Species ranges, source habitats, and special habitat features—Group 11 consists of the three-toed woodpecker and white-winged crossbill, both of which occur at upper elevations throughout the basin. The range of the three-toed woodpecker is somewhat broader than that of the crossbill, occupying a greater portion of western Montana and central Oregon (fig. 33). The three-toed woodpecker is a year-round resident of the basin, whereas the white-winged crossbill is primarily a winter migrant, although occasional summer flocks have been observed (Harrington-Tweit and Mattocks 1985).

Source habitats for group 11 are late-seral subalpine and montane forests. Source habitats shared in common by the two species are old forests of lodgepole pine, grand fir-white fir, and Engelmann spruce-subalpine fir. The three-toed woodpecker also uses whitebark pine and mountain hemlock, and the white-winged crossbill occurs in western larch and Pacific silver fir-mountain hemlock (vol. 3, appendix 1, table 1).

Specific habitats used by the three-toed woodpecker are mature and overmature stands with bark beetles, disease, and heart rot (Goggans and others 1988) and recent stand-replacing burns with abundant wood-boring insects (Caton 1996, Hutto 1995). Three-toed woodpeckers forage predominantly on wood-boring beetle larvae (Stallcup 1962) and are attracted to areas with high concentrations of beetles, particularly in spruce and lodgepole pine (Bock and Bock 1974, Hogstad 1976, Villard 1994). Snags, a special habitat feature used for nesting (vol. 3, appendix 1, table 2), generally fall within the diameter range of 22 to 50 cm (9 to 20 in) (Bull 1980, Lester 1980). Because snags are used for foraging as well as nesting, large burns and beetle-infested stands are strongly favored for breeding over unburned or noninfested stands (Caton 1996, Goggans and others 1988). The period when burns and beetle-infested stands are useful for foraging is limited to about 5 yr, because beetles no longer use snags after they have dried out (Bull 1980). For nesting, however, the presence of heartrot may be required for cavity excavation (Goggans and others 1988), and fire-killed conifers generally do not develop this stage of decay until more than 5 yr postfire (Caton 1996). Older snags within burns or beetle outbreaks generally satisfy nesting requirements.

Crossbills are highly dependent on conifer cone crops and congregate where seed production is locally abundant (Benkman 1992). The initiation of reproduction is triggered by abundance of conifer seeds. Nesting has been recorded every month of the year and occurs whenever the seed intake rate is sufficient for egg formation in females (Benkman 1990).

Broad-scale changes in source habitats—Trends in habitat availability for group 11 differ geographically. Historically, source habitats likely were distributed throughout most of the mountainous regions of the basin but generally occupied <25 percent of any given watershed (fig. 34A). Current source habitats seem to have roughly the same geographic distribution, but the amount of habitat in the northern portion of the ranges of the species generally declined, whereas habitat in the south increased (fig. 34B). Basin-wide, source habitats increased moderately or strongly in 38 percent of the watersheds and decreased moderately or strongly in 54 percent (fig. 35). The ERUs that support significant amounts of habitat for the group and had moderately or strongly increasing trends in more than 50 percent of watersheds were the Southern Cascades, Upper Klamath, Blue Mountains, and Central Idaho Mountains (fig. 35). The ERUs for which moderate or strong declines were projected in more than 50 percent

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Figure 33—Ranges of species in group 11 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 34—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 11 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 35—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 11, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

of watersheds were the Northern Cascades, the Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and the Snake Headwaters (fig. 35).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats—Ecologically significant increases were projected by Hann and others (1997) for late-seral montane forests in all four ERUs in which source habitat increased in more than 50 percent of watersheds. For the five ERUs for which source habitats were projected to decline in more than 50 percent of watersheds, ecologically significant declines were projected in late-seral subalpine forests in the Northern Cascades; for late-seral montane forests in the Lower Clark Fork; and for both late-seral subalpine and late-seral montane forests in the Northern Glaciated Mountains, Upper Clark Fork, and Snake Headwaters.

Condition of special habitat features—Trends in snag availability within group 11 source habitats are unknown at the broad scale. Densities of large-diameter snags (>53 cm [21 in] d.b.h.) likely declined basin-wide from historical to current levels (Quigley and others 1996). The trend in smaller snags (22 to 50 cm [9 to 21 in]) used by three-toed woodpeckers is, however, unknown.

Other factors affecting the group—Three-toed woodpeckers are adapted to shifting their foraging areas to coincide with high concentrations of wood-boring beetles (Koplin 1969). Availability of this shifting food resource could be affected by salvage logging of large burns and beetle-infested stands, and maintenance of conifer stands in vigorous condition through silvicultural thinning.

Population status and trends—There are insufficient sightings in the BBS data records to determine population trends for either white-winged crossbills or three-toed woodpeckers within the basin. Summarized across the West, however, three-toed woodpecker occurrences on 14 BBS routes have declined an average of 0.7 percent annually between 1966 and 1995 ($n = 14$, $P < 0.05$; Sauer and others 1996).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integration of potential resource objectives for group 11 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The following issues were identified from our analysis of source habitat trends and from the findings of current research on group 11 species:

1. Decline in late-seral subalpine and montane forests. Cover types with basin-wide decline are western larch and whitebark pine. Declines of Engelmann spruce-subalpine fir are most notable in northern portions of the basin.

2. Potential decline in key components of the shifting food and nesting resource, which is characterized by large areas of conifer trees infected with bark beetles, disease, or heart rot, or in the early stages of decay.

Potential strategies—The following strategies could be used to maintain habitat in the southern and western portions of the basin and to reverse broad-scale declines in the northern and eastern regions:

1. (To address issue no. 1) Basin-wide, maintain remaining old forests of western larch and whitebark pine, and actively manage to promote their long-term sustainability.

2. (To address issue no. 1) In the Northern Glaciated Mountains, Upper Clark Fork, and Snake Headwaters ERUs, accelerate development of old-forest conditions in montane and subalpine forests within areas currently dominated by mid-seral stages.

3. (To address issue no. 2) Throughout the ranges of the species, manage watersheds to maintain foraging and nesting habitat, with the location of that habitat shifting through time. For three-toed woodpeckers, maintain stands that have experienced beetle outbreaks and stand-replacing burns.

Practices that support strategies— The following practices would be effective in implementing the strategies listed above:

1. (In support of strategies no. 1 and no. 2) Use under-story thinning and prescribed burns, or both, to enhance development and sustainability of western larch and whitebark pine old forests.

2. (In support of strategy no. 3) Maintain some large (>214 ha [528 acres]) (Goggans and others 1988) forest stands with bark beetle outbreaks for at least 5 yr, until beetle occupancy diminishes.

3. (In support of strategy no. 3) Where suitable nesting and foraging trees are underrepresented, retain mature and old trees susceptible to bark beetle infestations, disease, and heart rot, or in the early stages of decay.

4. (In support of strategy no. 3) Allow wildfires to burn in some forests with high fire risk to produce stand-replacing conditions, and avoid postfire salvage logging in portions of large burned forests to maintain contiguous burned stands of at least 214 ha (528 acres) (Goggans and others 1988) for about 5 yr postfire.

Group 12—Woodland Caribou

Results

Species ranges and source habitats— Group 12 consists of the woodland caribou, a year-round resident of the basin. Woodland caribou have never been widely distributed in the basin (fig. 36). They are currently restricted to an area within the Northern Glaciated Mountains that includes parts of northeastern Washington, northern Idaho, and northwestern Montana. Evidence of their continued persistence in Montana is scant (USDI Fish and Wildlife Service 1994). The suspected historical range of the woodland caribou (ICBEMP 1996i) included parts of five ERUs: Northern Glaciated Mountains, Lower Clark Fork, Central Idaho Mountains, and small portions of the Columbia Plateau and Upper Clark Fork (fig. 36). Woodland caribou were federally listed as endangered in 1984.

Source habitats for woodland caribou are late-seral subalpine and montane forests (vol. 3, appendix 1, table 1). In total, five cover type-structural stage combinations provide source habitats for the woodland caribou. These are western redcedar/western hemlock old-forest single- and multi-storied stands; grand fir-white fir old-forest single- and multi-storied stands; and Engelmann spruce-subalpine fir old-forest multi-storied stands (vol. 3, appendix 1, table 1).

Broad-scale change in source habitats— This analysis of source habitats was based on the historical caribou range. Source habitats were projected to occur in five ERUs: the Columbia Plateau, the Northern Glaciated Mountains, the Lower Clark Fork, the Upper Clark Fork, and the Central Idaho Mountains (fig. 37). Source habitats in the Upper Clark Fork and Columbia Plateau were scarce (fig. 37).

Basin-wide, the trend in source habitats for caribou (historical to current periods) was mixed with 53 percent of watersheds projected with moderately or strongly negative trends and 41 percent with moderately or strongly positive trends (fig. 38). The three ERUs that supported significant caribou habitat each displayed a different trend. Trend in the Northern Glaciated Mountains was predominantly negative with a moderately or strongly negative trend projected for 65 percent of watersheds (fig. 38). For the Lower Clark Fork, a strongly positive trend was projected for 50 percent of watersheds and a strongly negative trend for 38 percent (fig. 38). Finally, a mixed trend was projected for the Central Idaho Mountains with watersheds split almost evenly among those showing a moderately or strongly negative trend (58 percent) and those showing a moderately or strongly positive trend (52 percent) (fig. 38).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— The predominantly negative trend for source habitat in the Northern Glaciated Mountains resulted largely from a strong decline in the old-forest multi-story stage of Engelmann spruce-subalpine fir (vol. 3, appendix 1, table 4). In the Lower Clark Fork ERU, the decrease in Engelmann spruce-subalpine fir old forest was offset by increases in western redcedar-western hemlock and grand fir-white fir old forests (vol. 3, appendix 1, table 4). In the Central Idaho Mountains, western redcedar-western hemlock, grand fir-white fir, and

Figure 36—Ranges of species in group 12 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 37—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 12 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 38—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 12, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of >60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Engelmann spruce-subalpine fir old forests all increased (vol. 3, appendix 1, table 4), apparently masking geographic variation and resulting in the mixed trend of watersheds increasing and decreasing (fig. 38).

Other factors affecting the group— Analyses by Zager and others (1995) indicated that adult mortality most limits population growth in the Selkirk population, and that at least 30 percent of this mortality is predator related. They suggested that high mortality rates may be associated with an increasing population of mountain lions, that in turn responded to expanding moose and white-tailed deer populations.

Woodland caribou populations are also subject to high rates of neonatal mortality, often approaching 50 percent. Calves typically make up 30 percent of the population at birth, but by recruitment age (1 yr) they typically make up <20 percent of the population (Scott and Servheen 1985).[3]

Both roads and human disturbance have been documented as causes of direct mortality for woodland caribou. Fatal collisions with automobiles occur on open roads in woodland caribou habitat (Scott and Servheen 1985). Ahigh percentage of the annual mortality in the 1980s was attributed to illegal harvest by hunters and poachers (Scott and Servheen 1985). Caribou mortality due to illegal shootings has decreased since the species was federally listed as endangered in 1984, but illegal shooting has not been eliminated. Road densities and the potential for human disturbance have both increased from historical to current periods. In woodland caribou range, current average road densities are estimated to be moderate to high (Hann and others 1997).

High levels of disturbance by snowmobiles can cause caribou to abandon portions of their range, although low levels of snowmobile use are believed to be compatible with caribou occupancy of an area (Simpson 1987).

Population status and trends— Historically, caribou were distributed throughout the Northeastern, North-Central, and Northwestern United States. Their range within the basin included northwestern Montana and Idaho south to the Salmon River (USDI Fish and Wildlife Service 1994). By the 1960s, their range in the United States was restricted to the Selkirk Mountains of northeastern Washington and northern Idaho (USDI Fish and Wildlife Service 1994). The reduction in the range of the caribou was probably due to a combination of habitat fragmentation (resulting from both fires and timber harvest) and excessive mortality from overharvest and vehicle collisions.

In the 1950s, the Selkirk population of caribou in northeastern Washington, northern Idaho, and southeastern British Columbia was estimated at about 100 animals (Evans 1960, Flinn 1956). By the early 1980s, this population had declined to 25 to 30 animals whose distribution centered around Stagleap Provincial Park, British Columbia (Scott and Servheen 1985). The population in Idaho was augmented with animals from British Columbia three times between 1987 and 1990. The result was the establishment of a herd in the Idaho portion of the Selkirk Mountains. Populations continue to decline, however (see footnote 3; Zager and others 1995). Additional augmentation efforts occurred in the Washington portion of the Selkirks in 1996 and 1997.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integration of potential resource objectives for group 12 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The primary issues for woodland caribou are reported in the Selkirk Mountain Woodland Caribou Recovery Plan (USDI Fish and Wildlife Service 1994).

1. Reductions in source habitat in key portions of caribou range.

2. Illegal shooting, including accidental shooting by deer and elk hunters.

3. Predation by mountain lions, bears, wolves, and coyotes.

4. Mortality from vehicle collisions.

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[3] Personal communication. 1997. Wayne Wakkinen, regional wildlife biologist, Idaho Department of Fish and Game, HCR 85, Box 323-J, Bonners Ferry, ID 83805.

5. Displacement resulting from other human disturbance (for example, snowmobiles [Simpson 1987]).

Potential strategies—The U.S. Fish and Wildlife Service has established the following strategies that would provide recovery benefits for woodland caribou:

1. (To address all issues) Maintain the two existing caribou herds in the Selkirk ecosystem, and establish a third herd in the western portion of the Selkirk Mountains in eastern Washington.

2. (To address issue no. 1) Provide for at least 179 415 ha (443,000 acres) of suitable and potential caribou habitat in the Selkirk Mountains to support a self-sustaining population.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above (taken from the Selkirk Mountain Woodland Caribou Recovery Plan [USDI Fish and Wildlife Service 1994]):

1. (In support of strategy no. 1) Reduce the impacts of poaching and hunting through outreach programs, restriction of access, and more effective law enforcement.

2. (In support of strategy no. 1) Reduce impacts of caribou-vehicle collisions by identifying areas where collisions are most likely and taking corrective actions (for example, reducing vehicle speeds, rerouting or closing roads, or increasing driver awareness.).

3. (In support of strategy no. 1) Identify most important additional sources of mortality by following radio-collared animals. Reduce other causes to the extent possible, recognizing that some mortality is unavoidable (for example, predation by other listed species).

4. (In support of strategy no. 1) Reduce impacts because of genetic and demographic influences by continuing augmentation and monitoring the success of augmentation efforts (but see Zager and others [1995] for cautions concerning the prognosis for augmentation efforts).

5. (In support of strategy no. 2) Maintain existing late-seral montane and subalpine forests within the areas designated to support caribou herds. Accelerate the development of old-forest conditions in currently mid-seral stands within these areas.

6. (In support of strategy no. 1) Evaluate the effects of roads, motorized vehicles, and recreational activities on caribou. Where such uses are not compatible with recovery (for example, where intensive snowmobile use is displacing caribou) implement standards (such as access timing or area closures) to address the issues.

Group 13—Northern Flying Squirrel

Results

Species ranges, source habitats, and special habitat features—This group consists of the northern flying squirrel, which is a year-round resident of the basin. Flying squirrels occur throughout forested portions of the basin (fig. 39). Source habitats for this species include old-forest and unmanaged young-forest stages of subalpine, montane, lower montane, and riparian woodland cover types (vol. 3, appendix 1, table 1). The understory reinitiation stage of most of these types also is shown as source habitat (vol. 3, appendix 1, table 1; ICBEMPc). This stage is characterized by varying levels of canopy closure, and may contain large trees and other structures (vol. 1, table 4; Hann and others 1997) characteristic of northern flying squirrel habitat (Carey 1995). Because the understory reinitiation stage is highly variable (Hann and others 1997), however, its suitability as source habitat for flying squirrels is also variable.

Two special habitat features have been identified for northern flying squirrels (vol. 3, appendix 1, table 2). Flying squirrels nest in cavities that result from either damage to trees or excavation by woodpeckers (Carey 1995). Thus, snags are a special habitat feature, although squirrels also use cavities in live trees and external stick nests (Carey 1995, Waters and Zabel 1995). In a study in western Oregon, Carey (1991) found that snags containing nests average 89 cm (35 in) d.b.h. Down woody material is also an important

VIEW

MAP

Figure 39—Ranges of species in group 13 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

feature of flying squirrel habitat (Carey 1991), presumably because of its role in supporting lichens and fungi that are the principle components of the diet of squirrels.

Broad-scale changes in source habitats— Historically, source habitats likely occurred throughout the forested portions of the basin (fig. 40A). Changes from historical have resulted in a reduction in the concentration of habitat across much of the range of the squirrel, with areas of increased habitat in the northeastern, central, and southwestern portions of the basin (figs. 40B, C). Overall, habitat has declined moderately or strongly in nearly 60 percent of watersheds in the basin and increased moderately or strongly in 27 percent of watersheds (fig. 41).

In eight ERUs, source habitat declined moderately or strongly in more than 50 percent of watersheds. These ERUs are the Northern Cascades, Southern Cascades, Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, Upper Clark Fork, Lower Clark Fork, and Snake Headwaters. Source habitat increased moderately or strongly in more than 50 percent of watersheds in the Upper Klamath and had mixed trends in the Central Idaho Mountains. Only relatively small amounts of habitat are present in the remaining three ERUs.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Ecologically significant decreases were projected (Hann and others 1997) for late-seral lower montane forests in seven of the eight ERUs for which source habitat declined in more than 50 percent of watersheds. The exception was the Snake Headwaters where significant declines were projected in late-seral montane and subalpine forests but not in late-seral lower montane forests. In addition to the declines in late-seral lower montane forests, there were declines in late-seral montane and late-seral subalpine forests in the Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork

Figure 40—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 13 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 41—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 13, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

(Hann and others 1997). Declines in late-seral subalpine forests also contributed to the decreases in source habitat in the Northern Cascades and Blue Mountains.

Unmanaged young forest and understory reinitiation stages declined throughout the basin, including substantial losses in unmanaged young forest in the Northern Cascades and Upper Snake for cover types used as source habitat by northern flying squirrels (vol. 3, appendix 1, table 4). An exception to this general pattern of decreases was increases in understory reinitiation in the Northern Glaciated Mountains and Lower Clark Fork. These increases likely account for the areas of increasing source habitat concentration that were projected (fig. 40) within these ERUs, which otherwise displayed general declines in source habitat. Because these mid-seral stages, and particularly the understory reinitiation stage, are quite variable, these projected increases merit further evaluation at a finer scale.

In the Upper Klamath, the only ERU for which an increase in source habitat was projected in more than 50 percent of watersheds, there were ecologically significant increases in late-seral lower montane, montane, and subalpine forests (Hann and others 1997).

Condition of special habitat features— Densities of large-diameter snags (≥53 cm [21 in] d.b.h.) likely declined basin-wide from historical to current levels (Quigley and others 1996, USDA Forest Service 1996).

Other factors affecting the group— Forest management practices may have a significant effect on the hypogeous sporocarps of mycorrhizal fungi, a principal food source for flying squirrels. In a study in the Klamath Mountains, hypogeous sporocarps were nearly absent from clearcuts and were strongly associated with coarse woody debris in late seral forests (Clarkson and Mills 1994). The negative association with clearcuts was thought to be due to microclimatic conditions and the effects of postharvest slash burns (Clarkson and Mills 1994). In a study in northeastern California, flying squirrel abundance was associated with the frequency of hypogeous sporocarps (Waters and Zabel 1995), but no correlation was found between sporocarp abundance and either thinning or broadcast burning (Waters and others 1994, cited in Waters and Zabel 1995). This study, however, did not examine sporocarp abundance in relation to clearcuts versus mature forests.

Population status and trends— No population trend information is available for northern flying squirrels within the basin.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integration of potential resource objectives for group 13 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues— The following issues were identified from the results of our analysis and other empirical research:

1. Widespread loss of old forests and associated structures (snags, logs, and cavities).

2. Reduced availability of remnant large trees and snags in all seral stages (Hann and others 1997).

3. Negative effect of forest management activities on fungus and lichen diversity and abundance (Carey 1991).

Potential strategies— The following strategies could be used to reverse broad-scale declines in source habitats and populations:

1. (To address issues nos. 1-3) Maintain existing late-seral forests and encourage the development of appropriate habitat structures (snags, decayed down wood, and abundance of fungi and lichens) in mid-seral forests in all cover types used as source habitats, particularly in the northern half of the basin (Northern Cascades, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs).

Practices that support strategies— The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) In the northern basin, give high priority to retention of old forests that have relatively low risk of loss through catastrophic fire. Priority should be given to large blocks having high interior-to-edge ratios and few large openings.

Figure 42—Ranges of species in group 14 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

2. (In support of strategy no. 1) Actively recruit snags and logs from green trees to increase the representation of old-forest structures (snags and logs) in mid-seral stands and in old forests where snags and logs are in low density or absent.

3. (In support of strategy no. 1) Manage early- and mid-seral stands for increased vegetative diversity in order to encourage fungus and lichen diversity and abundance (Carey 1991).

Group 14—Hermit Warbler

Results

Species ranges and source habitats— Group 14 consists of the hermit warbler, a migrant that breeds in the basin and winters in high-elevation forests in Mexico and Central America. Most of the range of the hermit warbler occurs outside the basin along the west coast of British Columbia, Washington, Oregon, and California, overlapping the basin only along the crest of the Cascade Range (fig. 42) primarily in three ERUs: the Northern Cascades, Southern Cascades, and Upper Klamath.

Habitat for hermit warblers is characterized by medium to large conifers (>31 cm [12.2 in] d.b.h.) (Morrison 1982). Source habitats within the basin include the old-forest and young-forest structural stages of interior Douglas-fir, red fir, grand fir-white fir, and Sierra Nevada mixed conifer (vol. 3, appendix 1, table 1). Both managed and unmanaged young forest support source habitat.

Broad-scale changes in source habitats— Source habitats for hermit warblers occur along the crest of the Cascade Range (fig. 43). Within this area, source habitat was projected to have increased moderately or strongly in over 75 percent of watersheds (fig. 44). Habitat decreased moderately or strongly in only 17 percent of watersheds. Source habitat increased moderately or strongly in 62 percent of watersheds in the Northern Cascades, in 90 percent of watersheds in the Southern Cascades, and in 100 percent of watersheds in the Upper Klamath (fig. 44).

Figure 43—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 14 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 44—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 14, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— In the Northern Cascades, the increase in source habitat was due to increases in managed young-forest stages of interior Douglas-fir and grand fir-white fir (vol. 3, appendix 1, table 4). In the Southern Cascades, increasing source habitat was associated with increases in interior Douglas-fir and grand fir-white fir old forests and interior Douglas-fir managed young forest (vol. 3, appendix 1, table 4). In the Upper Klamath, increases were driven by increasing old-forest stages of interior Douglas-fir and grand fir-white fir (vol. 3, appendix 1, table 4).

Other factors affecting the group— Hermit warblers forage along conifer branches, and sometimes deciduous trees and shrubs, for beetles, caterpillars, small flying insects, and spiders (Terres 1991). Thus, measures taken to control insects may impact hermit warblers.

The hermit warbler winters in high-elevation forests in Mexico and Nicaragua and sparingly into Costa Rica (Sharp 1992). Impacts to wintering habitats may negatively affect the species.

Population status and trends— There are insufficient data in the BBS information to be able to predict a population trend for the hermit warbler across the basin (Saab and Rich 1997). The BBS data analyzed within other geographic boundaries (Sauer and others 1996), however, showed an increasing trend in hermit warbler populations in eastern Oregon and Washington (7.6 percent per year, $n = 7$, $P < 0.01$, 1966 to 1979).

Management Implications

No significant issues were identified for hermit warblers or their habitat.

Group 15—Pygmy Shrew and Wolverine

Results

Species ranges, source habitats, and special habitat features—This group consists of the pygmy shrew and wolverine, which are year-round residents of the basin. Wolverines occur in parts of all ERUs in the basin, although they are absent from the middle portion of the Columbia Plateau, and the south-central portion of the basin (fig. 45). The range of the pygmy shrew is restricted to the northeastern portion of the basin, primarily within the Northern Glaciated Mountains and Lower Clark Fork ERUs (fig. 45).

Both species should be considered generalists. Source habitats for pygmy shrews include virtually all structural stages of all subalpine and montane forests with the exception of Sierra Nevada mixed conifer (vol. 3, appendix 1, table 1). All stages of the shrub-herb-tree regeneration type also serve as source habitat for pygmy shrews. Source habitats for wolverines include alpine tundra and all subalpine and montane forests (vol. 3, appendix 1, table 1). Within the forest types, all structural stages except the closed canopy stem exclusion stage provide source habitat.

Wolverines are predominantly scavengers, especially in winter when their diets consist primarily of ungulate carcasses (Banci 1994). In summer, they use a wider variety of foods including small mammals, birds, carrion, and berries (Weaver and others 1996). Copeland (1996) found that carrion-related food supplied 46 percent of wolverine diets in Idaho during both summer and winter. Banci (1994) suggested that diversity of habitats and foods is important to wolverines.

Several special habitat features have been identified for wolverines (vol. 3, appendix 1, table 2). Natal dens in Idaho were primarily located in subalpine cirque basins in isolated talus surrounded by trees (Copeland 1996). There is also evidence that wolverines use down logs and hollow trees for denning (Copeland 1996; Pulliainen 1968, as cited in Banci 1994), and cavities in live trees also may be used (Ognev 1935, cited in Banci 1994; Pulliainen 1968). Both talus and areas associated with large, fallen trees were used as maternal den sites in Idaho (Copeland 1996).

No special habitat features were identified for the pygmy shrew.

Broad-scale changes in source habitats— Historically, source habitats likely occurred throughout the forested portions of the basin, with some of the greatest concentrations in the northeast (fig. 46A).

Figure 45—Ranges of species in group 15 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 46—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 15 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 47—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 15, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

From historical to current times, source habitat has increased in the central and western portions of the basin and undergone minor decreases in the north (fig. 46B).

Basin-wide, source habitat was projected to have increased moderately or strongly in 56 percent of watersheds and to have decreased moderately or strongly in 22 percent (fig. 47). Within the nine ERUs that support significant amounts of source habitat (fig. 47), five (Northern Cascades, Southern Cascades, Columbia Plateau, Blue Mountains, and Central Idaho Mountains) have undergone moderate or strong increases in more than 50 percent of watersheds, one (Upper Clark Fork) has undergone decreases in 50 percent or more of watersheds, and three (Northern Glaciated Mountains, Lower Clark Fork, and Snake Headwaters) have had mixed trends.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Causes for source habitat increases and decreases differed across ERUs (Hann and others 1997). Community types that most influenced habitat increases were early seral montane in the Northern Cascades, late-seral subalpine in the Southern Cascades, mid-seral montane in the Columbia Plateau, mid- and late-seral montane in the Blue Mountains, and early-seral subalpine and late-seral montane in the Central Idaho Mountains. In the Upper Clark Fork, community types that contributed most to the decline in habitat were early- and late-seral montane.

Condition of special habitat features—Densities of large-diameter snags (>53 cm [21 in] d.b.h.) and of large, remnant trees likely declined basin-wide from historical to current levels (Hann and others 1997, Hessburg and others 1999, Quigley and others 1996). Trends in snag abundance ultimately affect the availability of large down logs and cavities, whereas the decrease in large, remnant trees would likely translate to a decrease in large, hollow trees. Talus likely exists currently where it existed historically.

Other factors affecting the group— The clearcut method of timber harvest can negatively affect wolverines. Snow-tracking and radio telemetry in Montana indicated that wolverines avoided recent clearcuts and burns (Hornocker and Hash 1981). Copeland (1996), however, found that wolverines in Idaho commonly crossed natural openings, burned areas, meadows, or open mountain tops.

Populations of wolverines can be impacted by fur harvesting if trapping is not carefully regulated (Banci 1994). Within the basin, trapping is allowed only in Montana, and most of the harvest is believed to be incidental in traps set for other carnivores (Banci 1994).

Copeland (1996) found that human disturbance near natal denning habitat resulted in immediate den abandonment but not kit abandonment. Disturbances that could affect wolverine are heli-skiing, snowmobiles, backcountry skiing, logging, hunting, and summer recreation (Copeland 1996, Hornocker and Hash 1981, ICBEMP1996f). Wolverine densities in Montana, however, did not differ between the wilderness and nonwilderness portions of one study area, nor was their behavior or habitat use different, based on snow tracking and radio telemetry (Hornocker and Hash 1981). In addition, Hornocker and Hash (1981) concluded that movements of wolverines in Montana were not affected by highways.

Weaver and others (1996) argued that wolverines are less resilient than other large carnivores due to their low lifetime reproductive capability, susceptibility to natural fluctuations in scavenging opportunities, and vulnerability to trapping. They suggested that wolverines, along with grizzly bears, have a greater requirement for large, contiguous reserves than do other large carnivores such as gray wolves and mountain lions.

No information is available on other factors that might affect the pygmy shrew.

Population status and trends—Hash (1987) described a contraction in the North American range of the wolverine beginning around 1840 with the onset of extensive exploration, fur trade, and settlement. State records suggest very low wolverine numbers in Montana, Idaho, Oregon, and Washington from the 1920s through 1950s, with increases in wolverine sighting since the 1960s (Banci 1994). The increases in Montana (Newby and McDonald 1964, Newby and Wright 1955) and in Washington (Johnson 1977) may have resulted from dispersal from Canada.

Throughout its range, the pygmy shrew is considered rare (Feldhamer and others 1993), and basin-wide trends in pygmy shrew populations are unknown.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integration of potential resource objectives for group 15 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The following issues were identified from the results of our analysis and other empirical research:

1. Loss of montane and subalpine old-forests and associated structures (snags, logs, and cavities), particularly in the northern portion of the basin.

2. Low population numbers.

3. Increased negative effects from humans, resulting from higher road densities, increased technological advances in vehicular capabilities, and interest in winter recreation.

Potential strategies—The following strategies could be used to reverse broad-scale declines in source habitats and populations:

1. (To address issue no. 1) Increase the representation of late-seral stage forests in all cover types used as source habitats, particularly in the northern half of the basin (Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs).

2. (To address issues no. 2 and no. 3) Identify refugia for long-term management of wolverine (Banci 1994).

3. (To address issues no. 2 and no. 3) Provide adequate links among existing wolverine populations. These dispersal corridors likely do not require the same habitat attributes needed to support self-sustaining populations (Banci 1994).

4. (To address issue no. 3) Reduce human disturbances, particularly in areas with known or high potential for wolverine natal den sites (subalpine talus cirques).

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) In the northern basin, retain existing old forests and identify mid successional forests where attainment of old-forest conditions can be accelerated.

2. (In support of strategy no. 1) Actively recruit snags and logs from green trees to increase the representation of old-forest structures (snags and logs) in mid-seral stands and in old forests where snags and logs are uncommon or absent.

3. (In support of strategy no. 1) Retain slash piles and decks of cull logs to substitute for down logs over the short term.

4. (In support of strategy no. 2) Maintain current wilderness areas and other congressionally designated reserves as refugia for wolverine, and reduce human disturbances near den sites in these areas.

5. (In support of strategy no. 2) Identify existing areas with the following desired conditions, or manage selected areas to create the following desired conditions for species strongholds: large, contiguous blocks of forest cover with abundant snags and large logs and low road densities with connectivity to subalpine cirque habitats required for denning, security, and summer foraging habitat.

6. (In support of strategy no. 3) Identify isolated populations and unoccupied habitats and use interagency planning to develop broad-scale links over the long term.

7. (In support of strategy no. 4) Minimize new construction of secondary roads and close unneeded roads after timber harvests.

No explicit recommendations are available in the literature or are any available from our results for the pygmy shrew.

Group 16—Lynx

Results

Species ranges, source habitats, and special habitat features—The lynx, a year-round resident of the basin, is the only member of group 16. The range of the lynx includes the northern, eastern, and central portions of the basin (fig. 48). There are limited

Figure 48—Ranges of species in group 16 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

records of lynx occurring in the Southern Cascades ERU (McKelvey and others 1999), but these records were not included in the range map delineated by Marcot and others (in prep.). In March 2000, the U.S. Fish and Wildlife Service determined the lynx to be a threatened species pursuant to the Endangered Species Act of 1973 (U.S. Government 2000a).

Primary habitat for lynx is found in subalpine and montane forests that are cold or moist forest types (vol. 3, appendix 1, table 1; McKelvey and others 1999). Within the montane forest community, source habitats are provided by all vegetation types except Pacific silver fir-mountain hemlock, red fir, and Sierra Nevada mixed conifer. Within the subalpine forest community, only Engelmann spruce-subalpine fir provides source habitat. Lynx habitat includes various structural stages (Koehler and Aubry 1994, Ruggiero and others 1999).

Lynx forage primarily in early-seral forests and in some mid-seral forests that support high numbers of prey; lynx also use late-seral forests for denning and rearing young as well as for hunting alternative sources of prey (Ruggiero and others 1999). Consequently, source habitats for lynx are provided by most of the coniferous forest structural stages with the exception of old-forest single-storied stands (vol. 3, appendix 1, table 1). Riparian woodlands and shrublands are also source habitats.

Hollow down logs are a special habitat feature for lynx (vol. 3, appendix 1, table 2); logs are used both as den sites and resting places (ICBEMP 1996e, Koehler 1990).

Broad-scale changes in source habitats— Basin-wide, amounts of source habitats for lynx increased moderately or strongly in 47 percent of watersheds and decreased in 23 percent from historical to current

Figure 49—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 16 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 50—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 16, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of >20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

periods (figs. 49 and 50). Habitat increased in more than 50 percent of the watersheds in two ERUs, the Blue Mountains and the Northern Glaciated Mountains (fig. 50). Trends were mixed in the remaining ERUs that contain significant habitat: Northern Cascades, Upper Clark Fork, Lower Clark Fork, Snake Headwaters, and Central Idaho Mountains (fig. 50).

Interpreting Results

Composition and structure associated with changes in source habitats—A strong increase in mid-seral montane forests, along with increases in early- and mid-seral subalpine forests (Hann and others 1997), accounted for the increasing source habitat trend in the Northern Glaciated Mountains. Increases in mid- and late-seral montane forests and early- and mid-seral subalpine forests (Hann and others 1997) contributed to the overall increase in source habitats in the Blue Mountains. Mid-seral montane and subalpine forests also increased in the Lower Clark Fork, Upper Clark Fork, and Snake Headwaters ERUs; however, these increases were offset by decreases in early-seral montane forests and late-seral montane and subalpine forests (Hann and others 1997). In the Northern Cascades, increases in early-seral montane and subalpine forests were offset by decreases in mid- and late-seral subalpine forests (Hann and others 1997). There were increases in early- and late-seral montane and subalpine forests in the Central Idaho Mountains (Hann and others 1997), but these increases were not widespread enough to result in an overall moderate or strong ERU trend.

Condition of special habitat features—Hann and others (1997) reported a decrease in abundance and occurrence of large down logs in areas of traditional forest management. Large down logs are used by lynx for denning and rearing young (Ruggiero and others 1999).

Other factors affecting the group—Trapping can be a significant source of mortality for lynx (Bailey and others 1986, Carbyn and Patriquin 1983, Mech 1980, Nellis and others 1972, Parker and others 1983, Ward and Krebs 1985). Trappers are capable of removing from 60 to 80 percent of the individuals in a given lynx population (Bailey and others 1986, Parker and others 1983). Incidental takes of lynx during bobcat and coyote trapping seasons may be cause for concern, especially with low-density lynx populations. Other forms of human disturbance also affect lynx. According to Koehler and Brittell (1990), minimal human disturbance is important to denning site selection. Winter recreation may have a significant effect on lynx populations. The packing effect of snowmobile trails may open areas of deep snow to foraging from other predators such as bobcats and lynx (Kohler and Aubry 1994, Ruggiero and others 1999). In the north Cascades, snowmobiling and other winter recreation have increased in the past decade, with suspected negative effects on lynx.[4] The increase in interactions between human and lynx, primarily because of increased use of off-highway vehicles (including snowmobiles), may result in increased lynx mortality from intentional and unintentional shooting and collisions with vehicles (Koehler and Brittell 1990). Highways could also pose barriers to lynx movement or increase mortality from vehicle collisions (Ruediger 1996, Terra-Berns and others 1997).

Lynx populations are closely tied to snowshoe hare population trends, especially north of the basin (Butts 1992, Murray and Boutin 1991, Parker and others 1983, Weaver 1993). Lynx populations in the basin, however, may not be as cyclic as those at more northern latitudes (Brittell and others 1989, Koehler 1990). Within the basin, several other predators (bobcat, red fox, and some hawk and owl species) compete with lynx for snowshoe hare as prey, unlike areas to the north; many of these competing predators possibly respond more positively to human-induced habitat alterations (Roloff 1995). This increased competition for prey may increase the vulnerability of lynx (Witmer and others 1998) as well as limit the size of lynx populations (Boutin and others 1986, Keith and others 1984).

Forest management practices have varying effects on both lynx and lynx prey habitat (Ruggiero and others 1999). Lynx do not hunt in large, open areas with little or no cover (Koehler 1990, Koehler and Brittell 1990), making large clearcut blocks potential barriers to movement (Koehler and Aubry 1994). Early-seral habitats created by fire or logging, however, are essential to maintain foraging areas for lynx prey, principally snowshoe hare (Koehler and Aubry 1994, Koehler and Brittell 1990). Koehler and Aubry (1994) proposed that frequent, small patches of habitat

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[4] Personal communication. 1997. Robert Naney, wildlife biologist, Okanogan National Forest, 1240 South Second Avenue, Okanogan, WA 98840-9723.

alteration that mimic natural disturbance patterns would be beneficial. Post-clearcut areas may not become suitable for snowshoe hare habitat for more than 10 years and may not become optimal hare habitat for another 20 years (Koehler and Aubry 1994). Relatively small patches of old forest (1 ha [2.5 acres]) are needed for denning, though these areas must be near and connected to good foraging habitat (Koehler and Brittell 1990). Travel corridors generally have a closed-canopy cover >2 m high (6.5 ft.) (Brittell and others 1989).

Population status and trends—Empirical data for distribution of lynx within the basin are scarce, and data on abundance of lynx populations are not available. McKelvey and others (1999) recently summarized all known lynx locations in the United States, which provides a framework for designing and conducting future surveys and demographic studies of lynx populations.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 16 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The following issues for lynx were taken from the literature.

1. The lack of empirical information on population ecology, foraging ecology, den site characteristics, habitat relations at the landscape scale, and distribution and status in the basin (Ruggiero and others 1999).

2. Altered mosaic of source habitats because of fire suppression and logging (Hann and others 1997).

3. Negative effects of human activities on lynx (Koehler and Aubry 1994).

4. The peninsular and disjunct distribution of suitable lynx habitat in the western mountains (Koehler and Aubry 1994), and the associated potential for population isolation or limited metapopulation structure to cause local or regional extirpations (Ruggiero and others 1999).

Potential strategies—

1. (To address issue no. 1) Develop an interagency research, inventory, and monitoring effort aimed at gathering information on population ecology, foraging ecology, den site characteristics, habitat relations at the landscape scale, and distribution and status in the basin.

2. (To address issue no. 2) Restore fire as an ecological process or use other forest management practices in montane and upper montane community types to provide for a suitable mosaic of early-seral habitat rich in shrubs and well connected to late-seral habitat with abundant large down logs.

3. (To address issue no. 3) Design silvicultural treatments at a landscape scale with the needs of snowshoe hare and other lynx prey as one consideration.

4. (To address issue no. 3) Provide areas of high-quality lynx habitat that are protected from human disturbance (Koehler and Aubry 1994).

5. (To address issue no. 4) Develop a strategy to allow for interactions among lynx populations, including the provision of travel corridors (Koehler 1990) and broader landscape connectivity.

6. (To address issue no. 4) Develop a strategy to allow for population reintroductions as appropriate.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategies no. 2 and no. 3) Management of stand dynamics for lynx and snowshoe hares focuses on the creation of early and late old-forest structural stages consistent with historical variability. In designing forest landscapes, give management consideration to habitats for alternate prey species such as red squirrel, voles, and mice in addition to denning habitat for lynx. Down wood is an important denning habitat component. When thinning stands to meet timber management objectives, stands should either be thinned early before they are recolonized by snowshoe hares or thinned when they are older (for example, 30 to 40 yr) and are little used by hares.

2. (In support of strategy no. 4) In areas of known or suspected lynx populations, close roads and areas to all vehicles as needed to minimize human disturbance, limit potential increase in competing predators, and provide for landscape connectivity among and within populations. Improve highway passage by using fencing and overpasses and underpasses.

3. (In support of strategies no. 5 and no. 6) Identify areas that currently support high-quality lynx habitat, have low road densities, and are sites of recent lynx observation. Identify such sites as species strongholds, and use them as the backbone of a metapopulation strategy (see vol. 1).

Group 17—Blue Grouse (Summer) and Mountain Quail (Summer)

Results

Species ranges, source habitats, and special habitat features—Group 17 consists of summer habitats for both blue grouse and mountain quail. The range of the blue grouse includes the western, northern, central, and eastern portions of the basin (fig. 51). The range of the mountain quail includes southern Washington, Oregon, and western Idaho (fig. 51; Ehrlich and others 1988). Blue grouse are ground nesters that forage primarily on seeds, berries, and insects; the young feed heavily on insects (Ehrlich and others 1988). Mountain quail are also ground nesters and feed primarily on bulbs, greens, and insects (Ehrlich and others 1988).

Source habitats for group 17 include all structural stages except stem exclusion of interior Douglas-fir, Sierra Nevada mixed conifer, and Pacific and interior ponderosa pine (vol. 3, appendix 1, table 1). In addition, blue grouse source habitats also include western larch, aspen, mixed-conifer woodlands, antelope bitterbrush-bluebunch wheatgrass, and wheatgrass bunchgrass. Chokecherry-serviceberry-rose is also source habitat for both species.

A special habitat feature for the mountain quail is riparian shrub (vol. 3, appendix 1, table 2). Mountain quail within the basin primarily are found within 100 to 200 m (328 to 656 ft) of a water source (Brennan 1989). The blue grouse (summer) is considered a contrast species as it is typically found at the interface of forest and open areas (Zwickel 1992; vol. 3, appendix 1, table 2).

Broad-scale changes in source habitats—Source habitats for blue grouse (summer) and mountain quail (summer) occur primarily in the forested ERUs across the basin (fig. 52A and 52B). The overall trend in source habitats since historical times has been neutral (fig. 53), with increasing trends occurring primarily in the western and southeastern part of the basin, and more decreasing trends occurring in the northeast part of the basin. The ERUs with increasing trends are the Southern Cascades, Upper Klamath, Northern Great Basin, Upper Snake, and Snake Headwaters. The ERUs with decreasing trends are the Lower Clark Fork, Upper Clark Fork, and Central Idaho Mountains. The remaining ERUs are overall neutral (Northern Cascades, Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, and Owyhee Uplands).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats—Increases in source habitats in the Northern Cascades are primarily because of increases in managed young forests of interior Douglas-fir and interior ponderosa pine, whereas a similar decline occurred in old-forest ponderosa pine (vol. 3, appendix 1, table 4). Increases in source habitats in the Southern Cascades, Upper Klamath, and Blue Mountains, and southern portions of the Columbia Plateau are due primarily to increases in old forest. Decreases in source habitats in much of the northeastern part of the basin are due to declines in both late- and early-seral community types.

The primary changes in source habitats in the Upper Snake were an increase in wheatgrass bunchgrass (vol. 3, appendix 1, table 4). Hann and others (1997), however, suspect that in some areas that show increases in upland herblands (including wheatgrass bunchgrass), these areas may in fact be areas of early-seral forests attributable to relatively recent timber harvest or large-scale wildfires, and were misclassified as upland herbland. In such a case, recent timber harvest or wildfire may have increased the quantity and quality of source habitat because of potential increases in

VIEW

MAP

Figure 51—Ranges of species in group 17 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

VIEW
MAP

Figure 52—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 17 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 53—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 17, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

shrubs. Increases in wheatgrass bunchgrass, however, also may be attributable to increases in exotic wheatgrasses such as crested wheatgrass, which does not provide source habitat for blue grouse. The increase in source habitat in the Snake Headwaters is primarily due to an increase in both early- and mid-seral interior Douglas-fir (vol. 3, appendix 1, table 4).

Condition of special habitat features— Basin-wide analysis of riparian vegetation found significant changes, including widespread declines in riparian shrublands (Quigley and others 1996). Because of the scale of our analysis and the fine-scale nature of riparian shrubland habitats, likely the results of our analysis do not reveal the true loss in this important habitat component for mountain quail. Water impoundments, grazing, residential developments, and agricultural activities can alter the extent, composition, and structure of mountain quail habitat (Brennan 1990, Murray 1938, Vogel and Reese 1995). Remaining habitat in the basin is fragmented, and populations exist often in islands of habitat connected by narrow corridors of vegetation (Vogel and Reese 1995).

Because the blue grouse (summer) is a forest-open areas contrast species, the scale of this analysis does not allow determination of change in the juxtaposition of these contrasting habitats. Thus, this special habitat feature is not accounted for in the results presented above, and a finer scale analysis is needed to fully evaluate the status of their source habitats. A loss of interspersion of early- and late-seral stages of forest partly because of altered fire regimes was identified by Lehmkuhl and others (1997) as a reason for a declining trend since the historical period of both habitat and populations of the blue grouse.

Other factors affecting the species— Some mountain quail populations migrate to lower elevations to winter (Brennan 1990, Ehrlich and others 1988, Leopold and others 1981). Winter habitat availability may be more limited than summer habitat because of severe winter weather in some mountainous areas (Edminster 1954). Low-elevation riparian shrub habitat is especially important during severe winters. Hydroelectric impoundments along the Columbia River and its tributaries have flooded thousands of acres of low-elevation winter habitat for mountain quail (Brennan 1990). One of the last remaining Idaho populations can be found along the Salmon River drainage in an area that experiences mild winters, thought to be one of the important variables for the continued presence of quail in this area (Brennan 1989).

Both blue grouse and mountain quail most often are found in areas with a high abundance of shrubs, which most likely are used for cover as well as forage (Brennan and others 1987, Zwickel 1992). Traditional forest managers commonly replanted harvested areas, thus hastening the rate of succession and shortening the time that a stand remains in the early-seral stage (Hann and others 1997). This practice, coupled with ground-disturbing site preparation before planting, often eliminates the herb, forb, and shrub structures from stands. Management activities such as salvage logging and planting in postfire habitats also may shorten the duration of these early-seral, shrub-dominated sites.

Grazing of domestic livestock may negatively impact blue grouse (Mussehl 1963, Zwickel 1972), as well as mountain quail (Brennan 1990).

The frequency and areal extent of wildfires declined since the early to mid 1900s because of suppression activities (Hann and others 1997). With the increased fuel loads in fire-suppressed areas, however, the trend since 1960 has changed, and the current extent of wildfires is approaching that of the early 1900s. This increase in postfire areas should benefit both blue grouse and mountain quail if these fires result in an increase in shrub vegetation.

Both species are negatively affected by human disturbance, primarily during the nesting/brood-rearing season (ICBEMP 1996h). The human population in the basin is estimated at 3 million, which is a substantial increase from the pre-European settlement period (McCool and others 1997). This change in population increases human encounters, thus having a potentially negative effect on both blue grouse and mountain quail. In particular, the introduction of human residents to an area also introduces domestic cats, an effective predator of mountain quail (Edminster 1954, Jewett and others 1953, McLean 1930.)

There are open hunting seasons for blue grouse throughout the basin, whereas hunting for mountain quail is only allowed in some parts of Oregon.

Population status and trends—Blue grouse still occupy most of their original range, although historical populations may have been stronger in some areas (Zwickel 1992). Although mountain quail populations to the west of the basin seem to be stable, populations in the basin have experienced dramatic declines (Brennan 1990, Robertson 1989, Washington Department of Wildlife 1993a).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 17 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Issues identified for group 17 were based on our analysis of source habitats as well as knowledge of finer scale habitat features for these species:

1. Decline in late- and early-seral source habitats, particularly in the northeastern part of the basin.

2. Changes in vegetation composition and structure of understory shrub habitat.

3. Loss of riparian shrubs.

4. Increased interaction with humans.

5. Isolated and disjunct populations of mountain quail vulnerable to extinction by stochastic events (that is, demographic, environmental, or genetic stochasticity).

Proposed strategies—

1. (To address issue no. 1) Maintain and restore late-seral montane and lower montane forests.

2. (To address issues no. 1 and no. 2) Increase the representation of shrub-dominated early seral forests.

3. (To address issues no. 1 and no. 2) Restore fire as an ecological process in the montane and lower montane community groups.

4. (To address issue no. 3) Maintain and restore riparian shrubland habitats, including protecting existing areas from the encroachment of exotics.

5. (To address issue no. 3) Reduce habitat degradation by livestock grazing in areas currently occupied by mountain quail.

6. (To address issue no. 4) Restrict human access in areas of known nesting use by blue grouse and mountain quail.

7. (To address issue no. 5) Expand the current range of mountain quail within their historical range.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Maintain existing old forests until mid-seral forests have developed into old forests at a level that is within the range of historical variability.

2. (In support of strategy no. 2) Leave some postfire areas unaltered to regenerate naturally.

3. (In support of strategy no. 3) Use prescribed fire to enhance growth and regeneration of understory or mountain shrub development. Avoid burning during the nesting season, as fires can cause direct mortality to mountain quail (Clark 1935, McLean 1930, Spaulding 1949).

4. (In support of strategy no. 4) Reduce exotic weed invasions by plantings of native shrub and herbaceous vegetation in riparian shrubland habitats.

5. (In support of strategy no. 5) Remove or explicitly control the timing and intensity of grazing to discourage weed invasions and to minimize losses and allow for restoration of native riparian and mountain shrubs.

6. (In support of strategy no. 6) Reduce road densities and timing of management activities to reduce human interactions with these species, especially

Figure 54—-Ranges of species in group 18 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

during the nesting and brooding season. In addition or as an alternative to reductions in road density, implement seasonal road closures during nesting and brooding periods.

7. (In support of strategy no. 7) Reintroduce and augment populations of mountain quail after habitat enhancement.

Group 18—Lazuli Bunting

Results

Species ranges and source habitats— Group 18 consists of the Lazuli bunting, a migratory breeder that occurs throughout the basin (fig. 54). Source habitats for Lazuli buntings are grass-forb-shrub edges, burns, early-seral stages of conifer forest, and dense, low vegetation along streams (Sharp 1992). Hutto (1995) found that Lazuli buntings demonstrated a strong positive response to early successional burned forests, resulting from stand-replacing fires that occurred in a broad spectrum of coniferous forest types across western Montana and northern Wyoming. This bunting was also a common nesting species in recently burned ponderosa pine/Douglas-fir forests of western Idaho (Saab and Dudley 1998). The Lazuli bunting is a shrub-nesting insectivore, foraging primarily by gleaning off foliage (Ehrlich and others 1988).

Source habitats analyzed in this report are the stand-initiation stage of the montane, lower montane, riparian woodland terrestrial communities and also chokecherry-serviceberry-rose (vol. 3, appendix 1, table 1). Among landscape and microhabitat features of cottonwood forests in eastern Idaho, the most important predictor of Lazuli bunting occurrence was shrub density and cover (Saab 1999). Other significant predictors of their occurrence included herbaceous ground cover and willow subcanopies, providing foraging and nesting habitat, respectively. Additionally, their relative abundance was significantly reduced in forest patches managed for grazing compared with unmanaged patches (Saab 1996, 1998). In cottonwood forests of western Montana, the abundance of Lazuli buntings also was reduced in heavily grazed areas, as compared to lightly grazed areas (Mosconi and Hutto 1981).

Broad-scale changes in source habitats—Historically source habitats for group 18 were broadly distributed throughout the mountainous regions of the basin, though most watersheds with source habitats contained less than 25 percent area in source habitats (fig. 55A). Currently, source habitats are more patchily distributed and absent from many watersheds that historically contained these habitats (fig. 55B).

The trend in source habitats was negative to strongly negative for nearly 60 percent of the watersheds in the basin (figs. 55C and 56). About 33 percent of the watersheds basin-wide had positive trends in source habitats (fig. 56). Eight ERUs had negative to strongly negative trends, including the Upper Klamath, Northern Great Basin, Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and Upper Snake. Trends were neutral in the Southern Cascades and Owyhee Uplands. Three ERUs, the Northern Cascades, Snake Headwaters, and Central Idaho Mountains, had positive trends.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats—For the ERUs with positive trends, increased area of various cover types, especially Douglas-fir, Englemann spruce, lodgepole pine, and aspen, were responsible for the trend (vol. 3, appendix 1, table 4). For the eight ERUs with negative trends, the loss of early-seral Douglas-fir, lodgepole pine, interior ponderosa pine, and western larch contributed most to the trend. Nearly 100 percent of the western larch stand-initiation stage was eliminated in these ERUs.

In addition, basin-wide declines have occurred in riparian woodlands at the broad scale (Hann and others 1997). Smaller patches of riparian vegetation, especially riparian shrublands, have declined in extent basin-wide because of disruption of hydrologic regimes from dams, water diversions, and road construction. Additionally, grazing and trampling of riparian vegetation by livestock, and increased recreational use along stream courses have reduced riparian habitats (USDA Forest Service 1996). Low-elevation wetlands in Idaho are considered “endangered” based on a 85- to 98-percent decline since European settlement (Noss and others 1995)

Other factors affecting the group—Traditional forestry practices commonly tried to accelerate the regeneration process in harvested areas by planting, thus hastening the rate of succession and shortening the time that a stand remained in the early-seral stage (Hann and others 1997). This practice coupled with ground-disturbing site-preparation activities before planting often eliminated the herb, forb, and shrub structure from stands. Planting in postfire habitats also shortens the duration of the stand-initiation stage. Salvage logging in postfire habitats may reduce the availability of tall structures used for singing perches. Hutto (1995) found that the relative abundance of many bird species, including the Lazuli bunting, differed between recently burned and recently harvested forests. Composition of trees, snags, and shrubs subsequent to a burn can differ depending on fire intensity and postfire timber harvest.

According to Hann and others (1997), the frequency and areal extent of wildfires declined since the early to mid 1900s because of suppression activities. With the increased fuel loads in fire-suppressed areas, however, the trend since 1960 has changed, and the current extent of wildfires is approaching the early 1900s. This increase in postfire areas should benefit Lazuli buntings if these fires result in an increase in shrub vegetation.

Lazuli buntings are Neotropical migratory birds. The availability of suitable habitats used during migration, as well as their winter habitat, are critical components. Status of habitats, effects of nonhabitat factors on populations, and management practices in migratory and wintering areas are, however, unknown.

Population status and trends—Recent BBS data indicate that the population was stable from 1968 to 1994 ($n \geq 14$; $P < 0.10$) across the basin (Saab and Rich 1997). Sauer and others (1996) identified increasing trends for Lazuli buntings in the western United States from 1980 to 1995 (+2.9 percent per yr, $n = 147$; $P < 0.01$).

Figure 55—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 18 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 56—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 18, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 18 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The results of our habitat trend analysis and the literature suggest the following issues are of high priority for group 18:

1. Altered frequency of stand-replacing fires.
2. Loss of shrub-dominated early-seral vegetation types.
3. Loss and degradation of riparian vegetation.

Potential strategies—The issues suggest the following broad-scale strategies would be effective in supporting the long-term persistence of the Lazuli bunting. Strategies would apply basin-wide.

1. (To address issue no. 1) Restore fire as an ecological process in the montane and lower montane community groups. Natural fire frequencies and intensities should be considered where appropriate.
2. (To address issue no. 2) Increase the representation of shrubs in the early-seral stages of forest communities.
3. (To address issue no. 3) Reduce impacts to shrubs from grazing, recreation, and other activities.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Leave some postfire and postharvest areas unaltered to regenerate naturally.
2. (In support of strategy no. 2) Use prescribed fire to increase the representation of shrubs in the early-seral stages of forest communities.
3. (In support of strategy no. 3) Remove or explicitly control the timing and intensity of grazing to develop and promote the long-term persistence of shrub communities.
4. (In support of strategy no. 3) Restrict activities in riparian areas that negatively affect riparian vegetation. Areas that currently support healthy shrub communities should be a priority for conservation.

Group 19—Gray Wolf and Grizzly Bear

Results

Species ranges and source habitats—Group 19 consists of the grizzly bear and gray wolf. Historically these two species ranged across most of the basin (fig. 57), although use of lower elevations within the Northern Great Basin and Owyhee Uplands ERUs was probably incidental. This distribution has been greatly reduced, and both species currently persist only in small, disjunct populations. Gray wolf populations occur in western Montana, central Idaho, and western Wyoming; grizzly bear populations remain in the northern Cascades, northern Idaho, western Montana, and western Wyoming (fig. 57).

The grizzly bear was listed as federally threatened under the ESAon July 28, 1975. The original recovery plan was approved in 1982 and amended in 1993. The northern Rocky Mountain gray wolf was listed as endangered on June 4, 1973, and a recovery plan was released in 1987 (USDI Fish and Wildlife Service 1987). Wolves have been state protected in Montana since 1975 and in Idaho since 1977 (USDI Fish and Wildlife Service 1987).

Source habitats for group 19 span a broad elevational range and include all terrestrial community groups except exotic herbland and agriculture. About 80 percent of all possible cover type-structural stage combinations are source habitats (vol. 3, appendix 1, table 1).

Source habitats for wolves must include suitable denning and rendezvous sites and a sufficient, year-round prey base of ungulates and alternate prey (USDI Fish and Wildlife Service 1987). Den sites are used for rearing pups and are typically near forested cover and removed from human activity. Wolves are sensitive to human disturbance near dens from mid-April to July

Figure 57—-Ranges of species in group 19 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

(Weaver and others 1996). Rendezvous sites are resting and gathering areas used by wolf packs after the pups are mobile and typically include meadow vegetation and adjacent forest with resting sites under trees (USDI Fish and Wildlife Service 1987). Home ranges can be exceedingly large, based on estimates from radio telemetry. In Minnesota for example, home range estimates ranged from 49 to 135 km^2 (19 to 52 mi^2) (Van Ballenberge and others 1975), and in Alberta, winter home ranges varied between 357 and 1779 km^2 (138 and 687 mi^2) (Fuller and Keith 1980). The principal foods of wolves in the Rocky Mountains are deer, elk, and moose (USDI Fish and Wildlife Service 1987; Weaver 1994, cited in Weaver and others 1996).

Grizzly bear habitat selection is affected by (1) abundance and quality of foods; (2) gender-specific orientation to different nutrients; (3) reproductive status of females and concerns about security of dependent young; (4) presence and identity of other bears, especially adult males; and (5) presence of humans and prior contact with humans.[5] Grizzly bears are omnivorous, but their use of certain high-quality foods with limited spatial or temporal distribution often results in seasonal shifts in habitat selection (Hamer and Herrero 1987; Mace and others 1996; Mattson and others 1991a, 1991b; McLellan and Hovey 1995; Servheen 1983). Also, food availability fluctuates among years, and habitat selection may therefore differ from one year to the next (Green and others 1997; Mattson and others 1991a, 1991b; McLellan and Hovey 1995).

A selection process also seems to be used for the location of dens for hibernation and the birth and rearing of young. Typical dens are either dug by bears or occur in natural cavities in subalpine, montane, and rock community groups. Den sites tend to be clustered, thereby suggesting that certain areas possess more favorable combinations of environmental factors for denning (USDI Fish and Wildlife Service 1993). Grizzly home ranges encompass large areas. For example, based on several studies, annual home ranges of males in the Northern Continental Divide Ecosystem are between 165 and 1406 km^2 (64 and 543 mi^2), with an average of 489 km^2 (189 mi^2) (USDI Fish and Wildlife Service 1993).

Broad-scale changes in source habitats— Source habitats for the grizzly bear and gray wolf likely occurred throughout the basin historically (fig. 58A). The current extent of habitat, albeit largely unoccupied, is similar to the historical distribution except for the Columbia Plateau, Lower Clark Fork, and Upper Clark Fork ERUs, where habitat is more patchily distributed than it was historically (fig. 58B).

Basin-wide, the overall trend in source habitats for group 19 was neutral (fig. 58C). Source habitats remained relatively stable in 9 of 13 ERUs (figs. 58C and 59). Fifty percent of all watersheds, located primarily in the southern half of the basin and along the western and northern borders, showed no trend in habitat (fig. 59). Source habitats were projected to have decreased in four ERUs: the Columbia Plateau, Lower Clark Fork, Upper Clark Fork, and Upper Snake (fig. 59).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Despite the overall neutral trend for source habitats for group 19, many of the terrestrial communities were projected to have changed dramatically from historical conditions. In general, mid-seral forests increased in areal extent basin-wide, whereas both early- and late-seral forests declined (Hann and others 1997). Some forest cover types, including western white pine, whitebark pine, western larch, and limber pine no longer occur in stands large enough to map at the broad scale, whereas Pacific silver fir-mountain hemlock and western redcedar-western hemlock increased, respectively, 1,700 and 853 percent basin-wide (Hann and others 1997).

Within nonforest terrestrial communities, upland herbland and upland shrubland both strongly declined, whereas three new terrestrial communities, urban, agriculture, and exotic herbland, have emerged since the historical period (Hann and others 1997). Examples of declining nonforest cover types are native forb and mountain big sagebrush, which declined, respectively, by 91 and 34 percent basin-wide (Hann and others 1997). Within the four ERUs having overall declining trends in source habitats for group 19, declines were mostly in western white pine, whitebark pine, western larch, limber pine, big sagebrush, and native forb (vol. 3, appendix 1, table 4).

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[5] Personal communication. 1998. David Mattson, U.S. Geologial Survey, Forest and Rangeland Ecosystem Science Center and Department of Fish and Wildlife Resources, University of Idaho, Moscow, ID 83844-1136.

Figure 58—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 19 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 59—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 19, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Other factors affecting the group— Human-caused mortality is the major factor limiting the recovery of wolves and grizzly bears (Fritts and Mech 1981; Knight and others 1988; Mattson and others 1996a, 1996b; Pletscher and others 1997; USDI Fish and Wildlife Service 1987, 1993). About 84 percent of all known mortalities of wolves on the Montana-British Columbia-Alberta border were human caused, primarily legal shootings in Canada (Pletscher and others 1997). In the northern Rockies, between 1974 and 1996, 85 to 94 percent of all deaths of marked grizzly bears >1 year old were due to humans (Mattson and others 1996a).

For wolves, human-caused losses are due to shooting, trapping, and vehicle accidents (Fritts and others 1985). Six of the nine mortalities that occurred in the first 20 months after the reintroduction into Yellowstone National Park were human caused: three wolves were illegally shot, one was killed by Animal Damage Control personnel after repeated sheep depredations, and two were killed by vehicles (Bangs and Fritts 1996). In many cases, wolf mortalities are related to real and perceived depredations of livestock.

For grizzly bears, human-caused mortalities stem from (1) direct human-bear conflicts in wilderness areas and parks (for example, hikers, photographers, or hunters); (2) attraction of grizzly bears to improperly stored food or garbage; (3) attraction of grizzly bears to improperly disposed dead livestock; (4) chance interactions between livestock and grizzly bears; (5) increased human occupancy of grizzly bear habitat, causing increased interactions and stress; and (6) hunting (USDI Fish and Wildlife Service 1993). Legal hunting of grizzly bears no longer occurs in the basin, but grizzly bears are taken by poachers and occasionally are mistakenly shot during the black bear hunting season.

Wolves, particularly juveniles, are susceptible to canine parvovirus and distemper, and these diseases could affect recovery in the northern Rocky Mountains if not monitored (USDI Fish and Wildlife Service 1987). Parasites and diseases do not appear to be significant causes of natural mortality of grizzly bears (Jonkel and Cowan 1971, Rogers and Rogers 1976, both cited in USDI Fish and Wildlife Service 1993).

Both species are negatively affected by roads. Roads per se are not a physical barrier; wolves use gated roads as travel corridors (Thurber and others 1994), and grizzly bears in Montana exhibit neutral or positive selection for areas with roads having <10 vehicles per day (Mace and others 1996). Roads, however, usually increase human presence and the likelihood of negative contacts. A disproportionate number of human-caused mortalities occur near roads, both for wolves (Mech 1970, as cited in Frederick 1991) and grizzly bears (Mattson and others 1996a). These mortalities are mostly legal and illegal shootings resulting from human access provided by roads (Mace and others 1996, McLellan and Shackleton 1988); vehicle collisions also play a role (Bangs and Fritts 1996, Knight and others 1988). Thurber and others (1994) cited three studies (Jensen and others 1986, Mech and others 1988, Thiel 1985) indicating wolf packs would not persist where road densities exceeded about 1.0 mi per mi^2 (0.6 km per km^2).

An additional, indirect effect of roads is that road avoidance leads to underutilization of habitats that are otherwise high quality. Mace and others (1996) found that grizzly bears in Montana avoided roads having >10 vehicles per day. In southeastern British Columbia, grizzly bears underutilized about 9 percent of available habitats by avoiding areas 100 m (328 ft) from roads, regardless of traffic volume (McLellan and Shackleton 1988). Several other studies have documented road avoidance by grizzly bears in or near the basin (Green and others 1997, Kasworm and Manly 1990, Mattson and Reinhart 1997, Mattson and others 1987). Similar effects have been observed with wolves: packs in the Great Lakes region avoided habitats with high road and human densities even though densities of deer, a principal prey, were also high in these areas (Mladenoff and others 1995). In northern Montana, wolf travelways were at least 4 to 22 km (2.5 to 13.6 mi) from the nearest driveable road, which precluded their use of otherwise high-quality habitats and food resources (Singer 1979).

Road access also increases the likelihood of habituation to humans. Individual wolves and grizzly bears can become accustomed to human presence, leading to nuisance situations that can result in the death of the habituated animal (Mattson and others 1992, Meagher and Fowler 1989).

The neutral trends in source habitats projected for the basin do not reflect loss of habitat effectiveness because of roads and human activities. Road densities in the basin have substantially increased from historical levels and are estimated to be moderate to high in most ERUs (Hann and others 1997). Moreover, the human population in the basin has increased and is estimated currently at 3 million (McCool and others 1997). The increase in road densities and human population are believed responsible for the unoccupied state of many source habitats of grizzly bears and wolves in the basin. For example, Merrill and others (1999) included roads, level of human activity, and distance and size of nearby human populations in their model of environmental suitability for grizzly bears in Idaho.

The demographic impact of human-caused mortality is intensified for grizzly bears by their low reproductive rate. Litters range from one to four cubs with an average of two, and females generally do not begin to reproduce until 5.5 yr old (USDI Fish and Wildlife Service 1993). Each female has the limited potential of adding three to four females to a population during her lifetime (USDI Fish and Wildlife Service 1993). Using this demographic information in conjunction with behavioral plasticity in food acquisition and dispersal capabilities, Weaver and others (1996) concluded that grizzly bears have fairly low resiliency to human disturbances, whereas gray wolves, based on these same factors, are moderately resilient.

Lack of connectivity among habitat reserves is a major factor affecting the long-term persistence of grizzly bears, and perhaps also wolves (Noss and others 1996). Source habitats are currently fragmented by human disturbances to a level where interchange within the entire regional population occurs rarely if at all (Noss and others 1996). Small, isolated populations are susceptible to extirpation from inbreeding, chance breeding events (for example, no female births in a given year), and environmental uncertainty (for example, drought or disease) (Shaffer 1981). This appears to be a concern for small, isolated grizzly bear populations (Allendorf and others 1991, cited in Mattson and others 1996b). Insufficient connectivity among local populations reduces the likelihood of recolonization once a population has been extirpated. The Bitterroot ecosystem is an example of a recent extirpation with extremely low probability of recolonization because of lack of connectivity with other grizzly bear populations (Merrill and others 1999).

Ultimately, human attitudes towards wolves and grizzly bears are what will ensure their survival or extirpation (Bangs and Fritts 1996, Mattson and others 1996a). Many of the negative effects of roads and human activities could be diminished through changes in human attitudes and behavior (Mattson and others 1996a, 1996b).

Population status and trends—Wolf populations were reduced to near extinction within the basin during the 1800s to early 1900s (USDI Fish and Wildlife Service 1987). Wolf numbers have increased, how ever, within the last 10 years. In addition to natural recolonizations of historical habitats in Washington, Idaho, and northwestern Montana (Marcot and others 1997), wolves have been reintroduced to central Idaho and the Yellowstone area as nonessential experimental populations (Federal Register 1994) beginning in 1995. Natural and experimental populations are currently doing well in all three areas identified for recovery: northwestern Montana, north-central Idaho, and the Greater Yellowstone Ecosystem. As of 1999 (USDI Fish and Wildlife Service 1999), northwestern Montana had about 65 wolves and 5 confirmed breeding pairs; central Idaho contained 140 wolves and 10 confirmed breeding pairs; and the Yellowstone ecosystem contained about 120 wolves and 8 breeding pairs.

Between 1800 and 1975, grizzly bear populations in the lower 48 States receded from estimates of over 100,000 to <1,000 bears (USDI Fish and Wildlife Service 1993). Extirpations within the basin include Utah (1923) and Oregon (1931) (USDI Fish and Wildlife Service 1993). The Interagency Grizzly Bear Committee[6] identified five recovery zones south of Canada where grizzly bears and grizzly habitat are managed for recovery, and within which the population parameters will be monitored (Interagency Grizzly Bear Committee 1998). The recovery zones are referred to as ecosystems to emphasize the ecological rather than jurisdictional nature of their boundaries (USDI Fish and Wildlife Service 1993). Four of the recovery zones are within the basin—the Northern Cascades, Selkirk, Cabinet-Yaak, and Northern Continental Divide ecosystems—and the fifth, the Yellowstone ecosystem, occurs on the eastern border

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[6] The Interagency Grizzly Bear Committee is composed of top officials from the U.S. Department of the Interior, Fish and Wildlife Service, National Park Service, Bureau of Land Management, Bureau of Indian Affairs; U.S. Department of Agriculture, Forest Service; state fish and game agencies of Montana, Wyoming, Idaho, and Washington; and management authorities from British Columbia and Alberta.

of the basin. The Selway-Bitterroot ecosystem is under consideration as a recovery zone, as outlined in the Final Environmental Impact Statement for Grizzly Bear Recovery in the Bitterroot Ecosystem (USDI Fish and Wildlife Service 2000).

Grizzly bear population estimates currently are available only for the Northern Continental Divide Grizzly Bear ecosystem (440 to 680 bears) (USDI Fish and Wildlife Service 1993) and the Yellowstone ecosystem (280 to 610 bears) (Eberhardt and Knight 1996). The Selkirk Mountains and Cabinet-Yaak ecosystems are believed to have breeding populations based on sightings of females with young, but populations within each ecosystem may be less than 20 grizzly bears (Knick and Kasworm 1989, Wielgus and Bunnell 1995). Population status within the Northern Cascades is unknown (Interagency Grizzly Bear Committee 1998, USDI Fish and Wildlife Service 1993). No grizzly bears currently live in the Bitterroot Mountains of Idaho (Interagency Grizzly Bear Committee 1998).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 19 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The following issues have been identified as major challenges to the conservation of the grizzly bear and gray wolf:

1. Excessive mortality from conflicts with humans.

2. Excessive mortality related to the presence of roads (accidents, poaching, and increased conflicts).

3. Displacement from suitable habitats because of human activities.

4. Isolation of populations within each recovery area.

The goal of the revised Grizzly Bear Recovery Plan is to identify actions necessary for the conservation and recovery of the grizzly bear and to remove the grizzly bear from threatened status in each recovery zone (USDI Fish and Wildlife Service 1993). The goal of the recovery plan for gray wolves is to remove the Northern Rocky Mountain wolf from the endangered and threatened species list by securing and maintaining a minimum of 10 breeding pairs of wolves in each of the three recovery areas for a minimum of 3 successive years (USDI Fish and Wildlife Service 1987).

Potential strategies—The following strategies could be used in the Northern Cascades, Northern Glaciated Mountains, Upper Clark Fork, Lower Clark Fork, Central Idaho Mountains, and Snake Headwaters ERUs to support recovery of the gray wolf and grizzly bear:

1. (To address issue no. 1) Reduce the prevalence of conflict situations and the number of human-caused mortalities of bears and wolves. Provide secluded habitats that reduce the potential for conflicts with humans.

2. (To address issue no. 2) Develop a policy for road construction, maintenance, and obliteration on public lands within gray wolf and grizzly bear recovery areas and in source habitats that surround and could potentially connect these habitats.

3. (To address issue no. 3) Reduce human activities in important grizzly bear foraging areas and around known wolf dens.

4. (To address issue no. 4) Provide interregional habitat connectivity across all ERUs with wolf and bear populations (Northern Cascades, Northern Glaciated Mountains, Upper Clark Fork, Lower Clark Fork, Central Idaho Mountains, and Snake Headwaters).

Practices that support strategies— Action items and practices for the recovery of the gray wolf and grizzly bear are in the Northern Rocky Mountain Wolf Recovery Plan (USDI Fish and Wildlife Service 1987), the Grizzly Bear Recovery Plan (USDI Fish and Wildlife Service 1993), the Interagency Grizzly Bear Guidelines (Interagency Grizzly Bear Committee 1986) and the Grizzly Bear Compendium (LeFranc and others 1987). The following practices have been drawn from these documents as examples and would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Alter the timing and location of livestock grazing to reduce the need for wolf and grizzly bear depredation control.

2. (In support of strategy no. 1) Implement sanitation practices, including law enforcement to support these practices, to minimize the likelihood of grizzly bear attraction to human food, garbage, and dead livestock.

3. (In support of strategy no. 1) Increase extent and scope of public education programs regarding the role of human-bear and human-wolf conflicts in the conservation of these species.

4. (In support of strategies no. 1 and no. 2) Minimize or avoid road construction within unroaded areas in grizzly bear ecosystems and wolf recovery areas. Obliterate or restrict use of roads in important seasonal habitats, such as low-elevation riparian areas (spring habitat for grizzly bears).

5. (In support of strategies no. 1 and no. 3) Reduce or temporarily discontinue activities such as livestock grazing, timber harvests, backcountry use, mining, and oil and gas development in important grizzly bear foraging areas during peak foraging periods. Restrict human access near wolf dens from April 15 to July 1.

6. (In support of strategy no. 4) Use concepts described in Noss and others (1996) to create habitat connectivity among recovery areas. Identify existing and potential dispersal corridors for wolves and bears, and seek opportunities with all landowners and affected parties to modify the timing, intensity, and location of human activities within these corridors.

Group 20—Mountain Goat

Results

Species ranges, source habitats, and special habitat features—Group 20 consists of the mountain goat, a year-round resident of the basin. Within the basin, the mountain goat occurs in the mountains of central and northeast Washington, northeast Oregon, central and northern Idaho, and western Montana. These areas correspond to five ERUs: the Northern Cascades, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, and Central Idaho Mountains (fig. 60). The range also includes small, bordering areas of the Southern Cascades and Columbia Plateau ERUs (fig. 60). Most populations are native, but mountain goats have been introduced into portions of Montana, and reintroduced into the Elkhorn and Blue Mountains of Oregon. Although the Hells Canyon population stems from a transplant, recent archeological evidence suggests historical occupancy of the Hells Canyon area and the Wallowa Mountains (Matthews and Coggins 1994).

Source habitats for mountain goats include 15 cover types within six community groups: alpine, subalpine forest, montane forest, lower montane forest, upland shrubland, and rock-barren (vol. 3, appendix 1, table 1). Mountain goats show no apparent preference for any cover type, as long as they occur on steep terrain or near cliffs and talus. Mountain goats seem to use all structural stages within forested cover types except for the stem-exclusion stage of montane and lower montane forests (vol. 3, appendix 1, table 1). Upland shrublands provide important foraging habitat, and forests provide both foraging habitat and protection from inclement weather (Johnson 1983).

Special habitat features identified for mountain goats are cliffs, talus, and seasonal wetlands (vol. 3, appendix 1, table 2). Cliffs and talus are central to mountain goat distribution and habitat use (Hjeljord 1973). Cliffs provide escape terrain from predators (Johnson 1983, Rideout 1978), and both cliffs and talus provide foraging areas with little competition from other herbivores (Rideout 1978).

Mountain goats forage on various plant species depending on local and seasonal availability. Grasses and sedges comprise a major portion of the diet in most locales (Adams and Bailey 1983, Hjeljord 1973, Saunders 1955), along with mosses, lichens, ferns, and shrubs (Rideout 1978). Mountain goats exhibit localized shifts in habitat use in response to changes in food availability because of snow accumulation, moisture, wind, and solar exposure (Rideout 1978). Mountain goats are subject to predation from mountain lions, golden eagles (Rideout 1978), wolves, and grizzly bears (Smith 1986, Smith and others 1992).

Broad-scale changes in source habitats—The following trends in source habitats for mountain goats were derived without reference to the proximity of cliffs and talus and therefore include habitat patches

Figure 60—Ranges of species in group 20 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

that are not available to mountain goats. Trends derived from a restricted subset of habitats near cliffs could differ substantially in magnitude from those reported here, but the general direction of the trends likely would remain the same.

The historical distribution of source habitats was essentially the same as it is now, occurring in the mountains of central and northeast Washington, northeast Oregon, central and northern Idaho, and western Montana (figs. 61A, and 61B). Because mountain goats use various cover types, trends in the extent of source habitats differed basin-wide. Trends were projected to be neutral in 32 percent of the watersheds and positive in 42 percent of the watersheds basin-wide (fig. 62). Positive trends were projected in more than 50 percent of watersheds in the Blue Mountains and Central Idaho Mountains ERUs, and declining trends were most prevalent in the Lower Clark Fork and Upper Clark Fork ERUs (figs. 61C and 62). All other ERUs with source habitats exhibited mixed trends.

Source habitats for mountain goats were most prevalent in the Northern Cascades ERU historically, and this has not changed. The area occupied by source habitats in this ERU comprised 51 percent of the area of watersheds included in mountain goat range during both time periods (vol. 3, appendix 1, table 3).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Neutral trends in source habitats were partly because alpine and rock-barren community groups did not change in areal extent from historical to current periods (Hann and others 1997; vol. 3, appendix 1, table 4). Within other community groups, neutral trends resulted from declines in some cover types that were offset by increases in other cover types used as source habitats. For example, in the Northern Cascades ERU, a major transition occurred from interior ponderosa pine to both interior Douglas-fir and grand fir-white fir (Hann and others 1997), but this resulted in static trends in habitat extent because all three cover types are source habitats (vol. 3, appendix 1, table 1).

Declining trends in the Lower Clark Fork ERU were due to total losses at the broad scale of old forests of interior ponderosa pine, as well as declines in the stand-initiation stage of lodgepole pine and Engelmann spruce-subalpine fir (vol. 3, appendix 1, table 4). Declines in the Upper Clark Fork were chiefly because of nearly total losses of interior Douglas-fir and interior ponderosa pine old forests (vol. 3, appendix 1, table 4). Although less extensive in area, strong declines in whitebark pine old forests also occurred in both the Lower and Upper Clark Fork ERUs (vol. 3, appendix 1, table 4). In the Central Idaho Mountains, increases in source habitat were primarily due to areal increases in Engelmann spruce-subalpine fir, grand fir-white fir, interior Douglas-fir, lodgepole pine, mountain mahogany, and shrub or herb-tree regeneration (vol. 3, appendix 1, table 4). Increases in the Blue Mountains were associated mostly with increases in grand fir-white fir (Hann and others 1997).

Condition of special habitat features— The areal extent of cliffs and talus has not changed between historical and current periods (Hann and others 1997). Seasonal wetlands are highly dependent on annual hydrologic cycles and therefore have fluctuated widely in occurrence and productivity over time.

Other factors affecting the group— Young of the year and yearlings incur the highest mortality rates, primarily because of harsh weather in conjunction with predation, internal parasites, and diseases (Johnson 1983). Adults are highly susceptible to hunting mortality, both legal and illegal (Kuck 1977, Matthews and Coggins 1994, Smith 1986, Swenson 1985).

Human activities disrupt mountain goats and can cause displacement from source habitats. Low-flying aircraft cause mountain goats to run, take alert defense postures, or take refuge under trees (Chadwick 1973). Road blasting and sonic booms also cause defensive reactions in mountain goats (Chadwick 1973). Mountain goats can become habituated to human disturbance, especially where they are not hunted, as in Glacier National Park (Pedivillano and others 1987, Singer and Doherty 1985), but more typically, mountain goats exhibit signs of stress when exposed to human disturbances. In Montana's Rocky Mountain Front, mountain goat reproduction and kid survival was lower in a herd exposed to much human activity (such as energy exploration, a downhill ski resort, and developed recreation) compared to a herd in a more remote area (Joslin 1986).

Figure 61—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 20 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 62—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 20, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Timber harvests can have both positive and negative effects on mountain goats. Overstory removal can increase forage productivity in areas where fire suppression has reduced the extent of open habitats (Johnson 1983). Sufficiently large stands of mature forests, however, must be retained for winter cover (Johnson 1983). Timber harvests also increase human access to mountain goat habitat through road construction (Chadwick 1973), and this has led to increased hunting mortality in some herds that were formerly less accessible (Johnson 1983).

Roads, particularly highways, also increase mortality rates through vehicle collisions (Singer 1978). In Glacier National Park, however, highway mortality was reduced by placing two highway underpasses on Highway 2 to allow goats to reach two mineral licks (Pedivillano and others 1987).

Many goat populations are small because of habitat fragmentation, hunting pressure, and the establishment of new herds with few individuals. A potential consequence of low numbers is a high probability of deleterious effects from inbreeding. For example, even after hunting of the Wallowa Mountain goat population was discontinued, the population remained static for many years until new genetic stock was introduced in the 1980s (Matthews and Coggins 1994).

Population status and trends—Mountain goat population trends differ across the basin. Populations in Washington[7] and Montana[8] have declined, whereas populations in the Wallowa and Elkhorn Mountains in northeastern Oregon have increased (Matthews and Coggins 1994). Native populations in Idaho have decreased, whereas introduced populations are stable or increasing.[9]

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 20 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Important issues affecting mountain goats were taken both from the literature and our habitat analysis.

1. Increased human disturbance in formerly isolated habitats.

2. Reduction in forage quantity and quality because of successional changes in source habitats from fire suppression.

3. Habitat fragmentation because of human land uses and successional changes in source habitats from fire suppression.

Potential strategies—

1. (To address issue no. 1) Reduce human activities, particularly where mountain goat herds are static or declining.

2. (To address issue no. 2) Restore quality and quantity of forage where forage has declined because of successional changes and changes caused by fire suppression.

3. (To address issue no. 3) Seek opportunities to reduce fragmentation in historical range caused by human land uses and fire suppression.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Incorporate mitigation measures for human activities within or adjacent to known mountain goat herds into all relevant planning documents.

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[7] Personal communication. 1997. Rolf Johnson, manager, deer and elk section, Washington Department of Fish and Wildlife, 600 Capitol Way North, Olympia, WA 98501.

[8] Personal communication. 1997. John McCarthy, special projects coordinator, Montana Department of Fish, Wildlife, and Parks, P.O. Box 200701, Helena, MT59620-0701.

[9] Personal communication. 1997. Lonn Kuck, wildlife game and research manager, Bureau of Wildlife, Idaho Department of Fish and Game, P.O. Box 25, Boise, ID 83707-0025.

Figure 63—Ranges of species in group 21 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

2. (In support of strategy no. 1) Carefully regulate frequency and height of low-flying aircraft over known mountain goat herds, including military exercises, helicopter logging, recreational flights, and wildlife surveys.

3. (In support of strategies no. 2 and no. 3) Use understory thinning and prescribed burns to improve the quantity and quality of forage, and increase links with isolated herds.

4. (In support of strategy no. 3) Use land acquisitions, exchanges, and easements to consolidate blocks of suitable mountain goat habitat, including blocks of currently unoccupied habitat.

Group 21—Long-Eared Owl

Results

Species ranges, source habitats, and special habitat features—Group 21 is comprised of the long-eared owl. Long-eared owls are year-round residents of the basin, but some individuals move long distances suggestive of migratory behavior during fall and spring (Marks and others 1994). The current range of the long-eared owl includes all 13 ERUs (fig. 63).

Source habitats for the long-eared owl include a broad range of vegetation types from mid-elevational forests to low-elevational shrublands. The six vegetation community groups in which source habitats occur are montane forests, upland woodlands, upland shrublands, upland herblands, riparian woodlands, and riparian shrublands (vol. 3, appendix 1, table 1). Source habitat cover types within the montane forest community include interior Douglas-fir, western larch, grand fir-white fir, Sierra Nevada mixed conifer, and red fir. Nearly all structural stages within these cover types except for managed young forests are considered source habitats.

Long-eared owls tend to nest and roost in dense vegetation, but they hunt almost exclusively in open habitats (Getz 1961, ICBEMP 1996h, Marks and others 1994, Thurow and White 1984). As such, they are considered a contrast species (vol. 3, appendix 1, table 2), requiring a juxtaposition of contrasting vegetative structure to meet all aspects of their ecology. Where forests are adjacent to open areas, trees are typically used for nest sites. Where forests are not present, nests are placed in tall shrubs (Holt 1997). This owl typically lays its eggs in abandoned stick nests of other species, especially common raven, American crow, and black-billed magpie nests (Marks and others 1994).

Broad-scale changes in source habitats—The historical distribution of source habitats was most concentrated in the Columbia Plateau, Northern Great Basin, and Owyhee Uplands (fig. 64A). The current distribution is about the same (fig. 64B), although significant declines have occurred in the northern half of the Columbia Plateau and in the eastern basin, and significant increases have occurred in the north, the central basin, and in the southwest (fig. 64C).

Trends in extent of source habitats are mixed across the basin: 29 percent of watersheds with source habitats showed no change in areal extent between the historical and current periods; 40 percent of watersheds had declining trends, and 31 percent had increasing trends (fig. 65). Four ERUs had declining and strongly declining trends in source habitats in >50 percent of watersheds. These were the Columbia Plateau (53 percent of watersheds), the Upper Clark Fork (75 percent of watersheds), the Upper Snake (76 percent of watersheds), and the Snake Headwaters (67 percent of watersheds). Increasing and strongly increasing trends occurred in >50 percent of watersheds in the Upper Klamath (63 percent of watersheds) and Blue Mountains (52 percent of watersheds) ERUs, and the Southern Cascades had increasing trends in 9 percent of watersheds (figs. 64C and 65).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats—Most vegetation types that provide source habitats for the long-eared owl have changed in extent from the historical period, but these changes have resulted in no net increase or decrease in source habitats.

Within the Upper Klamath, Blue Mountains, and Southern Cascades ERUs, increases in source habitats were largely due to increases in interior Douglas-fir, grand fir-white fir, juniper/sagebrush woodland, and big sagebrush (Hann and others 1997; vol. 3, appendix 1, table 4). Declines in the northern portion of the Columbia Plateau and the Upper Snake are primarily due to transitions from big sagebrush to agriculture and the conversion of many cover types in the upland shrubland and riparian shrubland community groups to exotic forbs-annual grass (Hann and others 1997; vol. 3, appendix 1, table 4). Declines in the Upper Clark Fork are due to increases in cropland and Engelmann spruce-subalpine fir cover types (Hann and others 1997), neither of which are source habitats for the long-eared owl, and declines in all structural stages of interior Douglas-fir (Hann and others 1997). Declines in the Snake Headwaters are due to transitions in both the upland herbland and upland shrubland communities to agriculture (Hann and others 1997).

Condition of special habitat features—No special habitat features were identified for the long-eared owl. The amount of edge habitat, however, may be a landscape-level variable of some importance to long-eared owls. The mid-scale analysis of vegetation changes in the basin (Hessburg and others 1999) indicated that the amount of edge increased significantly in 6 of 13 ERUs. Assuming that this scale of analysis is appropriate for long-eared owls, and assuming that interspersion of habitats is beneficial to this species, the increase in edge is considered a positive change in habitat condition.

Other factors affecting the group—The long-eared owl generally nests in trees, using stick nests created by other bird species, especially common raven, American crow, and black-billed magpie. Programs designed to reduce these species could therefore negatively affect the long-eared owl.

Little is known about effects of pesticides on this species. Henny and others (1984) discovered organochlorine residues in one-third of all long-eared owl eggs they examined.

Roads apparently do not impact long-eared owls. Mean distance to nearest road was not different for successful and unsuccessful nests (Marks 1986).

Population status and trends—Long-eared owls are common in most Western states, although they are considered rare in Montana (Craig and Trost 1979). Long-eared owl numbers appear to be stable in most states (Marti and Marks 1989). Within the basin, populations seem to attain peak densities in southern Idaho (Craig and Trost 1979).

VIEW

MAP

Figure 64—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 21 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 65—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 21, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 21 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The primary issue related to long-eared owl conservation is degradation and loss of native upland shrublands, riparian shrublands, and riparian woodlands.

Potential strategies—

1. Maintain and restore native upland shrublands, riparian shrublands, and riparian woodlands across the basin, particularly in the northern half of the Columbia Plateau and in the Upper Snake and Snake Headwaters ERUs.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. Limit livestock grazing and recreational activities in riparian shrublands and woodlands to allow growth of dense vegetation for nest sites.

2. Explore options under the Conservation Reserve Program (CRP) (Johnson and Igl 1995), or develop other incentive programs, to encourage restoration of agricultural areas to native cover types.

3. Restore native vegetation by appropriate treatments and seedings of native shrub, grass, and forb species.

Group 22—California Bighorn Sheep and Rocky Mountain Bighorn Sheep

Results

Species ranges, source habitats, and special habitat features—Group 22 consists of two subspecies of bighorn sheep, the California and Rocky Mountain bighorn sheep; both are year-round residents of the basin. Although they use similar habitats, the two subspecies are separated by disparate ranges of remnant populations and by different geographic areas that have been designated for their reintroduction. In general, California bighorn occur in the western and southern portions of the basin, and Rocky Mountain bighorn occupy the eastern and northern portions of the basin (fig. 66).

Historically, California bighorns occurred in central and southeastern Oregon, the eastern slope of the Cascade Range in Washington, northwestern Nevada, and the mountains of southwestern Idaho (fig. 66). Populations declined in the late 1800s, and bighorns were extirpated from all four states between 1900 and 1930 (Thorne and others 1985). Because of a series of reintroductions, California bighorns currently are found in many disjunct populations within their former range (fig. 66).

Rocky Mountain bighorns historically occurred in northeastern Oregon, central Idaho, Montana and Wyoming, and northeastern Nevada (Thorne and others 1985) (fig. 66). After a severe population decline in the early 1900s, bighorns remained in only a few isolated areas of their former habitat. The current range represents an increase in occupied habitat since that time, because of a combination of reintroductions and protection of remnant populations (Thorne and others 1985). Much of the historical range, however, is still unoccupied (fig. 66).

Source habitats for both subspecies are primarily in the alpine, subalpine, upland shrubland, and upland herbland community groups. Old-forest and stand-initiation stages of whitebark pine are source habitat, but only the stand-initiation stage of other forest cover types is used (vol. 3, appendix 1, table 1). Bighorn sheep prefer open habitats with short vegetation, both for high-quality forage (McWhirter and others 1992) and to maintain high visibility for predator avoidance (Risenhoover and Bailey 1985, Wishart 1978), and a negative correlation between forest cover and bighorn occurrence has been observed (Bentz and Woodard 1988). Postfire habitats can benefit bighorn sheep by improving forage quality (McWhirter and others 1992) and increasing visibility (Bentz and Woodard 1988).

In the basin, Rocky Mountain bighorn sheep exhibit more seasonal movements than do California bighorn sheep. Alpine and subalpine community groups are

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MAP

Figure 66—Ranges of species in group 22 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

primarily summer range for the Rocky Mountain subspecies, whereas upland herbland and shrubland are used in both seasons, depending on elevation (vol. 3, appendix 1, table 1).

Special habitat features identified for these two subspecies include cliffs, talus, and seasonal wetlands (vol. 3, appendix 1, table 2). The location of cliffs and talus ultimately defines the distribution of bighorn sheep because such features are essential for escape cover and the secure rearing of young (Wakelyn 1987). Cover types listed as source habitats (vol. 3, appendix 1, table 1) generally are not available to bighorns unless they are near cliffs.

Broad-scale changes in source habitats— The following trends in source habitats for bighorn sheep were derived without reference to the proximity of cliffs and talus and may not accurately represent changes in the more restricted subset of stands available to bighorns. Trends derived from a restricted subset of habitats could differ substantially in magnitude from those reported here, but the general direction of the trends likely would remain the same.

Source habitats (regardless of proximity to cliffs) currently occupy the same general geographic extent as the historical distribution of habitats but are less prevalent within each watershed (figs. 67A, and 67B), thereby resulting in overall negative trends in habitat extent. Many areas that formerly had bighorn sheep habitat in 25 to 50 percent of each watershed now meet source habitat conditions in less than 25 percent of each watershed, particularly in the central and northern regions of the basin (fig. 67B). Habitats declined in 57 percent of the watersheds throughout the basin and in most watersheds in five ERUs: the Blue Mountains, Northern Glaciated Mountains, Lower and Upper Clark Fork, and Upper Snake (fig. 68). Declining trends also were noted in the Northern and Southern Cascades, but these ERUs are on the western edge of the geographic range and contain little habitat (vol. 3, appendix 1, table 3). Most watersheds of the Northern Great Basin and Owyhee Uplands ERUs exhibited no change in the amount of source habitats, whereas watersheds in the Snake Headwaters exhibited mixed trends in habitat extent (fig. 68).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Declines in source habitats were due primarily to declines in big sagebrush, mountain big sagebrush, fescue-bunchgrass, interior ponderosa pine, native forb, western larch, wheatgrass bunchgrass, whitebark pine-alpine larch, and whitebark pine (vol. 3, appendix 1, table 4). A notable change that has affected bighorn sheep is the widespread conversion of native shrublands and grasslands to agricultural cover types (Hann and others 1997), particularly in historical winter range. Also, source habitats with high visibility for predator avoidance have been replaced by stands with reduced visibility, primarily through the transition of whitebark pine old forests to Engelmann spruce-subalpine fir and the transition of stand-initiation stage forest cover types to mid-seral stages (Hann and others 1997).

Condition of special habitat features— Cliffs and talus (represented by the community group rock-barren) have not changed between historical and current periods (Hann and others 1997). Cliffs and talus can be significantly altered through direct human disturbance such as blasting and road construction, but this type of activity generally has not occurred in remote areas currently used by bighorn. Seasonal wetlands are highly dependent on annual hydrologic cycles and therefore have fluctuated widely in occurrence and productivity over time.

Other factors affecting the group— Bighorn sheep are highly susceptible to pneumonia after exposure to bacteria (*Pasteurella* spp.), viruses (*Parainfluenza* type-3), lungworm, and stress agents (Foreyt 1994, Wishart 1978). Major reductions or total extirpation of bighorn herds because of pneumonia outbreaks are well documented (Cassirer and others 1996, Coggins 1988, Onderka and Wishart 1984, Spraker and others 1984). A recent episode of *Pasteurella*-associated pneumonia in the Hells Canyon area resulted in a known loss of 327 bighorn sheep between November 1995 and March 1996, which represented 50 to 75 percent of four herds in Oregon and Washington (Cassirer and others 1996).

Abundant circumstantial evidence indicates that domestic and exotic sheep are the source of nonendemic bacteria and viruses predisposing bighorn sheep to pneumonia (Coggins 1988, Foreyt and Jessup 1982,

Figure 67—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 22 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 68—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 22, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Martin and others 1996); moreover, direct evidence recently has been acquired through experimental contact between sheep and bighorns in enclosures (Foreyt 1994), and through bacterial swab cultures and DNA analysis of *Pasteurella* spp. collected from free-ranging bighorn sheep with pneumonia in Nevada and Oregon (Rudolph and others, in prep.). Domestic goats also may be reservoirs, although the evidence is less compelling. A feral goat was associated with diseased bighorn at the start of the outbreak in Hells Canyon and had genetically identical *Pasteurella* to one of the bighorn ewes; however, these bacteria were not common among bighorns sampled during the episode (Cassirer and others 1996; Rudolph and others, in prep.).

Bighorn sheep also are affected by grazing competition from livestock (USDI Bureau of Land Management 1995). Intensive grazing pressure that occurred between the late 1800s and early 1900s is believed a factor in the reduction in bighorn sheep populations of that era (Johnson 1983). Grazing competition with domestic sheep has been reduced in recent times because of efforts to maintain buffers between sheep and bighorns to reduce the potential for disease transmission. The leading source of grazing competition is from cattle (Blood 1961, Demarchi 1965, and Lauer and Peek 1976, as cited in Van Dyke and others 1983). Late winter grazing by cattle, however, has proven beneficial to the Lower Imnaha bighorn herd in Oregon.[10]

The condition of bighorn sheep habitats has been altered over the last century because of changes in historical fire regimes. Fire suppression has resulted in an increase in the density of trees of formerly open stands, reducing forage quality and causing bighorns to avoid these areas because of reduced visibility. Some cliff areas are currently inaccessible to bighorns because the stands of open timber through which bighorns formerly traveled have developed into dense stands that bighorns avoid (Wakelyn 1987). For the Rocky Mountain bighorn, fire-suppressed stands have created barriers between historical winter and summer range, thereby preventing occupancy of the total range even though each isolated range is currently suitable (Wakelyn 1987).

Some historical ranges have become fragmented by urban, mining, agricultural, and recreational developments (USDI Bureau of Land Management 1995). In some cases, this has created a barrier between seasonal ranges, as described above for fire-suppressed habitat. Additionally, fragmentation has resulted in habitat islands that can support only small, isolated herds (USDI Bureau of Land Management 1995).

Direct disturbance by humans can affect bighorn sheep by shifting their distribution (Hamilton and others 1982, Hicks and Elder 1979) and by increasing physiologic stress (MacArthur and others 1979). Hunted populations generally react more strongly than nonhunted populations (Hamilton and others 1982, Hicks and Elder 1979). Among the human activities that elicit the strongest negative response are low-flying aircraft (helicopters and military air exercises). Hiking in lambing areas is also disruptive to bighorns (USDI Bureau of Land Management 1995). The human population in the basin has increased from a relatively small number of native people to 3 million (McCool and others 1997); therefore, the number of human disturbances in bighorn sheep habitat likely has increased.

Population status and trends—Bighorn sheep populations declined substantially throughout their geographic range in the late 1800s and early 1900s. However, because of the establishment of hunting regulations, a better understanding of disease transmission, and concentrated reintroduction efforts throughout the West, bighorn numbers have steadily increased over the last 50 years (Thorne and others 1985). By 1995, many reintroductions of California bighorn resulted in the establishment of 6 herds in Idaho, 29 herds in Oregon, and 8 herds in Washington (USDI Bureau of Land Management 1995).

Populations of Rocky Mountain bighorn also have been widely reintroduced into their historical habitats within the basin. As of 1995, the reintroduced and native populations comprised 10 herds in Idaho, 9 herds in Oregon (1 extends into Washington), 3 additional herds in Washington, and 9 herds in Montana (USDI Bureau of Land Management 1995).

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[10] Personal communication. 1998. Victor Coggins, regional wildlife biologist, Oregon Department of Fish and Wildlife, 65495 Alder Slope, Enterprise, OR 97828.

Population trends differ by herd. Some reintroduced herds are still increasing; for example, the Pueblo Mountains herd in southeast Oregon currently numbers 130 and is still growing.[11] This herd was started with three reintroductions in 1976, 1980, and 1983 that totaled 40 animals (Coggins and others 1996). Some herds have static trends; for example, the Steens Mountain bighorn herd was started with 11 animals in 1960 (Coggins and others 1996) and increased to 275 (USDI Bureau of Land Management 1995), but currently numbers 250 and seems to be static for unknown reasons (see footnote 11). Several herds in the Hells Canyon area of Washington and Oregon have recently declined because of an outbreak of *Pasteurella*-associated pneumonia (Cassirer and others 1996).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 22 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues–Issues were taken from the literature and the results of our habitat analysis for these two subspecies.

1. Incompatibility with domestic sheep and possibly domestic goats because of the potential for disease transmission and competition for forage.
2. Reduction in forage quantity and quality because of successional changes in source habitats.
3. Habitat fragmentation (poor juxtaposition of seasonal ranges as well as isolation of small herds) because of successional changes in source habitats.
4. Habitat fragmentation because of agricultural, industrial, and recreational development.
5. Disturbance and habitat displacement because of human activities such as low aircraft fly-overs and hiking in lambing areas.

Potential strategies—

1. (To address issue no. 1) Actively control the potential for disease transmission and forage competition between bighorns and domestic livestock.
2. (To address issue no. 2) Restore quality and quantity of forage where forage has declined because of successional changes in vegetation.
3. (To address issue no. 3) Restore habitat links between summer and winter range and access to escape cover that have been lost because of changes in historical fire regimes.
4. (To address issue no. 4) Seek opportunities to reduce fragmentation in historical range caused by human land uses.
5. (To address issue no. 5) Reduce human activities in key foraging and lambing areas.

Practices that support strategies— The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Avoid direct contact between bighorn sheep and domestic sheep and goats. Guidelines established by the BLM for domestic sheep management in bighorn sheep habitats (USDI Bureau of Land Management 1995) recommend that buffers (having no domestic sheep or goats) are placed around bighorn sheep habitat and that bighorn sheep reintroductions do not occur in areas that have been grazed by domestic sheep or goats within the last 2 years.
2. (In support of strategy no. 1) Reduce forage competition with livestock by factoring bighorn sheep forage consumption into total forage utilization. Light to moderate cattle grazing during spring or early summer can be used to improve forage quality on bighorn sheep winter ranges (Bodie and Hickey 1980).
3. (In support of strategies no. 2 and no. 3) Use understory thinning and prescribed burns to improve the quantity and quality of forage and to restore open habitat links between winter and summer ranges and to provide access to cliffs that currently are inaccessible to bighorns.

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[11] Personal communication. 1998. Ron Garner, assistant district wildlife biologist, Oregon Department of Fish and Wildlife, P.O. Box 8, Hines, OR 97738.

4. (In support of strategy no. 4) Use land acquisitions, exchanges, and easements to consolidate blocks of suitable bighorn sheep habitat (USDI Bureau of Land Management 1995).

5. (In support of strategy no. 5) Incorporate mitigation measures into all planning documents for mines, highways, canals, and recreational developments within or adjacent to occupied bighorn sheep range to minimize human disturbance.

6. (In support of strategy no. 5) Regulate activities that cause unacceptable disturbance to bighorns, such as flights of low-flying aircraft and back country recreation.

Group 23—Rufous Hummingbird and Broad-Tailed Hummingbird

Results

Species ranges, source habitats, and special habitat features—Group 23 consists of the rufous hummingbird and the broad-tailed hummingbird, both of which are migratory breeders in the basin. The rufous hummingbird is distributed throughout forested portions of the basin (fig. 69), whereas the range of the broad-tailed hummingbird is restricted to small areas of Idaho and Montana (fig. 69). Both of these species are mostly associated with coniferous forests. The rufous hummingbird is found in 12 coniferous forest types and occurs in 53 combinations of forest types and structural stages. The broad-tailed hummingbird has source habitats in four coniferous types: Engelmann spruce-subalpine fir, interior Douglas-fir, grand fir-white fir, and interior ponderosa pine (vol. 3, appendix 1, table 1). Within the forest types, both species use old forests, understory reinitiation, and stand initiation. Source habitats for both species also include shrub-wetlands and aspen, and each species uses some woodland types. These species generally are found in more open forests, forests with openings, or in areas where open areas and forest habitats are adjacent because it is within these areas that the potential for deciduous shrubs and herbs is higher. Deciduous shrubs and herbs provide important foraging substrates (flowers) for these birds. Both species typically nest in conifers in areas that support an abundance of nectar-producing flowers, which serve as a foraging substrate. Nectar-producing flowers are a special habitat feature for hummingbirds (vol. 3, appendix 1, table 2).

Broad-scale changes in source habitats—Historically, source habitats for group 23 were broadly distributed throughout the mountainous regions of the basin (fig. 70A). Currently, source habitats are still widely distributed but more concentrated in fewer watersheds in most of the ERUs (fig. 70B).

Overall, the projected trend in source habitats for group 23 declined from historical to present. Basin-wide, about 36 percent of the watersheds had strong declines in source habitats, and 19 percent had moderate declines (fig. 71). Eight ERUs were projected to have moderate or strong declines in source habitats in more than 50 percent of watersheds (fig. 71). More than 50 percent of the watersheds in the Upper Klamath and Northern Great Basin were projected to have moderate or strong increases (fig. 71). The Northern Cascades, Snake Headwaters, and Central Idaho Mountains generally had no change in amount of source habitats (fig. 71).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— The increase in source habitats in the Upper Klamath and Northern Great Basin is directly related to an increase in late-seral montane forests (vol. 3, appendix 1, table 4). Decreases in source habitats in six ERUs are due primarily to reductions in late-seral ponderosa pine, western larch, and western white pine. Six ERUs (Southern Cascades, Upper Klamath, Columbia Plateau, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork) also showed substantial declines in early-seral forests, particularly ponderosa pine, western larch, and western white pine. Decreases in the Upper Snake resulted from declines in aspen (understory reinitiation) and chokecherry-serviceberry-rose. The decline in available source habitats in the Owyhee Uplands primarily was because of a decrease of about 2 percent in shrub-wetlands, but this figure may underrepresent the actual loss of habitat due to the small size of shrub-wetland patches relative to mapping unit size at the broad scale.

VIEW

MAP

Figure 69—Ranges of species in group 23 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 70—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 23 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 71—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 23, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Condition of special habitat features— An analysis of the abundance of nectar-producing flowers, the primary food source for these hummingbirds, is not possible at the scale of this analysis, and no information on condition or trend is available. The increasing trend in shade-tolerant, multi-storied stands likely decreased the abundance of forest-associated flowers by reducing the amount of sunlight needed for flower development.

Other factors affecting the group— Grazing has an overall negative impact on nectarivores because of these species' dependence on understory plants as a food source. Negative effects of grazing on broad-tailed hummingbirds have been documented in two studies (Page and others 1978, Schulz and Leininger 1991, cited in Saab and others 1995). Negative responses to grazing also were reported for the rufous hummingbird (Page and others 1978, cited in Saab and others 1995).

Because both species are Neotropical migratory birds, habitat used during migration and winter also may influence population trends. Russell and others (1994) observed that the quality of "stopover" habitats for migrant rufous hummingbirds differs greatly because of the natural variation in flowering, and found a positive correlation between variation in flowering and hummingbird survival. Little is known about the abundance or trend of wintering habitat of these species.

Population status and trends— Based on BBS data from 1968 to 1994, rufous hummingbirds in the basin have shown stable population trends (Saab and Rich 1997). There are insufficient BBS data for the broad-tailed hummingbird to analyze population trends within the basin (Saab and Rich 1997). Specialized monitoring techniques are needed to track population trends for both species of hummingbirds.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 23 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues— The following issues are drawn from our analysis of source habitat trends in combination with issues identified from other literature:

1. Decline in abundance of natural forest openings specifically within ponderosa pine, interior Douglas-fir, grand fir, and western larch. There also has been a nearly complete loss of open forests of western white pine (all structural stages).

2. Decline in abundance of forest-associated flowering plants because of exclusion of fire, establishment of shade-tolerant trees, and subsequent decrease in shrub and herbaceous understories.

3. Decline in abundance of understory flowering shrubs, particularly in riparian areas, because of cattle grazing.

Potential strategies— Habitat for rufous and broad-tailed hummingbirds would benefit from the following strategies that address the issues listed above:

1. (To address issue no. 1) Promote the development of forest openings and single-layered old-forest structures of ponderosa pine, interior Douglas-fir, grand fir, and western larch, particularly in the ERUs where source habitats have declined (Southern Cascades, Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork).

2. (To address issue no. 1) Increase the amount of early-seral forest in the ERUs where it has declined (Southern Cascades, Upper Klamath, Columbia Plateau, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork).

3. (To address issue no. 2) Restore fire as an ecological process to encourage development of forest openings and growth of shrubs and forbs.

4. (To address issue no. 3) Reduce impacts to flowering herbs and shrubs from grazing.

Practices that support strategies— The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Remove shade-tolerant understory trees to promote stand health and longevity in old-forest stands. Hand removal, or in some cases prescribed burning, may be effective.

2. (In support of strategies no. 2 and no. 3) Accelerate development of flowering shrubs and forbs with the use of prescribed underburning and thinning, or allow for natural wildfires to occur particularly in the following ERUs: Southern Cascades, Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, Upper Clark Fork, Owyhee Uplands, and the Upper Snake.

3. (In support of strategies no. 2 and no. 3) Select areas that have been burned by wildfire or harvested for timber, and try to extend the duration of the early-seral stage, which is rich in forbs and shrubs, by not planting conifers. Areas of primary importance are the Southern Cascades, Upper Klamath, Columbia Plateau, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork.

4. (In support of strategy no. 4) Remove or explicitly control the timing and intensity of grazing to develop and promote the long-term persistence of shrub communities.

Group 24—Sharptail Snake, California Mountain Kingsnake, and Black-Chinned Hummingbird

Results

Species ranges, source habitats, and special habitat features—Group 24 consists of three species that primarily depend on open forest and woodland habitats: the black-chinned hummingbird, the sharptail snake, and the California mountain kingsnake. The range of the black-chinned hummingbird covers the entire basin except the high elevations of the Cascade Mountains in both the Northern and Southern Cascades ERUs and the high elevations of the northern Rocky Mountains (fig. 72). Both species of snakes occur in scattered, isolated populations along the eastern slope of the Cascade Range (fig. 72). The two species of snakes are only known to occur in the same location near the Columbia River Gorge.

These three species primarily group together based on their consistent use of interior ponderosa pine, and interior Douglas-fir vegetation types in all structural stages except stem-exclusion, closed-canopy forests. They also use mixed-conifer woodlands and Oregon white oak (vol. 3, appendix 1, table 1).

The black-chinned hummingbird is the only member of the group whose source habitats include juniper, juniper/sagebrush, chokecherry-serviceberry-rose, mountain mahogany, shrub wetlands, and old-forest aspen (vol. 3, appendix 1, table 1). The sharptail snake uses more source habitats than the kingsnake, including nearly all seral stages of cottonwood-willow (also used by the black-chinned hummingbird), nearly all structural stages of western redcedar-western hemlock, and the stem-exclusion, closed-canopy, and stand-initiation structural stages of western larch (vol. 3, appendix 1, table 1).

Logs and talus are special habitat features for both species of snakes because of their dependency on moist environments (vol. 3, appendix 1, table 2). In the absence of nearby streams, microhabitats with higher moisture are found under logs and within talus (Brown and others 1995). These features also provide protection from predators and habitat for potential prey. Additionally, deciduous tree riparian is also a special habitat feature for the sharptail snake (vol. 3, appendix 1, table 2).

Nectar-producing flowers are considered a special habitat feature for the black-chinned hummingbird because of the dependence on nectar as a primary food source (vol. 3, appendix 1, table 2).

Broad-scale changes in source habitats—Because the distribution of the two species of snakes is restricted to a few disjunct locations, the results of our analysis for this group are primarily based on source habitats for the black-chinned hummingbird, which is widely distributed throughout the basin both historically (fig. 73A) and currently (fig. 73B). Source habitats are most abundant in northeastern Washington, the Upper Klamath, and central Oregon (figs. 73A, and 73B).

Overall, source habitats appeared to increase since the historical period, primarily in Oregon, Washington, and southeastern Idaho, whereas much of northern and central Idaho and Montana experienced declines (fig. 73C). About 53 percent of the watersheds basin-wide were projected to have increasing trends (fig. 74). The

Figure 72--Ranges of species in group 24 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 73—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 24 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 74—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 24, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

three ERUs with declining trends were Lower and Upper Clark Fork and Central Idaho Mountains (fig. 74), whereas mostly neutral trends were projected for the Blue Mountains and Northern Glaciated Mountains ERUs (fig. 74).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Changes in broad-scale habitat trends differed across the basin because of the wide array of cover types and structural stages used by group 24. Declining trends were fairly consistent for interior ponderosa pine old forest (both multi- and single-storied), and for stand-initiation stages of both ponderosa pine and Douglas-fir. Increases in habitat occurred in nearly all ERUs in both ponderosa pine and Douglas-fir young forests and in all woodland types (vol. 3, appendix 1, table 4). The increase in woodlands contributed substantially to the overall increase in source habitats, especially in rangeland-dominated ERUs (Upper Klamath, Northern Great Basin, Columbia Plateau, Snake Headwaters, and parts of the Blue Mountains). The increase in source habitats for group 24 closely reflects the increase in upland woodland reported for the basin (see map 3.58 in Hann and others 1997).

Condition of special habitat features— Trends in the condition of logs, talus, and flowers are not available at the broad scale. Activities that may negatively affect these variables include timber harvesting, road building, grazing, mining, and fire suppression. Timber harvesting and road building can lead to the direct removal of logs and flowers; mining can lead to disturbance of talus. Fire suppression can impact flower abundance by increasing forest canopy closure and reducing the amount of sunlight needed for flower development on herbaceous plants in the understory.

Other factors affecting the group— Humans have directly affected snakes through collection, harassment, and accidental mortalities. Because of its striking coloration, the California mountain kingsnake is in demand by collectors (ICBEMP 1996a). Humans also intentionally kill various snake species because of fear and hate, and are responsible for unintentional mortality caused by motorized vehicles (Brown and others 1995).

Population isolation was raised as a concern by the viability panel that evaluated sharptail snakes (ICBEMP 1996b). Although the viability panel did not evaluate the California mountain kingsnake, the same concerns and considerations are presumably important for this species because of its patchy and restricted range in the basin.

Because the black-chinned hummingbird is a Neotropical migrant, habitat used during migration and wintering habitat could impact its populations. In a study on migrating rufous hummingbirds, researchers found a correlation between abundance of nectar-producing flowers and hummingbird survival in habitat used during migration (Russell and others 1994). A similar correlation likely exists with black-chinned hummingbirds. Little is known about the abundance or trends of the wintering habitat of the black-chinned hummingbird.

Heavy grazing has had an overall negative impact on nectarivores by reducing the density of understory plants used as a food source (Saab and others 1995). Direct effects on the black-chinned hummingbird are unknown.

Population status and trends— There are no estimates of population change for either the sharptail snake or the California mountain kingsnake within the basin. According to Brown and others (1995), however, loss of snake habitat and population declines in snakes worldwide have increased because of the increased paving of roads, fast cars, intensive agriculture, urban sprawl, desertification of arid lands, deforestation of the tropics, pesticides, hobby collecting, rattlesnake “roundups,” and a general aversion to snakes. Sharptail snakes have declined in the Willamette Valley of Oregon, just west of the basin (Marshall and others 1996, Oregon Department of Fish and Wildlife 1987).

Population trend estimates for the black-chinned hummingbird in the basin are not available because of insufficient data from established BBS routes (Saab and Rich 1997). Specialized monitoring techniques would be needed to adequately measure population trends because they are difficult to detect (Saab and Rich 1997).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 24 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Although the results of our analysis show an increase in source habitats across the basin, other sources of information have indicated that habitat and populations have decreased since the historical period. The trend of special habitat features for these species may affect populations more strongly than the broad-scale changes in source habitats. The following are issues that relate to special habitat features and other management concerns:

1. Loss of down logs and surface litter used by snakes as a result of timber harvest.

2. Loss of habitat connectivity for snakes as a result of habitat loss and road construction.

3 Decline in availability of understory flowering shrubs, particularly in riparian areas, because of cattle grazing.

4. Decreases in natural forest openings and shrub understories because of exclusion of fire and invasions by shade-tolerant trees.

5. Collection of California mountain kingsnakes.

Potential strategies—The issues identified above suggest the following broad-scale strategies to maintain the long-term persistence of sharptail snakes, California mountain kingsnakes, and black-chinned hummingbirds:

1. (To address issue no. 1) Survey and manage for downed logs and litter for the two species of snakes.

2. (To address issue no. 2) Seek opportunities to improve connectivity between isolated populations of both the sharptail snake and California mountain kingsnake.

3. (To address issue no. 3) Maintain and restore flowering herbs and shrubs in areas that have been negatively affected by cattle grazing.

4. (To address issue no. 4) Restore fire as an ecological process, particularly in interior ponderosa pine and interior Douglas-fir plant communities, to encourage forest openings that are occupied by flowering shrubs and forbs.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Maintain and protect down logs at a level that is ecologically sustainable and meets the habitat requirements for snakes.

2. (In support of strategy no. 2) Close roads to minimize human disturbance and maximize dispersal capabilities, particularly in areas known to be occupied by either sharptail snakes or California mountain kingsnakes.

3. (In support of strategy no. 3) Remove or explicitly control the timing and intensity of grazing to develop and promote the long-term persistence of shrub communities.

4. (In support of strategies no. 3 and no. 4) Accelerate development of flowering shrubs and forbs by the use of prescribed underburning and thinning, or allow for natural wildfires to occur, particularly in the Douglas-fir and ponderosa pine plant communities. Highest priorities for following these practices are in the Lower Clark Fork, Upper Clark Fork, and Central Idaho Mountains ERUs.

Group 25—Northern Goshawk (Winter)

Results

Species ranges and source habitats—Group 25 consists of winter habitat for the northern goshawk. Summer habitat for the northern goshawk is described in group 5. During winter, the range of the goshawk is basin-wide (fig. 75). Throughout North America, little is known about goshawks in winter, but indications are that northern goshawks are partial migrants. Some of the population regularly winters outside the

Figure 75—Ranges of species in group 25 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

breeding area, whereas some do not migrate at all (Squires and Reynolds 1997). The degree to which goshawks migrate during winter may relate to prey availability. In the Yukon Territory in winter, goshawk-numbers fluctuate with snowshoe hare numbers (Doyle and Smith 1994). Some goshawks may travel short distances in winter to lower elevations or more open habitats (Squires and Reynolds 1997), and migrations may consist of predominately immature birds (Sibley 1993).

Source habitats are found in old forest and unmanaged young forests in montane, lower montane, and riparian woodland community groups and chokecherry-serviceberry-rose (vol. 3, appendix 1, table 1). Also, contrary to summer source habitats, winter source habitats include all of the upland woodland types.

Important attributes of goshawk prey habitat include snags, downed logs, woody debris, large trees, openings, herbaceous and shrubby understories, and an intermixture of various forest structural stages (Reynolds and others 1992).

Broad-scale changes in source habitats—Goshawk winter source habitats were projected to be broadly distributed, primarily throughout the forested areas of the basin, in historical times (fig. 76A). Source habitats are still widely available, although more disjunct in many areas, and there has been an increase in habitats in some areas that provided little or no source habitats historically (fig. 76B).

Trends in source habitat availability differed geographically (fig. 76C). Most areas with strong negative trends were in the northeast portion of the basin, within the Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork ERUs, where habitat loss was generally greater than 90 percent (figs. 76C and 77; vol. 3, appendix 1, table 3). A preponderance of watersheds in the Northern Cascades, Blue Mountains, Snake Headwaters, and Central Idaho Mountains ERUs had moderate and strong negative trends (fig. 77). The most significant gains in source habitats occurred in the Upper Klamath and Northern Great Basin ERUs (fig. 77). About 50 percent of the watersheds in the Columbia Plateau, Owyhee Uplands,

Figure 76—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 25 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 77—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 25, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of >20 percent but <60 percent; and -2 = a decrease of >60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

and Upper Snake ERUs also experienced strongly increasing trends (fig. 77). Trends in source habitats in the Southern Cascades showed a slight decrease (fig. 77).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— In areas with negative trends, projected declines occurred in nearly all source habitats, though predominately in the old-forest types (vol. 3, appendix 1, table 4). Some old-forest types increased in the Southern Cascades, Upper Klamath, and Blue Mountains ERUs. Further elaboration of the changes in old forest for the goshawk is found in the results for group 5, which includes goshawk (summer).

Large increases in juniper/sagebrush in the Upper Klamath, Northern Great Basin, Columbia Plateau, Blue Mountains, Upper Snake, and Snake Headwaters ERUs contributed to much of the increases in these ERUs or parts of these ERUs (fig. 77; vol. 3, appendix 1, table 4). Areas with increasing trends in source habitats correspond closely with the increases in upland woodlands as shown in map 3.58 in Hann and others (1997).

Other factors affecting the group— Little is known about population dynamics of goshawks, though it is thought that food availability may play an important role (Squires and Reynolds 1997). Goshawks prey primarily on relatively large-bodied mammals and birds, including tree squirrels, ground squirrels, lagomorphs, galliformes, corvids, piciforms, and passerines. Several studies have documented a positive relation of prey abundance with nest success (Doyle and Smith 1994, Linden and Wikman 1983, Ward and Kennedy 1996). Important components of habitat for many of the prey species listed above are snags, downed logs, woody debris, openings, large trees, herbaceous and shrubby understories, and interspersion of different vegetation structural stages (Reynolds and others 1992). In many areas in the basin, fire suppression, timber harvesting, and livestock grazing have resulted in a decrease in many of the attributes listed above as important characteristics of prey habitat for goshawks (Hann and others 1997).

Some evidence indicates that diet composition may change drastically during the nonbreeding season in Sweden, but winter food habits are unknown in North American populations (Squires and Reynolds 1987, Widen 1987).

Effects of falconry, shooting, and trapping of goshawks in North America are thought to be minimal (Squires and Reynolds 1987).

Human disturbance at nest sites can cause failure, but there is no information on the effects of human activities during the nonbreeding or winter season (Anon. 1989, Boal and Mannan 1994, Speiser 1992, Squires and Reynolds 1987).

Population status and trend— The BBS data for the goshawk were insufficient to determine population trends for the basin (Saab and Rich 1997) or for any state or physiographic region within the basin (Sauer and others 1996), because of low detection of goshawks by using the BBS survey method. Sufficient data, however, were available for western North America to indicate a stable trend in numbers between the years 1966 and 1995 (Sauer and others 1996).

A separate trend estimate was derived from fall migration counts conducted by Hawkwatch International at four locations in Utah and New Mexico. These data indicated an average rate of decline in migrating goshawks of about 4 percent annually between 1977 and 1991 (Hoffman and others 1992). The extent to which the migration data represented local declines near the survey stations was not determined.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 25 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues— Conservation issues for goshawk winter habitat, based on results of our analysis of source habitats in combination with empirical literature, include the following:

1. Reduction in the amount of old forests in the montane, lower montane, and riparian woodland community groups.

2. Possibly unsustainable conditions of old forests where there have been large transitions from shade-intolerant to shade-tolerant tree species. This issue stems from the exclusion of fire from many forested communities, which has resulted in increased susceptibility to stand-replacing fires (USDAForest Service 1996).

3. Loss of important attributes of prey habitat, including large trees, snags, downed logs, forest openings, and herbaceous and shrubby understories because of fire suppression, timber harvesting, and livestock grazing.

Potential strategies—Potential strategies that would be effective for maintaining source habitats for wintering goshawks within the basin are as follows:

1. (To address issues no. 1 and no. 2) Especially in the northern areas of the basin, promote greater diversity in forest structure at the landscape scale. Mid-seral stages currently predominate and do not provide source habitats. Maintain stands with active goshawk nests in old-forest condition, and identify opportunities to increase the representation of old forests in individual watersheds.

2. (To address issue no. 2) Reduce the risk of loss of habitat by focusing old-forest retention and restoration efforts on areas with low probability of stand-replacing fires. In ERUs where old-forest habitat has remained stable or increased from historical conditions, efforts could be focused on retaining existing habitat in areas with lower fire and insect risk while managing other areas to reduce risks of catastrophic loss of habitat.

3. (To address issues no. 1 and no. 3) Throughout the basin, provide for an abundant and sustainable prey base for goshawks by increasing the abundance of large trees, snags, downed logs, forest openings, and herbaceous and shrubby understories across the landscape.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) In the northern basin, identify representative stands of old forests for retention, and mid-successional stages for development into old-forest conditions. Priority should be given to large blocks having high interior-to-edge ratios and few large openings.

2. (In support of strategies no. 1 and no. 3) Actively recruit snags and logs from green trees to increase the representation of old-forest structures (snags and logs) in mid-seral stands and in old forests where snags and logs are in low density or absent.

3. (In support of strategy no. 2 and no. 3) Thin small-diameter trees, either through hand equipment or prescribed burns, to reduce fuel loading and increase herbaceous and shrubby understories for prey habitat and improve growth of overstory trees.

Group 26—Yuma Myotis, Long-Eared Myotis, Fringed Myotis, and Long-Legged Myotis

Results

Species ranges, source habitats, and special habitat features—Group 26 is comprised of four species of bats: the Yuma myotis, long-eared myotis, fringed myotis, and long-legged myotis. All four species are year-round residents of the basin, active from spring through fall and hibernating during winter. The species in group 26 are similar in their use of a broad range of forest and woodland habitats for foraging.

The ranges of the long-legged myotis and long-eared myotis encompass the entire basin (fig. 78). The Yuma myotis occurs across most of the basin except for an area in the southeast portion (fig. 78). The fringed myotis occurs in the western half of the basin and in the Upper Clark Fork ERU (fig. 78).

Source habitats shared by all members of group 26 are all cover types in the montane, lower montane, riparian woodland, and upland woodland community groups, and the mountain hemlock cover type in the subalpine community group (vol. 3, appendix 1, table 1). The long-eared myotis ranges somewhat higher than the other species and uses whitebark pine, whitebark pine-alpine larch, and Engelmann spruce-subalpine fir as source habitats. Source habitats for the

VIEW

MAP

Figure 78—Ranges of species in group 26 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Yuma myotis and long-eared myotis extend into big sagebrush, mountain big sagebrush, and low sage cover types (vol. 3, appendix 1, table 1).

The long-eared and fringed myotis forage primarily by hover-gleaning insects off of foliage (Barclay 1991, Nagorsen and Brigham 1993, Perkins 1996). The long-eared myotis consumes moths, beetles, and other insects (Whitaker and others 1977, 1981), and the fringed myotis consumes mostly beetles (Black 1974, cited in O'Farrell and Studier 1980). Surveys based on bat vocalizations indicate that in forested habitats, foraging is highest in clearcuts and mature stands, and low in precommercially thinned and young, unthinned stands (Erickson and West 1996). The Yuma myotis is primarily found in association with rivers, lakes, ponds, and streams, where it forages over water and eats midges and emergent aquatic insects (Whitaker and others 1977).

Several special habitat features were identified for group 26 (vol. 3, appendix 1, table 2). Large-diameter (>53 cm [21 in]) snags with exfoliating bark provide maternity roosts for the long-legged myotis (Nagorsen and Brigham 1993, Ormsbee and McComb 1998, Rabe and others 1998), the fringed myotis (Chung-MacCoubrey 1996, Rabe and others 1998), and the long-eared myotis (Chung-MacCoubrey 1996, Rabe and others 1998). Caves, mines, and buildings provide maternity roosts for the fringed myotis, Yuma myotis, and long-eared myotis (Christy and West 1993, Nagorsen and Brigham 1993). Caves and mines also are used as hibernacula by all four species (Nagorsen and Brigham 1993). Various structures are used for day and night roosts, including exfoliating bark, rock crevices, mines, caves, and buildings (Manning and Knox-Jones 1989, Nagorsen and Brigham 1993, O'Farrell and Studier 1980). Ormsbee and McComb (1998) found that snags extending above the canopy were most frequently used by long-legged myotis for day roosts.

Rabe and others (1998) suggested that snag-roosting bats may require higher densities of snags than cavity-nesting birds, because the stage at which snags are suitable for bat roosts (exfoliating bark) is extremely short lived, requiring the use of several snags over the course of a lifetime of a bat. Bats frequently shift maternity roosts, possibly to find snags with better thermal conditions when the bark on the previous roost is no longer suitable (Rabe and others 1998).

The presence of water is considered a special habitat feature for the Yuma myotis because it forages mostly by flying low over water (permanent or seasonal) and feeding on emerging aquatic insects (Whitaker and others 1977). Although less dependent on water, long-legged myotis (Ormsbee and McComb 1998) and long-eared myotis (Ports and Bradley 1996) forage over or near water, and the fringed myotis frequently forages over thickets along streams (Nagorsen and Brigham 1993). In shrubland habitats, nearby riparian woodlands may provide the only available roost sites. Thus, all species in group 26 have a strong association with water and riparian vegetation.

Broad-scale changes in source habitats—When the need for suitable roost sites is ignored, few changes have occurred in the extent of source habitats between historical and current periods (figs. 79A, B). Declining trends were most pronounced in the northern half of the Columbia Plateau and in the Upper Snake ERU, and increasing trends occurred mostly in the southern half of the Columbia Plateau, and in a few watersheds of the Northern Glaciated Mountains, Upper Klamath, Central Idaho Mountains, and Snake Headwaters ERUs (fig. 79C). Neutral trends in habitat extent were found in 59 percent of watersheds within the basin, and neutral trends predominated in all 13 ERUs (fig. 80). In most ERUs, the number of watersheds with increasing trends exceeded those with declining trends (fig. 80).

Interpreting Results

Composition and structure associated with changes in source habitats—Neutral trends in habitat extent reflect the ability of species in group 26 to use a wide variety of cover types and nearly all structural stages of forests as source habitats. The basin has experienced dramatic declines in old-forest structural stages of all forest cover types (Hann and others 1997; vol. 3, appendix 1, table 4). For group 26, however, these losses have been offset by increases in mid-seral stages that also serve as source habitats, as long as suitable roost sites are available.

Declines in the northern portion of the Columbia Plateau, the southern portion of the Central Idaho Mountains, and portions of the Owyhee Uplands and Upper Snake ERUs are due to losses of big sagebrush and mountain big sagebrush to agriculture (Hann and others 1997). Increases in the Northern Glaciated

Figure 79—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 26 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 80—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 26, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Mountains are due primarily to areal increases in managed young forests of interior Douglas-fir and interior ponderosa pine (vol. 3, appendix 1, table 4). Increases in the Central Idaho Mountains are due primarily to areal increases in managed young forests and understory reinitiation stages of several forest cover types, including Engelmann spruce-subalpine fir, interior Douglas-fir, grand fir-white fir, lodgepole pine, and western larch (vol. 3, appendix 1, table 4).

Within the riparian woodlands community group, old forests had strongly declining trends throughout the basin (vol. 3, appendix 1, table 4) and generally remain only in stands smaller than the 1-km^2 (0.4-mi^2) mapping unit used in this analysis. These losses occurred from changes in historical hydrologic regimes: reservoirs have eliminated many aspen and cottonwood-willow stands, a lowered water table has reduced others, and loss of periodic flooding has prevented establishment of seedlings (Merigliano 1996, Rood and Heinze-Milne 1989).

Condition of special habitat features—The number of caves has not changed significantly from historical to current times, but human disturbance from recreation has increased, causing some caves to be less available to hibernating bats. Mines proliferated in the early part of the historical period and provided additional habitat, but during the 1980s, thousands of abandoned mines throughout the West were closed with no input from biologists, thereby resulting in unknown loss of established roosts (Idaho State Conservation Effort 1995). The extent of cliffs and rocky areas has not changed since the historical period, but habitat quality of some cliffs has declined because of human disturbances (Lehmkuhl and others 1997).

Large-diameter snags >53 cm (21 in) have been reduced basin-wide in roaded areas with a history of timber sales (Hann and others 1997, Hessburg and others 1999, Quigley and others 1996). Consequently, the neutral trends in source habitats for the long-legged myotis may give a more positive assessment of habitat availability than is actually the case.

In addition to riparian woodlands large enough to map at the broad scale, smaller patches of riparian vegetation have declined in extent basin-wide, because of disruption of hydrologic regimes from dams, water diversions, and road construction, along with grazing and trampling of riparian vegetation by livestock and increased recreational use along stream courses (USDA Forest Service 1996). These fine-scale changes have caused additional declines in bat foraging habitat and potential roost sites.

Other factors affecting the group—Roost availability has greatly influenced the distribution of all Nearctic bat species (Humphrey 1975), and the conservation of group 26 bats is largely dependent on maintaining suitable roost sites. The most straightforward source of impact is destruction of the structure, that is, loss of snags through timber harvests, and removal of old buildings and bridges or closure of mines and caves for safety reasons (Perlmeter 1995, Pierson and others 1991). Perkins and Peterson (1997) attributed the low detection of bats in the Owyhee Mountains to the lack of suitable roosts, particularly in the form of cottonwood and juniper snags.

The second source of impact is disturbance of roosting bats, primarily by recreational activities in or near caves but also from mining, road construction, road access and any other activities near roosts (Pierson and others 1991). During winter, rising out of torpor requires a large caloric output, and repeated disturbances can drain the energy reserves of a bat and lead to starvation (Nagorsen and Brigham 1993). Recreational use of caves during the hibernation and nursery periods seriously affects persistence of individual colonies if disturbances are frequent (Nagorsen and Brigham 1993).

The third source of impacts at roost sites is purposeful killing of bats. Because of their high visibility at colonial roosts, bats have suffered high mortality rates; total loss of colonies has occurred from shooting by individuals who often are guided by negative folklore regarding bats (Nagorsen and Brigham 1993). Destruction of a single colony may represent a significant impact across large areas because of the patchy distribution of bats related to roost availability.

Roads may indirectly affect bat species by increasing human access to roost sites. Caves have become more accessible, thereby increasing the amount of human visitation and potential harassment of bats. The presence of roads increases the likelihood that snags will be cut for safety concerns or fuel wood (see Hann and others 1997). The additional loss of snags in areas where snag densities are currently low could limit populations of group 26 species.

Direct contact with pesticides can cause illness or death in bats. Although most organochlorine pesticides that cause accumulation of chemicals up the food chain have been banned or highly restricted in the United States, the relatively short-lived organophospates can provide high risks during application (Clark 1988). For example, a large die-off of bats observed in Arizona after application of methyl parathion was believed to be linked to direct contact with this chemical (Clark 1988).

Population status and trends—There are insufficient population data on any species in group 26 to determine population trends. In general, however, bats in the basin are believed to be declining because of increased human disturbance of roosts, declining snag densities, decrease of late-seral lower montane and montane forests, decreased acreage and quality of riparian areas, pesticide use, direct killing, and decreases in water quality (Lehmkuhl and others 1997).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 26 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Our results, combined with literature and other empirical information, suggest that the following issues are important for group 26:

1. Basin-wide loss of large-diameter snags (>53 cm [21 in]) for the long-legged myotis maternity roosts and day roosts.

2. Destruction of roosts, disturbance of roosting bats, or both.

3. Degradation and loss of native riparian vegetation.

4. Impacts of pesticides on bats and their prey.

5. Lack of information on hibernacula, including locations, special features, and numbers of bats associated with them.

6. Lack of population trend data.

Potential strategies—The following strategies could be used to maintain and improve habitat for these bat species:

1. (To address issue no. 1) Actively manage for the retention and recruitment of large-diameter snags in all forest cover types and structural stages.

2. (To address issue no. 2) Protect all roosts and reduce human disturbances near roosts.

3. (To address issue no. 3) Maintain and improve the condition of riparian and wetland vegetation for bat foraging areas.

4. (To address issue no. 4) Alleviate impacts of pesticides on bat populations.

5. (To address issues no. 5 and no. 6) In cooperation with other state, Federal, and tribal agencies, establish a coordinated approach to search for hibernacula, and to protect these sites.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Retain existing snags, particularly if >53 cm (21 in) and provide measures for snag replacement. Review existing snag guidelines or develop guidelines that reflect local ecological conditions and address snag numbers, diameter, height, decay class, species, and distribution. Retain snags in clusters to provide adjacent roosts for maternity colonies. Maintain snags at higher than historical levels to restore loss in previously harvested areas (ICBEMP 1996d).

2. (In support of strategy no. 1) Emphasize retention of snags that provide best solar exposure to bark or cavity roost sites (Betts 1996).

3. (In support of strategy no. 1) Reduce road densities in managed forests where snags are currently in low abundance. Close roads after timber harvests and other management activities, and minimize the period when such roads are open to minimize removal of snags along roads. In addition or as an alternative to road management, actively enforce fuel wood regulations to minimize removal of snags.

4. (In support of strategy no. 1) Restrict fuel wood permits to disallow snag cutting where snags are in low abundance, and particularly where existing roads cannot be closed. Blair and others (1995) recommend that public fuel wood harvest should be limited to trees <38 cm (15 in) d.b.h.

5. (In support of strategy no. 2) Monitor known roosts for potential human disturbances, and initiate closures of recreational or construction activity near roost sites.

6. (In support of strategy no. 2) If possible, stabilize old structures that are important for maternity roosts and hibernacula.

7. (In support of strategy no. 2) Survey caves, mines, and abandoned buildings before removal or closure, and protect roosting bats from human presence and disturbance. During closures, use specialized gates designed to allow continued use of mines and caves by bats (Pierson and others 1991).

8. (In support of strategy no. 2) Assure that construction of roads and rights-of-way are not going to cause siltation, slumping, or water run-off to enter cave habitats or alter other roosting structures (Perkins 1992-1994).

9. (In support of strategy no. 3) Identify areas of existing riparian and wetland habitats that are important bat foraging areas, and design conservation measures to protect and enhance foraging opportunities for bats.

10. (In support of strategy no. 3) Modify grazing practices to improve condition of degraded riparian areas for bat foraging and roosting.

11. (In support of strategy no. 3) Restore degraded areas by appropriate mechanical treatments and with seedings of appropriate native species.

12. (In support of strategy no. 4) Avoid pesticide use in areas of high bat foraging activity or near nursery colonies.

13. (In support of strategy no. 5) Use existing interagency cooperative agreements, or develop agreements where needed to conduct surveys for hibernacula.

14. (In support of strategy no. 5) Use individual project planning (such as timber sales, road construction, mineral extraction, or recreational development) as opportunities for conducting surveys for new roost sites and to assess population status of known roosts.

Group 27—Pine Siskin and Townsend's Big-Eared Bat

Results

Species ranges, source habitats, and special habitat features—Group 27 includes the pine siskin and the Townsend's big-eared bat, both of which are year-round residents of the basin. The pine siskin occurs throughout the basin except for low-elevation, nonforested areas, and the Townsend's big-eared bat is found basin-wide (fig. 81).

Both species are forest generalists within the subalpine, montane, upland woodland, and riparian woodland community groups. Most cover types within these community groups are source habitats for both species, but Engelmann spruce-subalpine fir is considered source habitat for only the pine siskin, whereas aspen is used only by the big-eared bat. Source habitat for both species was considered to be in all structural stages except the stem-exclusion and stand-initiation stages (vol. 3, appendix 1, table 1). Source habitats for the big-eared bat also include several cover types within the upland shrubland, upland herbland, and riparian shrubland community groups (vol. 3, appendix 1, table 1).

No special habitat features were identified for the pine siskin. Breeding takes place in various conifer species, including ornamentals, and foraging occurs in trees, shrubs, and grassy areas (Dawson 1997). Diet consists primarily of small seeds from annual plants, conifers, and deciduous trees (Dawson 1997). Pine siskin populations are highly irruptive on a continental scale, causing local abundance or scarcity of siskins from one year to the next, apparently in response to food availability (Bock and Lepthien 1976, Dawson 1997).

The Townsend's big-eared bat is colonial in its use of caves and cavelike structures for nursery colonies, day roosts, and hibernacula (Idaho State Conservation Effort 1995, Nagorsen and Brigham 1993; vol. 3,

Figure 81—Ranges of species in group 27 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

appendix 1, table 2). Big-eared bats do not roost in crevices like many other bat species but rather restrict their roosting sites to the ceilings of cavelike structures (caves, mines, and buildings), where they aggregate in large colonies. A stable, cold temperature and moderate airflow may be important criteria for hibernation (Genter 1986, Humphrey and Kunz 1976). The distribution of big-eared bats is patchy across the basin because of their restrictive roosting requirements.

The big-eared bat is a moth specialist (Idaho State Conservation Effort 1995; Nagorsen and Brigham 1993; Whitaker and others 1977, 1981). In central Oregon, they forage in sagebrush, bitterbrush, and open ponderosa pine forests (Dobkin and others 1995).

Broad-scale changes in source habitats— Source habitats were widespread across the basin historically, with greatest concentrations in the mountains of the Northern Cascades, Southern Cascades, Upper Klamath, Blue Mountains, Northern Glaciated Mountains, Upper Snake, and Snake Headwaters ERUs (fig. 82A). Extensive shrubland and grassland habitats suitable only for the big-eared bat occurred in the Columbia Plateau, Northern Great Basin, and Owyhee Uplands. The current extent of habitat is similar to the historical distribution (fig. 82B), although the abundance of habitat has changed in some areas. Watersheds with declining trends were primarily in the northern half of the Columbia Plateau, the Upper Snake, and Snake Headwaters ERUs (figs. 82C and 83). Watersheds with increasing trends were mostly in the Upper Klamath, Blue Mountains, Northern Glaciated Mountains, and Central Idaho Mountains (figs. 82C and 83). Basin-wide, the number of watersheds with declining, increasing, or static trends was nearly equal, representing 34, 34, and 31 percent of watersheds, respectively (fig. 83).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Mixed trends in habitat extent reflect the association of both species in group 27 with several cover types and nearly all structural stages of forests as source habitats. The basin has experienced dramatic declines in old-forest structural stages of all forest cover types (Hann and others 1997; vol. 3, appendix 1, table 4), but for group 27, these losses have been offset by increases in mid-seral stages that also serve as source habitats. Increases in the areal extent of habitats in the Upper Klamath were due to transitions from the fescue-bunchgrass cover type to mixed-conifer woodlands and an areal increase in the extent of interior Douglas-fir, historically less than 2 percent, but currently 15 percent of the ERU (Hann and others 1997). In the Blue Mountains, Northern Glaciated Mountains, and Central Idaho Mountains, increasing trends were largely due to increases in the areal extent of grand fir-white fir. Engelmann spruce-subalpine fir increased in the Central Idaho Mountains as well (Hann and others 1997; vol. 3, appendix 1, table 4).

Static trends in nonforested habitats are partially due to transitions from big sagebrush to juniper/sagebrush and juniper woodlands (Hann and others 1997), which have resulted in no net change in source habitats for the big-eared bat. Declines have occurred in the northern portion of the Columbia Plateau because of transitions from big sagebrush to agriculture (Hann and others 1997).

Condition of special habitat features—The number of caves likely has stayed the same from historical to present periods, but human disturbance from recreation has increased, thereby causing some caves to be abandoned by big-eared bats (Idaho State Conservation Effort 1995). Mines proliferated in the early part of the historical period and provided additional habitat, but during the 1980s, thousands of abandoned mines throughout the West were closed with no input from biologists, thereby resulting in unknown loss of established roosts (Idaho State Conservation Effort 1995).

Other factors affecting the group— Pine siskin foraging behavior, geographic location, and population levels are highly influenced by the combination of current population level and food availability—an abundance of seeds will cause the population to expand, and if the next year’s crop is unable to support the expanded population, the birds will move elsewhere (Bock and Lepthien 1976).

Because the distribution of Townsend’s big-eared bats is dependent on specialized roosting requirements, alterations and disturbances of any structures used for day roosts, nursery colonies, or hibernacula (caves, mines, old buildings) could affect the persistence of individual colonies. The most straightforward

Figure 82—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 27 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

VIEW FIGURE

Figure 83—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 27, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

source of impact is destruction of the structure, that is, removal of old buildings or closure of mines and caves for safety reasons (Pierson and others 1991).

The second source of impact is disturbance of roosting bats, primarily by recreational activities in or near caves but also from mining, road construction, and any other activities near roosts (Idaho State Conservation Effort 1995). Females at nursery colonies are alert and readily take flight if disturbed (Perkins and Schommer 1992), and frequent interruptions are known to result in abandonment of the roost site (Idaho State Conservation Effort 1995, Nagorsen and Brigham 1993). During winter, rising out of torpor requires a large caloric output, and repeated disturbances can drain the energy reserves of a bat and lead to starvation (Nagorsen and Brigham 1993). Recreational use of caves during the hibernation and nursery periods seriously affects persistence of individual colonies if disturbances are frequent (Idaho State Conservation Effort 1995, Nagorsen and Brigham 1993).

The third source of impacts at roost sites is purposeful killing of roosting bats (Idaho State Conservation Effort 1995). Because of their high visibility at colonial roosts, big-eared bats have suffered high mortality rates and sometimes total loss of a colony from shooting by individuals who often are guided by negative folklore (Nagorsen and Brigham 1993). Destruction of a single colony may represent a significant impact on big-eared bats across large areas because of the patchy distribution of bats related to roost availability.

The big-eared bat is negatively affected by the presence of roads. Increased road networks have made caves more accessible and have increased the amount of human visitation and potential harassment.

Because the big-eared bat is insectivorous, use of insecticides in foraging areas has the potential to impact bat species, primarily by reducing the prey base. For example, forest spraying for tussock and spruce budworm moths, although targeted at the larval stage of these insects, ultimately affects the number of flying adults and can cause a sufficient reduction in the prey base to suppress a year or two of Townsend's bat reproduction (Perkins and Schommer 1992). Also, exposure to insecticides can directly affect the health of bats. Although most organochlorine pesticides that cause accumulation of chemicals up the food chain have been banned in the United States or their use highly restricted, the relatively short-lived organophospates can cause illness or death to bats during application (Clark 1988).

Population status and trends—Population trends for the pine siskin are difficult to obtain because the irruptive tendencies of this species result in highly variable annual numbers at any given locale (Dawson 1997). The BBS data show no significant population trends in most states, Canadian provinces, or BBS physiographic regions because of wide fluctuations in numbers or insufficient routes to determine a trend (Sauer and others 1996). Two areas with significant annual declines from 1966 to 1995, however, have been reported, which reflect possible population trends in the basin: an annual decline of 4.5 percent ($n = 52$, $P < 0.01$) has occurred on BBS routes in Washington, and an annual decline of 4.1 percent ($n = 196$, $P < 0.01$) has occurred in USDI Fish and Wildlife Service Region 1 (five Western states) (Sauer and others 1996).

Wintering populations of the big-eared bat seem to have declined, based on a comparison of counts made at hibernacula in central Oregon in the 1960s compared to the 1980s (Perkins 1987). In general, several species of bats in the basin have declined because of increased human disturbance of roosts, declining snag densities, decrease of late-seral lower montane and montane forests, decreased acreage and quality of riparian areas, pesticide use, direct killing, and decreases in water quality (Lehmkuhl and others 1997).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 27 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Our results, combined with literature and other empirical information, suggest that the following issues are important for group 27:

1. Unknown causes for population declines of pine siskins.

2. Direct loss of big-eared bat roosts because of cave and mine closures and destruction of abandoned buildings.

3. Excessive disturbance of roosting bats because of human activities.

4. High mortality of roosting bats or total loss of colonies because of vandalism and shooting.

5. Reduction in bat prey base (moths) through excessive use of insecticides.

Potential strategies—Strategies for reversing the declining trends in pine siskin populations are difficult to formulate because of the irruptive nature of siskin populations at the continental scale. The following strategies have been identified to reverse broad-scale declines in populations of the big-eared bat:

1. (To address issue no. 2) Protect all known roost sites (nursery, day roosts, and hibernacula) of big-eared bats and restore historical roosts where feasible.

2. (To address issue no. 3) Reduce levels of human activities around known bat roosts.

3. (To address issue no. 4) Reduce vandal-related mortalities of roosting bats

4. (To address issue no. 5) Reduce impacts of insecticide use on principal prey of big-eared bats.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Survey all mines and caves scheduled for public closure for big-eared bats before closure. If roosting colonies are found, or if the structure has potential as a roosting colony, carry out the closure with gates that allow bats to enter and exit the structure. Unless superseded by other designs, use the bat gate designs in Tuttle and Taylor (1994), also presented in appendix B of Idaho's conservation strategy for Townsend's big-eared bat (Idaho State conservation Effort 1995). If possible, stabilize old structures that are important for maternity and hibernacula sites (Perkins 1992-1994).

2. (In support of strategy no. 2) Initiate seasonal public closures of caves used as big-eared bat roosts during critical time periods, by using signs, road closures, and bat gates.

3. (In support of strategy no. 2) Reduce surveys to the minimum needed for assessing colony health and population status. Coordinate research efforts to minimize entry of roosts for data collection.

4. (In support of strategy no. 3) Increase public education and awareness of bat ecology and the current conservation status of big-eared bats.

5. (In support of strategies no. 2 and no. 3) Reduce human access to bat roosting structures by closing roads that facilitate access to such habitat.

6. (In support of strategy no. 4) Avoid or minimize application of pesticides near bat roosts (Perkins 1992-1994). Utilize a 3.2-km (2-mi) "no-spray" buffer zone around roost sites (Idaho State Conservation Effort 1995). Within a 16-km (10-mi) radius of known roosts, use a strip-spraying technique to reduce the amount of area sprayed.

Group 28—Spotted Bat, Pallid Bat, and Western Small-Footed Myotis

Results

Species ranges, source habitats, and special habitat features—Group 28 consists of three bat species that generally are associated with low-elevation woodlands and shrublands: the spotted bat, pallid bat, and western small-footed myotis. The spotted bat and pallid bat occur in low numbers throughout eastern Washington and Oregon, and the spotted bat also occurs in eastern and southern Idaho (fig. 84). The small-footed myotis is somewhat more abundant and occurs throughout the basin except for high-elevation sites in the Cascade Range (fig. 84).

This analysis addresses year-round source habitat for all three species. The small-footed myotis is known to hibernate in the basin, but it is not known whether the spotted bat and pallid bat hibernate or leave the basin

Figure 84—Ranges of species in group 28 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

during winter (Nagorsen and Brigham 1993). With no migratory information, we have assumed that source habitats for all three species include winter hibernacula.

Cover types used as source habitats by all species in group 28 include interior ponderosa pine, juniper woodland, juniper/sagebrush, big sagebrush, mountain big sagebrush, and low sage (vol. 3, appendix 1, table 1). Additional cover types used as source habitats by one or two group members include cottonwood-willow (small-footed myotis), interior Douglas-fir and shrub wetlands (spotted bat), and salt desert shrub (spotted and pallid bats). Within interior ponderosa pine, the pallid bat is limited to old-forest structural stages, whereas the spotted bat and small-footed myotis also use young forest and understory reinitiation stages (vol. 3, appendix 1, table 1). All three species use both open- and closed-canopy structures of the shrub cover types.

A special habitat feature associated with all source habitats is the presence of cliffs or other rocky areas for roost sites (vol. 3, appendix 1, table 2). For the spotted and pallid bats, it is not necessary for roost structures to be adjacent to foraging areas because the spotted bat is known to travel up to 10 km (6.2 mi) between day roosts and feeding areas (Wai-Ping and Fenton 1989), and the pallid bat commutes up to 4 km (2.5 mi) (Nagorsen and Brigham 1993). Distances farther than these, however, would render shrub habitats unsuitable as source foraging areas. Commuting distances have not been reported for the small-footed myotis, but it seems to be versatile in its selection of roost sites, using boulders, vertical banks, and talus slopes in addition to cliffs (Nagorsen and Brigham 1993). Within this group, the spotted bat appears most limited in roost site selection, with all roosts reported in crevices of high cliffs (Nagorsen and Brigham 1993, Sarell and McGuinness 1993, Wai-Ping and Fenton 1989). The pallid bat primarily roosts in rock crevices but also uses tree cavities, buildings, and mines (Nagorsen and Brigham 1993).

The small-footed myotis and spotted bat are both aerial feeders, with diets that differ according to local prey availability (Nagorsen and Brigham 1993). In eastern Oregon, the small-footed myotis was reported to consume primarily moths, true bugs, and flies (Whitaker and others 1981). In eastern British Columbia, the spotted bat consumed mostly moths (Wai-Ping and Fenton 1989). The pallid bat can aerial feed, but mostly gleans prey from vegetation and the ground. In eastern Oregon, the diet was grasshoppers and moths (Whitaker and others 1981).

Broad-scale changes in source habitats—Historically, source habitats for group 28 were concentrated in the Columbia Plateau, Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs, and patchily distributed elsewhere in the basin (fig. 85A). The current distribution of habitats resembles the historical extent; significant losses of habitat in the Columbia Plateau and total loss of the former patchy habitats have occurred in the Upper Clark Fork ERU (fig. 85B). Trends in habitat extent were variable across the basin, but in general, habitats declined in the northern portion of the basin and were static to increasing in the south, except for the Snake Headwaters, a southern ERU with declining trends (fig. 85C).

About one-third of the watersheds within the basin had static trends in the areal extent of source habitats, but nearly half had declining or strongly declining trends (fig. 86). Eighty percent of watersheds in the Lower Clark Fork and 54 percent of watersheds in the Columbia Plateau had declining and strongly declining trends (fig. 86). Increasing and strongly increasing trends were projected in 43 percent of the watersheds in the Southern Cascades and 50 percent of the watersheds in the Upper Klamath (fig. 86). These represent the two ERUs with the highest percentages of increasing habitat extent for group 28.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats—Throughout the basin, declines in source habitats of shrubland bats were associated with declines in big sagebrush, mountain big sagebrush, and old-forest structural stages of interior ponderosa pine and interior Douglas-fir (vol. 3, appendix 1, table 4). Source habitats declined in the Columbia Plateau and Snake Headwaters because of the conversion of 46 and 41 percent of the big sagebrush cover type to agriculture within each ERU, respectively (Hann and others 1997). In the Lower Clark Fork ERU, 66 percent of the interior ponderosa pine cover type was replaced by grand fir-white fir (Hann and others 1997), a cover type that does not serve as source habitat for group 28.

Figure 85—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 28 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 86—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 28, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Increases in habitat extent generally were due to increases in juniper woodlands and juniper/sagebrush cover types (vol. 3, appendix 1, table 4). These increases often occurred in ERUs that experienced declines in native shrublands, resulting in overall mixed trends (for example, in the Owyhee Uplands) (vol. 3, appendix 1, table 4).

Condition of special habitat features—The extent of cliffs and rocky areas in the basin has not changed since the historical period, but the habitat quality of some cliffs has declined because of human disturbances (Lehmkuhl and others 1997).

Other factors affecting the group— Human disturbance can affect bat nursery colonies by disrupting young during the critical periods of growth and development. For spotted and pallid bats, nursery colonies are often inaccessible, and therefore disturbance potentials are low. The exception could occur if one or more rock climbing routes passed through a nursery colony and were visited frequently by climbers. Currently, no situation of this kind has been identified in the basin, but this may be due to a lack of monitoring rather than an absence of nursery colony-climber interactions.

Human activities can result in habitat degradation or disturbance at day roosts. Examples include road construction, dam building, mineral extraction, and the stabilizing of hazardous falling rocks above developments (Sarell and McGuinness 1993).

Direct contact with pesticides can cause illness or death in bats. Although most organochlorine pesticides that cause accumulation of chemicals up the food chain have been banned in the United States or their use highly restricted, the relatively short-lived organophospates can provide high risks during application (Clark 1988). For example, a large die-off of bats was observed in Arizona after the application of methyl parathion, and was believed to be linked to direct contact with this chemical (Clark 1988).

Pesticides also can impact bat populations by reducing the availability of arthropods that serve as prey. Bats in group 28 are impacted by the spraying of forests and agricultural crops for insect pests.

Population status and trends— Population estimates for bat species in the basin are either unknown or local in scale. Lehmkuhl and others (1997), however, reported that habitat conditions for most bat species have declined significantly from historical conditions because of the conversion of native vegetation to agriculture and urban development, increased human disturbance of roosts, reduced large snag densities, decreased acreage and distribution of late-seral montane and lower montane forests, and reduced acreage and quality of riparian areas.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 28 with broader, ecosystem-based objectives for all other resources.

Issues—Our results and the conclusions drawn from published literature suggest the following issues are important for group 28:

1. Loss of native shrub vegetation.

2. Disturbances at nursery and day roosts.

3. Impacts of pesticides on bats and their prey.

4. Lack of information on hibernacula, including locations, special habitat features, and numbers of bats associated with them.

5. Lack of population trend data.

Potential strategies—The following strategies could be used to maintain and improve habitat for these bat species:

1. (To address issue no. 1) Maintain and improve the condition of native shrublands to provide foraging areas.

2. (To address issue no. 2) Reduce human disturbances near known roosts.

3. (To address issue no. 3) Alleviate impacts of pesticides on bat populations.

4. (To address issues no. 4 and no. 5) In cooperation with other state, Federal, and tribal agencies, establish a coordinated approach to search for hibernacula.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Identify areas of existing native shrubland that could be managed for long-term persistence of native shrub cover types.

2. (In support of strategy no. 1) Explore options under the CRP (Johnson and Igl 1995), or develop other incentive programs to encourage restoration of agricultural areas to native cover types. Focus on areas that would increase patch size or links with existing source habitat patches.

3. (In support of strategy no. 1) Restore degraded areas by appropriate mechanical treatments and with seedings of native shrub, grass, and forb species.

4. (In support of strategy no. 2) Monitor known nursery roosts for potential disturbances, and initiate seasonal closures of recreational activity where appropriate. For example, seasonal restrictions on rock climbing would be appropriate if climbing routes passed through spotted bat nursery colonies.

5. (In support of strategy no. 2) Provide access for bats when mines are permanently closed.

6. (In support of strategy no. 2) Conduct surveys for bat roosts and hibernacula before road construction, mineral extraction, or slope stabilization where such activities are scheduled to occur near cliffs or caves with potential roosts. Provide mitigation or seasonal restrictions of potentially disturbing activities within the appropriate planning documents.

7. (In support of strategy no. 3) Avoid pesticide use in areas of high bat foraging activity or near nursery colonies.

8. (In support of strategy no. 4) Use existing interagency cooperative agreements, or develop agreements where needed to conduct surveys for hibernacula.

Group 29—Western Bluebird

Results

Species ranges, source habitats, and special habitat features—Group 29 consists of migratory breeding habitat for western bluebirds. Within the basin, western bluebirds are distributed across eastern Oregon and Washington, northern and western Idaho, and northwestern Montana (fig. 87). They are present in all ERUs except the Upper Snake and Snake Headwaters.

Western bluebirds use open forest stands and woodlands in combination with shrub and grass habitats. Specific source habitats (vol. 3, appendix 1, table 1) include old forest, single-storied western white pine and ponderosa pine; old-forest aspen; stand-initiation stages of most montane forest and lower montane forest community groups; juniper and white oak woodlands; the open-canopy low-medium shrub stage of most of the upland shrub community type; and native bunchgrasses and forbs. Additionally, burned pine forests created by stand-replacing fires likely are source habitats (Saab and Dudley 1998). Burned habitats, however, were not identified for this analysis.

Juxtaposition of forested and open areas is a necessary characteristic of source habitats for western bluebirds because they typically nest in tree cavities and forage for insects in adjacent openings (DeGraaf and others 1991; vol. 3, appendix 1, table 2). Because juxtaposition of cover types is important for nesting western bluebirds, they are considered a “contrast” species, and a finer scale analysis is needed to fully evaluate the status of their source habitats.

Western bluebirds are secondary cavity-nesters, so snags are a special habitat feature (vol. 3, appendix 1, table 2). They will use old woodpecker holes, natural cavities, and nest boxes (Brawn and Balda 1988, DeGraaf and others 1991). Their nests are located in open forests or at forest edges. In burned ponderosa pine forests of western Idaho, nesting western bluebirds favored partially salvage-logged compared to unlogged stands (0.44 nests per km surveyed [0.71 nests per mi] in logged vs. 0.16 nests per km [0.26 nests per mi] in unlogged) (Saab and Dudley 1998). Openings in partially logged, burned forests likely provided greater opportunities for aerial foraging by the bluebirds. In salvaged units, snag (>23 cm [9 in] d.b.h.)

Figure 87—Ranges of species in group 29 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

densities at bluebird nest sites (n = 65) averaged 65 ± 5.9 snags per ha (26.3 ± 2.4 snags per acre), and at nonnest random sites (n = 180) 31.4 ± 1.9 snags per ha (12.7 ± 0.8 snags/acre). Average diameter of nest trees in the burned forests of western Idaho was 34.8 ± 1.5 cm (13.7 ± 0.6 in).

Broad-scale changes in source habitats— Source habitats for western bluebirds declined strongly throughout most of the basin. Throughout the basin, source habitats for western bluebird had declined strongly in 50 percent of watersheds and moderately in another 25 percent of watersheds (figs. 88 and 89). The apparent strong negative trends were in seven ERUs: the Northern Cascades, Southern Cascades, Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, and Upper Clark Fork (fig. 89). More moderate declining trends were projected for the Upper Klamath and Central Idaho Mountains (fig. 89), whereas there was little change in source habitats from historical to current in the Northern Great Basin and Owyhee Uplands (fig. 89).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Hann and others (1997, see table 3.139) reported ecologically significant basin-wide declines for four of the terrestrial communities that support components of western bluebird source habitats. Communities that declined significantly were early-seral lower montane forest, late-seral lower montane single-layer forest, upland shrublands, and upland herblands. Of the terrestrial communities providing source habitats for bluebirds, only upland woodlands showed a basin-wide significant increase from historical to current (table 3.139 in Hann and others 1997). Decreases in habitats important to western bluebirds were also significant at the level of individual ERUs. The upland herb community declined significantly in all 11 ERUs within the range of the western bluebird, early-seral lower montane forest and late-seral lower montane single-layer forest declined in 10 ERUs, upland shrub declined in 8 ERUs, and early-seral montane forest declined in 6 ERUs (tables 3.141 through 3.165 in Hann and others 1997). Late-seral single-layer montane forest declined

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Figure 88—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 29 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 89—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 29, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

in four ERUs while increasing in five ERUs, and upland woodlands declined in three ERUs while increasing in six ERUs. Our evaluation at the broad scale did not assess the distribution of foraging habitat in relation to that for nesting habitat. Additional analysis of the juxtaposition of foraging with nesting habitats is needed at a finer scale of resolution. Results for source habitats shown here for both the current and historical time periods are likely overestimates as they do not take into account the need for juxtaposition of habitats.

Condition of special habitat features—Densities of large-diameter snags (>53 cm [21 in] d.b.h.) have declined basin-wide from historical to current levels (Hann and others 1997, Hessburg and others 1999, Quigley and others 1996). Trends in densities of smaller snags are variable (Hann and others 1997).

The scale of the analysis does not allow determination of change in the amount of edge or amount of edge habitat. Thus, this special habitat feature was not evaluated for changes in source habitats presented in the above results. Some levels of decrease in total habitat area may be associated with increases in edge habitat. Consequently, the large decreases reported here for western bluebird habitat may be somewhat mitigated by increases in edge as habitat blocks are harvested.

Other factors affecting the group— Some western bluebirds that breed in the basin migrate to California and Baja California in winter (DeGraaf and others 1991). Conditions on these wintering grounds could affect the status of populations in the basin. Western bluebirds respond positively to artificially constructed nest boxes in areas where the availability of cavities is limiting. In one study (Brawn and Balda 1988), bluebird densities increased from 8 to 31 pairs per 40 ha (100 acres) after the construction of nest boxes. Usurpation of nest cavities by Lewis' woodpeckers (Saab and Dudley 1995) could have negative effects on western bluebirds. Stress and elevated energetic costs could be associated with territorial encounters with Lewis' woodpeckers and potentially reduce reproductive success of western bluebirds.

Population status and trends— Saab and Rich (1997) reported that western bluebird populations in the basin were stable over the period 1968-94 based on BBS data. Stable population trends also have been reported for this western species throughout its range for the period 1966-96 (Sauer and others 1996). Specialized monitoring techniques may be needed for better estimates of bluebird population trends (Saab and Rich 1997).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 29 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Primary issues affecting source habitats of western bluebirds are as follows:

1. Reductions in snag densities.

2. Reductions in early- and late-seral montane and lower montane forests.

3. Possibly unsustainable conditions in late-seral montane and lower montane forests where there have been large transitions from shade-intolerant to shade-tolerant species.

4. Reductions and degradation of native upland shrublands and herblands.

Potential strategies—Habitat for western bluebirds could be improved by implementing the following strategies:

1. (To address issue no. 1) Maintain large remnant trees and snags in all seral stages of montane, lower montane, and woodland forests.

2. (To address issue no. 2) Maintain and restore early- and late-seral montane and lower montane forests where those types have been reduced in extent. Both the extent and pattern of these habitats are of concern because source habitats for western bluebirds are found in edge areas. Where possible, retention efforts for late-seral forests should be focused on areas where the potential for stand-replacing fires is low (USDAForest Service 1996).

3. (To address issue no. 3) Restore fire regimes that maintain a natural mosaic of shrublands and forests in those ERUs and portions of ERUs where substantial habitat remains (for example, Northern

Great Basin, Owyhee Uplands, southern portion of Columbia Plateau). In some areas, such strategies will result in temporary declines and periodic fluctuations in habitat abundance.

4. (To address issue no. 4) Restore native upland shrub and herblands.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Snag management practices could be designed to retain snags along forest edges in areas used by nesting western bluebirds, and artificial nest boxes could be used to help support western bluebird populations in areas where snags are not available as nesting structures.

2. (In support of strategy no. 1) In burned ponderosa pine-Douglas-fir forests selected for postfire salvage logging, retain about 65 snags per ha (26 per acre) of snags >23 cm (9 in) d.b.h.

3. (In support of strategies no. 2 and no. 3) Use wildfire and prescribed fire to restore natural forest openings and enhance shrub understories to attract insect prey.

4. (In support of strategy no. 3) Accelerate development of mid-successional stages of ponderosa pine to old forests by silvicultural treatments of prescribed underburning and thinning of small-diameter trees (<25 cm [9 in] d.b.h.).

5. (In support of strategy no. 4) Discourage spread of exotic plants by minimizing human-associated disturbance activities.

Group 30—Ash-Throated Flycatcher and Bushtit

Results

Species ranges, source habitats, and special habitat features—Group 30 consists of the bushtit and ash-throated flycatcher. The bushtit is a year-long resident in the basin, whereas the ash-throated flycatcher is a summer migrant. For both the ash-throated flycatcher and the bushtit, the basin constitutes the northern edge of their ranges. Both species have similar distributions within the basin, occurring along the western and southern extent of the basin (fig. 90).

The bushtit and ash-throated flycatcher depend on a similar mix of source habitats (vol. 3, appendix 1, table 1), including mixed-conifer woodlands, juniper/sagebrush woodlands, Oregon white oak, and mountain mahogany. Cottonwood/willow in the old-forest multi-storied structural stage also is considered source habitat for the ash-throated flycatcher.

Ash-throated flycatchers nest in cavities (either natural, woodpecker-excavated, or human-made [nest boxes]) of taller trees and snags (Austin and Russell 1972, Dunning and Bowers 1990, Sharp 1992). Snags were identified as a special habitat feature for ash-throated flycatchers (vol. 3, appendix 1, table 2). Bushtits place their nests in tall shrubs. Both species forage on arthropods.

Broad-scale changes in source habitats—Source habitats for this group historically were distributed within the western and southern parts of the basin, and watersheds with habitat appeared to be disjunct (fig. 91A). Currently, source habitats are more abundant and in some areas more continuous in distribution (fig. 91B). The largest concentration of both current and historical habitats is within the southern part of the Columbia Plateau (figs. 91A, B). The watersheds with increases in source habitats were most often the same as or adjacent to watersheds that supported source habitats historically (figs. 91A, B).

Overall, source habitats for this group strongly increased within the basin. Over 60 percent of the watersheds in the basin had strongly increasing trends, whereas about 17 percent had decreasing trends (fig. 92). Nearly 50 percent or more of the watersheds in seven of the nine ERUs with greater than 1 percent of the area as source habitats had strongly increasing trends since the historical period (fig. 92). These were the Upper Klamath, Northern Great Basin, Columbia Plateau, Blue Mountains, Owyhee Uplands, Upper Snake, and Snake Headwaters. Only the Northern Cascades had a greater number of watersheds with decreasing rather than increasing amounts of source habitat (fig. 92). The Southern Cascades generally had no net trend (fig. 92). The amount of source habitat in the Northern Glaciated Mountains is minimal (<1 percent of the ERU) (vol. 3, appendix 1, table 3).

Figure 90—Ranges of species in group 30 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 91—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 30 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 92—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 30, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats—The increasing trend in source habitats was attributed to increases in the juniper/sagebrush cover type (vol. 3, appendix 1, table 4). The extent of juniper/sagebrush woodlands has more than doubled in the basin, primarily because of excessive livestock grazing and fire suppression (Hann and others 1997).

Broad-scale trends in the other source habitat types, especially old-forest cottonwood-willow, Oregon white oak, and mountain mahogany, are difficult to determine at the 1-km^2 (0.4-mi^2) scale of analysis because of small patch size or linear configuration of these cover types across the basin.

Condition of special habitat features—The trend and condition of nest cavities for ash-throated flycatchers are unknown. Presumably, as the number of juniper trees increases, the aging of these junipers will produce natural cavities as snags develop and older branches fall off.

Other factors affecting the group—The primary prey for these species during the breeding season is insects (Ehrlich and others 1988, Sharp 1992). Native understory shrubs and grasses provide important substrates for production of insects, and excessive grazing can reduce or eliminate many of these key substrates for insects.[12]

A common management action is to reduce the densities of juniper especially where encroachment of or densities of junipers have increased. Removal of juniper may improve rangeland productivity and restore native biodiversity in some areas; however, management efforts to remove juniper trees would negatively affect source habitats for group 30.

Population status and trends—Data for ash-throated flycatchers and bushtits in the basin were insufficient to determine a population trend. Because both species have naturally low population numbers and narrow distributions, specialized monitoring techniques are required to estimate their numbers (Saab and Rich 1997).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 30 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Primary issues affecting source habitats for ash-throated flycatchers and bushtits are as follows:

1. For ash-throated flycatchers, loss of trees with natural cavities or trees suitable for excavation by other species because of juniper removal.

2. Degradation and loss of native understory shrubs and grasses that provide substrates for arthropod prey.

Potential strategies—The issues identified above suggest the following broad-scale strategies would be effective in contributing to the long-term persistence of bushtits and ash-throated flycatchers:

1. (To address issue no. 1) Retain contiguous blocks of mature juniper/sagebrush, especially in areas containing old junipers with cavities and hollow centers for potential nest sites of ash-throated flycatchers. Old-growth specimens usually have round or flat tops as compared to young, actively growing individuals that have a symmetrical, cone-shaped top (Oregon Department of Fish and Wildlife 1994)

2. (To address issue no. 2) Protect and restore native understory shrubs and grasses in source habitats.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Consider site-specific ecological potential and response to management before removing juniper trees.

2. (In support of strategy no. 1) Retain junipers with cavities and hollow centers that are potential nest sites for ash-throated flycatchers.

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[12] Personal communication. 1997. David Dobkin, wildlife biologist, High Desert Ecological Research Institute, 15 SWColorado, Suite 300, Bend, OR 97702.

3. (In support of strategy no. 1) Retain blocks of old-growth juniper during juniper control projects.

4. (In support of strategy no. 2) Restrict the use of herbicides, pesticides, and grazing in areas with contiguous blocks of source habitat that have intact native understories.

5. (In support of strategy no. 2) Restore native understories through seedings and plantings of native shrubs and grasses.

6. (In support of strategy no. 2) Minimize the likelihood of invasion of exotic vegetation by minimizing human-associated disturbances such as road building, motorized activity, grazing, and mining.

Group 31—Ferruginous Hawk, Burrowing Owl, Short-Eared Owl, Vesper Sparrow, Lark Sparrow, Western Meadowlark, and Pronghorn

Results

Species ranges, source habitats, and special habitat features—Group 31 consists of breeding habitat for the migratory ferruginous hawk, burrowing owl, vesper sparrow, lark sparrow, and western meadowlark, and year-round habitat for the short-eared owl and pronghorn. The short-eared owl, vesper sparrow, and western meadowlark are the most widely distributed species within this group (fig. 93), occurring throughout the basin. Less widely distributed are the burrowing owl and lark sparrow, which are both absent from the mountainous portions of central and northern Idaho (fig. 93). The ferruginous hawk uses less of the basin but is still widespread in the lower elevations (fig. 93). The least widely distributed species in this group is the pronghorn, which currently occupies most of the Northern Great Basin ERU, a large part of the Owyhee Uplands ERU, and small, disjunct areas over the southern half of the basin (fig. 93). In contrast, the historical range of the pronghorn included almost all of southern Idaho and eastern Oregon (fig. 93). Nelson (1925) stated that pronghorn historically occurred in Washington as well, but Yoakum (1978) disagreed. We have followed the recommendations of the latter author.

Source habitats for this group include various shrub, grass, and herbaceous cover types (vol. 3, appendix 1, table 1). All seven species have source habitats in big sagebrush and fescue-bunchgrass cover types, six share low sagebrush, and five have source habitats in juniper/sagebrush, mountain big sagebrush, native forb, and wheatgrass bunchgrass types. Whereas particular plant species may differ geographically, a key feature of this group is their preference for open cover types with a high percentage of grass and forbs in the understory. All species use the shrub component of the vegetation directly for nest sites, perch sites, or hiding cover. Pronghorn move into areas of higher shrub cover during winter. The ferruginous hawk is the only species that will use trees, especially junipers, which provide preferred nest sites in some geographic areas.

Burrowing owls depend on burrows and natural cavities in lava flows or rocky areas for nest sites; thus, burrows are a special habitat feature for this species (vol. 3, appendix 1, table 2). Burrows are almost always provided by burrowing mammals such as ground squirrels, marmots, prairie dogs, coyotes, and badgers, and the use of an area by owls may be closely tied to populations of these mammals (Haug and Oliphant 1990, Rich 1986, Thomsen 1971).

Populations (White and Thurow 1985) and productivity (Bechard and Schmutz 1995, Schmutz and Hungle 1989, Steenhof and Kochert 1985) of the ferruginous hawk fluctuate in response to prey population densities. Similarly, breeding populations of the short-eared owl are nomadic, and high densities of breeding birds may occur when rodent densities are high (Marti and Marks 1989). Thus, the status of all three raptors in this group is rather closely tied to the status of various mammal populations. Notably, these three raptor species are more tolerant of degraded shrub-steppe habitats with exotic vegetation than are other species in this group.

Significant correlations were documented between the coverage of grass and the densities of western meadowlark ($r = 0.62, P < 0.001$) and lark sparrow ($r = 0.37$, $P < 0.05$) (Wiens and Rotenberry 1981). Similar correlations occurred for the coverage of litter and these songbird species ($r = 0.36$, $P < 0.05$ and $r = 0.34$, $P < 0.05$, respectively).

Pronghorn may depend on free water during summers of dry years when they cannot meet water requirements from succulent forbs (Beale and Smith 1970,

Figure 93—Ranges of species in group 31 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

VIEW

MAP

Figure 93—Ranges of species in group 31 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 94—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 31 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Clemente and others 1995). In most years, however, availability of free water probably does not affect pronghorn habitat use (Deblinger and Alldredge 1991).

Broad-scale change in source habitats—Historically, source habitats for this group were widely available throughout the basin, but particularly in the Northern Great Basin, Columbia Plateau, Owyhee Uplands, and Upper Snake ERUs (fig. 94A). The most contiguous shrub-steppe habitat occurs at lower elevations, and source habitats for this group become less extensive at higher elevations. This is demonstrated by the narrow band of watersheds with 25 to 50 and 0 to 25 percent of area in source habitats within higher elevation ERUs (fig. 94B).

The projected extent of decreasing and strongly decreasing trends in source habitats was dramatic (fig. 94C). The Columbia Plateau and Upper Snake ERUs were dominated by decreasing trends, the latter having no watersheds with increasing trends. In contrast, large, contiguous portions of the Northern Great Basin and Owyhee Uplands ERUs, areas of higher elevation and precipitation, show a stable trend and continue to provide source habitats for this group.

Basin-wide, 54 percent of the watersheds had moderately or strongly declining trends in source habitats (fig. 95). The Columbia Plateau ERU historically provided the most watersheds with source habitats for this group (fig. 95), but over 72 percent of the watersheds in that ERU had moderately or strongly declining trends. The second most important ERU, the Owyhee Uplands, had stable trends in about 81 percent of its watersheds, but another 19 percent were moderately or strongly declining. The number of watersheds with moderately or strongly declining trends in source habitats outnumbered those with increasing trend in all other ERUs (fig. 95) except the Central Idaho Mountains.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats—The single largest loss in cover types within the basin has been the decline in big sagebrush (Hann and others 1997). Habitat losses were also significant for fescue-bunchgrass and wheatgrass bunchgrass (Hann and others 1997). This loss was most striking in the Columbia Plateau and Upper Snake ERUs (figs. 94C and 95). Other notable reductions include the near complete loss of source habitats in the Upper Clark Fork and Lower Clark Fork ERUs.

In the Columbia Plateau, major losses from historical conditions occurred in big and mountain sagebrush types, which declined by nearly half and over three-fourths, respectively (vol. 3, appendix 1, table 4). Native grass cover types also were heavily impacted, with a three-fourths decline in wheatgrass bunchgrass, and a nearly total loss of fescue-bunchgrass (Hann and others 1997). In the lower elevations of the Owyhee Uplands, big sagebrush was reduced by 25 percent (Hann and others 1997). Fescue-bunchgrass types had significant negative declines in nine ERUs (Hann and others 1997). Nearly all of the native forb cover type, source habitats for five of these species, was converted to other cover types (Hann and others 1997). Native forbs were projected to have covered a small portion of the basin historically but likely provided important local breeding habitats within larger blocks of more xeric vegetation.

In the Central Idaho Mountains ERU, nearly 33 percent of the watersheds had strongly increasing trends (fig. 95). This was attributed to large relative increases in juniper/sagebrush, juniper woodlands, and low sagebrush, all of which covered only a small fraction of the unit. A similar situation resulted in strongly increasing trends in the Northern Cascades, Blue Mountains, Northern Great Basin, and Snake Headwaters ERUs (fig. 95; vol. 3, appendix 1, table 4; Hann and others 1997). Any increases in wheatgrass bunchgrass or native forb cover types (vol. 3, appendix 1, table 4) should be viewed with caution because these cover types can be dominated by exotic vegetation, which is not considered source habitat for species of this group.

Several factors contributed to large-scale losses of sagebrush and fescue-bunchgrass habitats; foremost was conversion to agriculture. Agricultural lands have increased significantly in every ERU in the basin (vol. 3, appendix 1, table 4). In fact, the largest transitions among terrestrial communities from the historical to current periods were that of upland shrubland and upland herbland to agriculture (Hann and others 1997). This transition explains much of the pattern evident in figure 94C.

Figure 95—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 31, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

A second factor contributing to loss of sagebrush habitat was conversion of shrub-steppe vegetation to exotic forbs and annual grass. Notable portions of the Owyhee Uplands and Upper Snake ERUs underwent a conversion from upland shrubland to exotic herbland (Hann and others 1997). Conversion of native vegetation to exotics was augmented by the increased frequency of wildfire and by improper grazing (Quigley and others 1996, USDAForest Service 1996).

Condition of special habitat features—Burrowing owls rely on burrows provided by burrowing mammals for nest sites (Haug and Oliphant 1990, Rich 1986, Thomsen 1971). Populations of many burrowing mammals have declined because of various pest control programs, which may have reduced nest site availability for burrowing owls. No special habitat features were identified for other members of this group.

Other factors affecting the group—Losses of native perennial grass and forb understories within the sagebrush types, associated with intensive livestock grazing, cheatgrass invasions, and noxious weed invasions, are microhabitat changes that could not be evaluated by our broad-scale analysis. Because species in group 31 favor grass or shrub-grass types for nesting, foraging, or hiding, we know that the grass component of historical shrublands was important (for example, Wiens and Rotenberry 1981, Marti and Marks 1989). Removal of grass cover by livestock potentially has detrimental effects on the short-eared owl (Marti and Marks 1989). Finer scale analysis is needed to determine the extent of this problem because the broad-scale data may show source habitats in upland shrub types, where the shrubs are present but the understory is gone. The presence of livestock also may attract brown-headed cowbirds and subsequently increase the incidence of brood parasitism (Robinson and others 1995). The western meadowlark and vesper sparrow are common cowbird hosts, whereas the lark sparrow is only occasionally parasitized (Ehrlich and others 1988).

Ferruginous hawks prefer trees for nest sites, particularly junipers (Jasikoff 1982), and are most common in the juniper/sagebrush ecotone (Powers and others 1973, Smith and Murphy 1973, Thurow and others 1980). Expansion of juniper woodlands and juniper/sagebrush in the basin as a result of fire suppression likely has benefitted the species.

Fields of hay and cereal grains attract vesper sparrows (Perritt and Best 1989) and western meadowlarks (Lanyon 1994) for nesting, where nests, young, or adults may be destroyed during harvest. Short-eared owls and lark sparrows also likely are affected by this process. These fields function as sinks for local populations.

Species in this group evolved in shrub-steppe habitats where microbiotic crusts were broadly distributed (see Kaltenecker and Wicklow-Howard 1994). Microbiotic, or cryptogamic, crusts consist of lichens, bryophytes, algae, microfungi, cyanobacteria, and bacteria growing on or just below the soil surface in arid and semiarid environments (Kaltenecker and Wicklow-Howard 1994); these crusts developed without large herds of grazing ungulates (St. Clair and Johansen 1993). In addition, these crusts are projected to have been widely distributed throughout the source habitats for this group, particularly in the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs but also scattered in the Columbia Plateau ERU (Hann and others 1997, map 3.59). Increasing evidence suggests that microbiotic crusts improve soil stability, productivity, and moisture retention, moderate extreme temperatures at the soil surface, and enhance seedling establishment of vascular plants (Belnap and Gardner 1993, Harper and Pendleton 1993, Johansen and others 1993, St. Clair and others 1993), thus contributing to high ecological integrity of shrub-steppe habitats. Idaho BLM has recognized the potential importance of microbiotic crusts by proposing standards for rangeland health that include the maintenance of these crusts to ensure proper functioning and productivity of native plant communities (USDI Bureau of Land Management 1997). These crusts were widely destroyed by trampling during the excessive livestock grazing of the late 1800s and early 1900s (Daubenmire 1970, MacCracken and others 1983, Mack and Thompson 1982, Poulton 1955). Currently, high-intensity grazing and altered fire regimes modify shrub-steppe plant communities and threaten the maintenance and recovery of microbiotic crusts (Belnap 1995, Kaltenecker 1997, St. Clair and Johansen 1993).

Roads, human activities, and domestic dogs are known to impact ferruginous hawks, short-eared owls, burrowing owls (Bechard and Schmutz 1995, Green and Anthony 1989, Lokemoen and Duebbert 1976, Olendorff and Stoddart 1974, Ramakka and Woyewodzic 1993, Schmutz 1984, White and

Thurow 1985) and western meadowlarks (Lanyon 1994). Harassment of pronghorn by snowmachine and all-terrain vehicles stresses animals at all times of the year (Autenrieth 1978). Pronghorn also avoid sheep dogs (Yoakum and O'Gara 1990). Human disturbance might be especially significant for those species that are attracted to features of the agricultural-shrubland or agricultural-grassland contact zones; that is, burrowing owl, short-eared owl, and pronghorn.

Recreational shooting of marmots and ground squirrels impacts burrowing owls because the owls are accidentally or deliberately shot, whereas more general illegal shooting impacts short-eared owls (Marti and Marks 1989). Pesticide use leads to direct mortality in burrowing owls, short-eared owls (Marti and Marks 1989), and western meadowlarks (Griffin 1959) and an indirect loss in burrowing owls through a reduction in the populations of burrowing mammals.

Pronghorn movement is restricted or completely impeded by net-wire and other fences that prevent them from crossing beneath the lower strand (Helms 1978, Oakley and Riddle 1974, Yoakum 1980). Roads are readily crossed by pronghorn, but snow accumulating in roadside ditches also might present barriers to movement during winter (Bruns 1977).

Population status and trends—Based on BBS data summarized for the basin (Saab and Rich 1997), significant declines were reported for the period 1966-94 for western meadowlark (-0.8 percent per yr, $n \geq 14$, $P < 0.10$) and lark sparrow (-2.9 percent per yr, $n \geq 14$, $P < 0.05$). Saab and Rich (1997) identified western meadowlark and lark sparrow as two of 15 species that are of high concern to management under all future management themes for the basin. Vesper sparrow, burrowing owl, and ferruginous hawk had stable population trends within the basin for the same time period (Saab and Rich 1997). In physiographic region 89 (Columbia Plateau), which corresponds to much of the range of this group within the basin, trends over the period 1966-95 (Sauer and others 1996) were positive for the ferruginous hawk (+6.3 percent per yr, $n = 18$, $P < 0.05$).

Burrowing owl populations are increasing across the West (+6.3 percent per yr; $n = 116$, $P < 0.001$; Sauer and others 1996). No detectable trend was found for the short-eared owl in the basin (Saab and Rich 1997) or in physiographic region 89 (Columbia Plateau; Sauer and others 1996). Marti and Marks (1989) reported that short-eared owl numbers were stable, with fluctuating populations.

Burrowing owls, short-eared owls, and ferruginous hawks are not adequately monitored by the BBS technique so apparent population trends, or the lack thereof, for these species may not be reliable (Saab and Rich 1997).

An estimated 99 percent of the continental pronghorn population was killed by indiscriminate hunting between 1850 and 1900, but numbers have increased dramatically since then in Idaho and Oregon (Yoakum 1968, 1978, 1986a; Yoakum and O'Gara 1990). Populations reached peaks in 1989 of 21,800 in Idaho and 22,650 in Oregon (O'Gara 1996). The most recent estimates (1995) are 12,500 in Idaho and 17,122 in Oregon (O'Gara 1996).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 31 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The condition of the habitat for group 31 can be summarized by the composite ecological integrity ratings (Quigley and others 1996, p. 122) that show most of the habitat to have a "low" rating. Fescues and bunchgrasses—critical habitat components for this group— ". . . were irreversibly modified by extensive grazing in the late 1800s and early 1900s" (USDA Forest Service 1996, p. 51). Most of the current habitat for this group was classified into Rangeland Clusters 5 (generally corresponding to much of the Owyhee Uplands ERU) and 6 (generally the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs) where the primary risk to ecological integrity is "continued declines in herbland and shrubland habitats" (Quigley and others 1996, p. 112, 114). Further, Rangeland Cluster 6 has the additional risk of being ". . . highly sensitive to overgrazing and exotic grass and forb invasion" (Quigley and others 1996, p. 114). These widespread and overriding issues provide a clear statement of the problems facing this group over the long term.

Primary issues:

1. Permanent and continued loss of large acreage of shrub-steppe and fescue-bunchgrass habitat because of agricultural conversion, brush control, and cheatgrass invasion.

2. Soil compaction and loss of the microbiotic crust.

3. Adverse effects of human disturbance. For the burrowing owl, a primary issue is the loss of nesting burrows through poisoning and recreational shooting of burrowing mammals. For ground-nesting birds, the issue is nest mortality in agricultural fields from farm machinery during spring weed control and early harvests. For pronghorn, a primary issue is disruption of movement patterns because of fence constructions that inhibit passage. For all species in group 31, the issue is general disruption of breeding activity and movements because of human intrusion.

Potential strategies—

1. (To address issue no. 1) Identify and conserve large remaining areas (contiguous habitat >1000 ha [2,470 acres]) of shrub-steppe vegetation where ecological integrity is still relatively high, and manage to promote their long-term sustainability. Large contiguous blocks of public land in the Northern Great Basin and Owyhee Uplands are the most obvious sites. These generally include the subbasins in Rangeland Cluster 5 (Quigley and others 1996). These areas will provide long-term habitat stability for populations and provide the anchor points for restoration, corridor construction, and other landscape-level management.

2. (To address issue no. 1) Restore the grass and forb components of the shrub-steppe cover types to approximate historical levels throughout the basin.

3. (To address issue no. 2) Restore the microbiotic crust in ERUs where potential for redevelopment is high; that is, in areas near propagule sources that have suitable soil, vegetation, and climatic characteristics [see Belnap 1993, Belnap 1995, Kaltenecker 1997, Kaltenecker and Wicklow-Howard 1994]). Ecological reporting units with highest potential for redevelopment include the Northern Great Basin, Owyhee Uplands, Upper Snake, and to a lesser extent, the Columbia Plateau (map 3.59 in Hann and others 1997).

4. (To address issue no. 3) Maintain burrows for nesting and roosting by burrowing owls. Reduce mortality of ground-nesting birds in agricultural areas. Construct fences in pronghorn range that allow pronghorn passage. Minimize the adverse effects of human intrusion.

Practices that support strategies— The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Identify large areas of high ecological integrity to be managed for sustainability by analyzing current vegetation, precipitation patterns, elevation, temperature (Klemmedson and Smith 1964, Morrow and Stahlman 1984, Stewart and Hull 1949), and the presence of priority species in this group. These sites most likely will be successful on large areas of Federal land managed by BLM. Evaluation criteria for protection or enhancement include maintaining or increasing the size of smaller patches, preventing further habitat fragmentation, and protecting or increasing the size and integrity of corridors among patches, all in connection with the location of core areas.

2. (In support of strategy no. 1) Explore options under the CRP (Johnson and Igl 1995), or develop other incentive programs, to encourage restoration of agricultural areas to native cover types. Focus on areas that would increase patch size or links with existing source habitat patches.

3. (In support of strategy no. 2) Use fire prevention and suppression to retard the spread of cheatgrass in areas that are susceptible to cheatgrass invasion but currently are dominated by native grass species. Planting of fire-resistant vegetation through "green stripping" is being experimentally tested (Pellant 1994) and may be used to protect existing vegetation.

4. (In support of strategy no. 2) Restore selected areas of cheatgrass monocultures through seeding and other manipulations (Allen 1995, Daubenmire 1970, Evans and Young 1978, Hosten and West

1994, Kennedy 1994, Monsen and McArthur 1995, Ogg 1994, Whisenant 1995, Yoakum 1986b), where such restoration would increase the size of existing shrub-steppe patches or provide links between patches.

5. (In support of strategy no. 2) Restore native vegetation by appropriate treatments and seedings of native shrub, grass, and forb species.

6. (In support of strategy no. 2) Design livestock grazing systems to promote an abundance of forbs and grasses in the understory (Yoakum 1980).

7. (In support of strategy no. 3) Encourage the redevelopment of microbiotic crust by reducing or eliminating livestock grazing in areas where restoration of microbiotic crusts is encouraged (Mack and Thompson 1982, St. Clair and others 1993). Explore the use of ground-based and aerial soil inoculation to increase the speed and extent of dispersal of the organisms that create microbiotic crust (Belnap 1993).

8. (In support of strategy no. 4) Allow burrowing mammals such as ground squirrels and marmots to persist or expand to provide nesting burrows for burrowing owls (Coulombe 1971; Gleason and Johnson 1985; Rich 1984, 1986). Provide artificial burrows for burrowing owls where burrowing mammals must be controlled (Trulio 1995).

9. (In support of strategy no. 4) Modify agricultural practices to minimize direct mortality of nesting birds by delaying hay mowing until young birds are fledged (Clark 1975, Rodenhouse and others 1995, Vickery 1996). Avoid surface tillage for spring weed control. An alternative is to use the "undercutting" method, which is much less detrimental to meadowlarks (Rodgers 1983).

10. (In support of strategy no. 4) Control, reduce, or eliminate pesticide applications in and around agricultural areas, especially in the Columbia Plateau ERU where source habitats are small and virtually all surrounded by agricultural lands (USDAForest Service 1996). The Upper Snake ERU, and to a lesser extent the Owyhee Uplands, also have relatively many miles of interface with agricultural lands.

11. (In support of strategy no. 4) Avoid construction of net-wire and similar fences in pronghorn habitat or in pronghorn migration routes (Oakley and Ridle 1974). Modify existing fences and construct new fences in pronghorn range with the following specifications (these are standard policy on BLM lands occupied by pronghorns): bottom wire at least 41 cm (16 in) from the ground and smooth, not barbed; next wire up is 66 cm (26 in) from the ground; top wire is 91 cm (36 in) from the ground (Yoakum 1980).

12. (In support of strategy no. 4) Protect pronghorn winter ranges and fawning areas from intrusion by snowmobiles and all-terrain vehicles (Autenrieth 1978) through timed access control and area closures. Minimize access roads and, where possible, locate them on the periphery of the pronghorn use areas (Autenrieth 1978). Provide artificial nesting structures in areas away from human disturbance to attract ferruginous hawks to safer sites (Apple 1994, Niemuth 1992, Schmutz 1984). Protect burrowing owl nesting sites from disturbance by domestic dogs (Green and Anthony 1989, Martin 1983).

Group 32—Preble's Shrew, Uinta Ground Squirrel, White-Tailed Antelope Squirrel, Wyoming Ground Squirrel, Washington Ground Squirrel, Striped Whipsnake, Longnose Snake, Ground Snake, Mojave Black-Collard Lizard, and Longnose Leopard Lizard

Results

Species ranges, source habitats, and special habitat features—Group 32 consists of year-round habitat for the residents in this group: Preble's shrew, Uinta ground squirrel, white-tailed antelope squirrel, Wyoming ground squirrel, Washington ground squirrel, striped whipsnake, longnose snake, ground snake, Mojave black-collared lizard, and longnose leopard lizard.

***Mammals*—**Little is known about the Preble's shrew, but they may be widely distributed in the basin (fig. 96), based on records from the area's borders (Cornely and

Figure 96—Ranges of species in group 32 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 96—Ranges of species in group 32 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

VIEW

MAP

Figure 96—Ranges of species in group 32 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

others 1992, Zeveloff and Collett 1988). Among the four species of ground squirrels, the Uinta is restricted to the upper end of the Snake River drainage in the Snake Headwaters, Upper Snake, and Central Idaho Mountains ERUs (fig. 96). The range of the white-tailed antelope squirrel occurs in the Northern Great Basin and Owyhee Uplands ERUs and is nearly distinct from that of the Uinta ground squirrel (fig. 96). Two subspecies of the Wyoming ground squirrel occur in the basin, *Spermophilus elegans nevadensis* that overlaps with the antelope squirrel in the Owyhee Uplands, and *Spermophilus elegans aureus* that overlaps with the Uinta ground squirrel in northeastern Idaho (fig. 96). Finally, both the current and historical (fig. 96) range of the Washington ground squirrel is allopatric with the other three species, being confined almost entirely to the northern part of the Columbia Plateau ERU. The current range of the Washington ground squirrel is reduced and disjunct compared to the historical period.

Reptiles**—**The striped whipsnake is widely distributed at lower elevations in Washington, Oregon, and Idaho (fig. 96). Narrowly distributed and largely sympatric, the longnose snake and ground snake occur only in the Owyhee Uplands (fig. 96). The Mojave black-collared lizard has a distribution similar to the previous two species but has an additional portion of its range in the Northern Great Basin (fig. 96). Finally, the longnose leopard lizard is found largely in the Owyhee Uplands but has disjunct populations in the Northern Great Basin, Upper Snake, Columbia Plateau, and Southern Cascades ERUs.

Source habitats for group 32 include several shrub, grass, and herbaceous cover types (vol. 3, appendix 1, table 1). All 10 species have source habitats in big sagebrush, mountain big sagebrush, fescue-bunchgrass, and wheatgrass bunchgrass types. Ten species also have source habitats in low sage, whereas eight share juniper/sagebrush or mountain mahogany.

The striped whipsnake uses cliffs and talus where they occur in source habitats; these are special habitat features for this species (vol. 3, appendix 1, table 2). Preble's shrew requires a good understory of forbs and grasses and a dense overstory of sagebrush; it is associated with more mesic sites near ephemeral and perennial streams (Ports and George 1990). Down logs provide important foraging and hiding cover (vol. 3, appendix 1, table 2). Washington ground squirrels prefer deeper soils with less clay at 10 cm (4 in) and at 50 cm (20 in) compared to unoccupied sites (Betts 1990).

Talus slopes, canyon rims, and shadscale habitats are preferred over other types by ground snakes and collared lizards (Diller and Johnson 1982, Whitaker and Maser 1981). Collared lizards similarly prefer rock outcrops and sparse vegetation (Sanborn and Loomis 1979). Striped whipsnakes are much more apt to be encountered on canyon rims than on mid-slopes or in canyon bottoms (Gerber and others 1997).

Broad-scale changes in source habitats—Historically, source habitats for this group were projected to occur throughout the basin, with greatest concentrations in the Northern Great Basin, Columbia Plateau, Owyhee Uplands, and Upper Snake ERUs (fig. 97A). Substantial amounts of source habitats also occurred in the Blue Mountains, Northern Glaciated Mountains, Central Idaho Mountains, and Upper Klamath ERUs. Only the most mountainous and forested regions did not support members of this group.

The extent of decreasing and strongly decreasing trends in source habitats was dramatic (fig. 97C), particularly for the state of Washington, the northern half of Oregon, and the upper Snake River drainage. Nine ERUs had declining trends for most watersheds, whereas only two ERUs (Northern Great Basin and Owyhee Uplands) showed stable trends. The only noteworthy source habitat increases were in the Central Idaho Mountains (fig. 98).

Basin-wide, 56 percent of the watersheds showed a moderately or strongly declining trend in source habitats (fig. 98). The Columbia Plateau ERU historically provided the most watersheds with source habitats for this group (fig. 98). But over 83 percent of the watersheds in that ERU had moderately or strongly declining trends, and only about 5 percent were increasing. In the Blue Mountains, nearly 84 percent of the watersheds had moderately or strongly declining trends (fig. 98), and <4 percent were increasing. The Upper Snake ERU had no watersheds with increasing trends (fig. 98) and over 67 percent with moderately or strongly declining trends. In the Owyhee Uplands, over 81 percent of watersheds had stable trends, and 17 percent had moderately or strongly declining trends (fig. 98).

Figure 97—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 32 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 98—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 32, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Declines in source habitats were primarily due to reductions in the amount of big sagebrush, fescue-bunchgrass, wheatgrass bunchgrass, and interior ponderosa pine (Hann and others 1997). These losses were most striking in the Columbia Plateau and Upper Snake ERUs (fig. 97B; vol. 3, appendix 1, table 4). In the Columbia Plateau, big and mountain sagebrush types declined by nearly half and three-quarters, respectively, from historical conditions. Wheatgrass bunchgrass declined by three-fourths and fescue-bunchgrass was nearly eliminated (Hann and others 1997) in the Columbia Plateau.

Large-scale losses of sagebrush and native bunchgrass habitats were primarily due to conversion to agriculture. Basin-wide, the largest transitions among terrestrial communities from the historical to current periods were that of upland shrubland and upland herbland to agricultural (Hann and others 1997).

Another factor contributing to loss of sagebrush habitat is conversion of shrub-steppe vegetation to exotic forbs and annual grass. Substantial portions of the Owyhee Uplands and Upper Snake ERUs have undergone conversions from upland shrubland to exotic herbland (Hann and others 1997). Noteworthy increases in this cover type have occurred in all major shrub-steppe ERUs. Conversion of native vegetation to exotics is augmented by the increased frequency of wildfire and by improper grazing (Braun and others 1976, Daubenmire 1970, Evans and Young 1978, Quigley and others 1996, USDA Forest Service 1996).

Any increases in wheatgrass bunchgrass or native forb cover types (vol. 3, appendix 1, table 4) should be viewed with caution because these cover types can be dominated by exotic vegetation, which is not considered source habitat for species of this group. Additionally, in some cases the wheatgrass bunchgrass cover type was misclassified as an upland herbland group instead of an early-seral forest group that was created as a result of timber harvest or recent large-scale wildfires (see Hann and others 1997).

Relatively large increases have occurred in the source habitats of juniper woodlands (tripled), mountain mahogany (tripled), juniper/sagebrush (doubled), and low sage (one-third increase) in the Central Idaho Mountains (Hann and others 1997) (vol. 3, appendix 1, table 4).

Condition of special habitat features—The availability of mesic sites used by the Preble's shrew has declined as part of the general and widespread decline in riparian habitat conditions throughout the basin (Lee and others 1997, Quigley and others 1997). Cliffs and talus habitat for the striped whipsnake, although difficult to measure at the scale of this analysis, were estimated to be in much the same condition now as historically.

Other factors affecting the group— Poisoning and other eradication potentially affect populations of all four species of ground squirrels. Ground squirrels also are popular targets for recreational shooting. The Mojave black-collared lizard, longnose leopard lizards, and longnose snakes use small-mammal burrows for cover (Beck and Peterson 1995, Brown and others 1995, Nussbaum and others 1983, Pough 1973), and therefore could be indirectly affected by both poisoning and shooting. The effect of these factors on these species in the basin is unknown.

Accidental and deliberate mortality of snakes potentially increases with increased roading and traffic in the basin. Although the three species of snakes in this group may not be as frequently killed by vehicles as are some more common species (such as gopher snake and western rattlesnake), increasing human access to source habitats will predictably result in more deliberate killing of snakes. Currently, large areas of the Owyhee Uplands ERU support moderate to high road densities (see figs. 21 and 22 and "Species and Groups Affected by Factors Associated with Roads" in vol. 1).

The typical small size of Washington ground squirrel colonies makes them vulnerable to extirpation (Tomich 1982). Source habitats for this species were estimated to have undergone the fourth greatest decline among 91 broad-scale species of focus analyzed in this report (vol. 1, table 7). Washington ground squirrels may benefit from corridors of vegetation created by cultivation that allow exchange among colonies and general dispersal (Betts 1990).

Four of the reptilian species of this group (Mojave black-collared lizard, longnose leopard lizard, longnose snake, and ground snake), are located in isolated disjunct areas within the basin that make them vulnerable to extirpation.

Areas dominated by dense stands of cheatgrass or other exotic plants may preclude use by longnose leopard lizards (Stebbins 1985), longnose snakes (Beck and Peterson 1995), and collared lizards. In the Owyhee Uplands, areas with low vegetative cover and high amounts of bare ground or rock have the highest lizard densities (Whitaker and Maser 1981). In a study of off-road vehicle and grazing effects in the Mojave Desert in California, leopard lizards were found only in plots unused by off-road vehicles (compared with moderately and heavily used plots), and were absent from grazed plots (Busack and Bury 1974).

Because reptiles are increasingly popular as pets, all reptile species in this group, but particularly the lizards, are potentially affected by collecting (Lehmkuhl and others 1997). This impact will increase as the human population in the basin increases.

Soil compaction caused by livestock grazing could negatively affect both the longnose snake and ground snake. These burrowers benefit from loose, sandy, and friable soils (Beck and Peterson 1995, Nussbaum and others 1982).

Species in this group evolved in shrub-steppe habitats, where microbiotic crusts were broadly distributed (see Kaltenecker and Wicklow-Howard 1994). Microbiotic, or cryptogamic, crusts consist of lichens, bryophytes, algae, microfungi, cyanobacteria, and bacteria growing on or just below the soil surface in arid and semiarid environments (Kaltenecker and Wicklow-Howard 1994), and they developed without large herds of grazing ungulates (St. Clair and Johansen 1993). These crusts are projected to have been widely distributed throughout the source habitats for this group, particularly in the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs but also scattered in the Columbia Plateau ERU (Hann and others 1997, map 3.59). Increasing evidence indicates that microbiotic crusts improve soil stability, productivity, and moisture retention; moderate extreme temperatures at the soil surface; and enhance seedling establishment of vascular plants (Belnap and Gardner 1993, Harper and Pendleton 1993, Johansen and others 1993, St. Clair and others 1993), thus contributing to high ecological integrity of shrubsteppe habitats. The BLM in Idaho has recognized the potential importance of microbiotic crusts by proposing standards for rangeland health that include the maintenance of these crusts to ensure proper functioning and productivity of native plant communities (USDI BLM 1997). These crusts were widely destroyed by trampling during the excessive livestock grazing period of the late 1800s and early 1900s (Daubenmire 1970, MacCracken and others 1983, Mack and Thompson 1982, Poulton 1955). Currently, high-intensity grazing and altered fire regimes modify shrub-steppe plant communities and threaten the maintenance and recovery of microbiotic crusts (Belnap 1995, Kaltenecker 1997, St. Clair and Johansen 1993).

Population status and trends—Quantified population trends are not available for any of these species. The Washington ground squirrel has experienced range contraction (fig. 96), with 23 colonies in Washington and 12 in Oregon disappearing from 1980 to 1989. This area includes most of the colonies in the northern part of the basin (Betts 1990). This decline is wholly consistent with known habitat loss.

Lehmkuhl and others (1997) projected a decline from historical in populations of the Mojave black-collared lizard as a result of the cumulative effects of habitat loss because of agricultural conversion, exotic weed invasion, and reservoir development.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 32 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The condition of the habitat for group 32 can be summed up by the composite ecological integrity ratings (Quigley and others 1996) that show most of the habitat to have a “low” rating. Most of the current habitat for this group is classified into Rangeland Clusters 5 (generally corresponding to much of the Owyhee Uplands ERU) and 6 (generally the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs), where the primary risk to ecological integrity is "continued declines in herbland and shrubland habitats" (Quigley and others 1996). Further, Rangeland Cluster 6 has the additional risk of being ". . . highly sensitive to overgrazing and exotic grass and forb invasion" (Quigley and others 1996, p. 123). These widespread and overriding issues provide a clear statement of the problems facing this group over the long term. The results of our habitat trend analysis, combined with other literature cited here, suggest the following issues are of high priority for group 32:

1. Permanent and continued loss of large areas of shrub-steppe and fescue-bunchgrass habitat to agricultural conversion, brush control, cheatgrass invasion, and expansion of juniper woodlands and mountain mahogany.

2. Increased soil compaction and loss of the microbiotic crust.

3. Reduction in burrow availability for lizards and snakes.

4. Human-caused mortality and capture of reptiles for pets.

5. Loss of downed logs.

6. Loss of surface water and riparian vegetation.

Potential strategies—The following strategies could be used to reverse broad-scale declines in source habitats. These strategies should be applied basin-wide:

1. (To address issue no. 1) Identify and conserve remaining large areas of shrub-steppe, fescue-bunchgrass, wheatgrass bunchgrass, and other source cover types where ecological integrity is still relatively high (Gray and Rickard 1989, Rickard and Poole 1989, Schuler and others 1993, Smith 1994, Yoakum 1980). Large contiguous blocks of Federal land in the Northern Great Basin and Owyhee Uplands are the most obvious sites to consider. These generally include the subbasins in Rangeland Cluster 5 (Quigley and others 1996). However, native shrublands that currently exist on military lands in the state of Washington (Rickard and Poole 1989, Schuler and others 1993, Smith 1994) also are important. These core areas will provide long-term habitat stability for populations and provide the anchor points for restoration, corridor construction, and other landscape-level management.

2. (To address issue no. 1) Minimize further spread of juniper woodlands, juniper/sagebrush, and mountain mahogany that have expanded because of fire suppression, particularly in the Central Idaho Mountains and the Columbia Plateau.

3. (To address issue no. 2) Reduce causes of soil compaction, particularly within source habitats of the longnose snake and ground snake. This factor may be important in the Owyhee Uplands ERU in particular. Restore microbiotic crusts in ERUs with potential for redevelopment (that is, areas near propagule sources, and with suitable soil, vegetation, and climatic characteristics [see Belnap 1993, 1995; Kaltenecker 1997; Kaltenecker and Wicklow-Howard 1994]): Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs and, to a lesser extent, the Columbia Plateau ERU (Hann and others, map 3.59).

4. (To address issue no. 3) Maintain and restore small-mammal populations to provide burrows for the collared lizard, longnose leopard lizard, longnose snake, and ground snake.

5. (To address issue no. 4) Determine the impact of the capture of reptiles, especially lizards, for pets. Take action as necessary to allow wild populations to persist.

6. (To address issue no. 4) Reduce the direct and indirect effects of human disturbance on populations of species within group 32.

7. (To address issue no. 5) Increase the number of downed logs in the basin.

8. (To address issue no. 6) Improve the condition of riparian systems throughout the basin.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Identify large areas of high ecological integrity to be managed for long-term protection by analyzing current vegetation,

precipitation patterns, elevation, temperature (Klemmedson and Smith 1964, Morrow and Stahlman 1984, Stewart and Hull 1949), and the presence of priority species in this group. These sites are most likely to be successful on large areas of Federal land managed by BLM. Apply special management designations as necessary to protect these sites for the long term.

2. (In support of strategy no. 1) Explore options under the CRP (Johnson and Igl 1995), or develop other incentive programs, to encourage restoration of agricultural areas to native cover types. Focus on areas that would increase patch size or links with existing source habitat patches.

3. (In support of strategy no. 1) Avoid further loss of sagebrush cover through burning, plowing, seeding, and other brush "control" methods where sagebrush cover types are below historical levels.

4. (In support of strategy no. 1) Avoid further conversion of sagebrush and native grasslands to agricultural lands through policy and land management allocations. If conversion cannot be avoided, then tracts slated for conversion will have less impact if located so as to (a) minimize further fragmentation of shrub-steppe throughout the basin; (b) avoid further reducing the size of smaller, isolated patches, particularly in the Columbia Plateau ERU; and (c) avoid conversion in areas that currently occur in large blocks of moderate Composite Ecological Integrity (Quigley and others 1996), primarily in the Owyhee Uplands and Northern Great Basin ERUs.

5. (In support of strategy no. 1) Use fire prevention and suppression to retard the spread of cheatgrass in areas that are susceptible to cheatgrass invasion but currently are dominated by native grass species. Planting of fire-resistant vegetation through "green stripping" (Pellant 1994) should be examined for its value to protect existing vegetation as well as allow degraded sites a chance to recover.

6. (In support of strategy no. 1) Restore selected areas of cheatgrass monocultures through seeding and other manipulations (Allen 1995, Daubenmire 1970, Evans and Young 1978, Hosten and West 1994, Kennedy 1994, Monsen and McArthur 1995, Ogg 1994, Whisenant 1995, Yoakum 1986b) where such restoration would increase the size of existing shrub-steppe patches or provide links among patches.

7. (In support of strategy no. 1) Restore native vegetation by appropriate mechanical treatments and seedings of native shrub, grass, and forb species.

8. (In support of strategy no. 2) Apply wildland fire and grazing practices that arrest the advances of juniper woodlands in areas that historically did not support this vegetation type.

9. (In support of strategy no. 3) Reduce or eliminate livestock grazing in critical habitat for the ground and longnose snakes if soil compaction is found to contribute to population declines. Encourage the redevelopment of microbiotic crust by reducing or eliminating livestock grazing (Mack and Thompson 1982, St. Clair and others 1993). Explore the use of ground-based and aerial soil inoculation to increase the speed and extent of dispersal of the organisms that create microbiotic crust (Belnap 1993).

10. (In support of strategy no. 4) Allow burrowing mammals such as ground squirrels and marmots to persist or expand to provide burrows for the lizards in this group and for the longnose snake.

11. (In support of strategies no. 5 and no. 6) Minimize accidental and deliberate killing of snakes by vehicles and by humans on foot. Road densities, which provide an index to the potential for disturbance, reveal that the Owyhee Uplands, Northern Great Basin, and northern part of the Columbia Plateau ERUs are least susceptible to disturbance (Quigley and others 1996). Determine the direct effect of recreational shooting of ground squirrels on populations in this group. Effects may be serious only in local situations where the demand for this recreation and access to squirrels coincide. Washington ground squirrels are especially vulnerable because of their limited distribution and known losses to date. Avoid poisoning or otherwise controlling ground squirrel populations. Encourage and enforce laws that protect reptiles from collection.

12. (In support of strategy no. 8) Maintain strips of trees and snags along riparian corridors. Restore and enhance riparian and shoreline vegetation around permanent and seasonal water sources.

Group 33—Brewer's Sparrow, Lark Bunting, Sage Sparrow, Sage Thrasher, Sage Grouse, Pygmy Rabbit, and Sagebrush Vole

Results

Species ranges, source habitats, and special habitat features—Group 33 includes breeding habitat for the migratory Brewer's sparrow, lark bunting, sage sparrow, and sage thrasher, summer and winter range for the sage grouse, and year-round habitat for the pygmy rabbit and sagebrush vole. The basin encompasses a substantial portion of the entire range of all species in this group, with the exception of the lark bunting, which is peripheral to the basin, occurring only in the southeastern part of the basin (fig. 99). Both the pygmy rabbit and sage grouse (current range) have notable gaps in their distribution, with significant disjunct populations primarily in the Columbia Plateau ERU. The current range of the sage grouse also has disjunct populations occurring in the Upper Klamath and Snake Headwaters ERUs. In comparison, the historical range of the sage grouse (fig. 99) was substantially more extensive and included portions of the Columbia Plateau, Blue Mountains, Northern Glaciated Mountains, Lower Clark Fork, and the Upper Clark Fork ERUs, where the species does not occur today.

The seven species in this group have source habitats in two structural stages of big sagebrush and mountain big sagebrush: open canopy, low-medium shrub, and closed canopy, low-medium shrub (vol. 3, appendix 1, table 1). Four of the species (pygmy rabbit, sagebrush vole, sage grouse, and sage sparrow) also have source habitats in both structural stages of low sagebrush. Other habitats of importance are juniper/sagebrush (Brewer's sparrow, sage sparrow, sage thrasher) and the closed herb structural stage of big sagebrush (Brewer's sparrow, lark bunting, sage sparrow, and sage thrasher). Habitats used by only a single species in the group include mountain mahogany (Brewer's sparrow), salt desert shrub (sage sparrow), and herbaceous wetlands (sage grouse).

A special habitat feature for sage grouse during the brood-rearing period is riparian vegetation, especially wet meadows with forbs (vol. 3, appendix 1, table 2). Native forbs provide spring and summer food for hens and broods (Autenrieth and others 1982, Call 1979, Oakleaf 1971, Peterson 1970, Roberson 1986, Savage 1969, Wallestad and others 1975). Herbaceous vegetation is also important to sagebrush voles (Hall 1928) and pygmy rabbits (Lyman 1991), which augment their sagebrush diet with forbs and grasses. An understory composed of native grasses is believed important for most species in group 33 (Bock and Bock 1987, Connelly and others 1991, Cooper 1868, Dobler and others 1996, Gregg 1991, Hall 1928, Mullican and Keller 1986).

Bare ground is an important foraging substrate for sage sparrows and sage thrashers (Rotenberry and Wiens 1980). Brewer's sparrows, however, forage mostly in sagebrush. The value of bare ground to the other bird species in this group and the sagebrush vole is unknown. Because pygmy rabbits choose tall, dense sage for their burrows and foraging sites, we assume that vegetative cover that provides protection from predators is important (Lyman 1991) and that areas of bare ground would be avoided.

Broad-scale changes in source habitats— Source habitats for group 33 were historically widespread and continuous over much of the planning area (fig. 100A), particularly in the Columbia Plateau, Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs.

Basin-wide, nearly 48 percent of the watersheds showed a moderately or strongly declining trend in habitat, and declines exceeded increases in every ERU (fig. 101). Extensive habitat reductions were estimated in the Columbia Plateau and Upper Snake ERUs, with moderate declines in the Owyhee Uplands (figs. 100 and 101). Strongly increasing trends in habitat, however, were apparent in about 20 percent of watersheds in the Central Idaho Mountains and Columbia Plateau ERUs (fig. 101). Only the Northern Great Basin ERU has changed little from historical conditions (figs. 100 and 101).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— The single largest loss in cover types within the basin was the decline in big sagebrush (Hann and others 1997). Large-scale

VIEW

MAP

Figure 99—Ranges of species in group 33 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 99—Ranges of species in group 33 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 100—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 33 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 101—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 33, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

loss of sagebrush habitat was attributed to several factors. The first factor was conversion to agriculture. Agricultural lands have increased significantly in every ERU in the basin (Hann and others 1997). In fact, the largest transition of any terrestrial community was from upland shrubland to agriculture (Hann and others 1997). The ERUs with the biggest changes were the Columbia Plateau and Upper Snake. The former is now nearly half agricultural lands, whereas the latter is nearly one-third. These ERUs have had the greatest degree of conversion among all ERUs in the basin. Agriculture also now occupies over a tenth of the Owyhee Uplands ERU. Only the Northern Great Basin ERU has been relatively free of agricultural conversions.

A second factor contributing to loss of sagebrush habitat was conversion of shrub-steppe vegetation to exotic forbs and annual grass. Significant increases in this cover type occurred in all the major sagebrush ERUs. Exotic forbs and annual grass now occupy small portions of the Northern Great Basin, Columbia Plateau, and Owyhee Uplands ERUs, and over a tenth of the Upper Snake ERU (Hann and others 1997).

Increases in source habitats in the Central Idaho Mountains and Columbia Plateau ERUs were attributed to expansions of juniper/sagebrush and mountain mahogany cover types (Hann and others 1997).

Habitat condition for group 33 can be described by the composite ecological integrity ratings (Quigley and others 1996) that show most of the habitat to have a "low" rating. Most of the current habitat for this group was classified into Rangeland Clusters 5 (generally corresponding to much of the Owyhee Uplands ERU) and 6 (generally the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs), where the primary risk to ecological integrity is continued losses of herbland and shrubland habitats (Quigley and others 1996). Further, Rangeland Cluster 6 is vulnerable to overgrazing and to exotic grass and forb invasions (Quigley and others 1996).

Condition of special habitat features— Wet meadows and riparian vegetation, cover types used for brood-rearing by sage grouse, have declined substantially since historical times (Lee and others 1997, Quigley and others 1996).

Other factors affecting the group— Roading (Quigley and others 1996) has contributed to increased human disturbance in ERUs most important for sage grouse. Moderate road densities (0.4 to 1.0 km per km^2 [0.7 to 1.7 mi per mi^2]) are typical for the Northern Great Basin ERU, the Owyhee Uplands ERU, and the Upper Snake ERU. Roads and associated human disturbance can be especially harmful to sage grouse during the lekking and wintering periods. Habitat loss caused by roads is a direct effect.

The quality of soil may be important to the two burrowing species in this group (sagebrush vole and pygmy rabbit) because the soil must be capable of sustaining burrows. Weiss and Verts (1984) determined that burrow sites for pygmy rabbits are found in areas where soils are significantly deeper and looser than adjacent soils. Grazing, if not managed properly, can potentially damage pygmy rabbit habitat (Washington Department of Wildlife 1993b).

Voles seldom use compacted or rocky soil (Maser and others 1974) and may be absent from areas that have suffered soil erosion because of heavy livestock grazing (Maser and Strickland 1978).

Heavy livestock grazing could negatively impact other species in group 33 by altering the structure and composition of the soil and removing native herbaceous understory vegetation. Thus, areas that are currently judged to be source habitat because of the presence of sagebrush cover may not be currently suitable because of changes in soil or understory vegetation that cannot be mapped at the broad scale. Additionally, changes in natural wildfire regimes have contributed to invasions of exotic vegetation in native sagebrush habitats.

Species in this group evolved in shrub-steppe habitats, where microbiotic crusts were broadly distributed (see Kaltenecker and Wicklow-Howard 1994). Microbiotic, or cryptogamic, crusts consist of lichens, bryophytes, algae, microfungi, cyanobacteria, and bacteria growing on or just below the soil surface in arid and semiarid environments (Kaltenecker and Wicklow-Howard 1994); these crusts developed without large herds of grazing ungulates (St. Clair and Johansen 1993). In addition, these crusts are projected to have been widely distributed throughout the source habitats for this group, particularly in the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs but also scattered in the Columbia Plateau ERU (Hann and others 1997, map 3.59). Increasing evidence indicates that microbiotic crusts improve soil stability, productivity, and moisture retention; moderate extreme temperatures at the soil surface; and enhance seedling establishment of vascular plants (Belnap and Gardner 1993, Harper and Pendleton 1993, Johansen and others 1993, St. Clair and others 1993), thus contributing to high ecological integrity of shrub-steppe habitats. The BLM in Idaho has recognized the potential importance of microbiotic crusts by proposing standards for rangeland health that include the maintenance of these crusts to ensure proper functioning and productivity of native plant communities (USDI Bureau of Land Management 1997). These crusts were widely destroyed by trampling during the excessive livestock grazing of the late 1800s and early 1900s (Daubenmire 1970, MacCracken and others 1983, Mack and Thompson 1982, Poulton 1955). Currently, high-intensity grazing and altered fire regimes modify shrub-steppe plant communities and threaten the maintenance and recovery of microbiotic crusts (Belnap 1995, Kaltenecker 1997, St. Clair and Johansen 1993).

Little information is available on effects of landscape patterns on species in this group. Brewer's sparrows are known to have small territories, and individual pairs will occupy small patches of suitable habitat placed within a matrix of unsuitable vegetation. Sage thrashers also appear to use discontinuous, patchy habitats surrounded by other types but rarely occur as single pairs; the probability of habitat occupancy increases with shrub patch size (Knick and Rotenberry 1995). Sage sparrows seem to be both area sensitive and more social (Rich 1981) than the previous two species. Individual pairs essentially never occur alone. The species does not occupy small patches of habitat, and large patches of seemingly suitable habitat may be unoccupied. Thus, sage sparrows occur in large expanses of shrub-steppe where many pairs share adjacent territories (Knick and Rotenberry 1995) and apparently do not use slopes of greater than a few percent.

Disjunct patches of sagebrush that were previously connected to other patches may now be unsuitable source habitat for sage grouse because wintering flocks have large home ranges. Grouse select winter use sites based on snow depth and topography (Connelly 1982, Hupp 1987, Robertson 1991) where sagebrush is accessible. Sagebrush heights of 25 to 30 cm (10 to 12 in) and canopy cover of 10 to 25 percent, regardless of snow cover, are important for winter use by sage grouse. Because seasonal movements differ among regions and populations, this effect needs to be assessed case by case.

Populations of pygmy rabbits historically occurred in five counties in Washington, but current records indicate that populations occur in isolated fragments in only one county (Douglas) (Washington Department of Wildlife 1993b). These small, disjunct populations are susceptible to extirpation by habitat degradation and loss, as well as catastrophic events such as fire, disease, flooding, or intense predation.

The sage sparrow, Brewer's sparrow, and lark bunting are not frequently parasitized by brown-headed cowbirds (Ehrlich and others 1988). Both sparrows apparently accept the eggs (Rich 1978). The sage thrasher also is parasitized but rejects cowbird eggs (Rich and Rothstein 1985). Sage grouse using agricultural areas may be adversely affected by pesticide applications (Blus and others 1989, Post 1951, Ward and others 1942).

Population status and trends—Quantitative population trend data are available only for the bird species in group 33. No information is available for the pygmy rabbit, only anecdotal notes are available for the sagebrush vole and, because the lark bunting is peripheral to the basin, sample sizes for this species are inadequate.

Historical reports indicate that the sagebrush vole was abundant in grasslands around Walla Walla in 1868 (Cooper 1868), although it has not been found there since. Currently, other subspecies of this vole occur in higher elevation grasslands in Utah and California where sagebrush does not occur. This suggests that the species may occur today largely in shrub-steppe habitats because the large grasslands, which it may actually prefer, no longer exist. Thus, the species probably experienced substantial population declines.

Brewer's sparrow has the most clear population trend, decreasing 1.3 percent per yr ($n > 14$, $P < 0.01$) over the period 1968-94 and 4.3 percent per yr ($n > 14$, $P < 0.01$) over the period 1984-94 (Saab and Rich 1997) in the basin. This sparrow also is declining in Idaho (6.3 percent per yr, 1966-95; $n = 40$, $P < 0.01$) and in physiographic region 89 (Columbia Plateau; 5.2 percent decline over the same period, $n = 57$, $P < 0.01$) (Sauer and others 1996). Among 15 Neotropical migrants in the basin, Brewer's sparrow, sage sparrow, sage thrasher, and lark bunting were designated as species of high concern to management under all future management themes for the basin (Saab and Rich 1997).

Population trends for the sage sparrow and sage thrasher are not consistent with the population declines demonstrated by Brewer's sparrows and sage grouse. The sage sparrow shows no trend in the basin (Saab and Rich 1997) and a nonsignificant decline of -1.0 percent per yr (1966 to 1995, $n = 38$) in physiographic region 89 (Columbia Plateau; Sauer and others 1996). The sage thrasher also shows no trend in the basin (Saab and Rich 1997), a nonsignificant 1.1-percent decline per yr in Idaho ($n = 28$), a 2.1-percent per yr increase in Oregon ($n = 27$, $P < 0.01$), and a nonsignificant 0.8-percent increase in physiographic region 89 (Columbia Plateau; $n = 51$) over the period 1966-95 (Sauer and others 1996).

Sage grouse populations have shown significant, steep declines since the 1940s in Idaho,[13] Oregon (Crawford and Lutz 1985), and Washington (Tirhi 1995). The rates of decline in Idaho and Oregon are not significantly different.[14] Moreover, the rate of decline in Washington appears to be similar to that in Idaho and Oregon, thereby suggesting common, widespread factors affecting these populations. A complicating factor is that sage grouse in this geographic area may exhibit population cycles with a periodicity of around 10 years (Rich 1985, Willis and others 1993b). Thus, apparent trends over short periods should be regarded with caution. Populations in Washington were heavily impacted by habitat loss before surveys were established. Remaining populations now exist as isolated remnants (Tirhi 1995).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 33 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The results of our habitat trend analysis suggest the following issues are of high priority for group 33:

1. Permanent and continued loss of large areas of shrub-steppe habitat to agricultural conversion, brush control, and cheatgrass invasion.

2. Soil compaction, erosion, and loss of microbiotic crust.

3. Continued degradation of wet meadow and riparian vegetation adjacent to springs, seeps, and streams by improper grazing and, in some areas, spring development to provide livestock water supplies.

4. Adverse effects of human disturbance.

Potential strategies—The following strategies could be used to reverse broad-scale declines in source habitats:

1. (To address issue no. 1) Identify and conserve large remaining areas of shrub-steppe vegetation where ecological integrity is still relatively high (Gray and Rickard 1989, Rickard and Poole 1989, Schuler and others 1993, Smith 1994, Yoakum 1980). Basin-wide, maintain or restore 15 to 25 percent of sagebrush cover with heights of 36 to 79 cm (14 to 31 in) (Autenrieth 1981, Connelly and others 1991, Fischer 1994, Gregg 1991, Klebenow 1969, Patterson 1952, Peterson 1970, Wakkinen 1990, Wallestad 1975). In sage grouse winter range, maintain a mosaic of sagebrush height and cover classes to allow access to sagebrush with canopy cover of 10 to 25 percent and heights of 25 to 30 cm (10 in to 12 in) regardless of snow cover (Connelly 1982, Hupp 1987, Robertson 1991).

2. (To address issue no. 1) Restore native grass and forb understories to historical levels, where restoration potential exists, and retard the spread of nonnative vegetation.

3. (To address issue no. 2) Reduce and eliminate soil compaction and erosion to benefit both pygmy rabbits and sagebrush voles.

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[13] Personal communication. 1997. John Connelly, Upland Bird Research Coordinator, Idaho Department of Fish and Game, P.O. Box 25, Boise, ID 83707-0025.

[14] Personal communication. 1997. Terrell D. Rich, National Avian Ecologist, USDI Bureau of Land Management, 1387 S. Vinnell Way, Boise, ID 83709.

4. (To address issue no. 2) Restore microbiotic crusts in ERUs with potential for redevelopment (that is, areas near propagule sources, and with suitable soil, vegetation, and climatic characteristics [see Belnap 1993, Belnap 1995, Kaltenecker 1997, Kaltenecker and Wicklow-Howard 1994]): the Northern Great Basin, Owyhee Uplands, Upper Snake, and to a lesser extent, the Columbia Plateau (Hann and others, map 3.59).

5. (To address issue no. 3) Restore vegetation around springs, seeps, streams, meadows, and other riparian areas.

6. (To address issue no. 4) Minimize the adverse effects of human disturbance.

Practices that support strategies— The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Identify sites of high ecological integrity to be managed for long-term protection by analyzing current vegetation, precipitation patterns, elevation, temperature (Klemmedson and Smith 1964, Morrow and Stahlman 1984, Stewart and Hull 1949), and the presence of priority species in this group. These practices are most likely to be successful on large areas of Federal land managed by the BLM.

2. (In support of strategy no. 1) Explore options under the CRP (Johnson and Igl 1995), or develop other incentive programs, to encourage restoration of agricultural areas to native cover types. Focus on areas that would increase patch size or links with existing source habitat patches.

3. (In support of strategy no. 1) Avoid further loss of sagebrush cover through burning, plowing, seeding, and other brush "control" methods where sagebrush cover types are below historical levels.

4. (In support of strategy no. 1) Avoid further conversion of source habitats to agricultural lands, or strive to minimize the impacts of further conversions through landscape design, to minimize further fragmentation of shrub-steppe.

5. (In support of strategy no. 2) Use fire prevention and suppression to retard the spread of cheatgrass in areas that are susceptible to cheatgrass invasion but currently are dominated by native grass species. Planting of fire-resistant vegetation through "green stripping" (Pellant 1994) could be explored to evaluate its effectiveness in protecting existing native vegetation.

6. (In support of strategy no. 2) Restore selected areas of cheatgrass monocultures through seeding and other manipulations (Allen 1995, Daubenmire 1970, Evans and Young 1978, Hosten and West 1994, Kennedy 1994, Monsen and McArthur 1995, Ogg 1994, Whisenant 1995, Yoakum 1986b), where such restoration would increase the size of existing shrub-steppe patches or provide links among patches.

7. (In support of strategy no. 2) Plant perennial bunchgrasses or native forbs where these components of the habitat have been lost or reduced (Braun and others 1976, Daubenmire 1970, Evans and Young 1978, Yoakum 1986b). Criteria for enhancement include maintaining or increasing the size of smaller patches, preventing further habitat disassociation, and protecting or increasing the size and integrity of corridors among patches, all in connection with the location of sites with high ecological integrity as identified above.

8. (In support of strategies nos. 2-4) Modify grazing systems or reduce grazing use where native perennial bunchgrasses have been depleted.

9. (In support of strategy no. 4) Encourage the redevelopment of microbiotic crust by reducing or eliminating livestock grazing (Mack and Thompson 1982, St. Clair and others 1993). Explore the use of ground-based and aerial soil inoculation to increase the speed and extent of dispersal of the organisms that create microbiotic crust (Belnap 1993, 1994).

10. (In support of strategy no. 5) Protect existing riparian, spring, and seep sites of high ecological integrity from degradation, restore degraded sites, restore historical water tables in nonfunctioning riparian systems, and eliminate or greatly reduce water diversions. Seeding of native forbs, in particular, may be desirable in certain mesic areas to improve sage grouse brood-rearing habitat.

11. (In support of strategy no. 6) Protect sage grouse leks from human disturbance by designating leks and winter concentration sites as special management areas closed to public access, avoiding the placement of new roads or the improvement of existing roads in important sage grouse areas, and closing existing roads in sensitive areas.

12. (In support of strategy no. 6) Control, reduce, or eliminate pesticide use around agricultural areas adjacent to sage grouse habitat (Blus and others 1989, Post 1951, Ward and others 1942). Avoid use of toxic organophosphorus and carbamate insecticides in sage grouse brood-rearing habitats.

13. (In support of strategy no. 6) Restrict organized recreational events in sage grouse nesting, brood-rearing, and wintering habitats at the appropriate times of year (Call 1979, Roberson 1986).

Group 34—Kit Fox and Black-Throated Sparrow

Results

Species ranges, source habitats, and special habitat features—Group 34 consists of two shrubland species, the kit fox and black-throated sparrow. Both species occur in the most southern shrublands of the basin, and the black-throated sparrow also is found in south-central Washington (fig. 102). The kit fox is a year-round resident of the basin, whereas the black-throated sparrow is a summer resident, migrating to southern portions of its range and Baja California for the winter. The basin represents the northern periphery of the continental distribution for these species, both of which are more commonly associated with desert shrublands of southwestern North America.

Source habitats for both species are big sagebrush and salt desert shrub, and the black-throated sparrow also uses mountain big sagebrush (vol. 3, appendix 1, table 1). Structural stages within these cover types are open- and closed-canopy stages of low-medium shrubs. In southeastern Oregon and northern Nevada, black-throated sparrows are found predominantly in sites with higher shrub cover, greater maximum shrub height, and greater shrub species diversity than used by another shrub-steppe species, the sage sparrow (Wiens and Rotenberry 1981).

Aspecial habitat feature identified for the kit fox is the presence of burrows for den sites (vol. 3, appendix 1, table 2). Kit foxes often use the abandoned dens of other species, and most home ranges include several dens (Egoscue 1962). In addition to reproductive purposes, dens provide resting habitat that modifies the extremes of desert weather and furnishes protection from predators (Golightly and Ohmart 1983). No special habitat features have been identified for the black-throated sparrow.

Broad-scale changes in source habitats—Source habitats have undergone localized declines since historical times. Historically, source habitats were concentrated along the southeastern border of Oregon and southern border of Idaho, extending also into the portions of Nevada and Utah that are included in the basin (fig. 103A). Source habitats for the black-throated sparrow also occurred in south-central Washington. The current distribution of source habitats is roughly the same, but declines in habitat availability have occurred primarily in south-central Washington and south-central Idaho (fig. 103B).

The amount of source habitats was estimated as roughly the same as the historical extent in 65 percent of the watersheds in which these species occur, but 33 percent of the watersheds have exhibited declining trends (fig. 104). The greatest declines occurred in the Upper Snake ERU, where 29 of 55 watersheds had strongly declining trends (fig. 104). The Blue Mountains and Snake Headwaters ERUs also had strongly declining trends, but only three watersheds in each ERU provided source habitats historically, so the magnitude of change may not be significant. Habitat trends were mostly static in the Owyhee Uplands ERU, although 82 of the 256 watersheds with source habitats have declining trends (fig. 104).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats—The principal cause for observed declines in habitat availability is the alteration of sagebrush and salt desert shrub to other cover types, primarily agriculture, urban, juniper/sagebrush, and exotic forbs-annual grass. In the Columbia Plateau ERU, nearly one half of the big sagebrush cover type was converted to croplands (Hann and others 1997). Virtually all broad-scale patches of mountain big sagebrush in the Columbia

Figure 102—Ranges of species in group 34 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 103—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 34 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 104—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 34, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n.*

Plateau within the range of the black-throated sparrow were eliminated (vol. 3, appendix 1, table 4). In the Owyhee Uplands, the dominant cover type transition was from the big sagebrush cover type to croplands and exotic forbs-annual grass (Hann and others 1997). In the Upper Snake ERU, an estimated 41 percent of the sagebrush cover type was converted to croplands (Hann and others 1997).

Condition of special habitat features—No information is available to determine whether changes in availability of burrows for kit fox dens, or in soil conditions needed for burrow excavation, have occurred in the basin. Lack of suitable loose-textured soil for burrow construction may be a natural, limiting factor for kit foxes in southeastern Oregon (Keister and Immell 1994). The soil surface there is predominantly desert pavement, whereas soils near Fallon, Nevada, where higher densities of kit foxes occur than in Oregon, are typically sandy (Keister and Immell 1994). Land uses that increase soil compaction or cause the destabilization of dunes may inhibit burrow establishment.

Other factors affecting the group—The black-throated sparrow seems to show a positive numerical response to moderate livestock grazing (Bock and others 1984, cited in Saab and others 1995).

Because the kit fox is a predator, population health is affected by the availability of small-mammal prey, which in turn is affected by vegetation composition and structure. Land uses that do not directly affect kit foxes may nevertheless affect prey availability. Livestock grazing can impact small-mammal abundance and diversity (Bock and others 1984; Hanley and Page 1982, as cited in Horning 1994).

Kit foxes are vulnerable to poisoned baits placed for destruction of coyotes (Orloff and others 1986). They are also susceptible to hunting and trapping, usually as a nontarget species (DeStefano 1990). Coyote predation is a major cause of kit fox mortality in the San Joaquin Valley of California (White and others 1994), and is a potential limiting factor of kit foxes in the basin.

Population status and trends—Population trend data are not available for the black-throated sparrow within the basin. The only statistically significant population trend for the black-throated sparrow is based on numbers recorded on all BBS routes in North America with black-throated sparrow occurrences between 1966 and 1995. This survey-wide trend indicated a 4-percent annual decline across the range of the species over the 28-yr period ($n = 258$, $P < 0.05$; Sauer and others 1996). Occurrences of the black-throated sparrow on BBS routes within the basin are insufficient to conduct a statistically robust trend analysis (Saab and Rich 1997). Saab and Rich (1997), however, included the black-throated sparrow as one of 15 Neotropical migrants in the basin that are of high concern to management under all future management themes for the basin primarily because of its association with just four cover type-structural stage combinations. We know of no estimates of kit fox numbers within the basin.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 34 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Primary conservation issues for group 34 are as follows:

1. Loss of desert shrub habitat to other land uses and to shrub-control programs.

2. Degradation of desert shrub habitat quality through exotic weed invasions.

3. Effect of adverse land uses on understory vegetation that supports kit fox prey base.

4. Lack of information on the location and status of kit fox dens.

Potential strategies—Strategies for addressing the issues listed above include the following:

1. (To address issue no. 1) Maintain remaining native desert shrublands, especially in the Upper Snake ERU and in all watersheds within the Owyhee Uplands where strong negative trends have occurred.

2. (To address issue no. 2) Restore desired vegetation composition and structural attributes of shrublands that no longer meet source habitat conditions.

3. (To address issue no. 3) Avoid land use practices that potentially affect kit fox prey by reducing the grass-forb component of shrub communities.

4. (To address issue no. 4) Locate and protect active dens of the kit fox.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (To address strategy no. 1) Identify areas of native shrubland vegetation with high ecological integrity, particularly within the Columbia Plateau and Upper Snake ERUs, and actively manage to promote their long-term sustainability.

2. (To address strategy no. 2) Use prescribed burns, shrub planting, and exotic weed control to restore degraded shrublands, but avoid burning areas susceptible to invasion by noxious weeds.

3. (To address strategy no. 3) Adjust or maintain grazing management plans to promote long-term persistence of the grass and forb components of shrub communities.

4. (To address strategy no. 4) Conduct surveys for kit fox burrows, and provide protective measures for active burrows in all relevant planning documents.

Group 35—Loggerhead Shrike

Results

Species ranges and source habitats—Group 35 consists of breeding habitat for the loggerhead shrike. Range of the loggerhead shrike (fig. 105) includes most of the basin except for the mountainous portions of Idaho and Montana and the eastern slope of the Cascade Range. Outside the planning area, the species is widespread as a breeder or year-round resident in the United States and Mexico (Yosef 1996).

This shrike uses various woodland and shrub cover types including juniper, sagebrush, mountain shrub types, salt desert shrubs, and bitterbrush/wheatgrass (vol. 3, appendix 1, table 1). The common structural feature is a good component of woody vegetation in a landscape dominated by more open structure. Nests are typically placed in the taller woody vegetation, whereas the bird forages in open areas.

Broad-scale changes in source habitats—High percentages of contiguous watersheds with source habitats for the loggerhead shrike historically occurred in the Columbia Plateau, Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs (fig. 106A).

Basin-wide, moderate and strong declines (44 percent of watersheds) in source habitats exceeded moderate and strong increases (24 percent), but over 30 percent of watersheds showed no estimated change from the historical condition (fig. 107). Although declining trends in the Columbia Plateau seem to balance against increasing trends (fig. 107), these upward trends were due to large relative increases in vegetation that actually covered <8 percent of the ERU. The biggest losses occurred in the Upper Snake ERU (fig. 107), with over 57 percent of the watersheds showing strong decreases. In contrast, the Upper Klamath ERU was estimated to have nearly 62 percent of its watersheds strongly increasing in source habitats (fig. 107).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Among source habitats basin-wide, big sagebrush types have declined by one-third, the most serious habitat change for shrikes because of the total acreage affected (vol. 3, appendix 1, table 4; Hann and others 1997). Salt desert shrub and mixed-conifer woodlands also have declined substantially, one-third and one-half, respectively. Together, the latter declines affected only a small part of the basin (vol. 3, appendix 1, table 4; Hann and others 1997). The only other significant basin-wide changes have been increases in juniper/sagebrush, juniper woodlands, and mountain mahogany (Hann and others 1997). The latter three types combined, however, cover only a small percentage of the basin.

VIEW

MAP

Figure 105—Ranges of species in group 35 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

The largest changes have been in the Upper Snake and Columbia Plateau ERUs, where big sagebrush has declined by about 50 percent (vol. 3, appendix 1, table 4). In the Upper Clark Fork and Blue Mountains ERUs, mixed-conifer woodlands have declined by over four-fifths and one-half, respectively (Hann and others 1997). Declines in the Upper Clark Fork can be attributed to a near total loss of mixed-conifer woodlands, although this type historically only covered a small portion of the ERU. Increases in the southern Columbia Plateau are due to juniper/sagebrush, which more than doubled, and mountain mahogany, up nearly sixfold; these types together now are estimated to occupy nearly one-tenth of the ERU. Similarly, juniper/sagebrush in the Upper Klamath is estimated to have tripled, making the availability of source habitats there significantly greater (Hann and others 1997). Large increases in source habitats in the Northern Glaciated Mountains are most likely because of relatively large increases in mixed-conifer woodlands, though source habitat is limited in this ERU.

Large-scale loss of sagebrush habitats is due to several factors. The first factor is conversion to agriculture. Agricultural lands have increased significantly in every ERU in the basin (Hann and others 1997). In fact, the largest transition of any terrestrial community from historical to the current period was that of upland shrubland to agriculture (+9.0 percent), and the second largest was that from upland herbland to agriculture (+6.6 percent, Hann and others 1997). This transition, occurring in the fundamental source habitats for this group, explains much of the pattern evident in habitat trends for loggerhead shrike (fig. 106).

A second factor contributing to loss of sagebrush habitat is conversion of shrub-steppe vegetation to exotic forbs and annual grass. Increases in exotic cover types have occurred in all the major shrub-steppe ERUs. Substantial portions of the Owyhee Uplands and Upper Snake ERUs have undergone a conversion from upland shrubland to exotic herbland (Hann and others 1997).

Figure 106—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 35 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 107—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 35, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

The condition of the habitat for group 35 can be described by the composite ecological integrity ratings (Quigley and others 1996) that show most of the habitat to have a "low" rating. Most of the current habitat for this group was classified into Rangeland Clusters 5 (generally corresponding to much of the Owyhee Uplands ERU) and 6 (generally the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs), where the primary risk to ecological integrity is continued losses of herbland and shrubland habitats (Quigley and others 1996). Further, Rangeland Cluster 6 also is vulnerable to overgrazing and to exotic grass and forb invasions (Quigley and others 1996).

Other factors affecting the group— Shrikes prefer tall plants for nest sites, often choosing particularly tall individual big sagebrush plants or, more generally, sites with tall average shrub heights (for example, >1 m [3 ft]) (Leu 1995, Sharp 1992, Yosef 1996). This type of sagebrush community is apt to be a big sagebrush site with deeper soils and a slightly more mesic moisture regime. These sites are precisely where agricultural conversion has most commonly occurred in the past and where future risks of conversion remain the greatest (Hann and others 1997).

Shrikes also prefer to hunt from elevated perches such as fence posts, utility lines, and woody vegetation (Bohall-Wood 1987, Gawlik and Bildstein 1993, Yosef and Grubb 1992), and to restrict their foraging to an area within 10 m of such perches (Chavez-Ramirez and others 1994). Their use of any area may correspond directly to the availability of such perches. Young shrikes prefer to forage on bare ground and sites with little vegetative cover (Leu 1995). Foraging opportunities for young shrikes may be severely reduced because shrub-steppe habitats with natural openings of bare ground have been altered by exotic grasses (for example, cheatgrass) and forbs, creating a continuous vegetative layer (see Leu 1995).

In a study area generally corresponding to the Northern Great Basin ERU, shrike densities were negatively correlated with the cover of grass and positively correlated with woody cover, bare ground, and vegetation height (Rotenberry and Wiens 1980). Shrike densities were negatively correlated with those of Brewer's sparrow and positively correlated with those of rock wrens. Among habitat variables, shrikes were positively associated with the cover of rock and shrubs, and with shrub species diversity (Wiens and Rotenberry 1981).

Loggerhead shrikes evolved in shrub-steppe habitats, where microbiotic crusts were broadly distributed (see Kaltenecker and Wicklow-Howard 1994). Microbiotic, or cryptogamic, crusts consist of lichens, bryophytes, algae, microfungi, cyanobacteria, and bacteria growing on or just below the soil surface in arid and semiarid environments (Kaltenecker and Wicklow-Howard 1994); these crusts developed without large herds of grazing ungulates (St. Clair and Johansen 1993). In addition, these crusts were projected to have been widely distributed throughout the source habitats for this group, particularly in the Northern Great Basin, Owyhee Uplands and Upper Snake ERUs, but also scattered in the Columbia Plateau ERU (Hann and others 1997, map 3.59). Increasing evidence indicates that microbiotic crusts improve soil stability, productivity, and moisture retention; moderate extreme temperatures at the soil surface; and enhance seedling establishment of vascular plants (Belnap and Gardner 1993, Harper and Pendleton 1993, Johansen and others 1993, St. Clair and others 1993), thus contributing to high ecological integrity of shrub-steppe habitats. Idaho BLM has recognized the potential importance of microbiotic crusts by proposing standards for rangeland health that include maintaining these crusts to ensure proper functioning and productivity of native plant communities (USDI Bureau of Land Management 1997). These crusts were widely destroyed by trampling during the excessive livestock grazing of the late 1800s and early 1900s (Daubenmire 1970, MacCracken and others 1983, Mack and Thompson 1982, Poulton 1955). Currently, high-intensity grazing and altered fire regimes modify shrub-steppe plant communities and threaten the maintenance and recovery of microbiotic crusts (Belnap 1995, Kaltenecker 1997, St. Clair and Johansen 1993).

Conversion of native vegetation to exotics is augmented by the propensity of annuals, such as cheatgrass, to spread with wildfire and with improper grazing (Braun and others 1976; Daubenmire 1970; Evans and Young 1978; Quigley and others 1996, p. 123). Some losses of salt desert shrubs likely are due to selective grazing of palatable forbs in this cover type, combined with more xeric conditions that make vegetative resilience low.

Losses of pasture and old fields for wintering habitat in the Southeastern United States have affected shrike populations (Brooks and Temple 1990, Gawlik and Bildstein 1993). Loss of pasture and prairie habitats for breeding in Canada and the Eastern United States are widely cited as causes for population declines in those regions (Yosef 1996). These habitat losses have not been identified as limiting factors for shrike populations in the basin.

Because shrikes often forage and nest along roads (Blumton 1989, Craig 1978, Flickinger 1995, Yosef 1996), vehicular collisions may be an important source of mortality. Automobiles accounted for 29 percent of the observed fall and winter mortality of loggerhead shrikes in Virginia (Blumton 1989). Shrikes also may have been affected by DDT in the past and may suffer sublethal effects of certain insecticides, although the evidence is weak (Anderson and Duzan 1978, Grubb and Yosef 1994, Yosef 1996). Cowbird parasitism of nests does not appear to be a factor affecting productivity of loggerhead shrikes (Yosef 1996).

Population status and trends—Populations of loggerhead shrikes have been declining significantly in the basin, with a trend of -2.7 percent per yr ($n > 14$, $P < 0.05$) over the period 1968-94 (Saab and Rich 1997). The 1966-95 trend for BBS physiographic region 89 (Columbia Plateau) was -2.3 percent per yr ($n = 41$, $P < 0.05$; Sauer and others 1996). Saab and Rich (1997) included the loggerhead shrike as one of 15 Neotropical migrants in the basin that are of high concern to management under all future management themes for the basin.

Patterns of widespread declines throughout its range (Yosef 1996) suggest that either (1) habitat losses throughout its breeding range in various types of breeding habitat are similar, or (2) additional, more extensive factors are impacting the species, such as pesticides or wintering ground problems. These possibilities do not diminish the losses of source habitats in the basin but suggest that widespread population declines may be at least partly the result of a more pervasive cause.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 35 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The results of our habitat trend analysis suggest the following issues are of high-priority for group 35:

1. Permanent and continued loss of large acreage of big sagebrush cover types to agricultural conversion, brush control, reduction of microbiotic crusts, and cheatgrass invasion.

2. Adverse effects of human disturbance.

Potential strategies—The following strategies could be used to reverse broad-scale declines in source habitats:

1. (To address issue no. 1) Identify and conserve large remaining areas (contiguous habitat >1000 ha [2,470 acres]) of shrub-steppe vegetation where ecological integrity is still relatively high (Gray and Rickard 1989, Rickard and Poole 1989, Schuler and others 1993, Smith 1994, Yoakum 1980). Sites resistant to cheatgrass domination because of their moisture regime (>30 cm [12 in]) in the Upper Snake, Owyhee Uplands, Northern Great Basin, and Columbia Plateau ERUs are of highest priority.

2. (To address issue no. 1) Restore microbiotic crusts in ERUs with potential for redevelopment (that is, areas near propagule sources, and with suitable soil, vegetation, and climatic characteristics [see Belnap 1993, Belnap 1995, Kaltenecker 1997, Kaltenecker and Wicklow-Howard 1994]): the Northern Great Basin, Owyhee Uplands, Upper Snake, and, to a lesser extent, the Columbia Plateau (Hann and others, map 3.59).

3. (To address issue no. 1) Retard the spread of cheatgrass in native shrub-steppe vegetation communities.

4. (To address issue no. 2) Minimize adverse effects of human disturbance.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Protect and restore corridors and habitat blocks in areas of shrub-steppe that support large, contiguous areas of high ecological integrity so as to optimize long-term

conservation of shrikes. These practices are most likely to be successful on large tracts of Federal land managed by BLM.

2. (In support of strategy no. 1) Restore existing agricultural lands to native vegetation when possible. Sites where this might be especially useful are areas that were historically shrub-steppe and areas that would augment corridors among existing shrub-steppe patches.

3. (In support of strategy no. 1) Avoid further loss of sagebrush cover through burning, plowing, seeding, and other brush "control" methods where sagebrush cover types are well below historical levels.

4. (In support of strategy no. 1) Minimize the impacts of further agricultural conversions through landscape design. If conversion cannot be avoided, then tracts slated for conversion should be located to minimize further disassociation of shrub-steppe, to avoid reducing the size of isolated patches, and to avoid areas that are currently in large blocks of moderate Composite Ecological Integrity (Quigley and others 1996).

5. (In support of strategy no. 1) Restore native vegetation by appropriate mechanical treatments and seedings of native shrub, grass, and forb species (Allen 1995, Monsen and McArthur 1995, Whisenant 1995, Yoakum 1986b).

6. (In support of strategy no. 2) Encourage the redevelopment of microbiotic crust by reducing or eliminating livestock grazing (Mack and Thompson 1982, St. Clair and others 1993). Explore the use of ground-based and aerial soil inoculation to increase the speed and extent of dispersal of the organisms that create microbiotic crust (Belnap 1993, 1994).

7. (In support of strategy no. 3) Use fire prevention and suppression to retard the spread of cheatgrass in areas that are susceptible to cheatgrass invasion but currently are dominated by native grass species. Explore the effectiveness of planting fire-resistant vegetation through "green stripping" (Pellant 1994) to protect existing vegetation as well as allow degraded sites a chance to recover.

8. (In support of strategy no. 3) Restore selected areas of cheatgrass monocultures through seeding and other manipulations (Allen 1995, Daubenmire 1970, Evans and Young 1978, Hosten and West 1994, Kennedy 1994, Monsen and McArthur 1995, Ogg 1994, Whisenant 1995, Yoakum 1986b), where such restoration would increase the size of existing shrub-steppe patches or provide links among patches.

9. (In support of strategy no. 4) Minimize access to roads and, where possible, locate them on the periphery of areas known to have good shrike populations. Avoid construction of new roads or improvement of old roads in shrike habitat. Plan habitat enhancement projects for sites away from heavily traveled roads.

10. (In support of strategy no. 4) Avoid insecticide spraying during shrike breeding season.

Group 36—Columbian Sharp-Tailed Grouse (Summer)

Results

Species ranges, source habitats, and special habitat features—Columbian sharp-tailed grouse is a year-round resident that is distributed patchily in mesic shrubland and grassland types of the Upper Snake, Snake Headwaters, Central Idaho Mountains, Northern Glaciated Mountains, and Columbia Plateau ERUs (fig. 108). Only trends in summer habitat are evaluated here, because winter cover types (primarily riparian and upland shrub) occur in naturally small patches that could not be analyzed at the broad scale. During the late 1980s, early 1990s, 1996, and 1997, populations were augmented in Montana within the Northern Glaciated Mountains ERU and reintroduced in Oregon within the Blue Mountains ERU.

Summer source habitats of Columbian sharp-tailed grouse include open-canopied big, mountain, and low sagebrush cover types, wheatgrass and fescue bunchgrasses, herbaceous wetlands, upland or mountain shrub cover types of chokecherry-serviceberry-rose, and shrub wetland cover types (vol. 3, appendix 1, table 1) (Marks and Saab Marks 1987a, Meints and others 1992, Saab and Marks 1992). Within these habitats, sharptails only use areas where the annual

Figure 108—Ranges of species in group 36 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

precipitation is at least 30 cm (12 in) (Meints and others 1992), and where the topography is flat to rolling (<30 percent slope) (Saab and Marks 1992). During spring and summer, sagebrush and grasslands provide nesting and brood-rearing habitat, whereas mountain (upland shrub) and riparian shrubs are used for escape cover. Fall and winter habitats are primarily mountain shrub and riparian vegetation. Following those seasonal changes in habitat use, herbaceous vegetation and associated arthropods provide food for sharptails during spring and summer, whereas fruits and buds of woody vegetation, insects, and agricultural crops are consumed by grouse during fall and winter (Giesen and Connelly 1993).

During spring and summer in western Idaho, nesting and brood-rearing microhabitats used by sharptails are characterized by moderate vegetative cover (>60 percent), high structural diversity, and a high diversity of native herbaceous vegetation (Marks and Saab Marks 1987a, Saab and Marks 1992). Native perennials arrowleaf balsamroot and bluebunch wheatgrass were especially important nesting and brood-rearing cover during a drought year when many exotic annuals dried up and provided no cover (Saab and Marks 1992). Additionally, selected microhabitats in western Idaho were least modified by livestock grazing and near escape cover of mountain shrubs and riparian vegetation. Grouse broods in eastern Idaho preferred CRP lands over native shrublands or agricultural fields during summer (Sirotnak and others 1991). Seedings on CRP lands provide nesting cover and are often good sources of food if the seedings include alfalfa, *Tragopogon* species, and *Lactuca* species. Height of nest-brood cover was identified as a critical microhabitat feature and averaged 25 ± 16 cm (10 ± 6.3 in) in eastern Idaho (Meints and others 1992).

When native shrubland is used for nesting in Idaho, most nests are placed beneath a shrub (Marks and Saab Marks 1987a, Meints 1991). Thus, shrubs are a special habitat feature for this species (vol. 3, appendix 1, table 2). Shrub density at nests in eastern Idaho averaged 11,000 shrubs per ha (2.5 acres) compared to 5,000 shrubs per ha (2.5 acres) at independent, randomly located sites (Meints 1991). In a native grassland of northwestern Montana, preliminary data indicated that nests were placed beneath wheatgrass and fescue bunchgrasses.[15]

Spring and summer movements are typically within 1.0 to 2.5 km (0.63 to 1.6 mi) of dancing grounds (lek sites) (Saab and Marks 1992). Summer home ranges averaged 187 ± 114 ha (462 ± 282 acres) in western Idaho and 90 percent of all locations were within 1.2 km (0.75 mi) of a dancing ground (Saab and Marks 1992). Nests have been located <100 m (328 ft) (Marks and Saab Marks 1987a) to >3 km (1.9 mi) (Meints 1991) from lek sites, with most females nesting <1.6 km (1.0 mi) from the lek where they were trapped (Marks and Saab Marks 1987a, Meints 1991, Oedekoven 1985).

Winter habitat requirements seem more restricted than in other seasons (Giesen and Connelly 1993). Columbian sharptails in western Idaho wintered almost exclusively in mountain shrub or riparian cover types, the only cover types that provided food and escape cover regardless of snow depth (Marks and Saab Marks 1988). Fruits of Douglas hawthorn and buds of serviceberry and chokecherry were the main winter foods. Winter grouse locations in eastern Idaho averaged 90 m (295 ft) to riparian cover (Meints 1991). Movements of sharptails between breeding and wintering areas varied from 2.6 km (1.6 mi) in western Idaho (Marks and Saab Marks 1987a) to 20 km (12.5 mi) in southeastern Idaho (Meints 1991). Columbian sharptails apparently move farther to wintering habitats in regions lacking a broad distribution of winter food resources (Giesen and Connelly 1993).

Broad-scale change in source habitats—Historically, source habitats for Columbian sharp-tailed grouse were broadly distributed in eastern Washington and eastern Oregon, except in the Northern and Southern Cascades ERUs (fig. 109A). Historical source habitats were also in western portions of the Central Idaho Mountains, in the southern Owyhee Uplands, southern Snake Headwaters, and eastern portions of the Upper Snake and Snake Headwaters ERUs (fig. 109A).

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[15] Personal communication. 1997. Tim Thier, wildlife biologist, Montana Department of Fish, Wildlife, and Parks, P.O. Box 507, Trego, MT59934.

Figure 109—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 36 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 110—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 36, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

The current distribution of source habitats is limited and highly disjunct compared to historical patterns (fig. 109B). The western half of the Snake Headwaters and eastern Upper Snake ERUs currently provide the most contiguous habitat within the current range (figs. 108, 109B). In contrast, other remaining populations are restricted to small and isolated portions of the Central Idaho Mountains, Northern Glaciated Mountains, Columbia Plateau, Blue Mountains, and Lower Clark Fork ERUs (fig. 108). Breeding populations reintroduced to northeastern Oregon in the early 1990s occupy small areas near Enterprise in the Blue Mountains, and augmentations were conducted near Eureka, Montana, in the Northern Glaciated Mountains during the late 1980s and early 1990s (fig. 108).

Strong declines in source habitats were projected in over 60 percent of watersheds throughout the basin, whereas increases in habitat occurred in only 6 percent of watersheds (figs. 109C and 110). Eight of 11 ERUs with historical source habitats had strongly decreasing trends. The Northern Glaciated Mountains experienced the greatest declines, where 94 percent of the watersheds had strong decreases in source habitats (fig. 110).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— The open-canopy low-medium structural stage of mountain big sagebrush and big sagebrush experienced some of the greatest absolute declines on an ERU basis. The combined absolute decline for the open-canopy low-medium structural stage of these two sagebrush types declined in the Upper Snake (-40 percent), Owyhee Uplands (-20 percent), Columbia Plateau (-13 percent), Snake Headwaters (-7 percent), and Northern Great Basin (-2 percent) (vol. 3, appendix 1, table 4). In these open-canopied cover types, in the absence of fire, shrubs and trees eventually invade much of the area that was occupied by grasses and forbs.

In addition, large-scale losses of sagebrush habitats were attributed primarily to agricultural development. Agricultural lands have increased substantially in all ERUs within the basin (vol. 3, appendix 1, table 4). The largest conversions of terrestrial communities from historical to current levels were those of upland shrubland to agriculture and from upland herbland to agriculture (Hann and others 1997). These conversions were widespread within the historical range of sharptails and, in part, explained the broad-scale changes in their source habitats (fig. 109C).

Mountain shrub (chokecherry-serviceberry-rose) and shrub wetland terrestrial community groups are key components of sharp-tailed grouse habitat during late summer, fall, and winter. These cover types naturally occur in small patches and were difficult to map at the scale of this analysis. Therefore, accurate information was not available on habitat trends in mountain shrub and shrub wetlands.

Condition of special habitat features—Mesic sagebrush lands, mountain shrub (chokecherry-serviceberry-rose) communities, and riparian vegetation are special habitat features used by sharptails. Loss and degradation of these features, as a result of livestock grazing and agricultural conversions, were identified as factors contributing to the widespread population declines in Columbian sharp-tailed grouse within the basin (Marks and Saab Marks 1987a, 1988; Meints and others 1992; Saab and Marks 1992; Tirhi 1995). Additionally, losses of native perennial grasses and forb understories of the mesic sagebrush zones, because of livestock grazing and exotic grass invasions, are microhabitat features that could not be examined by the broad-scale analysis.

Other factors affecting the group— Livestock grazing is the dominant land use in occupied Columbian sharp-tailed grouse habitat. Habitat degradation by high-intensity livestock grazing (also by native ungulates) results in reductions or losses of native perennial grasses and forbs, necessary for grouse nesting and brood-rearing cover. Excessive grazing can alter the native vegetation by allowing invasions of exotic plants, including cheatgrass, medusahead, and mustards. Additionally, deciduous trees and shrubs, which are critical for sharptail escape cover and for winter food (Marks and Saab Marks 1987a, 1987b, 1988; Meints 1991; Tirhi 1995), may be reduced by intensive cattle browsing during late summer (Kovalchik and Elmore 1992).

Loss of lands managed under the CRP is potentially another factor influencing Columbian sharptails. In eastern Idaho, CRP lands provide important feeding, nesting, brood-rearing, and relatively mild winter habitat (Ulliman 1995). In Washington, however, CRP lands receive little use by sharptails (Schroeder 1994).

Although some females nest in CRP and other idle croplands, the most successful nests in Washington were built in native habitats of sagebrush or forbs mixed with grass (Schroeder 1994).

Herbicides and pesticides have been identified as potential threats to sharptails (Giesen and Connelly 1993). Herbicide spraying has negative effects on the species because of losses in herbaceous and woody vegetation that is used for nesting, brood-rearing, and wintering habitat. Pesticide spraying may have negative impacts by directly killing young or by reducing or eliminating insects used for food.

Fire can either enhance or degrade sharp-tail habitat, depending on the cover type, timing, frequency, intensity, size of burn (Giesen and Connelly 1993), soils, and precipitation. Many species of deciduous shrubs (for example chokecherry and rose) resprout after fire. In contrast, most sagebrush species do not resprout and may be eliminated by fires. Exotic vegetation can invade following fire, depending on the soils and precipitation.

Human disturbances related to the expansion of residential developments, increases in road densities, and associated recreational activities likely will exacerbate losses of suitable habitat within the historical range of Columbian sharp-tailed grouse (Giesen and Connelly 1993, Tirhi 1995).

Population status and trends— Sharptails currently occupy <5 percent of their historical range in the basin. The BBS data summarized for western North America indicate that population trends declined by an average of -7.7 percent annually between 1966 and 1995 ($n = 39$, $P < 0.05$; Sauer and others 1996).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 36 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues— The following issues were identified by our analysis of source habitat trends and from the findings of other studies on Columbian sharp-tailed grouse:

1. Fragmentation and loss of mesic shrubsteppe and steppe habitats by conversion to agriculture.
2. Alteration of shrub-steppe and steppe habitats by invasions of exotic forbs and grasses.
3. Degradation and loss of cover types within the shrub-steppe, steppe, mountain shrub, herbaceous wetlands, and shrub wetland community groups by excessive livestock grazing.
4. Loss of sagebrush cover because of burning, herbicide spraying, and other brush control methods.
5. Human disturbance of leks and wintering populations because of increased roading and human presence.
6. Increased application of pesticides in and near agricultural areas.
7. Loss of CRP lands by conversion back to active croplands.
8. Isolated and disjunct populations vulnerable to extinction by stochastic events (that is, demographic, environmental, or genetic stochasticity).

Potential strategies— The issues identified above suggest the following broad-scale strategies for the long-term persistence of Columbian sharp-tailed grouse:

1. (To address issue no. 1) Basin-wide, identify areas of mesic shrub-steppe vegetation with high ecological integrity and manage to promote their long-term sustainability.
2. (To address issue no. 2) Restore shrub-steppe and steppe habitats that have been altered by medusahead grass, cheatgrass, and exotic mustards, and focus on areas that would increase patch size or links with existing source habitat patches.
3. (To address issue no. 2) Protect shrub-steppe habitats against wildfire in areas vulnerable to invasion by exotic vegetation.
4. (To address issue no. 3) Reduce habitat degradation by livestock grazing in cover types within shrub-steppe, mountain shrub, riparian, grassland, and herbaceous wetland terrestrial community

groups that are currently occupied by sharptails, with a high potential of being recolonized by sharptails, or that have been identified for reintroductions.

5. (To address issue no. 4) Maintain sagebrush and mountain shrub cover, and increase these shrublands in areas where substantial losses have occurred because of brush control, especially in locations currently occupied by sharptails, with a high potential of being recolonized by sharptails, or in locations that have been identified for reintroductions.

6. (To address issue no. 7) Maintain CRP lands that are currently occupied by sharptails, lands that have a potential of being used by sharptails, or are near locations that have been identified for reintroductions.

7. (To address issue no. 8) Expand the current range of Columbian sharptails within their historical habitats.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategies no. 1 and no. 4) Establish special management areas for maintaining high-integrity shrublands where livestock grazing would be eliminated or restricted. Manage areas of at least 200 ha (494 acres) for summer nesting and brood-rearing habitat; suitable wintering habitats should be within 2.6 to 20 km (1.6 to 12.5 miles) of summer areas.

2. (In support of strategy no. 2) Restore degraded shrub-steppe, mountain shrub, and shrub wetland habitats by plantings of native shrub and herbaceous vegetation, and by prescribed fire (in areas not vulnerable to invasion by exotic plants).

3. (In support of strategy no. 2) Develop methods through ongoing or new research to restore shrub-steppe habitats altered by medusahead, cheatgrass, and exotic mustards.

4. (In support of strategy no. 3) Plant native vegetation that is naturally resistant to wildfire, and actively suppress wildfires in areas that are susceptible to postfire invasions of exotic vegetation.

5. (In support of strategy no. 4) Remove or explicitly control the timing and intensity of grazing to improve the ecological condition of degraded rangelands in locations occupied by sharptails, with a high potential of being recolonized by sharptails, or that have been identified for reintroductions.

6. (In support of strategy no. 5) Eliminate brush control for sagebrush and mountain shrubs in those areas currently occupied or with a high potential of being recolonized by sharptails, including the Snake Headwaters, Upper Snake, Central Idaho Mountains, Blue Mountains, and Columbia Plateau ERUs.

7. (In support of strategy no. 6) Promote the continuation and development of the CRP program, whereby private landowners are encouraged to reduce soil erosion and establish perennial cover, especially in the Upper Snake and Snake Headwaters ERUs.

8. (In support of strategy no. 7) Acquire lands that are currently occupied by sharptails but are not specifically managed for the grouse.

9. (In support of strategy no. 7) Reintroduce and augment sharp-tailed grouse populations after habitat enhancement.

Group 37—Grasshopper Sparrow, Clay-Colored Sparrow, and Idaho Ground Squirrel

Results

Species ranges, source habitats, and special habitat features—Group 37 consists of breeding habitat for the grasshopper sparrow and clay-colored sparrow, and year-round habitat for the Idaho ground squirrel. The breeding range of the grasshopper sparrow (fig. 111) includes most of the basin except for the Northern Great Basin, Upper Klamath, Southern Cascades, and Northern Cascades ERUs. The breeding range of the clay-colored sparrow (fig. 111), on the other hand, is restricted to the Northern Glaciated Mountains, Upper Clark Fork, and Snake Headwaters ERUs. Within the basin, ranges of these two sparrow

Figure 111—Ranges of species in group 37 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

species overlap only in Montana and Wyoming. Two subspecies of the Idaho ground squirrel occur in the basin, the northern Idaho ground squirrel (*Spermophilus brunneus brunneus*) and the southern Idaho ground squrriel (*Spermophilus brunneus endemicus*). Both of these subspecies are found only in western Idaho (fig. 111), and of the two subspecies, the northern is the more rare (Yensen 1991). The ecology and management concerns of the northern subspieces are the basis for most of the subsequent discussion of northern Idaho ground squirrel in this document.

Fescue-bunchgrass is the one cover type shared by all three species (vol. 3, appendix 1, table 1). Open-canopied mountain big sagebrush is source habitat used by the grasshopper sparrow and Idaho ground squirrel. Additionally, the open-canopied big sagebrush is source habitat for the ground squirrel. The clay-colored sparrow also has source habitats in chokecherry-serviceberry-rose and native forb cover types.

Neither sparrow has a clear preference for any special habitat features, but the clay-colored may be attracted to sites that have dense shrubs in a matrix of more open grasslandlike vegetation (Janes 1983). Idaho ground squirrels inhabit meadows, usually with shallow soils and small intrusions of deeper soil for nest burrows (USDA Forest Service and USDI Fish and Wildlife Service 1996).

Broad-scale changes in source habitats—Historically, source habitats for the sparrows in this group were widespread, but generally occupied <25 percent of most watersheds (fig. 112A). High percentages of contiguous watersheds with source habitats occurred in the northeast end and along the eastern edge of the Columbia Plateau ERU, and in the northern end of the Blue Mountains ERU. In the rest of the basin, however, large, contiguous source habitats of high ecological integrity were small and scattered. Nonetheless, the sparrows likely occupied relatively small patches of suitable habitat throughout their historical ranges.

Habitat loss has been obvious as both contiguous areas of source habitats and watersheds with relatively less habitat have greatly diminished (fig. 112B). The Columbia Plateau and Blue Mountains ERUs had strongly declining trends in source habitats for grasshopper sparrows. Similarly, the small but important source areas for the clay-colored sparrow in the northeastern portion of the basin and for Idaho ground squirrel in the center of the basin have decreased. Although much of the basin never had a high percentage of watersheds with source habitats, large acreages have been converted to landscapes with no habitat (fig. 112B).

Over 60 percent of the watersheds had strongly declining trends in source habitats basin-wide (fig. 113). Within the two ERUs that constitute the heart of the habitat for grasshopper sparrow, the Columbia Plateau and Blue Mountains, changes were markedly negative (fig. 113). Similarly, where the two sparrows occur together in the Northern Glaciated Mountains and Upper Clark Fork ERUs, trends were clearly declining (fig. 113). Source habitats for the ground squirrel were projected to have undergone the second greatest decline among 91 species evaluated (vol. 1, table 7). All three species in this group were in the habitat trend category with the greatest decrease in source habitats (vol. 1, table 7).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— The principal vegetation change corresponding to the negative trend in source habitats was in the fescue-bunchgrass cover type, which declined two-thirds from historical levels basin-wide (Hann and others 1997). The largest declines within the species ranges occurred in the Columbia Plateau and Northern Glaciated Mountains (>80 percent); Blue Mountains (75 percent); and Upper Clark Fork and Central Idaho Mountains (60 percent; vol. 3, appendix 1, table 4). The decrease in fescue-bunchgrass amounted to over 5 percent of all changes in the basin, an amount exceeded only by the decrease in big sagebrush (Hann and others 1997).

The open-canopy low-medium structural stage of mountain big sagebrush and big sagebrush experienced some of the greatest absolute declines on an ERU basis. The combined absolute decline for the open-canopy low-medium structural stage of these two sagebrush types declined in the Upper Snake (-40 percent), Owyhee Uplands (-20 percent), Columbia Plateau (-13 percent), Snake Headwaters (-7 percent), and Northern Great Basin (-2 percent) (vol. 3, appendix 1, table 4). In these open-canopied cover types, in

Figure 112—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 37 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 113—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 37, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

the absence of fire, shrubs and trees eventually invade much of the area that was occupied by grasses and forbs.

Basin-wide declines in mountain big sagebrush were substantial (Hann and others 1997) and resulted in critical losses of source habitats for the grasshopper sparrow and Idaho ground squirrel. Vegetation changes affecting Idaho ground squirrels may be difficult to discern for small meadows of sagebrush or native herbaceous cover types within ponderosa pine-dominated forests. This mosaic of habitats is not always detectable at the 1-km^2 (0.4-mi^2) pixel size that was used for evaluating habitat trends in this effort.

Increases in the Central Idaho Mountains were due to the large relative increase in native forbs, although this cover type occupies only a small fraction of the ERU (vol. 3, appendix 1, table 4).

Conversion of upland shrubland to agriculture affected 9 percent of the basin (Hann and others 1997). Major conversions in the Columbia Plateau, Owyhee Uplands, and Blue Mountains greatly affected this group. The basin-wide loss of fescue-bunchgrass and wheatgrass-bunchgrass cover types was largely because of conversion to agriculture. Transition of upland herbland to agriculture affected 7 percent of the basin, a conversion rate second only to that for upland shrubland (Hann and others 1997). Conversion in the Columbia Plateau and Blue Mountains was particularly high—up to 25 percent of upland shrublands. Basin-wide declines in mountain big sagebrush and native forbs also were attributed in part to agricultural conversion.

Habitat condition for group 37 can be described by the composite ecological integrity ratings (Quigley and others 1996) that show most of the habitat to have a “low” rating. Fescues and bunchgrasses, critical habitat components for this group, were irreversibly modified by high-intensity grazing in the late 1800s to early 1900s (USDA Forest Service 1996). Most of the current habitat for this group was classified into Rangeland Clusters 5 (generally corresponding to much of the Owyhee Uplands ERU) and 6 (generally the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs), where the primary risk to ecological integrity is continued losses of herbland and shrubland habitats (Quigley and others 1996). Further, Rangeland Cluster 6 is vulnerable to overgrazing and exotic grass and forb invasions (Quigley and others 1996).

Other factors affecting the group— Early season mowing of hayfields causes major nest failures in grassland-nesting species (Knapton 1994, Smith 1963). Where hayfields and similar agricultural lands have replaced native source habitats or are now located adjacent to such habitats, those sites likely serve as significant population sinks, particularly for grasshopper sparrows.

Grasshopper sparrow populations temporarily decline immediately after grassland fires (Bock and Bock 1992). Birds likely avoid recently burned areas because of the lack of grass cover, and they are expected to return to burned sites after grasses are restored. This sparrow also avoids areas where shrub cover exceeds 35 percent (Bock and Bock 1992, Smith 1963). Thus, fire plays a beneficial role in habitat management for this species.

Although clay-colored sparrows are sympatric with grasshopper sparrows in some regions, clay-coloreds prefer the other end of the grass-shrub gradient, becoming more common with increases in shrub cover and patches of shrubs (Knapton 1979, 1994; Owens and Myers 1973). Thus, clay-colored sparrows also will respond negatively, in the short term, to burning and may require more time to return to prefire population densities while shrubs become reestablished after fire (Pylypec 1991).

Species in this group evolved in shrub-steppe habitats, where microbiotic crusts were broadly distributed (see Kaltenecker and Wicklow-Howard 1994). Microbiotic, or cryptogamic, crusts consist of lichens, bryophytes, algae, microfungi, cyanobacteria, and bacteria growing on or just below the soil surface in arid and semiarid environments (Kaltenecker and Wicklow-Howard 1994); these crusts developed in the absence of large herds of grazing ungulates (St. Clair and Johansen 1993). In addition, these crusts are projected to have been widely distributed throughout the source habitats for this group, particularly in the Northern Great Basin, Owyhee Uplands, and Upper Snake ERUs but also scattered in the Columbia Plateau ERU (Hann and others 1997, map 3.59). Increasing evidence indicates that microbiotic crusts improve soil stability, productivity, and moisture retention; moderate extreme temperatures at the soil surface; and enhance seedling establishment of vascular plants (Belnap and Gardner 1993, Harper and Pendleton 1993, Johansen and others 1993, St. Clair and others 1993), thereby contributing to high ecological integrity of shrub-steppe habitats. Idaho BLM has recognized the potential importance of microbiotic crusts by proposing standards for rangeland health that include the maintenance of these crusts to ensure proper functioning and productivity of native plant communities (USDI Bureau of Land Management 1997). These crusts were widely destroyed by trampling during the excessive livestock grazing of the late 1800s and early 1900s (Daubenmire 1970, MacCracken and others 1983, Mack and Thompson 1982, Poulton 1955). Currently, high-intensity grazing and altered fire regimes modify shrub-steppe plant communities and threaten the maintenance and recovery of microbiotic crusts (Belnap 1995, Kaltenecker 1997, St. Clair and Johansen 1993).

Grazing may reduce or completely exclude grasshopper sparrow populations (Bock and Webb 1984, Saab and others 1995) because livestock remove grass, the main feature of a given site that attracts this species (Janes 1983).

The grasshopper sparrow may be area sensitive and more likely to occupy large tracts of habitat than small fragments (Samson 1980). Minimum area requirements in Maine are about 100 ha (247 acres) (Vickery and others 1994) and in Illinois are about 30 ha (74 acres) (Herkert 1994).

Although brown-headed cowbirds parasitize nests of grasshopper sparrows, the impact is believed to be generally low because of the cryptic nature of the nests of sparrows (Vickery 1996). Cowbirds also parasitize nests of clay-colored sparrows, which may accept or reject the eggs. The overall impact on this species is not known but may be lower than in many species, as cowbird parasitism accounts for only 22 percent of egg loss (Knapton 1994).

Idaho ground squirrels are threatened by sport shooting or “plinking” (Moroz 1995). Several sites occupied by the ground squirrels are regularly visited by shooters for this purpose. When populations are small, this activity could have a critical, detrimental impact. Increases in human occupation in the basin likely have caused an increase in human disturbance.

Idaho ground squirrels may experience competition with Columbian ground squirrels (Moroz 1995, USDAForest Service and USDI Fish and Wildlife Service 1996). Both species use similar habitats, but the Idaho ground squirrel tends to inhabit more xeric areas that cannot support Columbian ground squirrels. Columbian ground squirrels are larger and require larger areas with deeper soils. Although the Idaho ground squirrel can use the same habitats for hibernation, it may be competitively forced into the drier areas with more shallow soils. The shallow soil areas are more prone to fluctuating water tables and freezing during harsh winters, causing overwinter mortality in Idaho ground squirrels (Moroz 1995).

Low population numbers of the Idaho ground squirrel, probably no more than 600 to 800 individuals, make the species vulnerable to (1) genetic drift, inbreeding, and attendant loss of viability; (2) catastrophic invasions of predators, parasites, or diseases; and (3) extirpation because of natural population fluctuations (Moroz 1995). Populations are small and often isolated by several kilometers (Yensen 1991).

Poisoning through the use of rodenticides may negatively affect populations. Predation by domestic cats also is a concern (USDA Forest Service and USDI Fish and Wildlife Service 1996).

Forest encroachment into meadows due to fire suppression and natural succession may be a threat to Idaho ground squirrels (Moroz 1995). Encroachment on meadows, replacement of open forest stands with dense stands of trees, and human developments may have eliminated or reduced dispersal corridors (USDA Forest Service and USDI Fish and Wildlife Service 1996).

Population status and trends—Sample sizes for the clay-colored sparrow in the basin were insufficient to determine population trend (Saab and Rich 1997). The 1966-95 trend for BBS physiographic region 64 (Central Rocky Mountains) is +11.4 percent per yr ($n = 17$, $P < 0.05$), but the sample size is small (Sauer and others 1996).

Saab and Rich (1997) reported a stable population trend for the grasshopper sparrow in the basin but also stated that the species is not well monitored by the BBS technique and advised specialized monitoring. The trend for Washington is +7.5 percent per yr ($n = 18$, $P < 0.1$) and for physiographic region 89 (Columbia Plateau) is stable ($n = 24$, $P > 0.1$; Sauer and others 1996). Again, sample sizes are too small to provide definitive results.

There are 36 known historical and current population sites of northern Idaho ground squirrels (U.S. Government 2000b). Twenty-seven of these sites are currently occupied by northern Idaho ground squirrels, and the total population is estimated at less than 1,000 individuals. The northern subspecies was listed as threatened by the U.S. Fish and Wildlife Service in April, 2000 (U.S. Government 2000b).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 37 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—The results of our habitat trend analysis suggest the following issues are of high priority for group 37:

1. Continued loss of large acreage of fescue-bunchgrass and mountain big sagebrush cover types.
2. Loss of microbiotic crusts.
3. Undesired changes in shrub:grass ratios because of changes in historical fire regimes.
4. Direct mortality of ground nesting birds because of agricultural practices.
5. The disjunct nature of remaining habitat for grasshopper sparrow populations.
6. Loss of meadow habitat because of forest encroachment and human developments.
7. Loss of dispersal corridors for Idaho ground squirrel from replacement of open forest stands with dense stands and human developments.
8. Vulnerability to extinction of small, isolated populations of ground squirrels because of poisoning, shooting, predation, disease, or natural fluctuations.
9. Displacement from habitat because of interspecific competition.

Potential strategies—The following strategies could be used to reverse broad-scale declines in source habitats:

1. (To address issues no. 1 and no. 5) Identify and conserve remaining large areas of mountain big sagebrush and fescue-bunchgrass vegetation where ecological integrity is still relatively high (Bock and others 1993, Gray and Rickard 1989, Rickard and Poole 1989, Schuler and others 1993, Smith 1994, Yoakum 1980). The remaining blocks of habitat in the eastern Blue Mountains and southern Central Idaho Mountains ERUs (fig. 112) may serve as focal points for protection. For the clay-colored sparrow, only the small watersheds in the Upper Clark Fork and Northern Glaciated Mountains ERUs (fig. 112) can be expected to contribute to source habitats.

2 (To address issue no. 1) Restore native perennial bunchgrasses and avoid further depletion because of improper grazing (Braun and others 1976, Daubenmire 1970, Evans and Young 1978). Priority areas for the grasshopper sparrow are the eastern Blue Mountains and southern Central Idaho Mountains ERUs (fig. 113). For the clay-colored sparrow, priority areas are the Upper Clark Fork and Northern Glaciated Mountains ERUs.

3. (To address issue no. 2) Restore microbiotic crusts in ERUs with potential for redevelopment (that is, areas near propagule sources, and with suitable soil, vegetation, and climatic characteristics [see Belnap 1993, 1995; Kaltenecker 1997; Kaltenecker and Wicklow-Howard 1994]): the Northern Great basin, Owyhee Uplands, and Upper Snake ERUs and, to a lesser extent, the Columbia Plateau ERU (Hann and others 1997, map 3.59).

4. (To address issue no. 3) Use fire to obtain desired shrub:grass ratios. Enhance development of shrub communities, particularly mountain sagebrush and chokecherry-serviceberry-rose, in the Upper Clark Fork and Northern Glaciated Mountains ERUs. Maintain dense grassland cover in the eastern Blue Mountains and southern Central Idaho Mountains ERUs.

5. (To address issue no. 4) Minimize direct mortality of ground nesting birds in agricultural areas.

6. (To address issue no. 5) Maintain and restore the largest areas of native grassland habitats.

7. (To address issues no. 6 and no. 7) Maintain meadows and corridors currently used by Idaho ground squirrels. Restore potentially suitable meadows within the range of the species. Stop or reverse forest encroachment into meadows.

8. (To address issue no. 8) Prevent direct human-caused mortality of Idaho ground squirrels.

9. (To address issue no. 8) Restore populations of the Idaho ground squirrel.

10. (To address issue no. 9) Explore the removal of Columbian ground squirrels from adjacent habitats.

Practices that support strategies— The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Use landscape planning to avoid further reductions in the size of large blocks of mountain big sagebrush and fescue-bunchgrass within each watershed, particularly in the Blue Mountains and Central Idaho Mountains ERUs, where sizable blocks of source habitats are available.

2. (In support of strategy no. 1) Explore options under the CRP (Johnson and Igl 1995), or develop other incentive programs, to encourage restoration of agricultural areas to native cover types. Focus on areas that would increase patch size or links with existing source habitat patches.

3. (In support of strategies no. 2 and no. 3) Modify grazing systems or reduce grazing use where native perennial bunchgrasses have been depleted. The elimination of grazing may encourage the redevelopment of microbiotic crust (Mack and Thompson 1982, St. Clair and others 1993).

4. (In support of strategy no. 3) Explore the use of ground-based and aerial soil inoculation to increase the speed and extent of dispersal of the organisms that create microbiotic crust (Belnap 1993, 1994).

5. (In support of strategies no. 1, 3, and 4) Develop a prescribed burning program designed to increase native grass cover and reduce shrub cover (Vickery 1996) on limited acreages and in concert with strategy no. 1. For example, summer burns, which correspond to the period of increased natural lightning strikes, may be more beneficial for maintaining source habitats than burns at other times of the year (Shriver and others 1996); extensive, hot burns in shrub-steppe habitats are probably less beneficial than cooler, more controlled burns that leave some shrub cover (Bock and Bock 1987). In clay-colored sparrow habitats, fire control will allow development of the shrub component that this species prefers (Knapton 1994).

6. (In support of strategy no. 5) Where possible, avoid early season mowing of hayfields and other agricultural lands (Rodenhouse and others 1995, Vickery 1996). Defer mowing on publicly owned lands and develop incentives for private land owners (Vickery 1996). Avoid creating hayfields and similar crop fields adjacent to, or in the general area of, natural nesting habitats.

7. (To address strategy no. 6) A breeding site of 100 to 200 pairs in an area of source habitats 800 to 1400 ha (1,330 to 2,330 acres) is recommended to sustain a population of grasshopper sparrows (Delany and others 1995). Avoid fragmenting existing source habitats below this size and work to protect and restore other sites to at least this standard.

8. (In support of strategy no. 7) Maintain meadow and meadow-corridor habitats within ponderosa pine cover types for Idaho ground squirrels. Retard conifer invasion of meadows by thinning young trees from stands, prescribed burning, and controlled grazing (Moroz 1995). Replant with native grasses.

9. (In support of strategy no. 7) Develop livestock grazing practices that retain grass seed-heads available to ground squirrels (Moroz 1995).

10. (In support of strategy no. 7) Create new meadow habitats at suitable locations with various deep and shallow soils. Expand existing meadow habitats through practices in issue no. 6, with attention to corridors that could provide dispersal habitats for existing populations of Idaho ground squirrels.

11. (In support of strategy no. 8) Avoid use of rodenticides in occupied habitats of Idaho ground squirrels.

12. (In support of strategy no. 8) Control recreational uses such as off-road vehicles, roadside turnouts, and camping within meadow complexes occupied by Idaho ground squirrels. Encourage the public to avoid shooting, poisoning, or trapping the squirrel. Close important ground squirrel areas to discharge of firearms. Inform the public about this endemic Idaho species.

13. (In support of strategy no. 9) Reintroduce Idaho ground squirrels into suitable habitats.

14. (In support of strategy no. 10) Determine if removal or reduction of Columbian ground squirrel populations will provide more habitat for the Idaho ground squirrel.

Group 38—Black Rosy Finch and Gray-Crowned Rosy Finch

Results

Species ranges, source habitats, and special habitat features—Group 38 consists of the black rosy finch and the gray-crowned rosy finch, summer residents of alpine communities. The gray-crowned rosy finch occurs throughout the basin, whereas the black rosy finch is restricted to the eastern part of the basin (fig. 114). This analysis is focused on summer habitat only. Both finches winter in open habitats at lower elevations and occasionally are observed in towns.

Source habitats for group 38 are alpine tundra, barren rocky areas, and cliffs (vol. 3, appendix 1, table 1). Rosy finches nest primarily on cliffs in rocky crevices (French 1959), which are a special habitat feature used by these species. Both finches feed on seeds and insects (French 1959).

Broad-scale changes in source habitats—Source habitats coincide with the distribution of alpine tundra, both historically and currently (figs. 115A, and 115B). The greatest amount of source habitat occurs in the Rocky Mountains in Montana (fig. 115B). No change in amount of source habitats was projected for this group (figs. 115C and 116).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats—Neutral trends in source habitats were attributed to insignificant changes in the amount of alpine tundra since historical times (Hann and others 1997). These projections were limited by the coarse resolution of the data. Hann and others (1997) suspected that finer resolution data would indicate long-term degradation of soils and changes in the composition of vegetation resulting from excessive domestic sheep grazing within alpine environments. Thus, the projected neutral trend should be interpreted as describing habitat extent but not habitat quality.

Condition of special habitat features—Changes in the abundance of rocks and cliffs have not been documented but likely are insignificant.

Other factors affecting species within the group—Potential overgrazing by sheep and human recreational activities in alpine tundra could have a negative effect on habitat suitability for these species (ICBEMP 1996g, Lehmkuhl and others 1997). Rock climbing could cause local disturbances of nest sites.

Population status and trends—Trend data for populations of the black rosy finch or the gray-crowned rosy finch are not available. Low population numbers and limited habitat contribute to conservation concerns for both species (ICBEMP 1996g, Marshall and others 1996).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 38 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—Results of our habitat trend analysis do not lead to any management issues at the broad-scale. Expert opinions (ICBEMP 1996g, Lehmkuhl and others 1997), however, suggest the following issues may be important for the long-term viability of rosy finches:

Figure 114—Ranges of species in group 38 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Figure 115—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 38 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 116—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 38, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Figure 117—Ranges of species in group 39 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

1. Declines in quality of alpine vegetation in the basin because of past and current sheep grazing and recreational activities.

2. Disturbance to cliff and rock nest sites.

Potential strategies—The primary strategy for addressing issue no. 1 is to minimize negative effects of grazing and recreational activities in alpine tundra habitat. Because of lack of information on the degree of impacts to rock and cliff nest sites, no strategies are proposed for issue no. 2.

Practices that support the strategy— The following practices would be effective in implementing the strategy listed above:

1. Restrict human access and livestock use in heavily degraded areas of alpine tundra.

2. Modify grazing allotment plans and trail use regulations to prevent declines in good quality habitat.

3. Restore alpine areas that are in a degraded condition.

Group 39—Lewis' Woodpecker (Resident Population)

Results

Species ranges, source habitats, and special habitat features—Resident Lewis'woodpeckers are distributed in a small area of open woodlands in the northern end of the Southern Cascades and in southern portions of the Northern Cascades ERUs (fig. 117), along the eastern foothills of the Cascade Range. Birds use this area year-round, unlike migratory Lewis'woodpeckers described in group 2 that use the basin only during the breeding season. Source habitats of the resident Lewis'woodpecker include oak woodlands (vol. 3, appendix 1, table 1), parklike pine-oak, burned pine-fir forests, and cottonwood groves (Galen 1989). These vegetation types apparently were most abundant, historically and currently, in a small area within the northern portion of the Southern Cascades ERU (fig. 118).

Unlike most woodpecker species, Lewis'woodpecker is an aerial insectivore and requires openings for its foraging maneuvers. This woodpecker breeds in habitats that provide abundant insects (see group 2 for a broader discussion on migratory Lewis' woodpeckers) and winters in areas where temperatures are warm enough to support flying insects and where acorns are abundant. Acorns are harvested in fall and stored for winter use. Birds overwinter within the basin where a reliable acorn supply is available (Galen 1989).

Because this species has weak excavator morphology (Spring 1965), Lewis'woodpeckers typically require large snags in an advanced state of decay or trees with soft sapwood for ease of cavity excavation (Bock 1970, Raphael and White 1984, Saab and Dudley 1995, Tobalske 1997). Additionally, Lewis' woodpeckers usurp occupied cavities (Saab and Dudley 1995) or reuse old cavities created by strong excavators (that is, hairy woodpecker, black-backed woodpecker, and Northern flicker) or nest in natural cavities of trees (Bock 1970, Galen 1989, Saab and Dudley 1995, Tashiro-Vierling 1994, Vierling 1997).

Nest tree species of resident birds in the basin were primarily Oregon white oak and ponderosa pine, and less commonly Douglas-fir and cottonwood (Galen 1989). Snags and trees used for nesting are generally larger and more heavily decayed than expected based on availability of such snags (see group 2 for description of source habitats). In north-central Oregon, tree diameters at 23 nests in Oregon white oak ranged from 31.8 to 99 cm (12.5 to 39 in) and averaged 55. 9 cm (22 in); tree height ranged from 3.0 to 15.2 m (10 to 50 ft) and averaged 9.7 m (32 ft) (Galen 1989). Most of these nest trees, however, were living or had light decay. Heavily decayed trees, typical of nest trees elsewhere (see group 2 for source habitat description), were probably not necessary in north-central Oregon because nesting only occurred in preexisting cavities, and there was no evidence of Lewis' woodpeckers excavating new cavities (Galen 1989).

Nesting habitat in north-central Oregon was usually open pine-oak woodlands and burned coniferous forests (Galen 1989). Nests also were located in cottonwood groves and narrow oak groves adjacent to open areas. No nests were found in scrub-oak thickets along south-facing slopes, unburned coniferous forests, or clearcuts. Proximity to openness was considered a critical microhabitat feature for breeding habitat (Galen 1989). Open woodlands provide sufficient visibility and space for effective flycatching. Most nests (36 of 53) were located in areas with >75 percent open canopy. Snags were also an important component of nesting habitat. Snags were used for perching during the breeding season and for acorn storage during winter.

Nesting densities of resident woodpeckers in Oregon differed from one breeding pair per 8 ha (20 acres) of woodland to one breeding pair per 16 ha (40 acres), depending on suitable snags, trees, and cavities available for nesting (Galen 1989). Nesting habitat required for one pair of Lewis'woodpeckers was estimated at 10 ha (25 acres) of open pine-oak, oak, or cottonwood when these woodlands are adjacent to open areas of equal or greater size (Galen 1989).

Wintering habitat of resident Lewis' woodpeckers in the basin was associated with nest trees used during the breeding season (Galen 1989). Nearly 90 percent of 46 nests showed signs of wintering woodpeckers. Acorns were stored in nest trees or in adjacent snags, and oaks were nearby.

In foothills habitat of southeastern Colorado, acorns were the primary winter food source (Vierling 1997). Acorn crops were higher at occupied winter sites than at random sites. Availability of storage sites for mast was a critical feature of winter habitat (Vierling 1997). Storage trees were significantly taller (¯ = 17.5 m versus 10.9 m [57.8 ft vs. 36 ft]) and of larger diameter (¯ = 104.8 cm versus 61.7 cm [41.3 in versus 24.3 in]) than random trees (Vierling 1997). Crevices in dead and decaying trees, and the deep furrowed bark of cottonwoods, were important characteristics of acorn storage sites.

Broad-scale changes in source habitats— No apparent broad-scale changes occurred in breeding and wintering source habitats of resident Lewis'woodpeckers (figs. 118A, 118B, and 119).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Areal extent of oak woodlands, the only source habitats used for this group, was not estimated to have changed using the large pixel size of this analysis (vol. 3, appendix 1,

Figure 118—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 39 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 119—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 39, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of $\geq$60 percent; 1 = an increase of $\geq$20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of $\geq$20 percent but <60 percent; and -2 = a decrease of $\geq$60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

table 4; figs. 118A, and 118B). This cover type is limited within the basin and has had few threats of logging activities but greater threats by development and firewood cutting. Changes in oak woodlands may not be discernable where oaks occur in small stands or where they occur within conifer stands. Nearby pine-oak, burned pine forests, and cottonwood woodlands used by this resident population were not evaluated in the broad-scale analysis. Thus, a broad-scale analysis for this group has limited application.

Condition of special habitat features— Abundance of large, heavily decayed snags for nesting and acorn storage may have declined in the range of resident Lewis'woodpeckers within the basin. Densities of large-diameter snags ($\geq$53 cm [21 in] d.b.h.) have declined basin-wide from historical to current levels (Hann and others 1997, Hessburg and others 1999, Quigley and others 1996).

Oak mast-producing trees are critical for overwinter survival of Lewis'woodpeckers in the basin. A 500-yr-old Oregon white oak attains large diameters (58 to 89 cm d.b.h. [23 to 35 in]) on generally dry slopes that offer slower growing conditions (Galen 1989). Destruction of these old and mature trees by clearing for pastures and firewood cutting could jeopardize resident Lewis'woodpeckers.

Open woodlands that allow foraging maneuvers have probably decreased as a result of fire control practices. Historically, oak woodlands in Washington were maintained by frequent wildfires, and through controlled burning by early inhabitants (Ryan and Carey 1995). Oak woodlands currently are threatened by encroachment of ponderosa pine and Douglas-fir. Fire control also likely has reduced understory shrubs and associated arthropods that provide food during the breeding season. Additionally, understory shrubs may have been altered by disturbances of grazing practices and recreational activities.

Other factors affecting the group— Road densities have significantly increased throughout the basin (Hann and others 1997, Quigley and others 1996), allowing greater human access into forested regions and subsequent increases in snag removal for firewood. Salvage logging is another threat to snags that provide potential nest sites (Marshall and others 1996). Prolonged human presence at or near nest sites may cause abandonment (Bock 1970); however, stable populations coexist with park development and heavy tourist use during the breeding season in British Columbia (Siddle and Davidson 1991).

Chlorinated hydrocarbons, particularly DDT, which were formerly used as pesticides in fruit orchards and gardens, could have potentially negative effects on

Lewis'woodpeckers (Tobalske 1997) because these woodpeckers sometimes nest in agricultural settings (Sorensen 1986, Tashiro-Vierling 1994). Elevated energetic costs and stress may be associated with high rates of territorial encounters with European starlings, which could reduce reproductive success even if Lewis'woodpecker dominates the interaction (Siddle and Davidson 1991). Altered fire regimes and subsequent changes in the structure and composition of lower montane forests (Hann and others 1997) could reduce suitable oak woodlands for breeding and wintering Lewis'woodpeckers. Large cottonwoods, used for nesting and acorn storage, are threatened by altered hydrologic regimes, grazing practices, and urban development (Marshall and others 1996).

Population status and trends—No population trends are available for the resident Lewis'woodpeckers that occupy the eastern foothills of Mount Hood. Breeding Bird Surveys for the entire basin indicate that population trends have been stable during 1968-94 (Saab and Rich 1997), but any relation to the resident population is not known. Trend data generated by the BBS may be more adequate for monitoring populations of resident Lewis'woodpeckers than migratory populations (see group 2, "Population Status and Trends"). Dramatic cycles of population abundance related to local changes in habitat (Bock 1970) may not apply to resident birds that will use acorns as a year-round food source, supplemented by insects during the breeding season.

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 39 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

Issues—

1. Exclusion of fire in parklike oak and pine-oak woodlands and subsequent decreases in natural forest openings and shrubby understories because of invasions by conifers (Marshall and others 1996).

2. Losses of large oak trees for mast production because of firewood cutting, fire control, and pasture development.

3. Decline in availability of large, heavily decayed ponderosa pine for nesting and acorn storage sites.

4. Losses of large cottonwoods used for nesting and acorn storage (Marshall and others 1996).

5. Increase in application of agricultural insecticides.

Potential strategies—The issues identified above suggest the following broad-scale strategies for the long-term persistence of resident Lewis'woodpeckers in the northern portion of the Southern Cascades ERU.

1. (To address issue no. 1) Return natural fire regimes to oak and pine-oak woodlands.

2. (To address issues nos. 2–4) Retain large (>30 cm d.b.h. [12 in]), old snags and trees of Oregon white oak, ponderosa pine, and cottonwoods (Galen 1989).

3. (To address issues no. 3 and no. 4) Protect acorn storage sites in wintering areas (Galen 1989, Marshall and others 1996).

4. (To address issue no. 4) Maintain existing old-growth cottonwood forests and manage young forests for the long-term sustainability of cottonwood/riverine systems.

5. (To address issue no. 5) Avoid use of toxic chlorinated hydrocarbons and organophosphorus insecticides near Lewis'woodpecker nesting and wintering sites.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Maintain parklike oak and pine-oak woodlands by using silvicultural treatments of prescribed fire and thinning of small-diameter ponderosa pine (<30 cm [12 in]).

2. (In support of strategy no. 2) Retain all Oregon white oak and ponderosa pine trees or snags over 3 m (10 feet) tall and >30 cm (12 in) d.b.h. (Galen 1989). Management of 10-ha (25-acre) units having about 25 percent canopy cover will likely provide nesting habitat for one pair of Lewis' woodpeckers (see Galen 1989).

3. (In support of strategy no. 3) Control fuel wood permits for removal of oaks, pines, or cottonwood used for winter storage sites. Minimize the density of roads open to motorized vehicles. Close roads after timber harvest activities, and maintain short periods during which such roads are open to reduce removal of snags along roads. In addition or as an alternative to road management, actively enforce fuel wood regulations to minimize removal of snags.

4. (In support of strategy no. 4) Survey and map existing old forests of cottonwoods and reference their locations in land management planning documents. Monitor conditions of cottonwood stands to ensure that sufficient seedling or vegetative regeneration, or both, is occurring. Identify factors limiting regeneration so that appropriate corrective measures can be taken. For example, return natural hydrologic regimes to portions of large river systems that support cottonwood riparian woodlands (for example, the Columbia River).

5. (In support of strategy no. 5) Establish zones with no use of toxic agricultural insecticides near Lewis'woodpecker breeding and wintering habitats.

Group 40—Brown-Headed Cowbird

Results

Species ranges, source habitats, and special habitat features—Group 40 consists of the brown-headed cowbird, a migrant summer breeder found throughout the basin (fig. 120). The cowbird is considered a contrast species (vol. 3, appendix 1, table 2) because it requires a juxtaposition of contrasting vegetative structure to meet all aspects of its ecology. Foraging areas are in disturbed sites near livestock, and breeding areas generally are in forests and riparian areas where passerine densities are high (Robinson and others 1995). Source habitats for the brown-headed cowbird are the agricultural community type (vol. 3, appendix 1, table 1), and the presence of livestock is a special habitat feature. Additionally, the cowbird is dependent on the presence of active bird nests for parental care of their offspring. Nest parasitism by cowbirds has been documented for over 220 bird species, primarily passerine species, and at least 144 species have fledged cowbird young (Friedmann and Kiff 1985).

Although not mappable at the broad-scale of our analysis, horse corrals and pack stations in lower montane and montane community groups also provide source habitats. Associated breeding sites are located as far as 7 km (4.3 mi) (Rothstein and others 1987) from livestock areas, where cowbirds congregate to forage. Because of the presence of livestock areas, the distribution of source habitats is much greater than estimated by our broad-scale analysis.

Broad-scale changes in source habitats—Source habitats for the cowbird were probably not present in the basin historically (fig. 121A). Source habitats are now present in all ERUs and are particularly widespread in the Columbia Plateau and Upper Snake (fig. 121B). The trend in habitat availability has been strongly increasing basin-wide (figs. 121C and 122).

Interpreting Results

Composition and structure of vegetation associated with changes in source habitats— Increases in source habitats were primarily attributed to the conversion of native vegetation to agriculture. The establishment of the cropland-hay-pasture cover type occurred on sites previously dominated by the fescue-bunchgrass, big sagebrush, and native forb cover types (Hann and others 1997). Agriculture now covers >10 percent of the land area in five ERUs: Columbia Plateau (estimated 44 percent), Blue Mountains (estimated 17 percent), Northern Glaciated Mountains (estimated 12 percent), Owyhee Uplands (estimated 12 percent), and Upper Snake (estimated 33 percent; vol. 3, appendix 1, table 4).

Condition of special habitat features—The presence of livestock is strongly associated with agricultural land uses throughout the basin. Livestock areas suitable for cowbird foraging, therefore, have probably increased in proportion to the estimated increase in area used for agriculture. Moreover, livestock areas in the lower montane and montane community groups likely have increased from historical conditions because of the location of pack stations adjacent to wilderness areas and parks, and rural expansion into forested areas.

Figure 120—Ranges of species in group 40 within the basin (from Marcot and others, in prep.). For species whose ranges shifted significantly from historical conditions, separate maps are shown for historical and current ranges; otherwise, the current range map also denotes the historical range.

Other factors affecting the group—Because cowbirds rely on other bird species to raise their young, they are affected by the same factors that govern breeding success of their selected hosts. About 50 percent of cowbird eggs are lost to normal nest-related mortality such as weather and predation (Nice 1957). Additional losses depend on the behavioral responses of the host, including egg rejection, egg burial, and nest desertion (Friedmann 1929).

Microsite conditions affect cowbird densities and parasitism rates. Cowbird numbers and parasitism rates are higher near internal forest openings, powerline corridors, and streams and in small versus large woodlots (Robinson and others 1995). Forest fragmentation and high edge density are conducive to successful breeding by cowbirds (Robinson and others 1995).

Population status and trends—Cowbirds have undergone a dramatic range expansion across North America, both eastward and westward. Expansion into eastern forests occurred in the late 1700s; this expansion was brought about by forest clearing and increases in agriculture and livestock uses. Colonization westward into Washington and Oregon began a century later (Rothstein 1994); this range expansion was likely associated with the clearing of lands for agricultural and livestock uses. Population trends were stable basin-wide from 1966 to 1994 (Saab and Rich 1997). Within Oregon, BBS data suggested that populations have been decreasing by 4 percent annually from 1966 to 1995 ($n = 88$; $P < 0.05$; Sauer and others 1996).

Management Implications

The following issues, strategies, and practices may be useful to managers as a starting point for integrating potential resource objectives for group 40 with broader, ecosystem-based objectives for all other resources on FS- and BLM-administered lands in the basin.

VIEW MAP

Figure 121—Percentage of area identified as source habitats, historically (A) and currently (B), and the relative change in percentage of area of source habitats from historical to current periods (C), for group 40 within each of 2,562 watersheds in the basin. Relative change for each watershed is shown as one of five trend categories, where -2 = a decrease of ≥60 percent; -1 = a decrease of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; 1 = an increase of ≥20 percent but <60 percent; and 2 = an increase of ≥60 percent.

Figure 122—Percentage of watersheds within five trend categories of relative change in source habitats from historical to current periods for group 40, basin-wide and by ecological reporting units. Trend categories correspond to the following relative changes from historical to current periods: 2 = an increase of ≥60 percent; 1 = an increase of ≥20 percent but <60 percent; 0 = an increase or decrease of <20 percent; -1 = a decrease of ≥20 percent but <60 percent; and -2 = a decrease of ≥60 percent. Number of watersheds from which estimates were derived is denoted by *n*.

Issues—Issues primarily relate to the effect of nest parasitism by cowbirds on host species.

1. Reductions in nest success of host species, particularly state species of concern with known high parasitism rates.

2. Continued invasion of cowbirds into lower montane and montane community groups through the aid of small, remote livestock areas.

Potential strategies—

1. (To address issue no. 1) Minimize livestock concentrations in proximity to known source habitats for state and federally listed sensitive avian species.

2. (To address issue no. 1) Reduce parasitism rates on state species of concern.

3. (To address issue no. 2) Reduce opportunities for cowbird establishment in lower montane and montane community groups.

Practices that support strategies—The following practices would be effective in implementing the strategies listed above:

1. (In support of strategy no. 1) Consider the proximity of state species of concern before locating livestock-handling facilities on Federal land. Consider relocation of livestock facilities if such facilities exist in areas deemed important for recovery of an avian species of concern.

2. (In support of strategy no. 2) Intensively trap and remove cowbirds near nests of selected species of concern with high parasitism rates (Robinson and others 1995).

3. (In support of strategy no. 3) Delay annual establishment of livestock corrals within the lower montane and montane community groups during the early breeding season when cowbirds are actively seeking host nests (Kie 1991, Sanders and Flett 1989).

4. (In support of strategy no. 4) Consolidate remote livestock areas into fewer sites.

Abbreviations

Centimeter	(cm)
Hectare	(ha)
Inch	(in)
Kilometer	(km)
Meter	(m)
Mile	(mi)
Year	(yr)

References

Anon. 1989. Goshawk breeding habitat in lowland Britain. British Birds. 82: 56-67.

Adams, Layne G.; Bailey, James A. 1983. Winter forages of mountain goats in central Colorado. Journal of Wildlife Management. 47: 1237-1243.

Allen, Edith B. 1995. Restoration ecology: limits and possibilities in arid and semiarid lands. In: Roundy, Bruce A.; McArthur, E. Durant; Haley, Jennifer S.; Mann, David K., comps. Proceedings, wildland shrub and arid land restoration symposium; 1993 October 19-21; Las Vegas, NV. Gen. Tech. Rep. INT-315. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 7-15.

Allendorf, F.W.; Harris, R.B.; Metzgar, L.H. 1991. Estimation of effective population size of grizzly bears by computer simulation. In: Dudley, E.C., ed. The unity of evolutionary biology: Proceedings of the 4th international congress of systematic and evolutionary biology; 1990 July. College Park, MD: University of Maryland. Portland, OR: Dioscorides Press: 650-654. Vol. 2.

Anderson, W.L.; Duzan, R.E. 1978. DDE residues and eggshell thinning in loggerhead shrikes. Wilson Bulletin. 90(2): 215-220.

Apple, L. 1994. Moving time. Wyoming Wildlife. 58(8): 18-23.

Arthur, S.M.; Krohn, W.B.; Gilbert, J.R. 1989. Home range characteristics of adult fishers. Journal of Wildlife Management. 53(3): 674-679.

Aubry, K.B.; Houston, D.B. 1992. Distribution and status of the fisher in Washington. Northwestern Naturalist. 73: 69-79.

Austin, G.T.; Russell, S.M. 1972. Interspecific aggression of ash-throated flycatchers on Cassin's sparrows. Condor. 74(4): 481.

Autenrieth, R. 1978. Guidelines for the management of pronghorn antelope. In: Proceedings of the 8th biennial pronghorn antelope workshop; 1978 May 2-4; Jasper, AB. Jasper, AB: Alberta Recreation, Parks and Wildlife, Fish and Wildlife Division: 472-525.

Autenrieth, R.E. 1981. Sage grouse management in Idaho. Wildlife Bull. 9. Boise, ID: Idaho Department of Fish and Game. 239 p.

Autenrieth, Robert; Molini, William; Braun, Clait, eds. 1982. Sage grouse management practices. Tech. Bull. 1. Twin Falls, ID: Western States Sage Grouse Committee. 42 p.

Bailey, T.N.; Bangs, E.E.; Portner, M.F. [and others]. 1986. An apparent overexploited lynx population on the Kenai Peninsula, Alaska. Journal of Wildlife Management. 50: 279-290.

Baldwin, P.H. 1968. Predator-prey relationships of birds and spruce beetles. In: Proceedings of the North Central Branch, Entomological Society of America; 1968 March 27-29; Denver, CO. Lafayette, IN: Entomology Department, Purdue University. 23(2): 90-99.

Banci, Vivian. 1994. Wolverine. In: Ruggiero, Leonard F.; Aubry, Keith B.; Buskirk, Steven W. [and others], tech. eds. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the Western United States. Gen. Tech. Rep. RM-254. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 7-37. Chapter 2.

Bangs, Edward E.; Fritts, Steven H. 1996. Reintroducing the gray wolf to central Idaho and Yellowstone National Park. Wildlife Society Bulletin. 24(3): 402-413.

Barclay, R.M.R. 1985. Long- versus short-range foraging strategies of hoary (*Lasiurus cinereus*) and silver-haired (*Lasionycteris noctivagans*) bats and the consequences for prey selection. Canadian Journal of Zoology. 63(11): 2507-2515.

Barclay, R.M.R. 1986. The echolocation calls of hoary (*Lasiurus cinereus*) and silver-haired (*Lasionycteris noctivagans*) bats as adaptations for long- versus short-range foraging strategies and the consequences for prey selection. Canadian Journal of Zoology. 64(12): 2700-2705.

Barclay, R.M.R. 1991. Population structure of temperate zone insectivorous bats in relation to foraging behavior and energy demand. Journal of Animal Ecology. 60: 165-178.

Barnum, D.A. 1975. Aspects of western gray squirrel ecology. Pullman, WA: Washington State University. 58 p. M.S. thesis.

Beale, D.M.; Smith, A.D. 1970. Forage use, water consumption, and productivity of pronghorn antelope in western Utah. Journal of Wildlife Management. 34(3): 570-582.

Bechard, M.J.; Schmutz, J.K. 1995. Ferruginous hawk (*Buteo regalis*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 172. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists'Union. 20 p.

Beck, Jonathan M.; Peterson, Charles R. 1995. Movements and habitat selection of the longnose snake (*Rhinocheilus lecontei*) in southwestern Idaho. Tech. Bull. 95-18. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office. 30 p.

Belnap, J. 1995. Surface disturbances: their role in accelerating desertification. Environmental Monitoring and Assessment. 37: 39-57.

Belnap, Jayne. 1993. Recovery rates of cryptobiotic crusts: inoculant use and assessment methods. Great Basin Naturalist. 53: 89-95.

Belnap, Jayne. 1994. Potential role of cryptobiotic soil crusts in semiarid rangelands. In: Monsen, Stephen B.; Kitchen, Stanley G., eds. Proceedings, ecology and management of annual rangelands; 1992 May 18-21; Boise, ID. Gen. Tech. Rep. INT-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 179-189.

Belnap, Jayne; Gardner, John S. 1993. Soil microstructure in soils of the Colorado Plateau: the role of the cyanobacterium *Microcoleus vaginatus*. Great Basin Naturalist. 53: 40-47.

Benkman, C.W. 1990. Intake rates and the timing of crossbill reproduction. Auk. 107(2): 376-386.

Benkman, Craig W. 1992. White-winged crossbill. In: Poole, A.; Gill, F., eds. The birds of North America. No. 27. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union. 20 p.

Bentz, Jerry A.; Woodard, Paul M. 1988. Vegetation characteristics and bighorn sheep use on burned and unburned areas in Alberta. Wildlife Society Bulletin. 16(2): 186-193.

Betts, B. 1996. Bat maternity tree roosts and forest management. Blue Mountains Natural Resources Institute. Natural Resource News. Spring: 2-3.

Betts, B.J. 1990. Geographic distribution and habitat preferences of Washington ground squirrels (*Spermophilus washingtoni*). Northwestern Naturalist. 71(2): 27-37.

Black, Hal L. 1974. A north temperate bat community: structure and prey populations. Journal of Mammalogy. 55: 138-157.

Blackford, J.L. 1955. Woodpecker concentrations in burned forest. Condor. 57(1): 28-30.

Blair, Steve; Servheen, Greg; Gordon, Floyd [and others]. 1995. Saving all the pieces: the Idaho state conservation effort, white-headed woodpecker (*Picoides albolarvatus*) in Idaho. Boise, ID: Idaho Department of Fish and Game. 24 p.

Block, W.M.; Brennan, L.A. 1987. Characteristics of Lewis woodpecker habitat on the Modoc Plateau, California. Western Birds. 18(4): 209-212.

Blood, D.A. 1961. An ecological study of California bighorn sheep *Ovis canadensis californiana* (Douglas) in southern British Columbia. Vancouver, BC: University of British Columbia. 127 p. M.S. thesis.

Blumton, A.K. 1989. Factors affecting loggerhead shrike mortality in Virginia. Blacksburg, VA: Virginia Polytechnical Institute and State University. 85 p. M.S. thesis.

Blus, L.J.; Staley, C.S.; Henny, C.J. [and others]. 1989. Effects of organophosphorus insecticides on sage grouse in southeastern Idaho. Journal of Wildlife Management. 53(4): 1139-1146.

Boal, C.W.; Mannan, R.W. 1994. Northern goshawk diets in ponderosa pine forests on the Kaibab Plateau. Studies in Avian Biology. 16: 97-102.

Bock, C.E. 1970. The ecology and behavior of the Lewis woodpecker (*Asyndesmus lewis*). In: Zoology. 92/1970 May 25. Berkeley, CA: University of California Press. 91+ p.

Bock, C.E.; Bock, J.H. 1974. On the geographical ecology and evolution of the three-toed woodpeckers, *Picoides tridactylus* and *Picoides arcticus*. American Midland Naturalist. 92(2): 397-405.

Bock, C.E.; Bock, J.H. 1987. Avian habitat occupancy following fire in a Montana shrubsteppe. Prairie Naturalist. 19(3): 153-158.

Bock, C.E.; Bock, J.H. 1992. Response of birds to wildfire in native versus exotic Arizona grassland. Southwestern Naturalist. 37(1): 73-81.

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.; Bock, J.H.; Smith, H.M. 1993. Proposal for a system of Federal livestock exclosures on public rangelands in the western United States. Conservation Biology. 7: 731-733.

Bock, C.E.; Webb, B. 1984. Birds as grazing indicator species in southeastern Arizona. Journal of Wildlife Management. 48(3): 1045-1049.

Bock, Carl E. 1969. Intra- vs. interspecific aggression in pygmy nuthatch flocks. Ecology. 50(5): 903-905.

Bock, Carl E.; Lepthien, L.W. 1976. Synchronous eruptions of boreal seed-eating birds. American Naturalist. 110: 559-579.

Bodie, W.L.; Hickey, W.O. 1980. Response of bighorns to a rest-rotation grazing plan in central Idaho. Biennial symposium, Northern Wild Sheep and Goat Council. 2: 60-69.

Bohall-Wood, P. 1987. Abundance, habitat use, and perch use of loggerhead shrikes in north central Florida. Wilson Bulletin. 99(1): 82-86.

Boutin, S.C.; Krebs, J.; Sinclair, A.R.E.; Smith, J.N.M. 1986. Proximate causes of losses in a snowshoe hare population. Canadian Journal of Zoology. 64: 606-610.

Braun, Clait E.; Baker, Maurice F.; Eng, Robert L. [and others]. 1976. Conservation committee report on effects of alteration of sagebrush communities on the associated avifauna. Wilson Bulletin. 88: 165-171.

Brawn, J.D.; Balda, R.P. 1988. Population biology of cavity nesters in northern Arizona: do nest sites limit breeding densities? Condor. 90(1): 61-71.

Brennan, L.A. 1990. What happened to the mountain quail in Idaho? Quail Unlimited. 9: 42-43, 69.

Brennan, Leonard A. 1989. Report on a survey of mountain quail in eastern Oregon, southeastern Washington, western Idaho, and northern Nevada. Boise, ID: The Chukar Foundation. 58 p. 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.

Brennan, L.A.; Block, W.M.; Gutierrez, R.J. 1987. Habitat use by mountain quail in northern California. Condor. 89: 66-74.

Brittell, J.D.; Poelker, R.J.; Sweeny, S.J. [and others]. 1989. Native cats of Washington. Olympia, WA: Washington Department of Wildlife. 169 p. 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.

Brooks, B.L.; Temple, S.A. 1990. Habitat availability and suitability for loggerhead shrikes in the upper Midwest. American Midland Naturalist. 123: 75-83.

Brown, Herbert A.; Bury, R. Bruce; Darda, David M. [and others]. 1995. Reptiles of Washington and Oregon. Seattle, WA: Seattle Audubon Society. 176 p.

Bruns, E.H. 1977. Winter behavior of pronghorns in relation to habitat. Journal of Wildlife Management. 41(3): 560-571.

Bull, E.L. 1991. Summer roosts and roosting behavior of Vaux's swifts in old growth forests. Northwest Naturalist. 72(2): 78-82.

Bull, E.L.; Hohmann, J.E. 1993. The association between Vaux's swifts and old growth forests in northeastern Oregon. Western Birds. 24(1): 38-42.

Bull, E.L.; Holthausen, R.S. 1993. Habitat use and management of pileated woodpeckers in northeastern Oregon. Journal of Wildlife Management. 57(2): 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(4): 786-793.

Bull, E.L.; Wright, A.L.; Henjum, M.G. 1990. Nesting habitat of flammulated owls in Oregon. Journal of Raptor Research. 24(3): 52-55.

Bull, Evelyn L. 1980. Resource partitioning among woodpeckers in northeastern Oregon. Moscow, ID: University of Idaho. 109 p. Ph.D. dissertation.

Bull, Evelyn L.; Henjum, Mark G. 1990. Ecology of the great gray owl. Gen. Tech. Rep. PNW-GTR-265. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 39 p.

Bull, Evelyn L.; Partridge, Arthur D. 1986. Methods of killing trees for use by cavity nesters. Wildlife Society Bulletin. 14: 142-146.

Bull, Evelyn L.; Peterson, Steven R.; Thomas, Jack W. 1986a. 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, Evelyn L.; Thomas, Jack Ward; Horn, Kirk. 1986b. Snag management on National Forests in the Pacific Northwest—1984. Western Journal of Applied Forestry. 1(2): 41-43.

Bull, Evelyn L.; Twombly, Asa D.; Quigley, Thomas M. 1980. Perpetuating snags in managed mixed conifer forests of the Blue Mountains, Oregon. In: Proceedings, management of western forests and grasslands for nongame birds; 1980 February 11-14; Salt Lake City, UT. Gen. Tech. Rep. INT-86. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 325-336.

Busack, Stephen D.; Bury, R. Bruce. 1974. Some effects of off-road vehicles and sheep grazing on lizard populations in the Mojave Desert. Biological Conservation. 6: 179-183.

Buskirk, Stephen W.; Powell, RogerA. 1994. Habitat ecology of American martens and fishers. In: Buskirk, S.W.; Harestad, A.; Raphael, M.; Powell, R.A., eds. Martens, sables and fishers: biology and conservation. Ithaca, NY: Cornell University Press: 283-296.

Butts, T.W. 1992. Lynx (*Felis lynx*) biology and management: a literature review and annotated bibliography. Missoula, MT: U.S. Department of Agriculture, Forest Service, Northern Region. 1 vol.

Cade, B.S.; Hoffman, R.W. 1990. Winter use of Douglas-fir forests by blue grouse in Colorado. Journal of Wildlife Management. 54(3): 471-479.

Cade, B.S.; Hoffman, R.W. 1993. Differential migration of blue grouse in Colorado. Auk. 110: 70-77.

Call, Mayo W. 1979. Habitat requirements and management recommendations for sage grouse. Tech. Note. Denver, CO: U.S. Department of the Interior, Bureau of Land Management. 37 p.

Carbyn, L.N.; Patriquin, D. 1983. Observations on home range sizes, movements and social organization of lynx (*Lynx canadensis*) in Riding Mountain National Park, Manitoba. Canadian Field-Naturalist. 97: 262-267.

Carey, A.B. 1995. Sciurids in Pacific Northwest managed and old-growth forests. Ecological Applications. 5(3): 648-661.

Carey, Andrew B. 1991. The biology of arboreal rodents in Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-276. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 46 p. (Huff, Mark H.; Holthausen, Richard S.; Aubry, Keith B., tech. coords. Biology and management of old-growth forests).

Cassirer, E. Frances; Oldenburg, Lloyd E.; Coggins, Victor L. [and others]. 1996. Overview and preliminary analysis of a bighorn sheep die-off, Hells Canyon 1995-96. In: Proceedings of the 10th biennial symposium; 1996 April 29-May 3; Silverthorne, CO. Cody, WY: Northern Wild Sheep and Goat Council. 10: 78-86.

Caton, E. 1996. Effects of fire and salvage logging on the cavity-nesting bird community in northwestern Montana. Missoula, MT: University of Montana. 115 p. Ph.D. dissertation.

Chadwick, Douglas H. 1973. Mountain goat ecology-logging relationships in the Bunker Creek drainage of western Montana. Missoula, MT: School of Forestry, University of Montana; final report; State of Montana Project No. W-120-R-3, 4. No. 91.01; 228 p. M.S. thesis.

Chapin, T.G. 1995. Influence of landscape pattern and forest type on use of habitat by marten in Maine. Orono, ME: University of Maine. 131 p. M.S. thesis.

Chavez-Ramirez, F.; Gawlik, D.E.; Prieto, F.G.; Slack, R.D. 1994. Effects of habitat structure on patch use by loggerhead shrikes wintering in a natural grassland. Condor. 96(1): 228-231.

Christy, R.E.; West, S.D. 1993. Biology of bats in Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-308. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 28 p.

Chung-MacCoubrey, Alice L. 1996. Bat species composition and roost use in pinyon-juniper woodlands of New Mexico. In: Barclay, Robert M.R.; Brigham, R. Mark, eds. Bats and forests symposium; 1995 October 19-21; Victoria, BC. Victoria, BC: Ministry of Forests Research Program: 118-123.

Clark, D.R., Jr. 1988. Environmental contaminants and the management of bat populations in the United States. Gen. Tech. Rep. RM-166. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 409-413.

Clark, H.W. 1935. Fire and bird populations. Condor. 37: 16-18.

Clark, Richard J. 1975. A field study of the short-eared owl, *Asio flammeus* (Pontoppidan), in North America. Wildlife Monographs. 47: 1-67.

Clarkson, D.A.; Mills, L.S. 1994. Hypogeous sporocarps in forest remnants and clearcuts in southwest Oregon. Northwest Science. 68(4): 259-265.

Clemente, F.; Valdez, R.; Holechek, J.L. [and others]. 1995. Pronghorn home range relative to permanent water in southern New Mexico. Southwestern Naturalist. 40(1): 38-41.

Coffin, K.W.; Kujala, Q.J.; Douglass, R.J.; Irby, L.R. 1997. Interactions among marten prey availability, vulnerability, and habitat structure. In: Prouls, G.; Gryant, H.N.; Woodard, P.M., eds. *Martes*: taxonomy, ecology, techniques, and management: Proceedings of the 2d international *Martes* symposium; [Dates of meeting unknown]; [Meeting location unknown]. Edmonton, AB: The Provincial Museum of Alberta: 199-211.

Coggins, V.L. 1988. The Lostine Mountain bighorn sheep die-off and domestic sheep. In: Proceedings of the 6th biennial symposium; 1988 April 11-15; Banff, AB. Cody, WY: Northern Wild Sheep and Goat Council. 6: 66-76.

Coggins, Victor L.; Matthews, Patrick E.; Van Dyke, Walt. 1996. History of transplanting mountains goats and mountain sheep—Oregon. In: Proceedings of the 10th biennial symposium; 1996 April 29-May 3; Silverthorne, CO. Cody, WY: Northern Wild Sheep and Goat Council. 10: 190-197.

Connelly, J.W., Jr. 1982. An ecological study of sage grouse in southeastern Idaho. Pullman, WA: Washington State University. 84 p. Ph.D. dissertation.

Connelly, J.W.; Wakkinen, W.L.; Apa, A.D.; Reese, K.P. 1991. Sage grouse use of nest sites in southeastern Idaho. Journal of Wildlife Management. 55: 521-524.

Cooper, J.G. 1868. The fauna of Montana territory. American Naturalist. 2: 528-538.

Copeland, Jeffrey P. 1996. Biology of the wolverine in central Idaho. Moscow, ID: University of Idaho. 138 p. M.S. thesis.

Corn, J.G.; Raphael, M.G. 1992. Habitat characteristics at marten subnivean access sites. Journal of Wildlife Management. 56: 442-448.

Cornely, John E.; Carraway, L.N.; Verts, B.J. 1992. *Sorex preblei.* Mammalian species. No. 416: American Society of Mammalogists: 1-3.

Coulombe, H.N. 1971. Behavior and population ecology of the burrowing owl, *Speotyto cunicularia*, in the Imperial Valley of California. Condor. 73(2): 162-176.

Craig, R.B. 1978. An analysis of the predatory behavior of the loggerhead shrike. Auk. 95(2): 221-234.

Craig, T.H.; Trost, C.H. 1979. The biology and nesting density of breeding American kestrels and long-eared owls on the Big Lost River, southeastern Idaho. Wilson Bulletin. 91(1): 50-61.

Craighead, J.J.; Craighead, F.C. 1956. Hawks, owls, and wildlife. Harrisburg, PA: Stackpole Company. 443 p.

Crawford, J.A.; Lutz, R.S. 1985. Sage grouse population trends in Oregon, 1941-1983. Murrelet. 66(3): 69-74.

Crocker-Bedford, D.C. 1990. Goshawk reproduction and forest management. Wildlife Society Bulletin. 18: 262-269.

Daubenmire, R. 1970. Steppe vegetation of Washington. Bull. 62. Pullman, WA: State of Washington, Agricultural Experiment Station. 131 p.

Dawson, William R. 1997. Pine siskin (*Carduelis pinus*). In: Poole, Alan; Gill, Frank B., eds. The birds of North America. No. 280. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists'Union. 24 p.

Deblinger, R.D.; Alldredge, A.W. 1991. Influence of free water on pronghorn distribution in a sagebrush/steppe grassland. Wildlife Society Bulletin. 19(3): 321-326.

DeGraaf, R.M.; Scott, V.E.; Hamre, R.H. [and others]. 1991. Forest and rangeland birds of the United States: natural history and habitat use. Agric. Handb. 688. Washington, DC: U.S. Department of Agriculture. 625 p.

Delany, M.F.; Moore, C.T.; Progulske, D.R., Jr. 1995. Territory size and movements of Florida grasshopper sparrows. Journal of Field Ornithology. 66: 305-309.

Demarchi, R.A. 1965. An ecological study of the Ashnola bighorn winter ranges. Vancouver, BC: University of British Columbia. 103 p. M.S. thesis.

DeSante, D.F.; Pyle, P. 1986. Distributional checklist of North American birds. Volume 1: United States and Canada. Lee Vining, CA: Artemisia Press.

DeStefano, Stephen. 1990. Investigation of the status of kit foxes in southeastern Oregon. Tech. Rep. 90-5-01. Portland, OR: Oregon Department of Fish and Wildlife, Wildlife Diversity Program. 60 p.

Diem, K.L.; Zeveloff, S.I. 1980. Ponderosa pine bird communities. In: DeGraff, R.M.; Tilghman, N.G., eds. Management of western forests and grasslands for nongame birds. Gen Tech. Rep. INT-86. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 170-197.

Diller, Lowell V.; Johnson, Donald R. 1982. Ecology of reptiles in the Snake River Birds of Prey Area. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Snake River Birds of Prey Research Project. 107 p.

Dixon, R.; Saab, V. [In prep.]. Black-backed woodpecker. In: Poole, A.; Gill, F., eds. The birds of North America. Washington, DC: The American Ornithologists'Union; Philadelphia, PA: The Academy of Natural Sciences.

Dobkin, D.S.; Gettinger, R.D.; Gerdes, M.G. 1995. Springtime movements, root use, and foraging activity of Townsend's big-eared bat (*Plecotus townsendii*) in central Oregon. Great Basin Naturalist. 55(4): 315-321.

Dobler, F.C.; Eby, J.; Richardson, S.; Vander Haegen, M. 1996. Status of Washington's shrub-steppe ecosystem: extent, ownership, and wildlife/vegetation relationships. Res. Rep. Olympia, WA: Washington Department of Fish and Wildlife.

Dodge, W.E. 1977. Status of the fisher (*Martes pennanti*) in the conterminous United States. U.S. Department of the Interior. 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.

Doyle, F.I.; Smith, J.M.N. 1994. Population responses of northern goshawks to the 10-year cycle in numbers of snowshoe hares. Studies in Avian Biology. 16: 122-129.

Duncan, J.R. 1992. Influence of prey abundance and snow cover on great gray owl breeding dispersal. Winnipeg, MB: University of Manitoba. 127 p. Ph.D. dissertation.

Duncan, James R.; Hayward, Patricia H. 1994. Current management situation: great gray owls. In: Hayward, G.D.; Verner, J., tech. eds. 1994. Flammulated, boreal, and great gray owls in the United States: a technical conservation assessment. Gen. Tech. Rep. RM-253. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 159-175.

Dunning, J.B., Jr.; Bowers, R.K., Jr. 1990. Lethal temperatures in ash-throated flycatcher nests located in metal fence poles. Journal of Field Ornithology. 61(1): 98-103.

Eberhardt, L.L.; Knight, R.R. 1996. How many grizzlies in Yellowstone? Journal of Wildlife Management. 60(2): 416-421.

Edminster, F.C. 1954. American game birds of field and forest. Their habits, ecology and management. New York: Charles Scribner's Sons. 490 p.

Egoscue, Harold J. 1962. Ecology and life history of the kit fox in Tooele County, Utah. Ecology. 43(3): 481-497.

Ehrlich, Paul R.; Dobkin, David S.; Wheye, Darryl. 1988. The birder's handbook. New York: Simon and Schuster. 785 p.

Erickson, Janet L.; West, Stephen D. 1996. Managed forests in the western Cascades: the effects of seral stage on bat habitat use patterns. In: Barclay, Robert M.R.; Brigham, R. Mark, eds. Bats and forests symposium; 1995 October 19-21; Victoria, BC. Victoria, BC: Ministry of Forests Research Program: 215-227.

Evans, H.F. 1960. A preliminary investigation of caribou in the northwestern United States. Bozeman, MT: Montana State University. 145 p. M.S. thesis.

Evans, Raymond A.; Young, James A. 1978. Effectiveness of rehabilitation practices following wildfire in a degraded big sagebrush-downy brome community. Journal of Range Management. 31: 185-188.

Feldhamer, G.A.; Klann, R.S.; Gerard, A.S.; Driskell, A.C. 1993. Habitat partitioning, body size, and timing of parturition in pygmy shrews and associated soricids. Journal of Mammology. 74(2): 403-411.

Fischer, R.A. 1994. Sage grouse ecology final report. Boise, ID: Idaho Department of Fish and Game; job completion report; W-160-R-19; subproject 9. 91 p.

Flickinger, E.L. 1995. Loggerhead fatalities on a highway in Texas. In: Yosef, R.; Lohrer, F.E., eds. Shrikes (Laniidae) of the world: biology and conservation: 1st international shrike symposium; 1993 January 11-15; Lake Placid, FL. In: Proceedings of the Western Foundation of Vertebrate Zoology; 6(1): 67-69.

Flinn, P. 1956. Caribou of Idaho. Boise, ID: Idaho Department of Fish and Game. Unpublished report. On file with: Idaho Department of Fish and Game, P.O. Box 25, Boise, ID 83707. 79 p.

Foreyt, William J. 1994. Effects of controlled contact exposure between healthy bighorn sheep and llamas, domestic goats, mountain goats, cattle, domestic sheep, or mouflon sheep. In: Proceedings of the 9th biennial symposium: Northern Wild Sheep and Goat Council; 1994 May 2-6; Cranbrook, BC. Thermopolis, WY: Northern Wild Sheep and Goat Council. 9: 7-14. Proceedings available from: Foundation for North American Wild Sheep, 720 Allen Avenue, Cody, WY 82414.

Foreyt, William J.; Jessup, David A. 1982. Fatal pneumonia of bighorn sheep following association with domestic sheep. Journal of Wildlife Diseases. 18: 163-168.

Fortin, C.; Cantin, M. 1994. The effects of trapping on a newly exploited American marten population. 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: 179-191.

Frederick, Glenn P. 1991. Effects of forest roads on grizzly bears, elk, and gray wolves: a literature review. Publ. R1-91-73. Libby, MT: Kootenai National Forest. 53 p.

French, N.R. 1959. Life history of the black rosy finch. Auk. 76(2): 159-180.

Friedmann, H.; Kiff, L.F. 1985. The parasitic cowbirds and their hosts. Proceedings of the Western Foundation of Vertebrate Zoology. 2(4): 227-266.

Friedmann, J. 1929. The cowbirds, a study in the biology of social parasitism. Springfield, IL: C.C. Thomas. 421 p.

Fritts, Steven H.; Mech, L. David. 1981. Dynamics, movements, and feeding ecology of a newly protected wolf population in northwestern Minnesota. Wildlife Monographs No. 80. 79 p.

Fritts, Steven H.; Paul, William J.; Mech, L. David. 1985. Can relocated wolves survive? Wildlife Society Bulletin. 13(4): 459-463.

Fuller, Todd K.; Keith, Lloyd B. 1980. Wolf population dynamics and prey relationships in northeastern Alberta. Journal of Wildlife Management. 44(3): 583-602.

Galen, Christie. 1989. A preliminary assessment of the status of the Lewis'woodpecker in Wasco County, Oregon. Tech. Rep. 88-3-01. Portland, OR: Oregon Department of Fish and Wildlife, Nongame Wildlife Program.

Garrett, K.L.; Raphael, M.G.; Dixon, R.D. 1996. White-headed woodpecker. 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. 24 p.

Gawlik, D.E.; Bildstein, K.L. 1993. Seasonal habitat use and abundance of loggerhead shrikes in South Carolina. Journal of Wildlife Management. 57(2): 352-357.

Genter, D.L. 1986. Wintering bats of the upper Snake River plain: occurrence in lava-tube caves. Great Basin Naturalist. 46: 241-244.

Gerber, Mark F.; Munger, James C.; Ames, Aaron; Stewart, Jeff. 1997. Reptiles and amphibians in deep canyons: the Big Jacks and Little Jacks Creek drainages of Owyhee County, Idaho. Tech. Bull. 97-1. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office. 14 p.

Getz, L.L. 1961. Hunting areas of the long-eared owl. Wilson Bulletin. 73(1): 79-82.

Giesen, K.M.; Connelly, J.W. 1993. Guidelines for management of Columbian sharp-tailed grouse habitats. Wildlife Society Bulletin. 21(3): 325-333.

Gleason, R.S.; Johnson, D.R. 1985. Factors influencing nesting success of burrowing owls (*Athene cunicularia*) in southeastern Idaho. Great Basin Naturalist. 45(1): 81-84.

Goggans, R. 1986. Habitat use by flammulated owls in northeastern Oregon. Corvallis, OR: Oregon State University. 54 p. M.S. thesis.

Goggans, Rebecca; Dixon, Rita D.; Seminara, Claire. 1988. Habitat use by three-toed and black-backed woodpeckers, Deschutes National Forest, Oregon. Nongame Rep. 87-3-02. Portland, OR: Oregon Department of Fish and Wildlife; Deschutes National Forest, Nongame Wildlife Program. 49 p.

Golightly, R.T.; Ohmart, R.D. 1984. Metabolism and body temperature of two desert canids: coyotes and kit foxes. Journal of Mammalogy. 64(4): 624-635.

Gray, R.H.; Rickard, W.H. 1989. The protected area of Hanford as a refugium for native plants and animals. Environmental Conservation. 16: 215-216, 251-262.

Green, G.A.; Anthony, R.G. 1989. Nesting success and habitat relationships of burrowing owls in the Columbia Basin, Oregon. Condor. 91(2): 347-354.

Green, Gerald I.; Mattson, David J.; Peek, James M. 1997. Spring feeding on ungulate carcasses by grizzly bears in Yellowstone National Park. Journal of Wildlife Management. 61(4): 1040-1055.

Gregg, M.A. 1991. Use and selection of nesting habitat by sage grouse in Oregon. Corvallis, OR: Oregon State University. M.S. thesis.

Griffin, Daude N. 1959. The poisoning of meadowlarks with insecticides. Wilson Bulletin. 71(2): 193.

Grubb, T.C., Jr.; Yosef, R. 1994. Habitat-specific nutritional condition in loggerhead shrikes (*Lanius ludovicianus*): evidence from ptilochronology. Auk. 111(3): 756-759.

Hall, E.R. 1928. Notes on the life history of the sagebrush meadow mouse (*Lagurus*). Journal of Mammalogy. 9: 201-204.

Hamer, D.; Herrero, S. 1987. Grizzly bear food and habitat in the Front Ranges of Banff National Park, Alberta. International Conference on Bear Research and Management. 7: 199-214.

Hamilton, K.S.; Holl, S.K.; Douglas, C.L. 1982. An evaluation of the effects of recreational activity on bighorn sheep in the San Gabriel Mountains, California. Transactions of the annual meeting of the Desert Bighorn Council. 26: 50-55.

Hanley, T.A.; Page, J.L. 1982. Differential effects of livestock use on habitat structure and rodent populations in Great Basin communities. California Fish and Game. 68(3): 160-174.

Hann, Wendel J.; Jones, Jeffrey L.; Karl, Michael G. [and others]. 1997. Landscape dynamics of the basin. In: Quigley, Thomas M.; Arbelbide, Sylvia J., tech. eds. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 337-1055. Chapter 3.

Hargis, C.D. 1996. The influence of forest fragmentation and landscape pattern on American martens and their prey. Logan, UT: Utah State University. 154 p. Ph.D. dissertation.

Hargis, Christina D.; McCarthy, Clinton; Perloff, Richard D. 1994. Home ranges and habitats of northern goshawks in eastern California. In: The northern goshawk: ecology and management: Proceedings of a symposium of the Cooper Ornithological Society; 1993 April 14-15; Sacramento, CA. In: Studies in Avian Biology. Cooper Ornithological Society; 16: 66-75.

Harper, Kimball T.; Pendleton, Rosemary L. 1993. Cyanobacteria and cyanolichens: can they enhance availability of essential minerals for higher plants? Great Basin Naturalist. 53: 59-72.

Harrington-Tweit, Bill; Mattocks, Philip W., Jr. 1985. The nesting season June 1-July 31: northern Pacific Coast region. American Birds. 39(5): 953-956.

Hash, H.S. 1987. Wolverine. In: Novak, M.; Baker, J.A.; Obbard, M.E., eds. Wild furbearer management and conservation in North America. Toronto: Ontario Ministry of Natural Resources: 575-585.

Haug, E.A.; Oliphant, L.W. 1990. Movements, activity patterns, and habitat use of burrowing owls in Saskatchewan. Journal of Wildlife Management. 54(1): 27-35.

Hayward, G.D.; Escano, R.E. 1989. Goshawk nest-site characteristics in western Montana and northern Idaho. Condor. 91(2): 476-479.

Hayward, G.D.; Hayward, P.H.; Garton, E.O. 1993. Ecology of boreal owls in the northern Rocky Mountains. Wildlife Monographs. 124: 1-59.

Hayward, Gregory D. 1994a. Conservation status of boreal owls in the United States. In: Hayward, G.D.; Verner, J., tech. eds. 1994. Flammulated, boreal, and great gray owls in the United States: a technical conservation assessment. Gen. Tech. Rep. RM-253. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 139-153.

Hayward, Gregory D. 1994b. Conservation status of great gray owls in the United States. In: Hayward, G.D., Verner, J., tech. eds. 1994. Flammulated, boreal, and great gray owls in the United States: a technical conservation assessment. Gen. Tech. Rep. RM-253. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 202-206.

Hayward, Gregory D. 1994c. Review of technical knowledge: boreal owls. In: Hayward, G.D.; Verner, J., tech. eds. 1994. Flammulated, boreal, and great gray owls in the United States: a technical conservation assessment. Gen. Tech. Rep. RM-253. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 92-127.

Hayward, Gregory D. 1997. Forest management and conservation of boreal owls in north America. Journal of Raptor Research. 31(2): 114-124.

Heinemeyer, K.S. 1995. Conservation assessment for fisher (*Martes pennanti*) in Idaho. In: Idaho Department of Fish and Game; Nez Perce Tribe; Sawtooth National Forest. Forest carnivores in Idaho-Habitat conservation assessments (HCAs) and conservation strategies (CS). Idaho State Conservation Effort; Idaho Department of Fish and Game; Idaho Department of Parks and Recreation; Bureau of Land Management; U.S. Department of Agriculture, Forest Service, Northern Region and Intermountain Region; U.S. Department of the Interior, Fish and Wildlife Service: 28-41. Draft report dated April 21, 1995. 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.

Hejl, S.J. 1994. Human-induced changes in bird populations in coniferous forests in western North America during the past 100 years. In: Studies in Avian Biology. Cooper Ornithological Society; 15: 232-246.

Helms, B. 1978. Antelope and energy development. In: Proceedings of the 8th biennial pronghorn antelope workshop; 1978 May 2-4; Jasper, AB. Jasper, AB: Alberta Recreation, Parks and Wildlife, Fish and Wildlife Division: 206-215.

Henny, C.J.; Blus, L.J.; Kaiser, T.E. 1984. Heptachlor seed treatment contaminates hawks, owls, and eagles of Columbia basin, Oregon. Raptor Research. 18: 41-48.

Herkert, J.R. 1994. The effects of habitat fragmentation on Midwestern grassland bird communities. Journal of Ecological Applications. 4: 461-471.

Hessburg, Paul F.; Smith, Bradley G.; Kreiter, Scott D. [and others]. 1999. Historical and current forest and range landscapes in the interior Columbia River basin and portions of the Klamath and Great Basins. Part I: 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. (Quigley, Thomas M., tech. ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Hicks, Lorin L.; Elder, James M. 1979. Human disturbance of Sierra Nevada bighorn sheep. Journal of Wildlife Management. 43(4): 909-915.

Hines, James E. 1986. Social organization, movements, and home ranges of blue grouse in fall and winter. Wilson Bulletin. 98: 419-432.

Hjeljord, O. 1973. Mountain goat forage and habitat preference in Alaska. Journal of Wildlife Management. 37(3): 353-362.

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(4): 593-600.

Hoffman, N. 1997. Distribution of *Picoides* woodpeckers in relation to habitat disturbance within the Yellowstone area. Bozeman, MT: Montana State University. 74 p. M.S. thesis.

Hoffman, Stephen W.; DeRagon, William R.; Bednarz, James C. 1992. Patterns and recent trend in counts of migrant hawks in western North America. Washington, DC: U.S. Fish and Wildlife Service, Office of Migratory Bird Management. Unpublished report. On file with: Hawkwatch International, P.O. 35706, Suite 206, Albuquerque, NM 87176.

Hogstad, O. 1976. Sexual dimorphism and divergence in winter foraging behaviour of three-toed woodpeckers, *Picoides tridactylus*. Ibis. 118(1): 41-50.

Holt, Denver W. 1997. The long-eard owl (*Asio otus*) and forest management: a review of the literature. Journal of Raptor Research. 32: 175-186.

Horning, John. 1994. Grazing to extinction: endangered, threatened and candidate species imperiled by livestock grazing on western public lands. Washington, DC: National Wildlife Federation. 68 p.

Hornocker, M.G.; Hash, H.S. 1981. Ecology of the wolverine in northwestern Montana. Canadian Journal of Zoology. 59(7): 1286-1301.

Hosten, P.E.; West, N.E. 1994. Cheatgrass dynamics following wildfire on a sagebrush semidesert site in central Utah. In: Monsen, Stephen B.; Kitchen, Stanley G., eds. Proceedings, ecology and management of annual rangelands; 1992 May 18-21; Boise, ID. Gen. Tech. Rep. INT-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 56-68.

Humphrey, S.R. 1975. Nursery roosts and community diversity of nearctic bats. Journal of Mammalogy. 56(2): 321-346.

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.

Hupp, J.W. 1987. Sage grouse resource exploitation and endogenous reserves in Colorado. Fort Collins, CO: Colorado State University. 73 p. Ph.D. dissertation.

Hutto, R.L. 1995. Composition of bird communities following stand-replacement fires in northern Rocky Mountain conifer forests. Conservation Biology. 9(5): 1041-1058.

Idaho State Conservation Effort. 1995. Habitat conservation assessment and conservation strategy for the Townsend's big-eared bat. Unpublished report. Boise, ID: Idaho Department of Fish and Game. 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.

Interagency Grizzly Bear Committee. 1986. Interagency grizzly bear guidelines. 99 p. 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.

Interagency Grizzly Bear Committee. 1998. Grizzly update: news on grizzly bear recovery activities in the states of Wyoming, Montana, Idaho, and Washington from the Interagency Grizzly Bear Committee. 12 p. Unpublished report. On file with: Missoula, MT: U.S. Department of the Interior, Fish and Wildlife Service, Interagency Grizzly Bear Committee, University Hall, Room 309, University of Montana, Missoula, MT 59812.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996a. Expert panel notes for amphibians and reptiles terrestrial habitat assessment, fall 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996b. Expert panel notes for amphibian and reptile viability assessment, spring 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996c. Expert panel notes for bats and small mammals terrestrial habitat assessment, fall 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996d. Expert panel notes for bats and small mammals viability assessment, spring 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996e. Expert panel notes for carnivore terrestrial habitat assessment, fall 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996f. Expert panel notes for mammalian carnivore and ungulate viability assessment, spring 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996g. Expert panel notes for passerine and other birds viability assessment, spring 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996h. Expert panel notes for raptor and upland game bird viability assessment, spring 1996, for the interior Columbia basin. Unpublished notes. 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.

Interior Columbia Basin Ecosystem Management Project (ICBEMP). 1996i. Expert panel notes for ungulate terrestrial habitat assessment, fall 1996, for the interior Columbia basin. Unpublished notes. 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.

Janes, S.W. 1983. Status, distribution, and habitat selection of the grasshopper sparrow in Morrow County, Oregon. Murrelet. 64: 51-54.

Jasikoff, Thomas M. 1982. Habitat suitability index models: ferruginous hawk. FWS/OBS-82/10.10. Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service. 18 p.

Jensen, W.R.; Fuller, T.K.; Robinson, W.L. 1986. Wolf, *Canis lupus*, distribution on the Ontario-Michigan border near Sault Ste. Marie. Canadian Field-Naturalist. 100: 363-366.

Jewett, S.G.; Taylor, W.P.; Shaw, W.T.; Aldrich, J.W. 1953. Birds of Washington State. Seattle: University of Washington Press. 767 p.

Johansen, Jeffrey R.; Ashley, John; Rayburn, William R. 1993. Effects of rangefire on soil algal crusts in semiarid shrub-steppe of the lower Columbia basin and their subsequent recovery. Great Basin Naturalist. 53: 73-88.

Johnson, D.H.; Igl, L.D. 1995. Contributions of the Conservation Reserve Program to populations of breeding birds in North Dakota. Wilson Bulletin. 107: 709-718.

Johnson, R.E. 1977. An historical analysis of wolverine abundance and distribution in Washington, USA. Murrelet. 58: 13-16.

Johnson, Rolf L. 1983. Mountain goats and mountain sheep of Washington. Biol. Bull. No. 18. Olympia, WA: Washington State Game Department. 196 p.

Jones, Jeffrey L. 1991. Habitat use of fisher in north central Idaho. Moscow, ID: University of Idaho. 147 p. M.S. thesis.

Jones, Jeffrey L.; Garton, Edward O. 1994. Selection of successional stages by fishers in north-central Idaho. In: Buskirk, S.W., Harestad, A.; Raphael, M.; Powell, R.A., eds. Martens, sables, and fishers: biology and conservation. Ithaca, NY: Cornell University: 377-387.

Jonkel, Charles J.; Cowan, Ian McT. 1971. The black bear in the spruce-fir forest. Wildlife Monograph. No. 27. 57 p.

Joslin, G. 1986. Mountain goat population changes in relation to energy exploration along Montana's Rocky Mountain Front. In: Proceedings of the 5th biennial symposium of the Northern Wild Sheep and Goat Council; 1986 April 14-17; Missoula, MT. Thermopolis, WY: Northern Wild Sheep and Goat Council. 5: 253-271. Proceedings available from: Foundation for North American Wild Sheep, 720 Allen Avenue, Cody, WY 82414.

Kaltenecker, J.H. 1997. The recovery of microbiotic crusts following postfire rehabilitation on rangelands of the western Snake River plain. Boise, ID: Boise State University. 99 p. M.S. thesis.

Kaltenecker, J.; Wicklow-Howard, M. 1994. Microbiotic soil crusts in sagebrush habitats of southern Idaho. 48 p. (plus an appendix). Prepared for Eastside Ecosystem Management Project. 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.

Kasworm, Wayne F.; Manley, T.L. 1990. Road and trail influences on grizzly bears and black bears in northwest Montana. International Conference on Bear Research and Management. 8: 79-84.

Keister, George P.; Immell, David. 1994. Continued investigations of kit fox in southeastern Oregon and evaluation of status. Tech. Rep. 94-5-01. Portland, OR: Oregon Department of Fish and Wildlife, Wildlife Diversity Program. 33 p.

Keith, L.B.; Cary, J.R.; Rongstad, O.J.; Brittingham, M.C. 1984. Demography and ecology of a declining snowshoe hare population. Wildlife Monographs 90. 43 p.

Kennedy, Ann C. 1994. Biological control of annual grass weeds. In: Monsen, Stephen B.; Kitchen, Stanley G., eds. Proceedings, ecology and management of annual rangelands; 1992 May 18-21; Boise, ID. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 186-189.

Kennedy, Patricia L.; Ward, Johanna M.; Rinker, George A.; Gessaman, James A. 1994. Post-fledging areas in northern goshawk home ranges. In: The northern goshawk: ecology and management: Proceedings of a symposium of the Cooper Ornithological Society; 1993 April 14-15; Sacramento, CA. In: Studies in Avian Biology. Cooper Ornithological Society; 16: 75-82.

Kie, J.G. 1991. Wildlife and livestock grazing alternatives in the Sierra Nevada. In: Transactions of the Western Section of the Wildlife Society; 1991 February 7-9; Sacramento, CA. Sacramento, CA: The Western Section of the Wildlife Society. 27: 17-29.

Klebenow, D.A. 1969. Sage grouse nesting and brood habitat in Idaho. Journal of Wildlife Management. 33: 649-661.

Klemmedson, James O.; Smith, Justin G. 1964. Cheatgrass (*Bromus tectorum* L.). The Botanical Review. 30: 226-262.

Knapton, R.W. 1979. Optimal size of territory in the clay-colored sparrow, *Spizella pallida*. Canadian Journal of Zoology. 57(7): 1358-1370.

Knapton, Richard W. 1994. Clay-colored sparrow (*Spizella pallida*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 120. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union. 16 p.

Knick, S.T.; Rotenberry, J.T. 1995. Landscape characteristics of fragmented shrubsteppe habitats and breeding passerine birds. Conservation Biology. 9(5): 1059-1071.

Knick, Steven T.; Kasworm, Wayne. 1989. Shooting mortality in small populations of grizzly bears. Wildlife Society Bulletin. 17(1): 11-15.

Knight, R.R.; Blanchard, B.M.; Eberhardt, L.L. 1988. Mortality patterns and population sinks for Yellowstone grizzly bears, 1973-1985. Wildlife Society Bulletin. 16: 121-125.

Koehler, C.E.; Barclay, R.M.R. 1988. The potential for vocal signatures in the calls of young hoary bats (*Lasiurus cinereus*). Canadian Journal of Zoology. 66(9): 1982-1985.

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.; Brittell, J.D. 1990. Managing spruce fir habitat for lynx and snowshoe hares. Journal of Forestry. 88: 10-14.

Koehler, Gary M.; Aubry, Keith B. 1994. Lynx. In: Ruggiero, L.; Aubry, K.; Buskirk, S. [and others], tech. eds. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the Western United States. Gen. Tech. Rep. RM-254. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 74-98. Chapter 4.

Koplin, James R. 1969. The numerical response of woodpeckers to insect prey in a subalpine forest in Colorado. Condor. 71(4): 436-438.

Kovalchik, B.L.; Elmore, W. 1992. Effects of cattle grazing systems on willow-dominated plant associations in central Oregon. In: Symposium on ecology and management of riparian shrub communities; 1991 May 29-31; Sun Valley, ID. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 111-119.

Kuck, L. 1977. The impact of hunting on Idaho's Pahsimeroi mountain goat herd. In: Samuel, W.; MacGregor, W.G., eds. Proceedings of the 1st International Mountain Goat Symposium. 1: 114-125.

Lanyon, W. E. 1994. Western meadowlark (*Sturnella neglecta*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 104. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists'Union. 20 p.

Lauer, J.L.; Peek, J.M. 1976. Big game-livestock relationships on the bighorn sheep winter range, East Fork Salmon River, Idaho. Bull. No. 12. Moscow, ID: College of Forestry, Wildlife, and Range Science. 44 p.

Lee, Danny C.; Sedell, James R.; Rieman, Bruce E. [and others]. 1997. Broadscale assessment of aquatic species and habitats. In: Quigley, Thomas M.; Arbelbide, Sylvia J., tech. eds. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 1057-1496. Vol. 3. (Quigley, Thomas M., tech. ed. Interior Columbia Basin Ecosystem Management Project: scientific assessment).

LeFranc, Maurice N., Jr.; Moss, Mary Beth; Patnode, Kathleen; Sugg, William C., III, eds. 1987. Grizzly bear compendium. Washington, DC: Interagency Grizzly Bear Committee and the National Wildlife Federation. 540 p.

Lehmkuhl, John F.; Raphael, Martin G.; Holthausen, Richard S. [and others]. 1997. Historical and current status of terrestrial species and the effects of proposed alternatives. In: Quigley, Thomas M.; Lee, Kristine M.; Arbelbide, Sylvia J., tech. 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. Vol. 2.

Leopold, A.S.; Gutierrez, R.J.; Bronson, M.T. 1981. North American game birds and mammals. New York: Charles Scribner's Sons. 198 p.

Lester, Amy N. 1980. Numerical response of woodpeckers and their effect on mortality of mountain pine beetles in lodgepole pine in northwestern Montana. Missoula, MT: University of Montana. 103 p. M.S. thesis.

Leu, Matthias. 1995. The feeding ecology and the selection of nest shrubs and fledgling roost sties by loggerhead shrikes (*Lanius ludovicianus*) in the shrub-steppe habitat. Seattle: University of Washington. 90 p. M.S. thesis.

Linden, H.; Wikman, M. 1983. Goshawk predation on tetraonids: availability of prey and diet of the predator in the breeding season. Journal of Animal Ecology. 52: 953-968.

Linder, K.A. 1994. Habitat utilization and behavior of nesting Lewis' woodpeckers (*Melanerpes lewis*) in the Laramie Range, southeast Wyoming. Laramie, WY: University of Wyoming. 98 p. M.S. thesis.

Linkhart, B.D. 1984. Range, activity, and habitat use by nesting flammulated owls in a Colorado ponderosa pine forest. Fort Collins, CO: Colorado State University. M.S. thesis.

Lokemoen, J.T.; Duebbert, H.F. 1976. Ferruginous hawk nesting ecology and raptor populations in northern South Dakota. Condor. 78(4): 464-470.

Lyman, R.L. 1991. Late quaternary biogeography of the pygmy rabbit (*Brachylagus idahoensis*) in eastern Washington. Journal of Mammology. 72(1): 110-117.

MacArthur, R.A.; Johnson, 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.

MacCracken, J.G.; Alexander, L.E.; Uresk, D.W. 1983. An important lichen of southeastern Montana rangelands. Journal of Range Management. 36: 35-37.

Mace, Richard D.; Waller, John S.; Manley, Timothy L. [and others]. 1996. Relationships among grizzly bears, roads and habitat in the Swan Mountains, Montana. Journal of Applied Ecology. 33: 1395-1404.

Mack, R.N.; Thompson, J. N. 1982. Evolution in steppe with few large, hooved mammals. American Naturalist. 119: 757-773.

Maj, M.; Hillis, M.; Ormiston, J. 1995. Range permit reissuance-flammulated owl. 5 p. Unpublished memo. On file with: U.S. Department of Agriculture, Forest Service, Northern Region, P.O. Box 7669, Missoula, MT 59807.

Mannan, R.W. 1984. Summer area requirements of pileated woodpeckers in western Oregon. Wildlife Society Bulletin. 12(3): 265-268.

Manning, R.W.; Knox-Jones, J. 1989. *Myotis evotis*. Mammalian Species. No. 329. Baltimore, MD: American Society of Mammalogists. 5 p.

Marcot, B.G.; Wales, B.; Demmer, R. [In prep.]. Range maps of terrestrial species and communities in the interior Columbia River basin and northern portions of the Klamath and Great basins. Gen. Tech. Rep. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Marcot, Bruce G.; Castellano, Michael A.; Christy, John A. [and others]. 1997. Terrestrial ecology of the basin. In: Quigley, Thomas M.; Arbelbide, Sylvia J., tech. eds. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 1497-1713. Vol. 3. (Quigley, Thomas M., tech. ed. Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Marks, J.S. 1986. Nestsite characteristics and reproductive success of long-eared owls in southwestern Idaho. Wilson Bulletin. 98(4): 547-560.

Marks, J.S.; Evans, D.L.; Holt, D.W. 1994. Long-eared owl (*Asio otus*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 133. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists'Union. 24 p.

Marks, J.S.; Saab, V. 1987a. Habitat selection by Columbian sharp-tailed grouse in west-central Idaho. Boise, ID: U.S. Department of the Interior, Bureau of Land Management. 115 p.

Marks, J.S.; Saab Marks, V. 1987b. Influence of radio collars on survival of sharp-tailed grouse. Journal of Wildlife Management. 51(2): 468-471.

Marks, J.S.; Saab Marks, V. 1988. Winter habitat use by Columbian sharp-tailed grouse in western Idaho. Journal of Wildlife Management. 52(4): 743-746.

Marshall, D.B.; Chilcote, M.W.; Weeks, H. 1996. Species at risk: sensitive, threatened and endangered vertebrates of Oregon. 2d ed. Portland, OR: Oregon Department Fish and Wildlife. 1 vol.

Marshall, David B. 1992. Status of the black-backed woodpecker in Oregon and Washington. In: Proceedings of the Blue Mountains biodiversity conference; 1992 May 26-29; Portland, OR. Portland, OR: Portland Audubon Society. 13 p.

Marshall, J.T., Jr. 1988. Birds lost from a giant sequoia forest during fifty years. Condor. 90: 359-372.

Marti, C.D.; Marks, J.S. 1989. Medium-sized owls. In: Proceedings of the western raptor management symposium and workshop; 1987 October 26-28; [Meeting location unknown]. Boise, ID. National Wildlife Federation Science and Technical Series. 12: 124-133.

Martin, Kevin D.; Schommer, Tim; Coggins, Victor. 1996. Literature review regarding the compatibility between bighorn and domestic sheep. In: Proceeding of the 10th biennial symposium of the Northern Wild Sheep and Goat Council; 1996 April 29-May 3; Silverthorne, CO. Thermopolis, WY: Northern Wild Sheep and Goat Council. 10: 72-77.

Martin, S.J. 1983. Burrowing owl occurrence on white-tailed prairie dog colonies. Journal of Field Ornithology. 54(4): 422-423.

Maser, C.; Hammer, E.W.; Brown, C. [and others]. 1974. The sage vole, *Lagurus curtatus*, in the Crooked River National Grassland, Jefferson County, Oregon: a contribution to its life history and ecology. Saugetierkundliche Mitteilungen. 22: 193-222.

Maser, C.; Strickland, G.S. 1978. The sage vole, *Lagurus curtatus*, as an inhabitant of subalpine sheep fescue *Festuca ovina* communities on Steens Mountain: an observation and interpretation. Northwest Science. 52: 276-284.

Matthews, Patrick E.; Coggins, Victor L. 1994. Status and history of mountain goats in Oregon. In: Proceedings of the 9th biennial symposium of the Northern Wild Sheep and Goat Council; 1994 May 2-6; Cranbrook, British Columbia. Thermopolis, WY: Northern Wild Sheep and Goat Council. 9: 69-74. Proceedings available from: Foundation for North American Wild Sheep, 720 Allen Avenue, Cody, WY 82414.

Mattson, D.J.; Blanchard, B.M.; Knight, R.R. 1991a. Food habits of Yellowstone grizzly bears, 1977-1987. Canadian Journal of Zoology. 69: 1619-1629.

Mattson, D.J.; Reinhart, D.P. 1997. Excavation of red squirrel middens by grizzly bears in the whitebark pine zone. Journal of Applied Ecology. 34: 926-940.

Mattson, David J.; Blanchard, Bonnie M.; Knight, Richard R. 1992. Yellowstone grizzly bear mortality, human habituation, and whitebark pine seed crops. Journal of Wildlife Management. 56(3): 432-442.

Mattson, David J.; Gillin, Colin M.; Benson, Scott A.; Knight, Richard R. 1991b. Bear feeding activity at alpine insect aggregation sites in the Yellowstone ecosystem. Canadian Journal of Zoology. 69: 2430-2435.

Mattson, David J.; Herrero, Stephen; Wright, Gerald; Pease, Craig M. 1996a. Designing and managing protected areas for grizzly bears: how much is enough? In: Wright, R.G., ed. National parks and protected areas: their role in environmental protection. Cambridge, MA: Blackwell Science: 133-164.

Mattson, David J.; Herrero, Stephen; Wright, Gerald; Pease, Craig M. 1996b. Science and management of Rocky Mountain grizzly bears. Conservation Biology. 10(4): 1013-1025.

Mattson, David J.; Knight, Richard R.; Blanchard, Bonnie M. 1987. The effects of developments and primary roads on grizzly bear habitat use in Yellowstone National Park, Wyoming. International Conference on Bear Research and Management. 7: 259-273.

McCallum, D.A. 1994. Conservation status of flammulated owls in the United States. In: Hayward, G.D.; Verner, J., eds. Flammulated, boreal, and great gray owls in the United States: a technical conservation assessment. Gen. Tech. Rep. RM-253. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 74-79. Chapter 6.

McCallum, D.A.; Gehlbach, F.R. 1988. Nest-site preferences of flammulated owls in western New Mexico. Condor. 90(3): 653-661.

McCool, Stephen F.; Burchfield, J.A.; Allen, Stewart. 1997. Social assessment of the basin. In: Quigley, Thomas M.; Arbelbide, Sylvia J., tech. eds. An assessment of ecosystem components in the interior Columbia basin including portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 1872-2009. Chapter 7. Vol. 4. (Quigley, Thomas M., tech. ed. Interior Columbia Basin Ecosystem Management Project: scientific assessment).

McEllin, S.M. 1979. Nest sites and population demographies of white-breasted and pygmy nuthatches in Colorado. Condor. 81(4): 348-352.

McKelvey, Kevin S.; Aubry, Keith B.; Ortega, Yvette K. 1999. History and distribution of lynx in the contiguous United States. In: Ruggiero, Leonard F.; Aubry, Keith B.; Buskirk, Steven W. [and others]. 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: 207-264. Chapter 8.

McLean, D.D. 1930. The quail of California. California Division of Fish and Game. Game Bull. 2: 1-47.

McLellan, B.N.; Shackleton, D.M. 1988. Grizzly bears and resource-extraction industries: effects of roads on behaviour, habitat use, and demography. Journal of Applied Ecology. 25: 451-460.

McLellan, Bruce N.; Hovey, Fred W. 1995. The diet of grizzly bears in the Flathead River drainage of south eastern British Columbia. Canadian Journal of Zoology. 73: 704-712.

McWhirter, Doug; Smith, Scott; Merrill, Evelyn; Irwin, Larry. 1992. Foraging behavior and vegetation responses to prescribed burning on bighorn sheep winter range. In: Proceedings of the 8th biennial symposium of the Northern Wild Sheep and Goat Council; 1992 April 27-May 1; Cody, WY. Thermopolis, WY: Northern Wild Sheep and Goat Council. 8: 264-278. Proceedings available from: Foundation for North American Wild Sheep, 720 Allen Avenue, Cody, WY 82414.

Meagher, Mary; Fowler, Sandi. 1989. The consequences of protecting problem grizzly bears. In: Bromley, M., ed. Bear-people conflicts: Proceedings of a symposium on management strategies; [Dates of meeting unknown]; [Meeting location unknown]. Yellowknife, NWT: Northwest Territories Department of Renewable Resources: 141-144.

Mech, L.D. 1970. Implications of wolf ecology to management. In: Jorgensen, S.E.; Faulkner, L.E.; Mech, L.D., eds. Proceedings of symposium on wolf management in selected areas of North America; [Meeting dates unknown]; [Meeting location unknown]. [Place of publication unknown]: U.S. Department of the Interior, Fish and Wildlife Service: 39-44.

Mech, L.D. 1980. Age, sex, reproduction, and spatial organization of lynxes colonizing northeastern Minnesota. Journal of Mammology. 61: 261-267.

Mech, L.D.; Fritts, S.H.; Radde, G.L.; Paul, W.J. 1988. Wolf distribution and road density in Minnesota. Wildlife Society Bulletin. 16: 85-87.

Meints, D.R. 1991. Seasonal movements, habitat use, and productivity of Columbian sharp-tailed grouse in southeastern Idaho. Moscow, ID: University of Idaho. 74 p. M.S. Thesis.

Meints, D.R.; Connelly, J.W.; Reese, K.P. [and others]. 1992. Habitat suitability index procedure for Columbian sharp-tailed grouse. Idaho Forest Wildlife and Range Experiment Station Bulletin. Moscow, ID: University of Idaho, College of Forestry and Range Sciences. (vi): 1-27.

Merigliano, Michael. F. 1996. Ecology and management of the south fork Snake River cottonwood forest. Tech. Bull. 96-9. Boise, ID: U.S. Department of the Interior, Bureau of Land Management. 79 p.

Merrill, Troy; Mattson, David J.; Wright, R. Gerald; Quigley, Howard B. 1999. Defining landscapes suitable for restoration of grizzly bears *Ursus arctos* in Idaho. Biological Conservation. 87(2): 231-248.

Mikkola, H. 1983. Owls of Europe. Vermillion, SD: Buteo Books. 397 p.

Milne, K.A.; Hejl, S.J. 1989. Nest-site characteristics of white-headed woodpeckers. Journal of Wildlife Management. 53(1): 50-55.

Mladenoff, David J.; Sickley, Theodore A.; Haight, Robert G.; Wydeven, Adrian P. 1995. A regional landscape analysis and prediction of favorable gray wolf habitat in the northern Great Lakes region. Conservation Biology. 9(2): 279-294.

Monsen, Stephen B.; McArthur, E. Durant. 1995. Implications of early Intermountain range and watershed restoration practices. In: Roundy, Bruce A.; McArthur, E. Durant; Haley, Jennifer S.; Mann, David K., eds. Proceedings, wildland shrub and arid land restoration symposium; 1993 October 19-21; Las Vegas, NV. Gen. Tech. Rep. INT-315. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 16-25.

Moore, K.R.; Henny, C.J. 1983. Nest site characteristics of three coexisting accipiter hawks in northeastern Oregon. Raptor Research. 17: 65-76.

Moroz, Paul. 1995. Idaho ground squirrel, *Spermophilus brunneus brunneus*, habitat conservation assessment (HCA), conservation strategy (CS). Idaho Interagency Conservation/Prelisting Effort. Idaho Department of Fish and Game; Idaho Department of Parks and Recreation; U.S. Department of the Interior, Fish and Wildlife Service and Bureau of Land Management; U.S. Department of Agriculture, Forest Service. 26 p. Draft 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.

Morrison, Michael L. 1982. The structure of western warbler assemblages: ecomorphological analysis of the black-throated gray and hermit warblers. Auk. 99(3): 503-513.

Morrow, Larry A.; Stahlman, Phillip W. 1984. The history and distribution of downy brome (*Bromus tectorum*) in North America. Weed Science. 32 (Supplement 1): 2-6.

Mosconi, S.L.; Hutto, R.L. 1981. The effect of grazing on land birds of a western Montana riparian habitat. In: Peek, J.M.; Dalke, P.D., eds. Symposium on wildlife-livestock interactions; [Date of meeting unknown]; [Location of meeting unknown]. Moscow, ID: University of Idaho: 221-233.

Mullican, T.R.; Keller, B.L. 1986. Ecology of the sagebrush vole (*Lemmiscus curtatus*) in southeastern Idaho. Canadian Journal of Zoology. 64(6): 1218-1223.

Murray, D.L.; Boutin, S. 1991. The influence of snow on lynx and coyote movements: does morphology affect behavior? Oecologia. 88: 463-469.

Murray, T.B. 1938. Upland game birds in Idaho and their future. University of Idaho Bulletin. 33: 55-60.

Mussehl, T.W. 1963. Blue grouse brood cover selection and land use implications. Journal of Wildlife Management. 27: 547-556.

Nagorsen, D.W.; Brigham, R.M. 1993. Bats of British Columbia. Vancouver, BC: UBC Press. 164 p.

Nellis, Carl H.; Wetmore, Stephen P.; Keith, Lloyd B. 1972. Lynx-prey interactions in central Alberta. Journal of Wildlife Management. 36(2): 320-329.

Nelson, Edward W. 1925. Status of the pronghorned antelope, 1922-1924. Bull. 1346. Washington, DC: U.S. Department of Agriculture. 64 p.

Nero, R.W. 1980. The great gray owl-phantom of the northern forest. Washington, DC: Smithsonian Institution Press. 167 p.

Newby, R.E.; McDonald, J.J. 1964. Range extension of the wolverine in Montana. Journal of Mammalogy. 45: 485-486.

Newby, R.E.; Wright, P.L. 1955. Distribution and status of the wolverine in Montana. Journal of Mammalogy. 36: 248-253.

Nice, M.M. 1957. Nesting success in altricial birds. Auk. 74(3): 305-321.

Niemuth, N. 1992. Use of man-made structures by nesting ferruginous hawks in Wyoming. Prairie Naturalist. 24(1): 43.

Noss, R.F.; LaRoe, E.T., III; Scott, J.M. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Biological Report 28. Washington, DC: National Biological Service.

Noss, Reed F.; Quigley, Howard B.; Hornocker, Maurice G. [and others]. 1996. Conservation biology and carnivore conservation in the Rocky Mountains. Conservation Biology. 10(4): 949-963.

Nussbaum, Ronald A.; Brodie, Edmund D., Jr.; Storm, Robert M. 1983. Amphibians and reptiles of the Pacific Northwest. Moscow, ID: University of Idaho Press. 332 p.

Oakleaf, R.J. 1971. The relationship of sage grouse to upland meadows in Nevada. Federal Aid Project W-28-2. [Place of publication unknown]: Nevada Department of Fish and Game. 64 p.

Oakley, C.; Riddle, P. 1974. The impact of a severe winter and fences on antelope mortality in south central Wyoming. In: Proceedings of the 6th antelope states workshop; 1974 February 19-21; Salt Lake City, UT. [Place of publication unknown]: [Publisher unknown]: 155-171.

Oedekoven, O.O. 1985. Columbian sharp-tailed grouse population distribution and habitat use in south central Wyoming. Laramie, WY: University of Wyoming. 58 p. M.S. thesis.

O'Farrell, M.J.; Studier, E.H. 1980. *Myotis thysanodes*. Mammalian Species. No. 137. Baltimore, MD: American Society of Mammalogists. 5 p.

O'Gara, B.W. 1996. Estimates of pronghorn numbers. Unpublished report. On file with: School of Forestry, University of Montana, Missoula, MT 59812. 1 p.

Ogg, Alex G. 1994. A review of the chemical control of downy brome. In: Monsen, Stephen B.; Kitchen, Stanley G., eds. Proceedings, ecology and management of annual rangelands; 1992 May 18-21; Boise, ID. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 194-196.

Ognev, S.I. 1935. Mammals of USSR and adjacent countries. Vol. 3: Carnivora (Fissipedia and Finnepedia). Jerusalem, Israel. 641 p. [Translated by Israel Program for Scientific Translations. 1962].

Olendorff, R.R.; Stoddart, J.W., Jr. 1974. The potential for management of raptor populations in western grassland. In: Hamerstrom, F.N.; Harrell, B.N.; Olendorff, R.R., eds. Management of raptors: Proceedings of the conference on raptor conservation, part 4; 1973 March 22-24; Fort Collins, CO. Vermillion, SD: Raptor Research Foundation. Raptor Research Reports. 2: 44-48.

Onderka, D.K.; Wishart, W.D. 1984. A major die-off from pneumonia in southern Alberta. In: Proceedings of the 4th biennial symposium of the Northern Wild Sheep and Goat Council; 1984 April 30-May 3; Whitehorse, Yukon. Thermopolis, WY: Northern Wild Sheep and Goat Council. 4: 356-363.

Oregon Department of Fish and Wildlife. 1987. The herpetology of the Willamette Valley, Oregon. Nongame Wildlife Program Tech. Rep. 86-1-02. Portland, OR: Oregon Department of Fish and Wildlife. 79 p.

Oregon Department of Fish and Wildlife. 1994. Juniper woodland management. Portland: Oregon Department of Fish and Wildlife, Habitat Conservation Division. 26 p.

Orloff, S.; Hall, F.; Spiegel, L. 1986. Distribution and habitat requirements of the San Joaquin kit fox in the northern extreme of their range. Transactions of the Western Section of the Wildlife Society: 2260-2270.

Ormsbee, Patricia C.; McComb, William C. 1998. Selection of day roosts by female long-legged myotis in the central Oregon Cascade Range. Journal of Wildlife Management. 62(2): 596-602.

Owens, R.A.; Myres, M.T. 1973. Effects of agriculture upon populations of native passerine birds of an Alberta fescue grassland. Canadian Journal of Zoology. 51(7): 697-713.

Page, J.L.; Dodd, N.; Osbornde, T.O.; Carson, J.A. 1978. The influence of livestock grazing on non-game wildlife. CALNEVA Wildlife. [Place of publication unknown]: The Wildlife Society, Western Section: 159-173.

Parker, G.R.; Maxwell, J.W.; Morton, L.D. [and others]. 1983. The ecology of lynx (*Lynx canadensis*) on Cape Brenton Island. Canadian Journal of Zoology. 61: 770-786.

Patla, S. 1990. Northern goshawk monitoring report no. 2, 1989. [Not paged]. On file with: U.S. Department of Agriculture, Targhee National Forest, P.O. Box 208, St. Anthony, ID 83445.

Patterson, R.L. 1952. The sage grouse in Wyoming. Denver, CO: Sage Books. 341 p.

Pedivillano, Cathy; Wright, Gerald R. 1987. The influence of visitors on mountain goat activities in Glacier National Park, Montana. Biological Conservation. 39(1): 1-11.

Pekins, P.J.; Lindzey, F.G.; Gessaman, J.A. 1991. Physical characteristics of blue grouse winter use-trees and roost sites. Great Basin Naturalist. 51(3): 244-248.

Pellant, Mike. 1994. History and applications of the Intermountain greenstripping program. In: Monsen, Stephen B.; Kitchen, Stanley G., eds. Proceedings-ecology and management of annual rangelands; 1992 May 18-21; Gen. Tech. Rep. INT-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 63-68.

Perkins, J.M. 1992-1994. Results of summer bat surveys, Wallowa Valley Ranger District, Eagle Cap Ranger District, and the HCNRA of the Wallowa-Whitman National Forest, Wallowa County, OR. U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest. [Not paged]. 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.

Perkins, J.M.; Cross, S.P. 1988. Differential use of some coniferous forest habitats by hoary and silver-haired bats in Oregon. Murrelet. 69(1): 21-24.

Perkins, J.M.; Schommer, T. 1992. Survey protocol and an interim species conservation strategy for *Plecotus townsendii* in the Blue Mountains of Oregon and Washington. 23 p. 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.

Perkins, J. Mark. 1987. Distribution, status, and habitat affinities of Townsend's big-eared bat (*Plecotus townsendii*) in Oregon. Tech. Rep. 85-5-01. [Portland, OR]: Oregon Department of Fish and Wildlife, Nongame Wildlife Program.

Perkins, J. Mark. 1996. Does competition for roosts influence bat distribution in a managed forest? In: Barclay, Robert M.R.; Brigham, R. Mark, eds. Bats and forest symposium; 1995 October 19-21; [Meeting location unknown]. Victoria, BC: British Columbia Ministry of Forests Research Program: 164-172.

Perkins, J. Mark; Peterson, Joshua R. 1997. Bat distribution in the juniper woodlands of the Idaho Owyhee Mountains, summer 1996. Tech. Bull. No. 97-4. U.S. Department of the Interior, Bureau of Land Management, Idaho State Office. 12 p.

Perlmeter, S.I. 1995. Bats and bridges: a field study of the thermal conditions and social organization of night roosts in the Willamette National Forest. North York, ON: York University. 98 p. M.S. thesis.

Perritt, J.E.; Best, L.B. 1989. Effects of weather on the breeding ecology of vesper sparrows in Iowa crop fields. American Midland Naturalist. 121(2): 355-360.

Peterson, J.G. 1970. The food habits and summer distribution of juvenile sage grouse in central Montana. Journal of Wildlife Management. 34(1): 147-155.

Pierson, Elizabeth D.; Rainey, William E.; Koontz, Dolora M. 1991. Bats and mines: experimental mitigation for Townsend's big-eared bat at the McLaughlin mine in California. In: Corner, R.D.; Davis, P.R.; Foster, S.Q. [and others], eds. Proceedings V: issues and technology in the management of impacted wildlife; 1991 April 8-10; Snowmass Resort, CO. Boulder, CO: Thorne Ecological Institute: 31-42.

Pletscher, Daniel H.; Ream, Robert R.; Boyd, Diane K. [and others]. 1997. Population dynamics of a recolonizing wolf population. Journal of Wildlife Management. 61(2): 459-465.

Ports, M.A.; Bradley, P.V. 1996. Habitat affinities of bats from northeastern Nevada. Great Basin Naturalist. 56(1): 48-53.

Ports, M.A.; George, S.B. 1990. *Sorex preblei* in the northern Great Basin. Great Basin Naturalist. 50(1): 93-95.

Post, G. 1951. Effects of toxaphene and chlordane on certain game birds. Journal of Wildlife Management. 15(4): 381-386.

Pough, F.H. 1973. Lizard energetics and diet. Ecology. 54(4): 837-844.

Poulton, C.E. 1955. Ecology of the non-forested vegetation in Umatilla and Morrow Counties, Oregon. Pullman, WA: Washington State University. 166 p. Ph.D. dissertation.

Powell, RogerA.; Zielinski, William J. 1994. Fisher. In: Ruggiero, L.; Aubry, K.; Buskirk, S. [and others], tech. eds. The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the western United States. Gen. Tech. Rep. RM-254. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 38-73. Chapter 3.

Powers, L.R.; Howard, R.R.; Trost, C.H. 1973. Population status of the ferruginous hawk in southeastern Idaho and northern Utah. In: Murphy, J.R.; White, C.M.; Harrell, B.E., eds. Population status of raptors: Proceedings of the conference on raptor conservation techniques, part 6; 1973 March 22-24; Fort Collins, CO. Raptor Research Reports. 3: 153-157.

Pulliainen, E. 1968. Breeding biology of the wolverine (*Gulo gulo* L.) in Finland. Annales Zoologica Fennici. 5: 338-344.

Pulliam, H.R. 1988. Sources, sinks, and population regulation. American Naturalist. 132(5): 652-661.

Pulliam, H.R.; Danielson, B.J. 1991. Sources, sinks, and habitat selection: a landscape perspective on population dynamics. American Naturalist. 137:55-66.

Pylypec, B. 1991. Impacts of fire on bird populations in a fescue prairie. Canadian Field Naturalist. 105(3): 346-349.

Quick, H.F. 1956. Effects of exploitation on a marten population. Journal of Wildlife Management. 20(3): 267-274.

Quigley, Thomas M.; Haynes, Richard W.; Graham, Russell T., tech. eds. 1996. Integrated scientific assessment for ecosystem management in the interior Columbia basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-382. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 303 p. (Quigley, Thomas M., ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Quigley, Thomas M.; Lee, Danny C.; Haynes, Richard W. [and others]. 1997. Ecological integrity, socioeconomic resiliency, and trends in risk. In: Quigley, Thomas M.; Lee, Kristine M.; Arbelbide, Sylvia J., tech. 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: 835-896. Vol. 2.

Rabe, Michael J.; Morrell, Thomas E.; Green, Heather [and others]. 1998. Characteristics of ponderosa pine snag roosts used by reproductive bats in northern Arizona. Journal of Wildlife Management. 62(2): 612-621.

Ramakka, J.M.; Woyewodzic, R.T. 1993. Nesting ecology of ferruginous hawk in northwestern New Mexico. Journal of Raptor Research. 27(2): 97-101.

Raphael, M.G.; Jones, L.C. 1997. Characteristics of resting and denning sites of American martens in central Oregon and western Washington. In: Proulx, Gilbert; Bryant, Harold N.; Woodard, Paul M., eds. *Martes*: taxonomy, ecology, techniques, and management. Edmonton, AB: The Provincial Museum of Alberta: 146-165.

Raphael, M.G.; White, M. 1984. Use of snags by cavity-nesting birds in the Sierra Nevada. Wildlife Monographs. 86: 1-66.

Remington, T.E.; Hoffman, R.W. 1996. Food habits and preferences of blue grouse during winter. Journal of Wildlife Management. 60(4): 808-817.

Reynolds, R.T. 1983. Management of western coniferous forest habitat for nesting accipiter hawks. Gen. Tech. Rep. RM-102. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 6 p.

Reynolds, R.T.; Graham, R.T.; Reiser, M.H. [and others]. 1992. Management recommendations for the northern goshawk in the Southwestern United States. Gen. Tech. Rep. RM-217. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 184 p.

Reynolds, R.T.; Linkhart, B.D. 1990. Extra-pair copulation and extra-range movements in flammulated owls. Ornis Scandinavica. 21: 74-77.

Reynolds, R.T.; Linkhart, B.D. 1992. Flammulated owls in ponderosa pine: evidence of preference for old growth. In: Old-growth forests in the Southwest and Rocky Mountain regions: Proceedings of a workshop. Gen. Tech. Rep. RM-213. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 166-169.

Rich, T. 1981. Microgeographic variation in the song of the sage sparrow. Condor. 83(2): 113-119.

Rich, T. 1984. Monitoring burrowing owl populations: implications of burrow re-use. Wildlife Society Bulletin. 12(2): 178-180.

Rich, T. 1986. Habitat and nest-site selection by burrowing owls in the sagebrush steppe of Idaho. Journal of Wildlife Management. 50(4): 548-555.

Rich, T.D. 1978. Cowbird parasitism of sage and Brewer's sparrows. Condor. 80: 348.

Rich, T.D. 1985. Sage grouse population fluctuations: evidence for a 10-year cycle. Tech. Bull. 85-1. Boise, ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office. 29 p.

Rich, T.D.; Rothstein, S.I. 1985. Sage thrashers reject cowbird eggs. Condor. 87: 561-562.

Rickard, William H.; Poole, Linda D. 1989. Terrestrial wildlife of the Hanford Site: past and future. Northwest Science. 63(4): 183-192.

Rideout, Chester B. 1978. Mountain goat. In: Schmidt, John L.; Gilbert, Douglas L., eds. Big game of North America: ecology and management. Harrisburg, PA: Stackpole Books: 149-159.

Risenhoover, Kenneth L.; Bailey, James A. 1985. Foraging ecology of mountain sheep: implications for habitat management. Journal of Wildlife Management. 49(3): 797-804.

Roberson, Jay A. 1986. Sage grouse-sagebrush relationships: a review. In: McArthur, E. Durant; Welch, Bruce L., eds. Proceedings, symposium on the biology of *Artemisia* and *Chrysothamnus*; 1984 July 9-13; Provo, UT. Gen. Tech. Rep. INT-200. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 157-167.

Robertson, M.D. 1989. Statewide survey of mountain quail, 1989: a report on the status of mountain quail in Idaho. Boise, ID: Idaho Department of Fish and Game. 27 p. Unpublished report. On file with: Idaho Department of Fish and Game, P.O. Box 25, Boise, ID 83707.

Robertson, M.D. 1991. Winter ecology of migratory sage grouse and associated effects of prescribed fire in southeastern Idaho. Moscow, ID: University of Idaho. 88 p. M.S. thesis.

Robinson, Scott K.; Rothstein, Stephen I.; Brittingham, Margaret C. [and others]. 1995. Ecology and behavior of cowbirds and their impact on host populations. In: Martin, Thomas E.; Finch, Deborah M., eds. Ecology and management of Neotropical migratory birds. New York: Oxford University Press: 428-460.

Rodenhouse, H.L.; Best, L.B.; O'Connor, R.J.; Bollinger, E.K. 1995. Effects of agricultural practices and farmland structures. In: Martin, T.E.; Finch, D., eds. Ecology and management of Neotropical migratory birds. New York: Oxford University Press: 269-293.

Rodgers, R.D. 1983. Reducing wildlife losses to tillage in fallow wheat fields. Wildlife Society Bulletin. 11: 31-28.

Rogers, Lynn L.; Rogers, S.M. 1976. Parasites of bears: a review. In: Pelton, M.R; Lentfer, J.W.; Folk, G.E., eds. Bears: their biology and management. IUCN Publication New Ser. 40: 411-430.

Roloff, G. 1995. Canadian lynx (*Felis lynx*) in Idaho: habitat conservation assessment and conservation strategy. [Boise, ID: Idaho Department of Fish and Game]. 13 p. Unpublished report. On file with: Idaho Department of Fish and Game, P.O. Box 25, Boise, ID 83707.

Rolseth, S.L.; Koehler, C.E.; Barclay, R.M.R. 1994. Differences in the diets of juvenile and adult hoary bats, *Lasiurus cinereus*. Journal of Mammalogy. 75(2): 394-398.

Rood, Stewart B.; Heinze-Milne, Sig. 1989. Abrupt downstream forest decline following river damming in southern Alberta. Canadian Journal of Botany. 67: 1744-1749.

Rosenberg, K.V.; Raphael, M.G. 1986. Effects of forest fragmentation on vertebrates in Douglas-fir forests. In: Verner, J.; Morrison, M.L.; Ralph, C.J., eds. Wildlife 2000: modeling habitat relationships of terrestrial vertebrates. Madison, WI: University of Wisconsin Press: 263-272.

Rotenberry, John T.; Wiens, John A. 1980. Habitat structure, patchiness, and avian communities in North American steppe vegetation: a multivariate analysis. Ecology. 61: 1228-1250.

Rothstein, S.I. 1994. The cowbird's invasion of the Far West: history, causes and consequences experienced by host species. Studies in Avian Biology. 15: 301-315.

Rothstein, S.I.; Verner, J.; Stevens, E.; Ritter, L.V. 1987. Behavioral differences among sex and age classes of the brown-headed cowbird and their relation to the efficacy of a control program. Wilson Bulletin. 99(3): 322-337.

Rudolph, Karen M.; Cowan, Lisa M.; Hunter, David L. [and others]. [In prep.]. Sharing of *Pasteurella* spp. among free-ranging bighorn sheep, domestic sheep, and feral goats.

Ruediger, Bill. 1996. The relationship between rare carnivores and highways. In: Evink, G.; Garrett, P.; Berry, J., eds. Transportation and wildlife: reducing wildlife mortality and improving wildlife passageways across transportation corridors. Proceedings of a symposium; 1996 April 3-May 2; Orlando, FL. Orlando, FL: Department of Transportation, Federal Highway Administration: 24-38.

Ruggiero, Leonard F.; Aubry, Keith B.; Buskirk, Steven W. [and others]. 1999. 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. 480 p.

Russell, R.W.; Carpenter, F.L.; Hixon, M.A.; Paton, David C. 1994. The impact of variation in stopover habitat quality on migrant rufous hummingbirds. Conservation Biology. 8(2): 483-490.

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 Agriculture, Forest Service, Pacific Northwest Research Station. 36 p.

Saab, V.A. 1996. Influences of spatial scale and land-use practices on habitat relationships of breeding birds in cottonwood riparian forests. Boulder, CO: University of Colorado. 140 p. Ph.D. dissertation.

Saab, V.A. 1998. Effects of recreational activity and livestock grazing on habitat use by breeding birds in cottonwood forests along the South Fork Snake River. Tech. Bull. No. 98-17. BLM-ID-PT99-004-1150. Boise, ID: U.S. Department of the Interior, Bureau of Land Management. 48 p.

Saab, V.A. 1999. Importance of spatial scale to habitat use by breeding birds in riparian forests: a hierarchical analysis. Ecological Applications. 9(1): 135-151.

Saab, V.A.; Bock, C.E.; Rich, T.D.; Dobkin, D.S. 1995. Livestock grazing effects in western North America. In: Martin, Thomas E.; Finch, Deborah M., eds. Ecology and management of Neotropical migratory birds. New York: Oxford University Press: 311-353.

Saab, V.A.; Marks, J.S. 1992. Summer habitat use by Columbian sharp-tailed grouse in western Idaho. Great Basin Naturalist. 52(2): 166-173.

Saab, V.A.; Rich, T.D. 1997. Large-scale conservation assessment for Neotropical migratory land birds in the interior Columbia River basin. Gen. Tech. Rep. PNW-399. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 57 p. (Quigley, Thomas M., ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Saab, V.; Dudley, J. 1995. Cavity use and nest usurpation by Lewis'woodpecker. On file with: U.S. Department of Agriculture, Forest Service, Intermountain Research Station, 316 E. Myrtle, Boise, ID 83702.

Saab, Victoria A.; Dudley, Jonathan G. 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.

Samson, F.B. 1980. Island biogeography and the conservation of prairie birds. In: Proceedings, 7th North American prairie conference; 1980 August 4-6; Springfield, MO. Springfield, MO: Southwest Missouri State University: 293-305.

Sanborn, S.R.; Loomis, R.B. 1979. Systematics and behavior of collared lizards (*Crotaphytus, Iguanidae*) in southern California. Herpetologica. 35(2): 101-106.

Sanders, S.D.; Flett, M.A. 1989. Montane riparian habitat and willow flycatchers: threats to a sensitive environment and species. Gen. Tech. Rep. PSW-110. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 262-266.

Sarell, M.J.; McGuinness, K.P. 1993. Rare bats of the shrub-steppe ecosystem of eastern Washington. Gen. Tech. Rep. Olympia, WA: Washington Department of Wildlife.

Sauer, J.R; Peterjohn, B.J.; Schwartz, S.; Hines, J.E. 1996. The North American breeding bird survey home page; version 95.1. Laurel, MD: Patuxent Wildlife Research Center. http://www.mbr-pwrc.usgs.gov/bbs/bbsold.html.

Saunders, Jack K., Jr. 1955. Food habits and range use of the Rocky Mountain goat in the Crazy Mountains, Montana. Journal of Wildlife Management. 19(4): 429-437.

Savage, D.E. 1969. The relationships of sage grouse to upland meadows in Nevada. Transactions, The Wildlife Society, California-Nevada Section. [Place of publication unknown]: The Wildlife Society. 16: 8-17.

Schmutz, J.K. 1984. Ferruginous and Swainson's hawk abundance and distribution in relation to land use in southeastern Alberta. Journal of Wildlife Management. 48(4): 1180-1187.

Schmutz, J.K.; Hungle, D.J. 1989. Populations of ferruginous and Swainson's hawks increase in synchrony with ground squirrels. Canadian Journal of Zoology. 67(10): 2596-2601.

Schroeder, M.A. 1994. Productivity and habitat use of Columbian sharp-tailed grouse in north-central Washington. Yakima, WA: Washington Department of Fish and Wildlife; progress report.

Schuler, Carol A.; Rickard; William H.; Sargeant, Glen A. 1993. Conservation of habitats for shrubsteppe birds. Environmental Conservation. 20 (1): 57-64.

Schulz, T.T.; Leininger, W.C. 1991. Nongame wildlife communities in grazed and ungrazed montane riparian sites. Great Basin Naturalist. 51: 286-292.

Scott, M.D.; Servheen, G. 1985. Caribou ecology. Boise, ID: Idaho Department of Fish and Game. Job completion rep., Proj. No. W-160-R-11. 136 p.

Servheen, Christopher. 1983. Grizzly bear food habits, movements, and habitat selection in the Mission Mountains, Montana. Journal of Wildlife Management. 47(4): 1026-1035.

Shaffer, Mark L. 1981. Minimum population sizes for species conservation. BioScience. 31(2): 131-134.

Sharp, B.E. 1992. Neotropical migrants on National Forests in the Pacific Northwest: a compilation of existing information. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, P.O. Box 3623, Portland, OR 97208.

Shriver, W.G.; Vickery, P.D.; Hedges, S.A. 1996. Effects of summer burns on Florida grasshopper sparrows. Florida Field Naturalist. 24(3): 68-73.

Shump, K.A.; Shump, A.U. 1982. *Lasiurus cinereus.* Mammalian Species. No. 185. Baltimore, MD: American Society of Mammalogists. 5 p.

Sibley, David. 1993. Birds of Cape May. 1st ed. Cape May Point, NJ: New Jersey Audubon Society, Cape May Point Bird Observatory. 145 p.

Siddle, C.; Davidson, G. 1991. Status of the Lewis'woodpecker (*Melanerpes lewis*) in British Columbia. Report commissioned by Wildlife Branch, Ministry of Environment, Victoria, British Columbia, Canada. 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.

Simpson, K. 1987. The effects of snowmobiling on winter range use by mountain caribou. Wildlife Working Rep. WR-25. Nelson, BC: Ministry of Environment and Parks. 13 p.

Singer, F.J. 1978. Behavior of mountain goats in relation to U.S. Highway 2, Glacier National Park, Montana. Journal of Wildlife Management. 42(3): 591-597.

Singer, F.J. 1979. Status and history of timber wolves in Glacier National Park, Montana. In: Klinghammer, E., ed. The behavior and ecology of wolves. New York: Garland STPM Press: 19-42.

Singer, F.J.; Doherty, J.L. 1985. Movements and habitat use in an unhunted population of mountain goats, *Oreamnos americanus.* Canadian Field Naturalist. 99: 205-217.

Sirotnak, J.M.; Reese, K.P.; Connelly, J.; Radford, K. 1991. Characteristics of Conservation Reserve Program fields in southeastern Idaho associated with upland bird and big game habitat use. Federal Aid Wildlife Restoration Project W-160-R. Boise, ID: Idaho Department of Fish and Game.

Smith, Christian A. 1986. Rates and causes of mortality in mountain goats in southeast Alaska. Journal of Wildlife Management. 50(4): 743-746.

Smith, D.G.; Murphy, J.R. 1973. Breeding ecology of raptors in the eastern Great Basin of Utah. Science Bulletin Biological Series. Vol. 18, No. 3. Provo, UT: Brigham Young University. 76 p.

Smith, Kirby G.; Urquhart, Martin A.; Festa-Bianchet, Marco. 1992. Preliminary observations of timing and causes of mountain goat kid mortality in west-central Alberta. In: Proceedings of the 8th biennial symposium of the Northern Wild Sheep and Goat Council; 1992 April 27-May 1; Cody, WY. Thermopolis, WY: Northern Wild Sheep and Goat Council. 8: 293-304.

Smith, Michael Roger. 1994. Evaluating the conservation of avian diversity in eastern Washington: a geographic analysis of upland breeding birds. Seattle: University of Washington. 89 p. M.S. thesis.

Smith, R.L. 1963. Some ecological notes on the grasshopper sparrow. Wilson Bulletin. 75(2): 159-165.

Sorensen, E. 1986. A precipitous decline in Lewis'woodpecker in Salt Lake and Davis counties. Utah Birds. 2: 45-54.

Spaulding, E.S. 1949. The quails. New York: The MacMillan Co. 123 p.

Speiser, R. 1992. Notes on the natural history of the northern goshawk. Kingbird. 42: 133-137.

Spraker, T.R.; Hibbler, C.P.; Schoonveld, G.G.; Adbetm, W.S. 1984. Pathological changes and micro-organisms found during a stress related die-off. Journal of Wildlife Diseases. 20: 319-327.

Spring, Lowell W. 1965. Climbing and pecking adaptations in some North American woodpeckers. Condor. 67(6): 457-488.

Squires, J.R.; Reynolds, R.T. 1997. Northern goshawk (*Accipiter gentilis*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 298. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union. 32 p.

Squires, J.R.; Ruggiero, L.F. 1996. Nest-site preference of northern goshawks in south central Wyoming. Journal of Wildlife Management. 60(1): 170-177.

St. Clair, L.L.; Johansen, J.R. 1993. Introduction to the symposium on soil crust communities. Great Basin Naturalist. 53(1): 1-4.

St. Clair, Larry L.; Johansen, Jeffrey R.; Rushforth, Sammuel R. 1993. Lichens of soil crust communities in the Intermountain area of the western United States. Great Basin Naturalist. 53: 5-12.

Stallcup, P.L. 1962. Contents of 103 stomachs of northern three-toed woodpeckers. Journal of the Colorado-Wyoming Academy of Sciences. 5: 39.

Stebbins, Robert C. 1985. Peterson field guides: western reptiles and amphibians. Boston, MA: Houghton Mifflin Co. 336 p.

Steenhof, K.; Kochert, M.N. 1985. Dietary shifts of sympatric buteos during a prey decline. Oecologia. 66(1): 6-16.

Stewart, George; Hull, A.C. 1949. Cheatgrass (*Bromus tectorum* L.) an ecologic intruder in southern Idaho. Ecology. 30: 58-74.

Swenson, Jon E. 1985. Compensatory reproduction in an introduced mountain goat population in the Absaroka Mountains, Montana. Journal of Wildlife Management. 49(4): 837-843.

Tashiro-Vierling, K.Y. 1994. Population trends and ecology of the Lewis'woodpecker (*Melanerpes lewis*) in southeastern Colorado. Boulder: CO: University of Colorado. 73 p. M.A. thesis.

Terra-Berns, Mary; Call, Paula; Harris, Chuck [and others]. 1997. Canada lynx in Idaho: past, present, and future. Boise, ID: Idaho Department of Fish and Game. 79 p.

Terres, John K. 1991. The Audubon Society encyclopedia of North American birds. New York: Wings Books. 1109 p.

Thiel, R.P. 1985. Relationship between road densities and wolf habitat suitability in Wisconsin. American Midland Naturalist. 113: 404-407.

Thomas, Jack Ward; Anderson, Ralph G.; Maser, Chris; Bull, Evelyn L. 1979. Snags. In: Thomas, Jack Ward, tech. ed. Wildlife habitats in managed forests, the Blue Mountains of Oregon and Washington. Agric. Handb. No. 553. Washington, DC: U.S. Department of Agriculture: 60-77.

Thompson, I.D. 1994. Marten populations in uncut and logged boreal forests in Ontario. Journal of Wildlife Management. 58(2): 272-280.

Thompson, I.D.; Colgan, P.W. 1987. Numerical responses of martens to a food shortage in north central Ontario. Journal of Wildlife Management. 51(4): 824-835.

Thomsen, Lise. 1971. Behavior and ecology of burrowing owls on the Oakland Municipal Airport. Condor. 73(2): 177-192.

Thorne, E. Tom; Hickey, William O.; Stewart, Shawn T. 1985. Status of California and Rocky Mountain bighorn sheep in the United States. In: Hoefs, Manfred, ed. Wild sheep: distribution, abundance, management, and conservation of the sheep of the world and closely related mountain ungulates. Northern Wild Sheep and Goat Council Special Report: 56-74.

Thurber, Joanne M.; Peterson, Rolf O.; Drummer, Thomas D.; Thomasma, Scott A. 1994. Gray wolf response to refuge boundaries and roads in Alaska. Wildlife Society Bulletin. 22(1): 61-68.

Thurow, T.L.; White, C.M. 1984. Nesting success and prey selection of long-eared owls along a juniper/sagebrush ecotone in south central Idaho. Murrelet. 65(1): 10-14.

Thurow, T.L.; White, C.M.; Howard, R.P.; Sullivan, J.F. 1980. Raptor ecology of Raft River Valley, Idaho. Idaho Falls, ID: U.S. Department of Energy, Idaho Operations Office. 45 p.

Tirhi, M.J. 1995. Washington state management plan for sharp-tailed grouse. Olympia,WA: Washington Department of Fish and Wildlife, Wildlife Management Program. 99 p.

Tobalske, B.W. 1997. Lewis'woodpecker (*Melanerpes lewis*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 284. Philadelpia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists'Union. 28 p.

Tomich, P. Quentin. 1982. Ground squirrels (*Spermophilus beecheyi* and allies). In: Chapman, Joseph A.; Feldhamer, George A., eds. Wild mammals of North America: biology, management, and economics. Baltimore, MD: Johns Hopkins University Press: 192-208.

Trulio, L.A. 1995. Passive relocation: a method to preserve burrowing owls on disturbed sites. Journal of Field Ornithology. 66(1): 99-106.

Tuttle, M.D.; Taylor, D.A.R. 1994. Bats and mines. Resource Publ. No. 3. [Place of publication unknown]: Bat Conservation International, Incorporated.

Ulliman, M.J. 1995. Rangewide conservation assessment for Columbian sharp-tailed grouse and its habitats. Moscow, ID: University of Idaho, Department of Fish and Wildlife Resources. 96 p.

U.S. Department of Agriculture, Forest Service. 1996. Status of the interior Columbia basin: summary of scientific findings. Gen. Tech. Rep. PNW-GTR-385. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station; U.S. Department of the Interior, Bureau of Land Management. 144 p.

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management. 1997a. Eastside draft environmental impact statement. Boise, ID: Interior Columbia Basin Ecosystem Management Project. [Irregular pagination]. 2 vol.

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management. 1997b. Upper Columbia River basin draft environmental impact statement. Boise, ID: Interior Columbia Basin Ecosystem Management Project. [Irregular pagination]. 2 vol.

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Fish and Wildlife Service. 1996. Conservation agreement: *Spermophilus brunneus brunneus*, northern Idaho ground squirrel. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest; Boise, ID: U.S. Department of the Interior, Fish and Wildlife Service. 14 p.

U.S. Department of the Interior, Bureau of Land Management. 1995. Mountain sheep ecosystem management strategy in the 11 western states and Alaska. BLM/SC/PL-95/001+6600. 79 p. [Place of publication unknown]: U.S. Department of the Interior, Bureau of Land Management.

U.S. Department of the Interior, Bureau of Land Management. 1997. Proposed standards for rangeland health and guidelines for livestock grazing management. BLM/ID/PT-97/002+4120. Boise ID: U.S. Department of the Interior, Bureau of Land Management, Idaho State Office.

U.S. Department of the Interior, Fish and Wildlife Service. 1987. Northern Rocky Mountain wolf recovery plan. Denver, CO: U.S. Department of the Interior, Fish and Wildlife Service. 119 p.

U.S. Department of the Interior, Fish and Wildlife Service. 1993. Grizzly bear recovery plan. Missoula, MT. 181 p.

U.S. Department of the Interior, Fish and Wildlife Service. 1994. Selkirk mountain woodland caribou recovery plan. Portland, OR: U.S. Department of the Interior, Fish and Wildlife Service. 59 p.

U.S. Department of the Interior, Fish and Wildlife Service. 1999. Rocky Mountain wolf recovery, 1999 annual report. 17 p. Available: http://www.r6.fws.gov/wolf/annualrpt99.

U.S. Department of the Interior, Fish and Wildlife Service. 2000. Grizzly bear recovery in the Bitterroot ecosystem. Final environmental impact statement. Missoula, MT: U.S. Department of the Interior, Fish and Wildlife Service. 792 p.

U.S. Government. 1994. Rules and regulations. Federal Register 59: 224 (Nov. 22, 1994): 60252- 60281.

U.S. Government. 2000a. Rules and regulations. Federal Register 65: 58 (March 24, 2000): 16052-16086.

U.S. Government. 2000b. Rules and regulations. Federal Register 65: 66 (April 5, 2000): 17779- 17786.

Van Ballenberghe, Victor; Erickson, Albert W.; Byman, David. 1975. Ecology of the timber wolf in northeastern Minnesota. Wildlife Monographs. No. 43. 43 p.

Van Dyke, WalterA.; Sands, Alan; Yoakum, Jim [and others]. 1983. Wildlife habitats in managed rangelands—the great basin of southeastern Oregon—bighorn sheep. Gen. Tech. Rep. PNW-GTR-159. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station; U.S. Department of the Interior, Bureau of Land Management. 39 p.

Van Horne, B.; Bader, A. 1990. Diet of nestling winter wrens in relationship to food availability. Condor. 92(2): 413-420.

Vickery, P.D. 1996. Grasshopper sparrow (*Ammodramus savannnarum*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 239. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists'Union. 24 p.

Vickery, P.D.; Hunter, M.L., Jr.; Melvin, S.M. 1994. Effect of habitat area on the distribution of grassland birds in Maine. Conservation Biology. 8: 1087-1097.

Vierling, K.T. 1997. Habitat selection of Lewis' woodpeckers in southeastern Colorado. Wilson Bulletin. 109: 121-130.

Villard, P. 1994. Foraging behavior of black-backed and three-toed woodpeckers during spring and summer in a Canadian boreal forest. Canadian Journal of Zoology. 72(11): 1957-1959.

Vogel, C.A.; Reese, K.P. 1995. Mountain quail status report: a document preliminary to a habitat conservation assessment for mountain quail. 33 p. 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.

Wai-Ping, V.; Fenton, M.B. 1989. Ecology of the spotted bat (*Euderma maculatum*) roosting and foraging behavior. Journal of Mammalogy. 70(3): 617-622.

Wakelyn, Leslie A. 1987. Changing habitat conditions on bighorn sheep ranges in Colorado. Journal of Wildlife Management. 51(4): 904-912.

Wakkinen, W.L. 1990. Nest site characteristics and spring-summer movements of migratory sage grouse in southeastern Idaho. Moscow, ID: University of Idaho. 57 p. M.S. thesis.

Wallestad, Richard. 1975. Male sage grouse responses to sagebrush treatment. Journal of Wildlife Management. 39(3): 482-484.

Wallestad, Richard; Peterson, Joel G.; Eng, Robert L. 1975. Foods of adult sage grouse in central Montana. Journal of Wildlife Management. 39(3): 628-630.

Ward, J.C.; Martin, M.; Allred, W. 1942. The susceptibility of sage grouse to strychnine. Journal of Wildlife Management. 6(1): 55-57.

Ward, J.M.; Kennedy, P.L. 1996. Effects of supplemental food on size and survival of juvenile northern goshawks. Auk. 113: 200-208.

Ward, Richard M.P.; Krebs, Charles J. 1985. Behavioural responses of lynx to declining snowshoe hare abundance. Canadian Journal of Zoology. 63(12): 2817-2824.

Washington Department of Wildlife. 1993a. Distribution status of the mountain quail (*Oreortyx pictus*) in Washington. Olympia, WA: Washington Department of Wildlife. 22 p. Unpublished report. On file with: Washington Department of Wildlife, 600 Capitol Way North, Olympia, WA 98501-1091.

Washington Department of Wildlife. 1993b. Status of the western gray squirrel (*Sciurus griseus*) in Washington. Olympia, WA: Washington Department of Wildlife. 33 p. Unpublished report. On file with: Washington Department of Wildlife, 600 Capitol Way North, Olympia, WA 98501-1091.

Waters, J.R.; McKelvey, K.S.; Zabel, C.J.; Oliver, W.W. 1994. The effects of thinning and broadcast burning on sporocarp production of hypogeous fungi. Canadian Journal of Forest Research. 24: 1516-1522.

Waters, J.R.; Zabel, C.J. 1995. Northern flying squirrel densities in fir forests of northeastern California. Journal of Wildlife Management. 59(4): 858-866.

Weaver, J. 1993. Lynx, wolverine and fisher in the western United States. Missoula, MT: Northern Rockies Conservation Cooperative. 132 p. Unpublished report. On file with: Northern Rockies Conservation Cooperative, P.O. Box 8594, Missoula, MT 59807.

Weaver, J. 1994. Ecology of wolf predation amidst high ungulate diversity in Jasper National Park, Alberta. Missoula, MT: University of Montana. Ph.D. dissertation.

Weaver, John L.; Paquet, Paul C.; Ruggiero, Leonard F. 1996. Resilience and conservation of large carnivores in the Rocky Mountains. Conservation Biology. 10(4): 964-976.

Weckwerth, R.P.; Hawley, V.D. 1962. Marten food habits and population fluctuations in Montana. Journal of Wildlife Management. 26(1): 55-74.

Weckwerth, R.P.; Wright, P.L. 1968. Results of transplanting fishers in Montana. Journal of Wildlife Management. 32(4): 977-980.

Weiss, N.T.; Verts, B.J. 1984. Habitat and distribution of pygmy rabbits (*Sylvilagus idahoensis*) in Oregon. Great Basin Naturalist. 44(4): 563-571.

Whisenant, Steven G. 1995. Landscape dynamics and arid land restoration. In: Roundy, Bruce A.; McArthur, E. Durant; Haley, Jennifer S.; Mann, David K., eds. Proceedings, wildland shrub and arid land restoration symposium; 1993 October 19-21; Las Vegas, NV. Gen. Tech. Rep. INT-315. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 26-34.

Whitaker, J.O., Jr.; Maser, C. 1981. Food habits of seven species of lizards from Malheur County, southern Oregon. Northwest Science. 55(3): 202-208.

Whitaker, J.O., Jr.; Maser, C.; Cross, S.P. 1981. Food habits of eastern Oregon bats, based on stomach and scat analyses. Northwest Science. 55(4): 281-292.

Whitaker, John O., Jr.; Maser, Chris; Keller, Laurel E. 1977. Food habits of bats of western Oregon. Northwest Science. 51(1): 46-55.

White, C.M.; Thurow, T.L. 1985. Reproduction of ferruginous hawks exposed to controlled disturbance. Condor. 87(1): 14-22.

White, P.J.; Ralls, Katherine; Garrott, Robert A. 1994. Coyote-kit fox interactions as revealed by telemetry. Canadian Journal of Zoology. 72: 1831-1836.

Widen, P. 1987. Goshawk predation during winter, spring, and summer in a boreal forest area of central Sweden. Holarctic Ecology. 10: 104-109.

Widen, P. 1989. The hunting habitats of goshawks, *Accipiter gentilis* in boreal forests of central Sweden. Ibis. 131: 205-231.

Wielgus, Robert B.; Bunnell, Fred L. 1995. Tests of hypotheses for sexual segregation in grizzly bears. Journal of Wildlife Management. 59(3): 552-560.

Wiens, John A.; Rotenberry, John T. 1981. Habitat associations and community structure of birds in shrub-steppe environments. Ecological Monographs. 5(1): 21-41.

Willis, Edwin O.; Snow, David W.; Stotz, Douglas F. [and others]. 1993a. Olive-sided flycatchers in southeastern Brazil. Wilson Bulletin. 105: 193-194.

Willis, M.J.; Keister, G.P., Jr.; Immell, D.A. [and others]. 1993b. Sage grouse in Oregon. Wildl. Res. Rep. 15. Portland, OR: Oregon Department of Fish and Wildlife, Wildlife Research Section. 56 p.

Wishart, William. 1978. Bighorn sheep. In: Schmidt, John L.; Gilbert, Douglas L., eds. Big game of North America: ecology and management. Harrisburg, PA: Stackpole Books: 161-171.

Witmer, G.W.; Martin, S.K.; Sayler, R.D. 1998. Forest carnivore conservation and management in the interior Columbia basin: issues and environmental correlates. Gen. Tech. Rep. PNW-420. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 51 p. (Quigley, Thomas M., ed.; Interior Columbia Basin Ecosystem Management Project: scientific assessment).

Yensen, Eric. 1991. Taxonomy and distribution of the Idaho ground squirrel, *Spermophilus brunneus*. Journal of Mammalogy. 72(3): 583-600.

Yoakum, J. 1980. Habitat management guides for the American pronghorn antelope. Tech. Note 347. Denver, CO: U.S. Department of the Interior, Bureau of Land Management. 77 p.

Yoakum, James D. 1968. A review of distribution and abundance of American pronghorn antelope. Biennial Antelope States Workshop Proceedings. 3: 4-14.

Yoakum, James D. 1978. Pronghorn. In: Schmidt, John L.; Gilbert, Douglas L., eds. Big game of North America. Harrisburg, PA: Stackpole Books: 103-121.

Yoakum, James D. 1986a. Trends in pronghorn populations: 1800-1983. In: Proceedings of the Twelfth Pronghorn Antelope Workshop. [Meeting dates unknown]: [Meeting location unknown]. Jasper, AB: Alberta Recreation, Parks and Wildlife, Fish and Wildlife Division: 12: 77-85.

Yoakum, James D. 1986b. Use of *Artemisia* and *Chrysothamnus* by pronghorns. In: McArthur, E. Durant; Welch, Bruce L., eds. Proceedings of a symposium on the biology of *Artemisia* and *Chrysothamnus*; 1984 July 9-13; Provo, UT. Gen. Tech. Rep. INT-200. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 176-185.

Yoakum, Jim D.; O'Gara, Bart W. 1990. Pronghorn/livestock relationships. In: McCabe, R.E., ed. Transactions of the 55th North American wildlife and natural resources conference; 1990 March 16-20; Denver, CO. Washington, DC: Wildlife Management Institute: 475-487.

Yosef, R. 1996. Loggerhead shrike (*Lanius ludovicianus*). In: Poole, A.; Gill, F., eds. The birds of North America. No. 231. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union. 28 p.

Yosef, R.; Grubb, T.C., Jr. 1992. Territory size influences nutritional condition in nonbreeding loggerhead shrikes (*Lanius ludovicianus*): a ptilochronology approach. Conservation Biology. 6(3): 447-449.

Younk, James V.; Bechard, Marc J. 1994. Breeding ecology of the northern goshawk in high-elevation aspen forests of northern Nevada. The northern goshawk: ecology and management: Proceedings of a symposium of the Cooper Ornithological Society; 1993 April 14-15; Sacramento, CA. In: Studies in Avian Biology. Cooper Ornithological Society; 16: 119-121.

Zager, P.L.; Mills, L.S.; Wakkinen, W.; Tallmon, D. 1995. Woodland caribou: a conservation dilemma. Endangered Species Update. 12 (10, 11): 1-4.

Zeveloff, Samuel I.; Collett, Farrell R. 1988. Mammals of the intermountain west. Salt Lake City, UT: University of Utah Press. 365 p.

Zwickel, F.C. 1972. Some effects of grazing on blue grouse during summer. Journal of Wildlife Management. 36: 631-634.

Zwickel, Fred C. 1992. Blue grouse, *Dendragapus obscurus*. In: Poole, Alan F.; Stettenheim, Peter; Gill, Frank B., eds. The birds of North America. No. 15. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists'Union. 28 p.

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