

**WILDLIFE USE OF MANAGED FORESTS
IN WASHINGTON: A REVIEW**

By

Larry L. Irwin
Joseph B. Buchanan
Tracy L. Fleming
Steven M. Speich



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EXECUTIVE SUMMARY

The 1987 Timber-Fish-Wildlife Agreement included adaptive management processes to provide for wildlife diversity on state and private forests in the State of Washington. This paper presents a literature review that provides a basis for monitoring terrestrial wildlife on managed forests. The purposes of the review included the following: (a) develop a list of parameters which influence wildlife habitat use and population dynamics in managed forests; (b) examine applicable habitat classification systems; and (c) develop a study plan for monitoring wildlife in managed forests.

The review was predicated upon emerging concepts in landscape ecology and new technology for evaluating wildlife habitats. Chapter 2 describes the scientific basis for wildlife-habitat relationships, recognizing that developing a greater understanding of wildlife response to natural and human-induced disturbances is a valid process for understanding forestry-wildlife interactions. In Chapter 3, a managed forest is defined as a shifting mosaic of dynamic forest patches subject to human-directed changes and random natural disturbance. Chapter 3 then evaluates several approaches for monitoring wildlife responses to managed forests, including indicator-species concepts, guilds, wildlife habitat models, and habitat classification systems. New research is suggested to construct management guilds as a means of monitoring wildlife responses. The management guilds would be coupled with development of an operational Geographic Information System (GIS).

The literature review on wildlife-habitat relationships provided details which were used to construct a matrix of parameters that would be useful in monitoring. The monitoring program, described in Chapter 4, will benefit by classifying managed forest mosaics using geomorphic land units, which incorporate landforms, soils, vegetation, and climatic influences on wildlife diversity and distribution. For some applications, particularly sensitive species, new technology is required for development, primarily including wildlife-habitat models that incorporate life-history attributes and risk analysis.

Chapter 5 describes adaptive concepts which employ the GIS capability and forest management practices in experiments that will result in greater understanding of wildlife-habitat relationships and provide the basis for modifying management policy. A management-experimentation program is suggested that simultaneously will evaluate management effectiveness and answer important ecological questions. The basic strategy includes a thorough quantification of variation in habitat conditions, both at the stand and landscape level. Several topics for research investigation were outlined.

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CHAPTER I. INTRODUCTION AND SCOPE OF REVIEW

Recent forest-practice rules for the State of Washington require more comprehensive consideration of a variety of forest resources. These rules stem from intensive negotiations between disparate interest groups. The negotiations followed breakthrough efforts for resource integration by Thomas (1979) and Brown (1985), and resulted in the Timber-Fish-Wildlife Agreement (TFW) in March 1987. The TFW Agreement initiated a process that aims to provide fish and wildlife habitats in managed forests on state and private lands in Washington. Fueled by improved communication, smoother coordination, and genuine cooperation, TFW embraces concepts within adaptive environmental assessment and management (Holling 1978, Walters 1986).

The salient points of the Agreement include: (a) monitoring on-site forest/wildlife interactions and (b) employing scientifically-driven management experiments, both of which are to be used in feedback loops to assess biological assumptions and improve management. Such processes are expected to reduce opportunity costs, improve scientific understanding and stimulate additional management options. Cooperators and the public hope the TFW agreement becomes a formula for sustained forestry, satisfactory wildlife populations, and supportable management programs.

Herein, we report a literature review aimed to provide a scientific basis for implementing the TFW Agreement as it relates to terrestrial vertebrate wildlife. We reviewed literature that discussed wildlife responses to habitat conditions within managed forest stands and mosaic landscapes. The review had three primary purposes:

- (a) develop a list of biological and physical parameters which influence wildlife presence, distribution, and population dynamics within managed forest mosaics in the State of Washington;
- (b) examine available habitat classification schemes which have applicability to Washington's forests; and
- (c) develop a study plan for monitoring and evaluating forest practices.

The list of parameters, particularly those which are controllable in forest management, and an examination of available technology for evaluating habitat conditions, comprise

the basis of a suggested program for monitoring and research evaluation of wildlife responses to forestry practices. The process we discuss should stimulate the articulation of explicit management-experiment hypotheses. Field tests of the hypotheses will improve management and scientific understanding of wildlife responses.

The TFW Cooperative Monitoring, Evaluation, and Research Committee (CMERC) agreed that, for wildlife in managed forests, goals and objectives should include optimizing wildlife species- and wildlife habitat diversity. Thomas (1979) and Brown's (1985) efforts led to several working hypotheses for providing habitat and species diversity in managed forests. We hope this document will aid the TFW Program's quest for continued development of adaptive management strategies to accommodate biological diversity.

The process of simultaneously achieving wildlife diversity and wood production goals is based on coordinated manipulation of several variables which influence wildlife diversity and population response (Hall and Thomas 1979):

- (a) schedules of silvicultural treatments;
- (b) arrangement of stands in time and space;
- (c) stand conditions;
- (d) size of treatment area; and
- (e) land-type.

Hall and Thomas (1979) and Brown (1985) emphasized stand-level contributions to habitat diversity as a means for providing species richness, and paid particular attention to the spatial arrangement of stands, amount of edge and degree of contrast between stands.

We add to the previous efforts in two significant ways. First, we consider landscape influences on wildlife distribution and population dynamics, which: (a) account for, and derive understanding from, natural disturbance patterns; and (b) integrate variation in spatial relationships among wildlife populations. Secondly, we outline quantitative processes for testing various working hypotheses and evaluating wildlife-habitat models.

Assumptions

Management for species richness requires specific forest manipulations that provide necessary habitat conditions over time and space. We assume that an acceptable and sustained flow of wood products can result from judiciously-applied practices that simultaneously maintain wildlife diversity over a managed

landscape. We also assume that TFW cooperators can agree on a specific definition of diversity as well as objectives for optimal wildlife diversity within the broader concept of biological diversity.

We assume that nearly all wildlife habitat management on state and private forestlands will occur in coordination with timber management practices, and that timber production has been and will continue to be the dominant land use. Further, we assume purposeful incentives will develop that encourage forestry practices that maintain and enhance a variety of forest benefits and resources, recognizing tremendous variability in opportunities and conditions.

Finally, we assume that wildlife distribution and population dynamics on state and private forestlands are significantly influenced by landscape processes and activities on federal forestlands, and, for migrant species, by influences on lands outside the state.

Methods and Scope

The scope of our review included choosing examples that would aid in development of a generally applicable program of monitoring and research evaluation of forest management treatments and experiments. We did not attempt to provide coverage of all species and all possible treatment combinations.

We canvassed published literature describing the process of habitat selection and its relationship to population regulation in terrestrial vertebrate wildlife in managed forests of North America. For species or groups which have not been examined in managed forests, we inferred relationships from studies of habitat use and selection. This included a general review of landscape ecology and patch dynamics (e.g., size and spatial and temporal arrangement of harvest units) which influence wildlife diversity.

Also, we examined contributions to understanding population dynamics in managed forests within Washington and the Pacific Northwest. A few European references were included. We included a review of available land and habitat classification systems which have applicability to Washington's forests. We excluded an examination of riparian-zone relationships.

Further, we included reviews of relevant wildlife-habitat models. We also contacted supervisors of research institutions in the region, requesting information on their programs of wildlife and forestry research.

We developed a matrix of parameters which influenced use and populations in managed forests. This list was reduced to a selected group of parameters which we believe can be used in a monitoring program. Therefore, our work was conducted along 2

independent fronts: (1) an examination of responses of wildlife to forest conditions which resulted in the matrix of parameters; and (2) an evaluation of literature relevant to development of a sustainable program of monitoring the results of forest-management experiments.

We used boldface type in the text to indicate technical terms which we define in the Glossary. We listed scientific names of species of plants and animals in Appendix E. In general, we followed the format of the Journal of Wildlife Management, except that English units of measurements were used.

CHAPTER II. SCIENTIFIC BASIS FOR WILDLIFE-HABITAT RELATIONSHIPS

Adaptive management principles predict that wise forest-resource management will be served best through imaginative approaches which can be supported by broad consensus. Such consensus agreement requires knowledge of basic determinants of habitat selection and factors which regulate wildlife populations. Therefore, we describe the theoretical and empirical basis for wildlife-habitat relationships as a means for understanding wildlife response to habitat changes within managed forests. Such a discussion is essential to an adequate description of management relationship, and, more importantly, to the formulation of testable hypotheses and design of monitoring programs to evaluate management success (Karr and Freemark 1985).

Habitat Selection

The theoretical basis for evaluating habitat selection among vertebrate wildlife is well developed (e.g., Fretwell and Lucas 1970, Cody 1985, Emlen 1985, Rosenzweig 1985, Stephens and Krebs 1986, Morris 1988). Habitat choice is determined by availability of food resources, nest sites and mates, refuge from predators and weather, abundance of conspecific and interspecific competitors, parasitism and diseases, and other factors that influence reproductive success or survival. According to theory, habitat quality generally should decline with increasing population density (Fretwell and Lucas 1970, Emlen 1985, Rosenzweig 1985). Habitat selection, then, depends primarily upon the basic suitability of different habitats, discounted by density-dependent demands of conspecific animals already present (Morris 1988). Density-dependent interactions between species will modify habitat choice further.

Habitat selection on a local level may be based on the particular genetic make-up of the individual (Wecker 1963), a specific search image, early-learned experience, or any combination of these factors (Klopfer 1969). Habitat selection, then, is assumed to be a product of evolutionary processes that ensure that individuals seek and find the particular environments to which they are adapted (Ruggiero et al. 1988). This does not necessarily imply that animals always find the conditions that maximize reproduction. On the contrary, habitat selection

optimizes among costs and benefits associated with conditions that favor reproduction (e.g., food supplies) and factors that cause mortality (e.g., predators, harsh weather).

Immediate stimuli, or proximate factors, may be "summed" by an animal, such that every habitat occupied need not possess all features of optimum conditions. Occupancy implies only that the combined effects of individual factors exceed the behavioral threshold to settle in a site. For this reason, habitat selection data alone cannot provide inferences to optimal habitat conditions, or even conditions in which a species can maintain populations over time. Therefore, conclusions of habitat preference and requirement, as inferred from habitat selection data, should be considered preliminary until experimental evidence is available.

The specific set of stimuli that induces animals to select a particular habitat or habitat condition may or may not be related to survival and reproduction (Hilden 1965). Thus, proximate factors in selecting habitats may be associated or correlated with the ultimate factors which actually have molded the species' adaptive traits. Physical structures have long been considered as important proximate factors stimulating habitat selection for many species. The structural features act directly in providing shelter, nesting substrate, or protection from predators. Or, they can act indirectly, by providing cues to the potential availability and diversity of food (Rotenberry 1981).

Johnson (1980) identified hierarchical levels of habitat selection to aid design and interpretation of field studies. First-order selection includes the geographic range occupied by a species. Selection of a home range by an individual or social group constitutes 2nd-order selection. Third-order selection refers to use vs. availability of habitat conditions within the home range. Finally, selection of feeding, resting, or denning/nesting sites is 4th-order selection. In field studies, 3rd- and 4th-order selection are most often considered in inferring habitat preference and requirement. But, as mentioned above, experimentation is required for conclusive evidence.

Landscape Ecology

Creating a consensus understanding of wildlife responses to managed forest systems requires knowledge of relationships with natural systems. Washington's forests evolved with natural disturbances which will continue to change the face of the forest landscape (e.g., wildfire, windthrow, flooding, erosion, ice-storms, insect epidemics, forest diseases). Heterogeneity and change, therefore, must be recognized as fundamental features, not only of managed forest systems, but also of the natural environment (Karr and Freemark 1985).

The concept of a natural forest-disturbance regime should aid development of a management process for understanding

wildlife use of man-dominated forests (Runkle 1985). The fact that adaptations of vertebrates and vertebrate assemblages exhibit recurrent patterns to natural disturbance and heterogeneity led Karr and Freemark (1985) to attempt a synthesis for examining vertebrate responses to forest management treatments. We incorporated parts of their synthesis in this report. Urban et al. (1987) also point out that studies within managed landscapes may provide the basis for understanding both landscape ecology and management because of the ability to learn from management experiments.

Understanding managed system responses becomes quite complex, when viewed within the context of a forest ecosystem which has inherent dynamics (Pickett and White 1985). However, Urban et al. (1987) describe a hierarchical rule-set for simplifying landscape ecology, which should result in insights for managing forest landscapes.

As applied to landscape ecology, the hierarchical system provides general guidance for defining functional relationships. This aids studies for predictions of how management alters biological processes in forest systems. There are 4 levels within the hierarchy: landscape, watershed, stand, and forest gap. These levels correspond to similar scaling within wildlife systems: ecosystem (or metapopulation), wildlife community, population (or social group), and individual animal (Table 1).

In this report, we follow the convention adopted by many papers we reviewed, which applied the term landscape to all relationships above the stand level. Thus, watershed-level relationships can be discussed as part of the landscape.

Table 1. Hierarchical perspective provides guidelines for defining functional relationships of managed forest mosaics, and defines pathways that relate components at different scales (modified from Urban et al. 1987).

Hierarchy level	Scale	Wildlife level	Scale (no.)
landscape	>10,000 ac	ecosystem/metapopulation	1,000s
watershed	100s-1000s ac	community/population	100s-1,000s
stand	1s-10s ac	niche/social group	2-100
gap	0.05-0.25 ac	individual animal	1

Forest systems are mosaics of patches. Each patch follows its own pattern of development and dynamics, which, in turn, is influenced by the nature of the mosaic itself, i.e., landscape arrangement of the patches. Individual animals and wildlife populations respond to the temporal and spatial arrangement of

such patch mosaics in varying manners. Some animals spend their entire lives or generations within single patches, others use some patch types for reproduction and others for feeding, while others use patches within a mosaic in relative selective fashion, distinguishing among different patches. Still others may drift over different forest mosaics. Communities and ecosystems contain populations exhibiting diverse patch-use strategies, which are influenced by management (Figure II-1).

Populations and Patches.--Habitat patches with conditions in which individuals survive but are unable to reproduce are known as population sinks. Occupancy of sinks depends upon immigrating individuals from reproductive populations, or sources. Source-sink relationships may provide significant consequences to landscape stability (Wiens 1985): (a) the dynamics of the source populations may dominate those of the much larger metapopulation; (b) source populations, by providing most dispersers, may contribute differentially to the overall gene pool; and (c) if density-dependent factors impose limits on the subpopulations occupying source patches, many of the individuals produced there may be forced to disperse elsewhere, some to sink patches. The sink populations thus may be more dense at times than the source populations (Van Horne 1983). Time lags in response to changes in patch quality will obscure such relationships even further.

Patches and Community Structure.--Communities represent aggregations of species that are integrated over many dynamic patches in space, so community dynamics result from the behavior of their constituent populations. Each species is immersed in different patches with different dynamics and, with other species, integrate patch dynamics and interactions over differing scales of space and time. This creates difficulty for investigations and simplified explanations of community dynamics that could guide forest management (Wiens 1985).

Nonetheless, the relationship of patch structure with community features has received considerable theoretical attention at the community level, particularly with respect to how patchiness enhances the persistence of predator/prey interactions, by providing prey with spatial refugia from predation, or the coexistence of competitors, by permitting spatial differentiation of resource use. For example, Litvaitis et al. (1985) found that snowshoe hares survived bobcat predation during cyclic lows by occupying refugia-type habitats not available to all individual hares during periods of population abundance.

Patches and Habitat Fragmentation.--Patch dynamics assume a very real and practical significance in relation to management of wildlife populations, habitats, or forest landscapes. Conservation topics increasingly contain references to fragmentation of natural forest habitats and its effects, how fragments should be arrayed with respect to one another, and what their optimal area might be (Harris 1984). Discussions involving

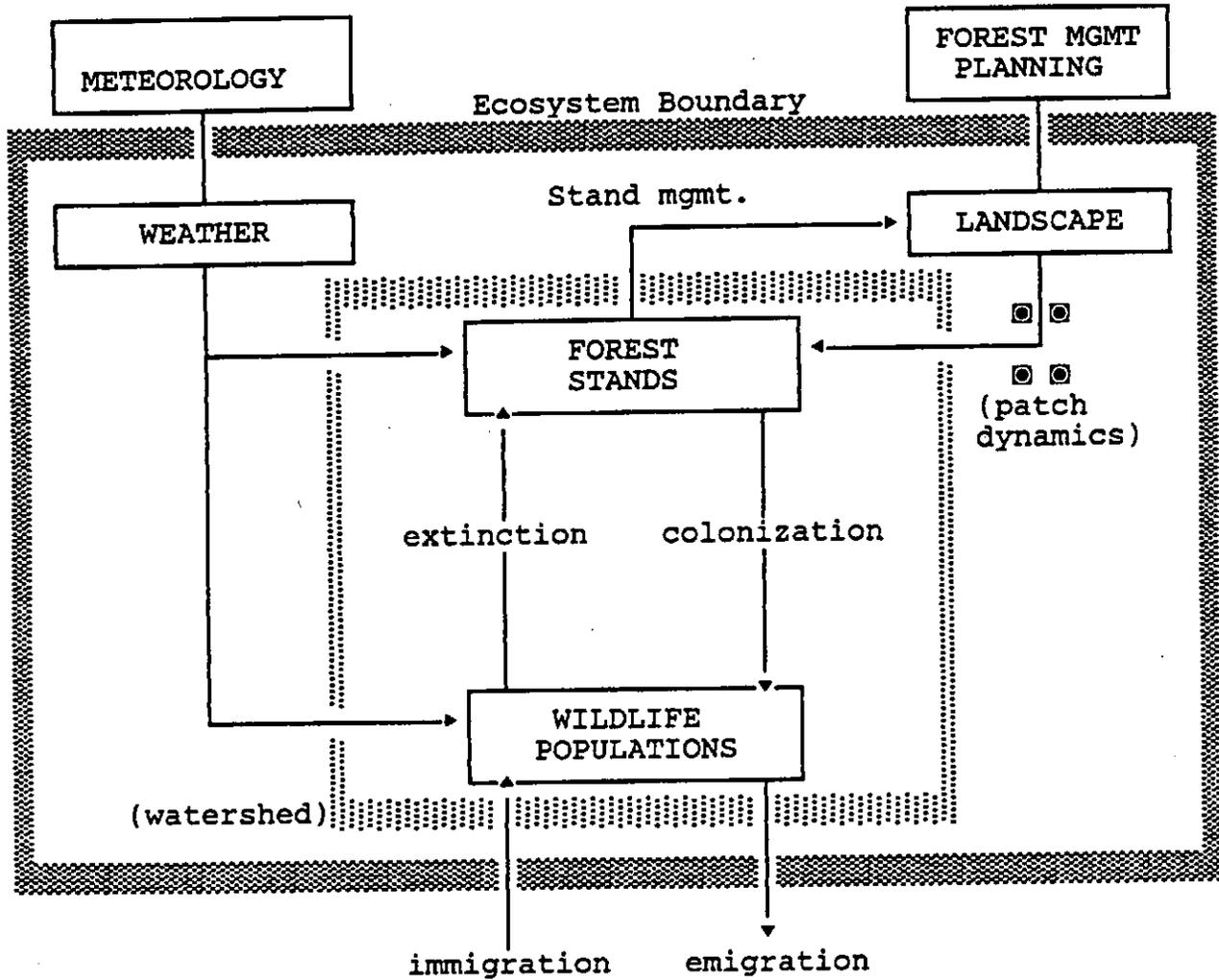


Figure II-1. Populations are linked to landscape dynamics through 2 feedback loops, forest stand x population interactions, and stand x landscape mosaic interactions. Wildlife population dynamics result from within-stand processes as well as temporal and spatial distribution of stands in the managed landscape mosaic.

fragmentation show considerable controversy, and no clear consensus has emerged (Verner 1986). However, clearly it is inappropriate to apply general theory to management situations in which details of the patch dynamics of the environment or the patch responses of organisms are of critical importance (Wiens 1985).

The reason that different groups of vertebrates differ in the way they respond to habitat fragmentation is related to the numerous differences in the natural histories of these animals (Urban and Shugart 1986). This point seems critical because fragmentation reduces the average size of patches of a given habitat, increases distance between patches, decreases the ratio of interior to edge area within patches, and temporarily increases the landscape diversity of an area by creating new patches that undergo succession.

Therefore, for species which are sedentary and disperse over short distances, or for populations having low recruitment rates, the rate of colonization of patches created by local extinction or habitat changes will be low. Habitat specialization and low population densities further reduce patch colonization rates. These influences/attributes increase the probability that a population in a patch will undergo local extinction. With increasing fragmentation of a landscape stochastic, or random disturbances become more important and may increase the likelihood of local extinctions. With a reduction in the overall number of suitable patches in a region, regional extinction thus also becomes, at least theoretically, more probable (Wiens 1985).

Because of the greater role of random-disturbance effects (e.g., wildfire) and the inevitable time lags in recruitment and patch colonization, populations and communities in a fragmented landscape are unlikely to be in overall equilibrium. And theory or management practices founded upon equilibrium or deterministic assumptions are not likely to perform very well. So, while there is clearly a linkage between population density and patterns of patch occupancy and spatial distribution, that relationship likely is more complex and multifactorial than existing theory might lead us to believe.

Nevertheless, Crow (1989) points out that landscape ecology, which emphasizes spatial patterns, provides a useful context for considering vertebrate diversity in managed forest landscapes (Fig. II-2). Biological diversity, a broader concept than discussed here, considers genetics, species, and ecosystem elements which can operate both within landscapes (local diversity) and among landscapes (regional diversity). The TFW Agreement primarily considers species diversity within managed forest landscapes, but should recognize that broader contexts exist. We anticipate that adaptive management strategies ultimately will incorporate biological diversity.



Fig. II-2. Monitoring wildlife populations in managed forests requires considerations of spatial arrangement of numerous patches, each with its own internal structure and dynamics.

CHAPTER III. REVIEW OF APPROACHES FOR MONITORING WILDLIFE RESPONSES TO MANAGED FORESTS

Monitoring Wildlife Response to Habitat Change

We define a managed forest as a shifting mosaic of dynamic forest patches subject to relatively systematic, human-directed changes and random probability of natural disturbance. Under this definition all successional types theoretically would be predictable and relatively permanent, although varying in size as well as temporally and spatially. So, one might view a fully-managed forest as a dynamic steady-state landscape under a relatively stable disturbance regime. The responses by wildlife would result from within-patch dynamics as well as the temporal and spatial distribution of managed patches, because these two aspects of management would be linked.

The simultaneous occurrence of local dynamics and broad-scale equilibrium underscores the central importance of scale hierarchies in the interpretation of wildlife responses to managed forests (Urban et al. 1987). Dynamics on 1 scale, (e.g., stand succession) may result in stasis on another (e.g., landscape). Therefore, we employ a hierarchal approach to interpretations of wildlife use of managed forests, following Hoekstra and Flather (1986). In general, wildlife present in a managed/treated stand are:

- (a) present at the time of disturbance as individuals living continuously in the stand;
- (b) those whose home ranges contain all or part of the treated stand (or site); and
- (c) those which recently immigrated into the area.

These relationships are coupled with the mobility of species and social mechanisms influencing dispersal. The severity and size of the disturbance determine which strategy is most likely to dominate and thereby influence the wildlife community. Forest treatments which are both very severe (e.g., complete canopy removal and complete removal of slash and debris) and applied over a large area can be expected to result in a protracted time for re-colonization for many wildlife species.

The literature reveals that wildlife management agencies are struggling to develop comprehensive programs for evaluating wildlife responses to actual and proposed forest habitat changes. In western Washington and Oregon alone there are some 460 species of vertebrates which use forests for all or part of their life cycles (Brown 1985). Reliable monitoring of population trends for all species across the State of Washington is technologically impossible. It is an understatement to say that development of a responsive program for monitoring wildlife will challenge the patience, energy and imagination of all involved. Thus, the TFW

cooperators are obliged to build a monitoring program using management experience, models, indices, and new research.

In this chapter we discuss several common approaches and tools for monitoring wildlife-habitat relationships. In general, the literature shows wildlife responses to habitat changes are evaluated by the following:

- (1) observing individual species' responses (Appendix A);
- (2) monitoring one or a few species that may indicate responses by other species (indicator species);
- (3) directly measuring species diversity (at 3 levels of resolution); or
- (4) evaluating habitat conditions which are assumed or are determined to be correlated with population performance (i.e., habitat models).

Biological Indicators

Recent planning guidelines for the U.S. Forest Service (36th Code of Federal Regulations, section 219) prescribe the use of management indicator species, which are selected because their population changes are believed to indicate the effects of management activities on other species. Presumably, by providing habitat required by indicator species, all other species dependent on the same limiting habitat conditions would be protected. The management indicator species concept seems to have arisen from guild theory and from the concept that some species have narrow niche tolerances (Graul et al. 1976). The concept of a guild was defined originally by Root (1967:335) as a "group of species that exploit the same class of environmental resources in a similar way". Trends in one member of a guild, or the indicator species, were assumed to reflect similar trends in all species in a guild.

In practice, guilds frequently are developed using a pre-determined scheme, such as grouping together species which have similar foraging or nesting patterns (Jaksic 1981, Severinghaus 1981). Once the impact of management on any 1 species is known for a guild, the impact on every other species presumably is known. It is an appealing concept for management due to apparent reduction in time and money required for monitoring.

However, results of field tests caution against widespread use of the guild-indicator species concept. For example, species within a guild may occupy habitats largely independently of other members of the guild (Wiens et al. 1986). Also, Mannan et al. (1984) found within-guild responses to managed forests varied considerably for 5 bird guilds in northeastern Oregon.

Furthermore, monitoring approaches that use indicator-

species concepts are conceptually invalid without confirmatory research (Landres 1983, Irwin and Cole 1987). Moreover, it may be prohibitively expensive to conduct statistically reliable surveys to monitor changes in abundance of even relatively common species (Verner 1984). Others have criticized widespread use of indicator species for evaluating wildlife community responses to forest management (Jaksic 1981, Verner 1984, Verner et al. 1985). Thus, monitoring should not rely on guild-indicators developed using pre-determined relationships.

Instead, Verner et al. (1985) recommend monitoring entire "management guilds", involving an assessment of the capability to support specified groups. A management guild is an ecological assemblage of species that respond in a similar way to a variety of changes likely to affect their environment. The trends in combined total counts of all species in each management guild would be evaluated before and after harvest operations (Verner et al. 1985). This process is more efficient and cost-effective than using indicator species. Because more individuals are counted for a whole management guild than for any single species within the guild, fewer counting stations are needed to obtain sample sizes large enough to detect statistically significant trends.

Further, the management-guild approach has other benefits (Verner et al. 1985):

- (1) field workers tally all species whenever a habitat is sampled, reducing the chance of the undetected loss of a rare species;
- (2) the concept contains various options for regrouping species to address specific questions, such as migrants vs. non-migrants or by territory size (to examine fragmentation effects); and
- (3) the variance in whole guild counts should be less.

DeGraaf and Chadwick (1984) presented a comparison of the relative abilities of bird species, both separately and grouped into nesting and foraging guilds, to classify several northeastern forest cover types and age classes. They used pre-assigned guilds with Bray-Curtis polar ordination to summarize bird and guild-community relationships. The ordinations arranged forest and general cover types by similarity of species or guild composition.

In DeGraaf and Chadwick's (1984) work, comparisons of ordination results across several cover types suggested that bird species reflect habitat differences to a greater degree than do either foraging or nesting guilds. Individual species tend to group habitats by age class, and to a lesser extent, by forest cover type. Their results suggest that use of guilds for habitat classification purposes is more reliable at the landscape level

than at the project (stand) or drainage level.

Guild analyses have been developed most frequently for avian communities. Jaksic (1981) noted that use of statistical procedures to define reliable guilds is helpful if all taxa are considered. Habitat assessments that employ such ecologically-oriented guilds use biological principles in a more integrative fashion than guild-indicators (Karr 1987). The management guild-ordination process, therefore, seems useful to TFW monitoring, although research is required to define the relationships and assemblages across all taxonomic groups.

Wildlife Habitat Evaluation Models

Wildlife habitat quality is defined in several ways, but it generally relates to an area's ability to supply the life requirements for a particular species or species group. Habitat and environmental assessment methods are used to quantify and evaluate the quality of habitat or environment for wildlife. This can be done using parameters that apply at either the stand or landscape scale. A commonly-used approach assigns a numerical value to an area based upon the quality and quantity of habitat that it contains.

Without exception, habitat evaluation methods are applied under the assumption that wildlife abundance or species diversity relates to habitat characteristics, and that habitat provides a measure of potential for wildlife. Of course, other factors (e.g., severe weather, predation, hunting) may determine whether this potential is realized. These factors make it difficult to predict wildlife population changes as a result of habitat changes.

Numerous models and procedures for predicting wildlife-habitat interactions have been developed recently from the Habitat Evaluation Procedures (HEP) process within the U.S. Fish and Wildlife Service (U.S. Fish Wildl. Serv. 1980). The foundation for HEP is the Habitat Suitability Index (HSI), which depicts relationships between habitat quality and sets of specific variables. HSI assumes variability in abundance of wildlife species or guilds is associated with structural characteristics of vegetation.

The HEP model partitions habitat into resources needed by the species or species group and subdivides each component into measurable physical, biological, or chemical variables. Experienced biologists subjectively weight these variables according to their significance and combine them into an HSI value which compares existing conditions with optimal conditions for the species or group/guild. The ratio of existing to optimal conditions expresses habitat suitability as an index with values between 0 and 1. The index is assumed to have a direct linear relation with carrying capacity. Basic tasks to be completed in a HEP analysis include: cover type mapping, model development (or

if a model exists, fine-tuning to a local area), measurement of habitat variables, and data analysis (Figure III-1).

Because HSI models are usually constructed for use throughout a species' range, HSI models need to be modified for optimal application in specific areas (Williams 1988). O'Neil et al. (1988) evaluated a procedure for modifying HSI models for local use, and discussed the need for validation tests to include a number of sites with a variety of conditions. Irwin and Cook (1985) and Cook and Irwin (1985) pointed out the value of sampling population and habitat conditions across a wide geographic area to validate an HSI model for pronghorns.

HSI models have proven highly variable in their ability to predict habitat quality accurately (Bayer and Porter 1988). So, HEP models should be considered working hypotheses which must be verified. Thus, management testing of HSI models is important. For example, Laymon and Reid (1986) found an HSI model for spotted owls in California did not perform well for predicting use of low-suitability habitat. Owls were using small (5-17 ac) patches for foraging within a larger mosaic of habitat. HSI models may be inappropriate for predicting habitat quality as a continuous variable, but suitable for broad, discrete habitat classes. The ultimate utility of HSI models for TFW applications may depend upon the guild-classification scheme that is employed.

Pattern Recognition Models (PATREC)

Pattern recognition models, or PATREC, have received increasing attention for evaluating wildlife habitats (Grubb 1988, Putman 1988). PATREC is based upon Bayesian statistics whose basic properties include a simple mathematical form which captures the process by which most biologists intuitively assess habitat conditions. PATREC uses information on the frequency with which specific habitat attributes occur among areas of a particular habitat suitability class, as well as comparable information on the frequency with which the same components occur among areas of other habitat suitability classes.

In PATREC frequencies of occurrence for the various habitat suitability classes are called conditional probabilities and habitat attributes are called diagnostic criteria. Diagnostic criteria and their associated conditional probabilities are used to evaluate an area of unknown quality by determining status (presence or absence) of habitat attributes, and then calculating the probability of the area being highly suitable with the use of Bayes' theorem and conditional probability values. The probability of having a highly suitable habitat can be used as an index of habitat quality.

Bayesian statistical inference is a mathematical technique used commonly when decisions must be made under conditions of uncertainty. So, investigators must: (1) estimate the

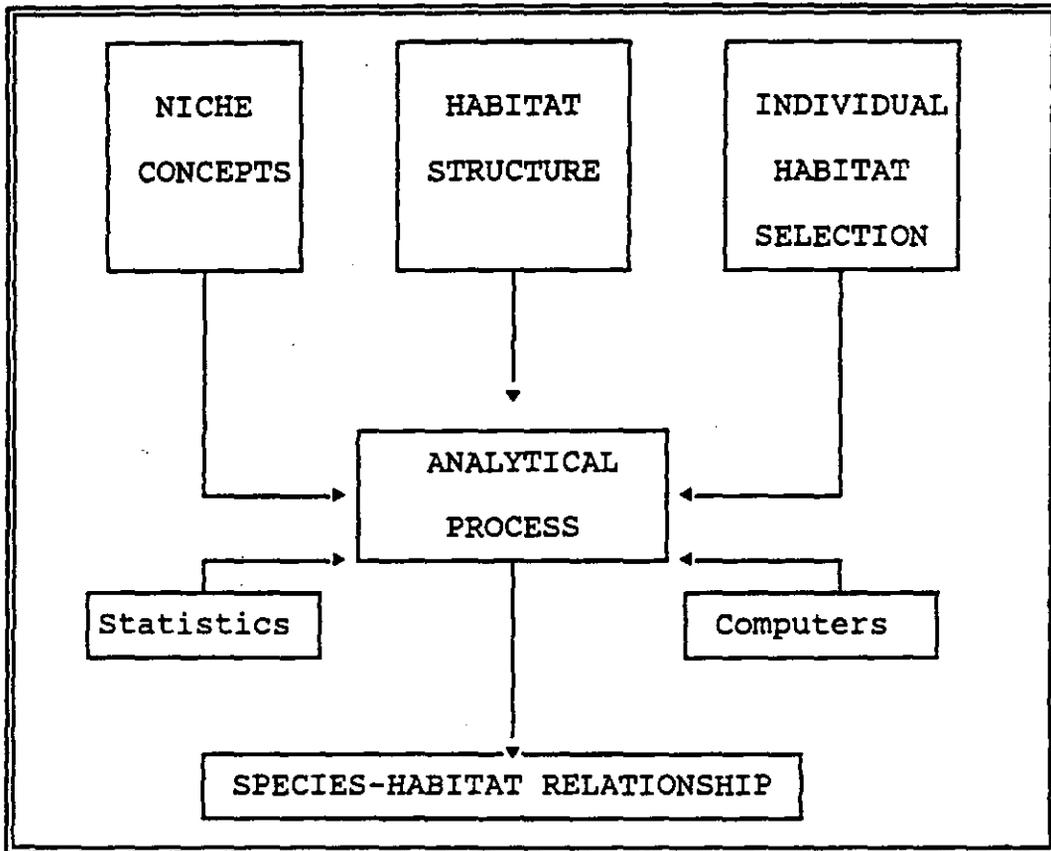


Figure III-1. Relationship between theory, observation, and habitat interaction for developing habitat relationships models for single species.

probability (called prior probability) that some condition exists or will exist in the future; (2) collect sample data related to the condition; and (3) revise the initial probability estimates to account for the sample results (called posterior probability). Habitat suitability of an area can then be estimated after the presence or absence of individual habitat attributes has been determined, and a few calculations are completed.

Grubb (1988) illustrated the use of PATREC with an example for bald eagle habitat in Arizona. He noted it also can be used to evaluate the cost of alternative habitat management procedures. PATREC models should be considered first generation models because habitat attributes should be independent of each other, and they rarely are in field situations. However, the seriousness of lack of independence is debated (Grubb 1988).

The model's strong points include the following:

- (a) PATREC is easily upgraded with new information, which should make it valuable to the TFW concept;
- (b) PATREC will help identify and clarify the relative importance of habitat attributes;
- (c) PATREC will increase understanding of what constitutes suitable habitat;
- (d) PATREC will improve ability to communicate understanding; and
- (e) PATREC should improve the ability to make acceptable management decisions.

Other Models

A class of models that can be used on microcomputers to assess forestry/wildlife relations includes area models (Marcot et al. 1988). These include cumulative effects or cumulative impacts models, and are designed to assess the combined effects on wildlife species from either a variety of management activities or activities conducted over a broad area. These models may include automated mapping systems (described below).

Raphael and Marcot (1986) evaluated the reliability of an untested Wildlife-Habitat-Relationships (WHR) matrix for mixed-evergreen forest of northwestern California. The WHR model related 4 levels of habitat suitability to each of 4 seral stages. The authors compared extensive wildlife surveys with a set of predictions generated by the model. They found that breeding birds and wintering birds differed from numbers predicted. The model incorrectly predicted change of abundance for pair-wise comparisons of successional stages. The model did, however, correctly predict general patterns of wildlife community similarity among seral stages for most species groups.

Raphael and Marcot (1986) concluded the WHR model was useful because some of its predictions were empirically correct. However, they recommended substantial revision to improve performance, particularly for between-stage comparisons. The model was more difficult to adjust for species predicted but not observed. They felt that WHR models were more reliable over a watershed scale or larger. They sampled only 1 of the model's 18 habitat types, at a cost of \$600,000, so they concluded that managers would likely rely on untested models. The high cost of research validation underscores the potential value to test such models via adaptive management concepts, in which model validation occurs as a result of management experience.

Thomas et al. (1986) discussed a habitat effectiveness model for big game that is probably most applicable in southeastern Washington. Their model weighted roads, and size and spacing of forest patches that provide food or cover on the scale of a 5,000-25,000-acre management area. They applied multi-spectral scanning of Landsat imagery, which allowed an evaluation of a variety of management alternatives, such as schedules for road construction. They indicated that an examination of preference and need was required and that it should include an analysis of open roads and traffic density. A follow-up effort developed a model for use in evaluating winter ranges (Thomas et al. 1988).

Recent research efforts in habitat relationships of large ungulates show increasing attention to physiological indicators as a means of evaluating herd health in relation to population density and habitat conditions. One technique uses diaminopimelic acid, or DAPA (Nelson and Davitt 1984, Leslie et al. 1989). Fecal nitrogen also may be a useful indicator (Leslie and Starkey 1984), although Hobbs (1987) urges caution in making interpretations.

More recent research by DelGuidice et al. (1988) and DelGuidice and Seal (1989) suggests that urinary chemical ratios (urea nitrogen:creatinine, plus electrolyte balance) promise a reliable means of evaluating winter population health in white-tailed deer and wolves. These techniques require field verification before they are accepted widely in management; several field-tests currently are underway.

Habitat Classification Schemes

Habitat classification systems are crucial to measuring and understanding wildlife species responses to habitat changes. In this section we discuss common classification systems which may be useful to the TFW monitoring program.

Where the internal stand characteristics and spatial and temporal properties among stands of the managed forest system can be measured, these measurements are obviously preferred over any habitat classification scheme. If one can measure size, distribution, frequency, rotation length, severity, and synergism

of forest treatments (including measures of central tendency, dispersion and frequency distributions), and evaluate wildlife responses at several levels of ecological organization, a new, managed-landscape classification system does not seem necessary. It would appear logical to use current classifications, including forest associations (habitat types) and successional stages, superimposing measures of the forest conditions and treatments.

In our view, however, there are valuable contributions which can be made with visionary attempts at classifying managed forest landscapes, using ecologically-based spatial/temporal relationships and geological information in addition to vegetation. In particular, it should be possible to develop a classification that incorporates functional relationships between habitat parameters and wildlife at the landscape- as well as stand level. Such a process is not expected to alter previous classification systems significantly; rather it should refine them to include the relevant parameters that influence wildlife populations and use.

We expect a classification system for managed landscapes that includes parameters that influence wildlife to stem from new models that link spatial and temporal variation with standard central-tendency statistics. We firmly believe that measuring wildlife responses to natural and man-induced habitat variation will provide the most significant opportunities for adaptive management. Thus, in the following paragraphs we develop a conceptual process that could lead to a managed landscape classification system that guides ecologically-based wildlife-habitat modeling and monitoring.

Habitat Type Classifications.--Ecological classifications have the greatest potential for developing broader compatibility between wildlife and forest management practices. At the stand level, wildlife respond to variations in vegetation structure and composition. Therefore, an understanding of wildlife responses to changes in forest environments requires a basic knowledge of vegetation potential and changes over time. Since the concept of forest association reflects natural patterns, such as the habitat type (Daubenmire and Daubenmire 1968), a list of available climax vegetation classification systems and associated successional pathways should aid implementation of the TFW agreement.

Daubenmire (1968) defined habitat type as all those land areas potentially capable of supporting similar plant communities at climax. Habitat types provide a permanent and ecologically based stratification system. They also serve as a reference point for successional modeling, because units within a single habitat type can be expected generally to show similar successional responses to management treatments or natural disturbances.

Daubenmire and Daubenmire (1968) and Franklin and Dyrness (1973) provide descriptions for climax forest associations in

Washington. Daubenmire and Daubenmire's (1968) descriptions in northeastern Washington and northern Idaho were modified recently by Cooper et al. (1987). Also, Henderson (1988) provided a description of vegetation associations on the Olympic National Forest, and Williams and Lillybridge (1983, 1987, 1988) provided descriptions of plant associations of the Okanogan, Colville, and Wenatchee National Forests which should apply to state and private lands.

Some (e.g., Pfister and Arno 1980) regard habitat types as units which integrate environmental factors in a fashion that permits relative comparisons of productivity. Mueller-Dombois and Ellenberg (1967) require that to function as a land classification system, habitat types should be more narrowly defined and include landscape features, productivity, and other management-oriented variables. We agree with the latter approach because of the influence of landscape features on wildlife distribution and abundance.

Successional Stage.--At the stand level, wildlife respond to structural and compositional conditions, so successional models correlate more closely with "cover type", a term used commonly in wildlife management to describe current vegetation conditions on a site. Several successional models are available which are appropriate for stand-level evaluations of wildlife use in managed forests in Washington. Irwin and Peek (1979a) used multiple regression to predict shrub growth and successional dynamics in intensively-managed forests in the grand fir-cedar-hemlock zone of northern Idaho and northeastern Washington. In their work, presence and biomass of important understory shrub species were related to a number of habitat factors and silvicultural treatments. Treatments ranged from clearcuts, to selective harvest and various slash-treatment methods. And Moeur (1985) presented a modeling algorithm for predicting shrub cover for use with a forest stand prognosis model for northern Idaho. Dyrness (1973), Arno et al. (1986), Keane (1987), and Schoonmaker and McKee (1988) provide successional descriptions which should be of general and specific importance to TFW.

The operational difficulty with most successional models is that they may not include the specific variables that influence wildlife habitat use or abundance (e.g., snags, downed logs). Also, successional models tend to be limited by the number of sites sampled, so they provide little insights into management procedures which result in variations that might broaden the scope of compatibility between forest practices and wildlife. Further, recent studies (e.g., Arno and Keane 1986, Keane 1987) note that variation in successional pathways occurs within individual habitat types. This variation must be more fully understood to develop ecologically-based successional models that could aid management.

Landtype Classification System.--Landtyping (Wendt et al. 1975, USFS 1976) seems an appropriate stratification process for

beginning to integrate wildlife-habitat relationships at the landscape level with those at the drainage and stand level. Landtype systems integrate soils, landforms, climate, and plant ecology as an aid to understanding resource relationships (Figure III-2). Hall and Thomas (1979) recognized the value of a landtype classification system in developing silvicultural strategies for wildlife in the Blue Mountains of eastern Washington and Oregon.

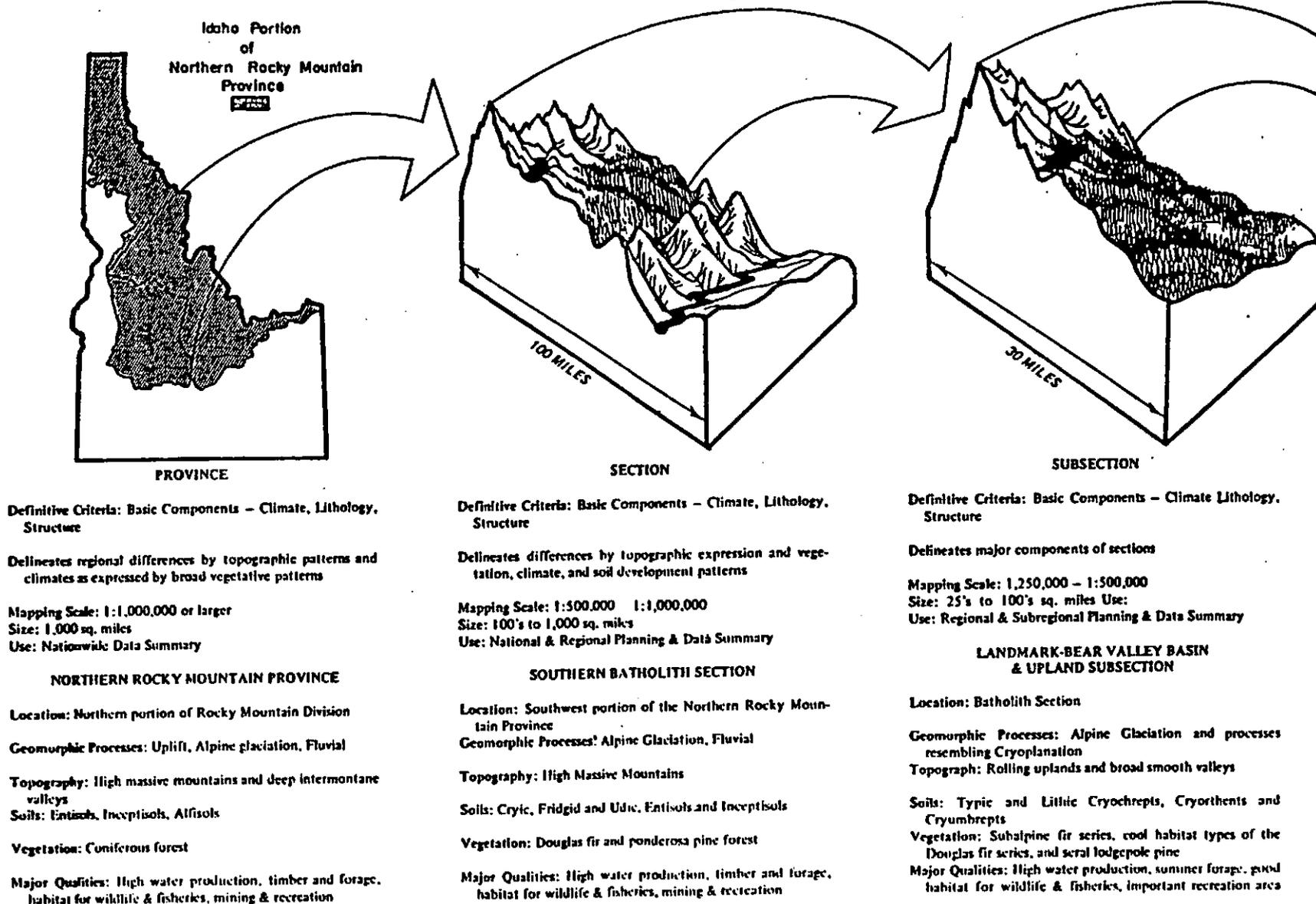
The landtyping stratification system has been developed based on the principle of recognizing the geomorphic nature of the landscape and the factors which determine behavior of ecosystems: inorganic materials, time, climate, vegetation, and animal life. Within a 7-layered classification, the landtype and landtype association levels have proven useful for National Forest comprehensive land-use planning in Idaho and Montana. A landtype association is a particular unit with characteristic soils and landforms, which are permanent elements of ecosystems that have predictable patterns.

We illustrate the landtype classification system using an hypothetical example of an elk herd in the Wenatchee National Forest. In winter elk generally occupy fluvial (sculpted by water-flow) slopes and canyonlands which contain the transition zone between Douglas-fir/Ponderosa pine forest and shrublands. In spring the elk might occupy low-elevation alluvial lands where green grasses are available earlier than in montane landtypes. By summer elk will have migrated to deeply-dissected fluvial lands, glaciated trough lands, or perhaps to glaciated volcanic headlands in the Alpine Lakes Wilderness.

The point is that a single forest management practice (e.g., clearcutting) within 1 habitat type (e.g., Douglas-fir/snowberry) has decidedly different relationships with elk depending upon the landtype association where the treatment occurred. Thus, models which predict wildlife habitat use in managed forests ought to account for landforms.

Several authors indicate current wildlife-habitat models cannot reliably compare the relative probability of use of different sites (e.g., Thomas et al. 1986, Wisdom et al. 1986). Such models compare the effects of alternative management designs over time at 1 site. A landtyping scheme would allow comparisons of HSI or other wildlife-habitat models among patches within a forest or physiographic province. Furthermore, the landtype association scheme should spawn research questions that predict wildlife presence or abundance on the basis of functional relationships among landscape habitat components. Quite obviously, landtype associations create, or at least influence, most of the natural heterogeneity that characterizes much of Washington's forest landscape. Thus, we suggest the TFW cooperators consider the Landtype System as a potential stratification to permit landscape ecology concepts to be applied and validated with management experience.

LAND SYSTEMS OF THE



Definitive Criteria: Basic Components – Climate, Lithology, Structure

Delineates regional differences by topographic patterns and climates as expressed by broad vegetative patterns

Mapping Scale: 1:1,000,000 or larger
Size: 1,000 sq. miles
Use: Nationwide Data Summary

NORTHERN ROCKY MOUNTAIN PROVINCE

Location: Northern portion of Rocky Mountain Division

Geomorphic Processes: Uplift, Alpine glaciation, Fluvial

Topography: High massive mountains and deep intermontane valleys

Soils: Entisols, Inceptisols, Alfisols

Vegetation: Coniferous forest

Major Qualities: High water production, timber and forage, habitat for wildlife & fisheries, mining & recreation

Definitive Criteria: Basic Components – Climate, Lithology, Structure

Delineates differences by topographic expression and vegetation, climate, and soil development patterns

Mapping Scale: 1:500,000 1:1,000,000
Size: 100's to 1,000 sq. miles
Use: National & Regional Planning & Data Summary

SOUTHERN BATHOLITH SECTION

Location: Southwest portion of the Northern Rocky Mountain Province

Geomorphic Processes: Alpine Glaciation, Fluvial

Topography: High Massive Mountains

Soils: Cryic, Frigidid and Udic, Entisols and Inceptisols

Vegetation: Douglas fir and ponderosa pine forest

Major Qualities: High water production, timber and forage, habitat for wildlife & fisheries, mining & recreation

Definitive Criteria: Basic Components – Climate Lithology, Structure

Delineates major components of sections

Mapping Scale: 1:250,000 – 1:500,000
Size: 25's to 100's sq. miles
Use: Regional & Subregional Planning & Data Summary

LANDMARK-BEAR VALLEY BASIN & UPLAND SUBSECTION

Location: Batholith Section

Geomorphic Processes: Alpine Glaciation and processes resembling Cryoplanation

Topograph: Rolling uplands and broad smooth valleys

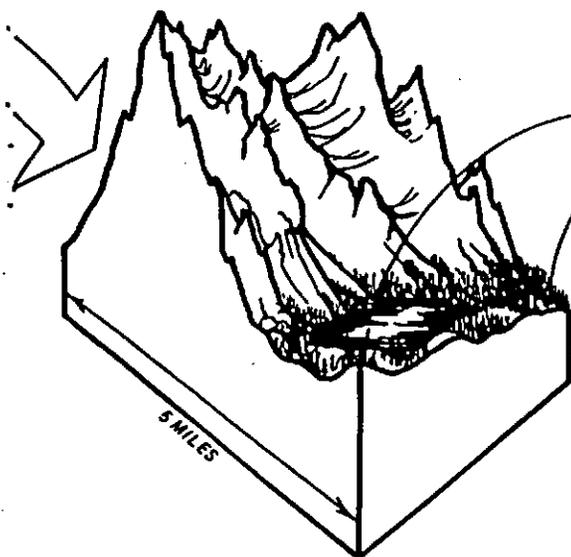
Soils: Typic and Litic Cryochrepts, Cryorthents and Cryumbrepts

Vegetation: Subalpine fir series, cool habitat types of the Douglas fir series, and seral lodgepole pine

Major Qualities: High water production, summer forage, good habitat for wildlife & fisheries, important recreation area

Fig. III-2. Landtype Classification of the Boise National Forest, Idaho (from Wendt et al. 1976).

BOISE NATIONAL FOREST



LANDTYPE ASSOCIATION

Definitive Criteria: Manifest Components – Soils, landform, biosphere

Delineates areas in similar stages of development which reflect hazards and/or capabilities

Mapping Scale: 1:60,000 – 1:125,000

Size: 1's to 25's sq. miles

Use: Data summary for allocation of resources

GLACIATED GRANITIC HEADLANDS

Location: High elevation areas

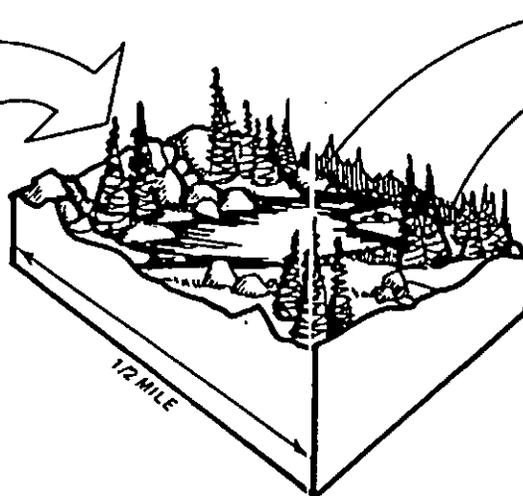
Geomorphic Processes: Alpine Glaciation

Topography: Steep rocky peaks, ridges, and cirque basins

Soils: Lithic and Typic Cryorthents and Cryochrepts of sandy-skeletal and loamy skeletal families

Vegetation: Subalpine fir/white bark pine, Subalpine fir/alksedge and Subalpine fir/grouse whortleberry habitat types

Major Qualities: High water production, recreation & wildlife, aesthetics



LANDTYPE

Definitive Criteria: Manifest Components – Soils, landforms, biosphere

Delineates the permanent elements of the ecosystem that have predictable behavior patterns

Mapping Scale: 1:30,000 – 1:60,000

Size: 1/10 to 2 sq. miles

Use: Comprehensive Planning

CIRQUE BASIN LANDS – DEEP SANDY and LOAMY SKELETAL SOILS

Location: Head of glaciated valleys

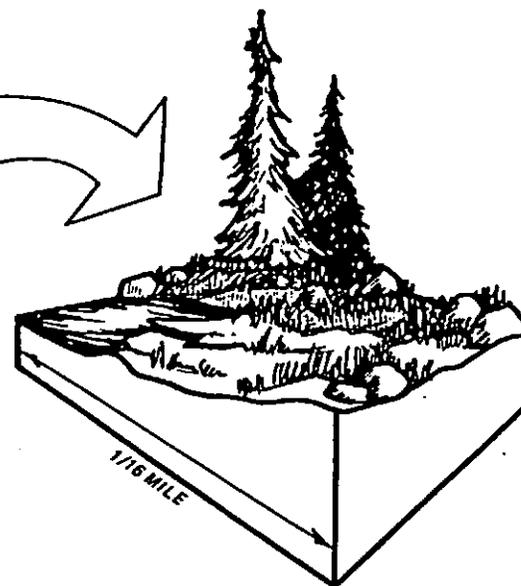
Geomorphic Processes: Scouring and removal of materials by ice

Topography: Gently sloping basins

Soils: Typic Cryochrepts loamy skeletal mixed, Typic Cryorthents loamy skeletal mixed

Vegetation: Subalpine fir/grouse whortleberry and Subalpine fir/alksedge

Major Qualities: Water flow regulator, recreation, wildlife, fisheries, aesthetics



LANDTYPE PHASE

Definitive Criteria: Manifest Components – Soils, landforms, biosphere

Delineates individual components of landtypes

Mapping Scale: 1:15,000 – 1:30,000

Size: 1/100 to 1/10 sq. mile

Use: Project Development and Detailed Planning

TYPIC CRYOCHREPTS LOAMY SKELETAL MIXED

Location: Depression and nearly level bottom areas

Geomorphic Processes: Deposition by past glacial melt water

Topography: Concave depressions

Soils: Typic Cryochrepts loamy skeletal mixed

Vegetation: Subalpine fir/grouse whortleberry

Major Qualities: Water flow regulator, Summer forage

SITE: Similar to high intensity detailed soil surveys. Provides precise understanding of ecosystem components

Fig. III-2, continued.

The landtype concept is not new, but has not been used widely in wildlife management. The major difficulty in applying the landtype concept in forest/wildlife management in the past has been the lack of a set of rules which might guide field studies as well as an inability to inventory and analyze the resources available at the landscape scale. Furthermore, associations between wildlife populations and landtypes would need to be validated following development of initial hypotheses and expert opinion, via management-experiments at the landscape level.

Fortunately, recent algorithms for landscape analysis, using Geographic Information Systems (GIS) and spatial statistics permit the development of new conceptual and analytic processes/models for evaluating wildlife/forestry relationships at the landscape level. These are developed in the next section.

Geographic Information Systems and Habitat Classifications

To simplify theory in wildlife ecology and for ease of management application, nature most often has been assumed to be spatially homogeneous. This assumption is precisely why such theory frequently has failed when taken into nature or when used as a basis for management systems applied to the real world (Wiens 1985). That assumption also has become a statistical constraint for many research designs that maximize variation between forest successional stages but minimize within-stage variation. So, there is clear urgency for theoretical and management models that realistically consider the spatial and temporal variation, or dynamics, of disturbed/managed systems and predict the consequences of these dynamics.

A rapid and simplified approach is needed for measuring and predicting habitat diversity on landscapes of varying sizes and types as an aid in monitoring and management decision-making and as an aid in linking resource classification systems with research studies of functional relationships. Geographic information systems, in concert with land and vegetation classification systems and computer technology can aid this development.

Geographic information systems store large volumes of map data, so they may provide useful sources of land-use data which can be used to automate application of wildlife-habitat models, such as HSI. Essentially any map, interpreted aerial photograph, or other information that can be referenced geographically, can be converted into machine-readable format and displayed and analyzed with the GIS (Mead 1982).

Analysis of geographic or spatial data includes overlaying and combining maps, summarizing the areas of different types (e.g., habitat types) mapped either on a single map or one created through combination. In a manual system, all these

summaries are obtained by hand using such equipment as dot grids, planimeters, and standard drafting techniques. In an automated system, these steps are all done, or can be done, by the computer. Whether or not a computer is used, the GIS is designed to organize acquired data and information, store it in such a way that users can efficiently update, retrieve, and analyze it, and apply it to a decision-making process (Mead 1982).

To conduct desired GIS analyses using a computerized information system, resource data must be entered into the computer in a form it understands. Information on wildlife habitat is entered as coordinate points, supplied to the computer as a series of digits. For example, a nest location for a bald eagle would be represented as a single pair of x,y coordinates. A stream would be represented as a series of x,y coordinates connected by arcs or straight lines. An area of forest vegetation also would be represented by a series of x,y coordinate pairs and arcs or straight lines which form a closed polygon. Areas also can be entered and/or stored in cell form, where a polygonal shape (e.g., lake or Upland Management Area) can be represented by a series of (usually) smaller rectangular cells which collectively have approximately the same shape and area.

. The application of one potential geographically-oriented technique which uses measures of interspersion and juxtaposition as components of a spatial diversity index was described by Heinen and Cross (1983). The technique uses a simplified, grid-cell approach for evaluating the potential of a site or landscape to provide habitat for wildlife species. The process develops indices for interspersion, juxtaposition, and spatial diversity, which is measured relative to species-specific responses to interspersion and juxtaposition.

Heinen and Cross' (1983) technique seems useful because it is easy to apply, is relatively inexpensive, involves simple calculations, and can incorporate the use of micro-computers. Moreover, it is versatile in that it works with species which require large homogeneous areas (low interspersion) as well as those which require a great deal of habitat diversity.

Following Heinen and Cross (1983), interspersion is calculated for grid-cells by counting the cells surrounding a centroid cell which contain different cover-type categories. Since 8 cells surround the centroid cell, the number of different cells is divided by 8, resulting in an index that ranges from 0 to 1. Juxtaposition is calculated by first identifying all combinations of edge types around the center cell. A numerical rating is given to each edge type, assigning a value of 1 to diagonal edges and 2 for vertical or horizontal edges. Relative weighting factors (ranging from 0 to 1) are assigned each type and represent the quality of different community junctions. The quality factor is multiplied by a numerical rating of each edge type to give a total value for each edge type. All values are

totaled and divided by 12 to allow the juxtaposition index for each centroid cell to range from 0 to 1.

An example calculation of juxtaposition (J) is:

Edge Type	Quantity	Quality	Total
A/A	4	0.2	0.8
A/B	5	0.5	2.5
A/C	3	0.6	1.8
Total			5.1

In this example, $J = 5.1/12 = 0.43$.

The spatial diversity (Sd) index used by Heinen and Cross (1983) is as follows:

$Sd_a = ([\sigma_a I_s] + [\alpha_a J_x])(E1_a)(E2_a)(E3_a)$, where A indicates a particular species, σ_a indicates the relative importance of interspersion, α_a indicates the relative importance of juxtaposition (σ_a and α_a each range between 0 and 1, but must sum to 1), and $E1_a$, $E2_a$, and $E3_a$ indicate exclusion factors, which also range from 0 to 1. An exclusion factor is defined as any habitat component with a positive or negative impact on a particular species. Using the previous example, the Sd index is as follows:

$$Sd = (0.5 \times 0.625) + (0.5 \times 0.43) = 0.528.$$

In this example, no exclusion factors were identified. Any number of exclusion factors may be used depending on the area and species being examined. For example, the presence of water within 1 mile may influence use of a forest by certain amphibians, so if a site was close to water, the exclusion index would be assigned a value of 1. Table III-1 provides several additional measures of spatial indices (Forman and Gordon 1986) that might be useful in monitoring or model development.

Williams et al. (1983) evaluated a GIS database for characterizing ruffed grouse breeding sites in Pennsylvania. And Donovan et al. (1987) evaluated a GIS-based source of habitat information for an assessment of nesting and brood-rearing habitat for wild turkeys. They measured optimal spatial arrangement using (1) the linear amount of edge/hectare, and (2) interspersion of 4 habitat types, measured as the minimal distance that encompassed all 4 important habitat types. The process was tested using actual distribution as compared to predicted HSI values.

The Donovan et al. (1987) HSI model, built from resource information system variables, was found useful in evaluating turkey brood-rearing habitat suitability. However, the authors believed that generalizing habitat requirements is necessary for GIS-based habitat modeling because of the high cost of constructing a GIS. Therefore, Donovan et al. (1987) felt that

Table III-1. Measures and equations for patch characteristics
which can be used in evaluations of patch use by wildlife

Equation	Description of parameters
	<u>Patch Shape</u>
$D = P/2\sqrt{A\pi}$, where	D = index to shape of patch P = perimeter of patch A = area of patch
	<u>Isolation</u>
$r_i = 1/n \sum_{j=1}^n d_{ij}$, where	r_i = index of isolation of patch i n = number of neighboring patches d_{ij} = distance between patch i and any neighbor patch j
	<u>Accessibility</u>
$a_i = \sum_{j=1}^n d_{ij}$, where	a_i = index of accessibility of patch d_{ij} = distance along a linkage, e.g., a forest corridor, between patch i and neighbor patches
	<u>Interaction among patches</u>
$I_i = \sum_{j=1}^n (A_j / d_j^2)$, where	I_i = interaction of patch i with n neighboring patches A_j = area of any neighboring patch j d_j = distance between edges of patch i and any patch j
	<u>Isolation of patches</u>
$D = \Sigma(\sigma_x^2 + \sigma_y^2)$, where	D = index of isolation of all patches present, with patches located on an x, y grid. The average location and variance for all patches are calculated for the y coordinate. σ_x^2 = variance of x coordinates σ_y^2 = variance of y coordinates
	<u>Dispersion of patches</u>
$R_C = 2d_C (\tau/\pi)$, where	R_C = index of dispersion d_C = average distance from a patch center (or centroid) to its nearest neighboring patch τ = the average density of patches. $R_C > 1$ (up to 2.149) for regularly-distributed patches. $R_C < 1$ is a measure of aggregation.

unless the specific habitat requirements of wildlife species can be associated with GIS-measured variables, the success of the modeling effort will be limited.

Another approach involving a pattern-recognition algorithm to track and recapture what a map looks like from digitized data was outlined by Potter and Kessell (1980). They presented a model that calculates pre- and post-disturbance mosaic diversity indices by habitat type and age class as well as by wildlife species importance values. It responds to any grid size and user-determined environmental stratification system.

Potter and Kessell's (1980) model can calculate the mosaic diversity of an area that is recorded as patches of specified size and will distinguish mosaic patches. The algorithm used is an optimal scanning procedure in which contiguous grid cells with similar characteristics (whether they be landtype, community type, age class, or species occurrence) are enumerated. It also has the potential of providing the manager with a tool for determining the resolution level at which data must be sampled to achieve a desired planning level.

Potter and Kessell (1980) used standard measures of within-community diversity (Shannon-Weaver 1949, Simpson 1949) over the entire mosaic to obtain a quantitative estimate of both the number and heterogeneity of patches and their utilization by a wide range of animal species. They tested the program using western Montana habitat types stratified into 6 age classes in conjunction with wildlife distribution and habitat use data. Such a model may serve as a 1st-approximation vehicle that can relate plant and animal populations to the spatial and temporal arrangement of managed forest communities.

A criticism of Potter and Kessell (1980) is that they used diversity indices, which are conceptually flawed (Karr 1987), because the interdependence of richness and abundance confounds resulting information on species function in a community. Also, to be of widespread use, their model would need to incorporate algorithms for successional development. Thus, their model, with certain modifications, could be useful to the TFW process.

GIS systems, combined with computers, expand the ability to integrate spatial and temporal (using successional models or management alternatives) information fully into research planning and land management. Berry (1987) discussed a fundamental approach to computer-assisted map analysis that treats entire maps (landscapes) as variables. The set of analytic procedures for processing mapped data forms a mathematical structure analogous to traditional statistics and algebra. The procedures discussed are available for personal computer environments.

For example, one might visualize a stack of "floating maps" with common spatial registration, allowing the user to "look" down and across the stack (to represent changes in both temporal and spatial factors). Berry (1987) noted spatial statistics seek to characterize the geographic pattern or distribution of mapped data, by describing spatial variation instead of distilling data using central-tendency statistics. Spatial statistics incorporate locational information in mapping the variation in values (e.g., location of high- and low-density blue grouse or goshawk populations relative to forest successional stages).

In GIS processes one can combine information provided by traditional statistics and spatial statistics for decision making. A multiplan spreadsheet package is available which allows users to define inter-relationships among variables (e.g., distribution and connectivity of patches and presence or abundance within a bird guild). Computer-assisted map analyses also force the user to consider carefully the nature of the data being processed, enabling decision makers to understand more fully the analytic process and comment on model weightings (such as HSI values) or erroneous assumptions. Thus, computer-assisted GIS analyses encourage broader involvement in the analytic process.

Expert Systems--Generating New Predictive Classifications

It may be useful for CMERC to consider developing expert systems designed for integrating wildlife-habitat relationship models with GIS programs. Expert systems are used frequently in the practice of medicine for disease-diagnosis. Expert systems may become the next generation of predictive modeling that rapidly integrates current knowledge with information from field studies. An expert system is a computer-based consultation program consisting of "facts" and expert knowledge to help classify, diagnose, or plan. In current programs of database/wildlife habitat capability systems, users must ask pertinent questions of the system and then develop lines of reasoning. In expert systems, the computer does much of the querying and reasoning by using built-in rules.

Marcot (1986a) described the basis for developing and applying expert systems to wildlife conservation problems. Expert systems use facts and "if-then" choices, or rules, to solve management problems. One can develop networks of hundreds of rule statements. An expert system keeps track of internal reasoning processes, handles uncertainty and rules of thumb in computations, and revises its own database and logic structure from experience. Predicting the response of wildlife species to habitat conditions and prescribing management activities for mitigation/enhancement are two functions of expert systems which can help biologists and planners.

Marcot (1986a) noted that an expert system that predicts wildlife response to habitat conditions should:

- (1) identify species which may occur together under general habitat conditions;
- (2) evaluate response of a species or a set of species to changes in habitat;
- (3) suggest which habitat attributes would best predict species' patterns of abundance;
- (4) allow users to offer information as well as prompt the user for information;
- (5) give a rationale for hypotheses or conclusions reached;
- (6) be designed to be updated with new facts and rules; and
- (7) prescribe habitat conditions and recommend methods for creating these conditions to maintain or enhance particular species.

Marcot (1986a) further pointed out that quality control should include field testing of predictions and peer review of adequacy and accuracy of the facts, reasoning processes, and controls used. Furthermore, validation also should include a test of the system's utility, that is, applicability in an actual management and decision-making environment. The greatest benefit to TFW would be in distributing existing expertise in narrowly-defined problem areas to users that require but lack such expertise. Therefore, it would be a tremendous tool for coalescing perspectives.

Davis et al. (1988) described an expert system of artificial intelligence that is adapted to the needs of natural resource managers. They described spatial inferencing for estimating fire danger in one area as a function of conditions there and in adjacent areas. Also, the Forestry Department of Australian National University recently developed a GIS with linear optimization models (FORPLAN). They plan to combine that system with an expert system that determines habitat quality of a site from vegetation structure and topography. A 2nd system integrates the expert system and database for a large number of sites in spatial arrays. It predicts wildlife distribution over large areas and compares results with areas of known distribution. Building the system helped to identify gaps in knowledge.

Expert systems contain tremendous potential for evaluation of wildlife-habitat relationships in managed forests. Further research would involve refining the ability of an expert system to "know" about spatial and temporal complexities of forest environments. Thus, Marcot (1986a) noted that expert systems would be most useful at the project or community level. Expert systems perhaps are not currently well-suited for problems of

habitat change at the landscape level, because of lack of knowledge on relationships, which precludes development of rule-sets. Developing an expert system on the scale of a state would require several years of research and development, so planners would need to carefully consider cost, need, and utility in the context of adaptive management concepts.

CHAPTER IV. SUGGESTED MONITORING PROGRAM

A broad, responsive monitoring program should be organized around clearly-defined, measurable objectives for wildlife diversity. It should be designed to evaluate hypotheses about relationships between wildlife and forest conditions, and how the relationships will be affected by alterations in forest conditions at each level of ecological organization. Such a program ideally should be designed to determine the benefits of forestry for wildlife as well as situations where forestry results in "impacts".

The TFW monitoring program should specify predictions that will be tested, describing the basis of the predictions and statements of confidence in their accuracy. Moreover, the plan should provide an objective basis for results to be used to evaluate management effectiveness and to modify management procedures if necessary (Figure IV-1).

Finally, the monitoring plan should recognize that state and private lands in the State of Washington do not exist in a vacuum: factors influencing wildlife on these lands may well be asserting their influence on lands managed by other jurisdictions (e.g., U.S. Forest Service, National Park Service, State Parks, Research Natural Areas, Wilderness, etc.). Therefore, it may be wise for the TFW Program to consider linking with federal agencies which also must monitor wildlife populations (e.g., U.S. Forest Service) and with universities conducting research on topics directly relevant to the program.

We believe the monitoring program should link measures of wildlife at the stand or project level with landscape concepts. The TFW cooperators might consider researching an overall strategy for monitoring wildlife that accounts for natural variation in population attributes, by incorporating landscape influences. However, it should be recognized that wildlife habitat assessment and monitoring remain in embryonic stages and that there are no easy ways around the inherent difficulties. In this chapter we suggest a program of monitoring which incorporates emerging concepts in landscape ecology and which blends with landscape experiments designed to improve management with experience.

In western Washington and Oregon there are some 460 vertebrate wildlife species which use forests for all or part of their life cycles (Brown 1985). These species must be accommodated within a complex set of forest conditions in a strongly seasonal environment. Substantial vegetative, geographic, edaphic, and climatic variability makes it difficult to generalize about monitoring wildlife responses to forestry practices. For example, certain habitat parameters may exert differential influences among the various physiographic provinces.

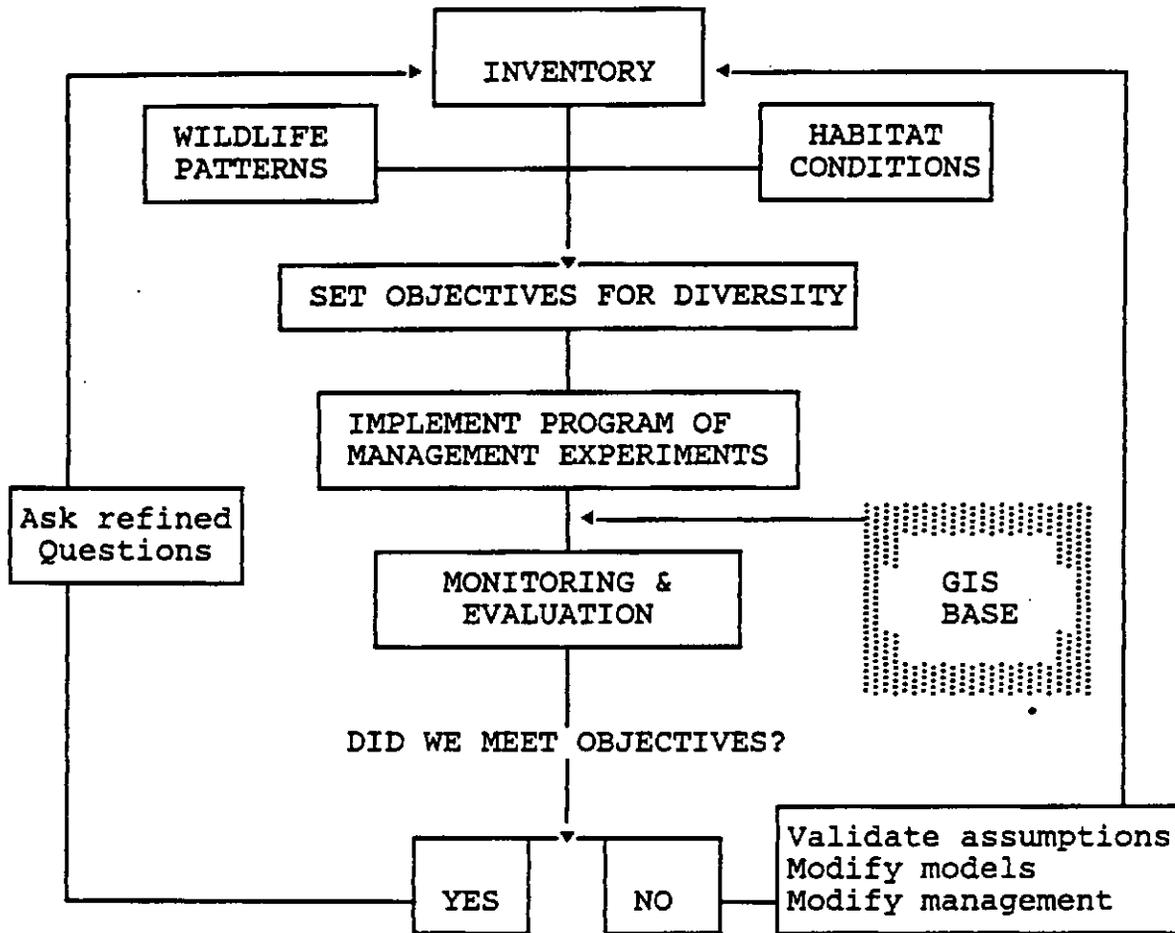


Figure IV-1. Adaptive Management Program involving feedback loops that modify forest management systems via management experiments designed to determine functional relationships between wildlife and habitat conditions within stands and landscapes of managed forest mosaics.

What variables to measure in an extensive program for monitoring wildlife responses depend upon the objectives for wildlife and habitat diversity. Thomas (1979) provided appendices of species' life forms expected in 6 seral stages of mixed forests communities in the Blue Mountains of Washington and Oregon, and Brown (1985) provided similar data for species in western Washington and Oregon. These data seem appropriate for a beginning basis for a monitoring program that includes forest-mosaic and stand objectives for wildlife diversity.

In addition, successful adaptive management requires identification of key wildlife/habitat relationships and a flexible program of monitoring and evaluation. The focus of these efforts involves evaluating vertebrate responses to various habitat parameters at the stand and at the drainage/landscape level. For these reasons, we developed two lists of parameters that influence wildlife use of managed forests from the literature review, which could be used in the monitoring program. We considered parameters from two levels of resolution: landscape (Table IV-1) and stand (Table IV-2).

We examined the matrix of parameters listed in Appendix B and scored each to reflect the following:

- (1) the number of vertebrate classes which have exhibited a relationship with the parameter in question;
- (2) whether the parameter had been identified as important to wildlife in the Pacific Northwest;
- (3) the documented strength of the relationship between species response and the parameter;
- (4) relative potential for use in wildlife-habitat models;
- (5) the type of effort required to gather data (e.g., are data available in GIS, or is additional field sampling required?);
- (6) the relative adaptability of the parameter to current timber stand inventory procedures. We summed the scores to evaluate the most important variables to be measured.

The most important habitat variables to be measured at the landscape level include stand age, edge, fragmentation indices, and stand area (Table IV-1). Other parameters listed in Table III-1 may assume greater importance as a result of ongoing research (Appendix D) and new research suggested in Chapter V.

Habitat parameters with the highest scores that relate to wildlife presence and diversity in managed stands were stand age, presence of hardwood vegetation, tree height, snag availability, vegetation cover, and food availability (Table IV-2). Other parameters that received relatively high scores but are perhaps in need of further documentation for use in monitoring wildlife

Table IV-1. Scores for determination of the relative importance of habitat variables at the landscape level.

Parameter	No. of Classes	Documented in PNW?	Parameter/response correlation	Adaptability to habitat models	Ease of measurement	Adaptable to standard inventory	Total score
stand age	1	1	3	3	2	1	11
edge	3	2	3	3	2	0	13
fragmentation	2	1	3	3	2	0	11
stand area	3	2	3	3	2	1	14
isolation	1	0	2	3	2	0	8
distance to water	2	1	2	2	2	0	9
moisture conditions	2	2	2	2	1	0	9
afforestation	1	0	2	1	2	0	6

Column scores are as follows: No. of Classes = no. of vertebrate classes (reptiles and amphibians, birds, mammals); Documented in PNW? = the no. of vertebrate classes for which research has been conducted in the Pacific Northwest (s. B.C., Canada, WA, OR, ID, n. CA); Parameter/response correlation = general strength of correlation (1 = weak, 2 = moderate, 3 = strong); Adaptability to habitat models (1 = low, 2 = moderate, 3 = high); Ease of measurement (1 = requires field work, 2 = field work not required); Adaptable to standard inventory (0 = no, 1 = yes).

Table IV-2. Scores for determination of the relative importance of habitat variables at the stand level.

Parameter	No. of Classes	Documented in PNW?	Parameter/response correlation	Adaptability to habitat models	Ease of measurement	Adaptable to standard inventory	Total score
habitat structure	1	0	3	3	1	1	9
stand age	3	3	3	3	1	1	14
patch diversity	1	1	3	2	2	1	10
tree species	1	1	2	3	1	1	9
presence of hardwoods	2	1	2	3	2	1	11
tree size (dbh)	1	1	3	3	1	1	10
tree density	2	1	2	3	1	1	10
basal area	1	0	2	3	1	1	8
tree height	1	1	3	3	2	1	11
canopy height	1	0	3	3	1	1	9
canopy cover	1	0	3	3	2	1	10
foliage density	1	0	1	3	1	1	7
tree condition	1	1	3	3	1	1	10
snag/tree size	1	1	3	3	1	1	10
snag availability	2	1	3	3	1	1	11
snag size (dbh)	1	1	3	3	1	1	10
snag condition	1	1	3	3	1	1	10
cavity availability	1	1	3	3	1	1	10
cavity height	1	0	3	3	1	1	9
hole size	1	0	3	3	1	1	9
cavity size	1	0	3	3	1	1	9
cavity permanence	1	1	3	3	1	1	10
mid-story cover	1	1	2	3	1	1	9
sapling cover	1	0	3	3	1	1	9
stem density	1	1	3	3	1	1	10
shrub density	1	1	3	3	1	1	10
vegetation cover	2	1	3	3	1	1	11
nitrogen availability	1	0	2	1	1	0	5
litter cover	1	1	3	3	1	1	10
downed logs	3	3	3	3	1	1	14
soil moisture	1	0	2	3	1	0	7
ambient temperature	1	1	2	3	1	1	9
cover/forage ratio	1	1	3	3	1	1	10
foraging substrate	1	0	2	3	1	1	8
food availability	2	2	3	3	1	0	11
human disturbance	2	2	2	2	1	1	10
distance to cover	1	0	3	3	1	1	9
nest availability	1	1	3	2	1	1	9
perch availability	1	0	3	3	1	1	9
distance to edge	1	0	3	3	2	1	10

Column scores are as follows: No. of Classes = no. of vertebrate classes (reptiles and amphibians, birds, mammals); Documented in PNW? = the no. of vertebrate Classes for which research has been conducted in the Pacific Northwest (s. B.C., Canada, WA, OR, ID, n. CA); Parameter/response correlation = general strength of correlation (1 = weak, 2 = moderate, 3 = strong); Adaptability to habitat models (1 = low, 2 = moderate, 3 = high); Ease of measurement (1 = requires field work, 2 = field work not required); Adaptable to standard inventory (0 = no, 1 = yes).

diversity include habitat structure, patch diversity, tree size, tree density, canopy cover, tree condition, snag size, snag condition, stem density, shrub density, and distance to edge.

Our scoring method may be limited in that lower scores may be a function of: (a) differential importance of parameters within vertebrate classes (e.g., snag condition is important to cavity users but not for elk), or (b) insufficient research. On the other hand, the vast majority of landscape and stand parameters we identified clearly have been associated strongly with vertebrate responses to forest management strategies. In addition, most parameters appear adaptable for use within models and are readily sampled in the field or from landscape information systems.

Monitoring also should provide special consideration of those species potentially in greatest jeopardy given prevailing population levels and land-use patterns. Managers must be concerned especially with plant- and animal habitat specialists, area-sensitive species that may decline with fragmentation, species of low mobility, and rare or endangered species prone to local extinction. While we did not review the literature seeking to define wildlife species which appear "sensitive" to changes in forest conditions, we acquired information for developing a list of forest-associated species which are of concern to the Washington Department of Wildlife (Table IV-3).

Species listed in Table IV-3 most likely cannot be monitored reliably at the stand level, due to small population size (in some cases), high expected variance, high cost of monitoring at statistically valid levels, and because their dynamics are expected to be lively. It follows that habitat measures are even less reliable indicators of effects of forest management on uncommon species. Therefore, it might be best to examine responses by sensitive species at the landscape scale. At the landscape scale, dynamics of sensitive species should be less variable, because non-breeding adults absorb most of the variance in local abundance and because population dynamics in patches (stands) are not in synchrony.

For these lists to be implemented it will be necessary to determine management guilds and conduct research to establish the relationships of the guilds to the parameters under a variety of forest management conditions and scenarios.

A responsive monitoring program could result from a hierarchical process of data acquisition. Information gathering might vary with the level of uncertainty and ability to gather precise information. For example, monitoring could determine trends accurately in at least a few species, rely on indices of population trends and habitat quality in other groups, and develop new models to predict responses by species of concern.

Table IV-3. Wildlife species of special interest in Washington which may be influenced by forest practices.

Common name	scientific name
Cope's giant salamander	Dicamptodon copei
Dunn's salamander	Plethodon dunni
Larch Mountain salamander	Plethodon larselli
Van Dyke's salamander	Plethodon vandykei
spotted frog	Rana pretiosa
western pond turtle	Clemmys marmorata
common loon	Gavia immer
great blue heron	Ardea herodias
black-crowned night heron	Mycticorax mycticorax
turkey vulture	Cathartes aura
osprey	Pandion haliaetus
bald eagle	Haliaeetus leucocephalus
northern goshawk	Accipiter gentilis
golden eagle	Aquila chrysaetos
merlin	Falco columbarius
peregrine falcon	Falco peregrinus
sandhill crane	Grus canadensis
marbled murrelet	Brachyramphus marmoratus
yellow-billed cuckoo	Coccyzus americanus
flammulated owl	Otus flammeolus
spotted owl	Strix occidentalis
great gray owl	Strix nebulosa
black swift	Cypseloides niger
Vaux's swift	Chaetura vauxi
Lewis' woodpecker	Melanerpes lewis
white-headed woodpecker	Picoides albolarvatus
three-toed woodpecker	Picoides tridactylus
black-backed woodpecker	Picoides arcticus
pileated woodpecker	Dryocopus pileatus
purple martin	Progne subis
western bluebird	Sialia mexicana
Keen's myotis	Myotis keenii
long-eared myotis	Myotis evotis
fringed myotis	Myotis thysonodes
long-legged myotis	Myotis volans
Townsend's big-eared bat	Plecotus townsendii
Pallid bat	Antrozous pallidus
red-tailed chipmunk	Tamias ruficaudis
western gray squirrel	Sciurus griseus
northern bog lemming	Synaptomys borealis
gray wolf	Canis lupus
grizzly bear	Ursus arctos

We believe that research is needed to develop management guilds, and TFW cooperators need to agree on how to incorporate mammals and herptiles. It should be useful to define faunal

assemblages via inventories of occurrence in each major physiographic province of the state. This would provide a basis for objectives for faunal species diversity. For determining management guilds for evaluation, Fry et al. (1986) recommend using the maximum number of species.

For some inferences, presence-absence data plus assessments of habitat quality may suffice, such as validated wildlife-habitat models that predict population responses resulting from habitat changes. For species which are difficult to evaluate, risk measurements, models, and expert opinion might be useful, especially using models that incorporate life-history attributes (Urban and Shugart 1986). Such a program should contain a process for addressing risk associated with managing while attempting to reduce uncertainty.

Risk Analysis.--At the time of a management decision, the decision-maker has some information concerning alternatives, states, and outcomes. This information generally is incomplete, so the manager must deal with uncertainty. Some key management decisions are calculated risks, or gambles, no matter how well biologists attempt to refine the information base (Walters 1986). Since managed forests are dynamic, managers must use a certain amount of professional judgment in evaluating all information in arriving at a decision. Although the risks of making decisions without complete information are fraught with practical difficulties, there are factors which allow some quantification of decision-making components. Such a quantification, in combination with habitat modeling that uses life-history attributes (Urban and Shugart 1986), may be useful in conducting forestry practices in areas which may contain wildlife species which are not numerous enough to allow precise estimates of population responses to management treatments.

Risk is defined as the uncertainty associated with estimates of outcomes (Lifson and Shaifer 1982). Risk analysis is a component of decision making theory that can aid in assessing and planning management programs which contain biological uncertainty. Two general types of uncertainty are described that relate to TFW monitoring of wildlife responses to managed forests. These include scientific uncertainty and decision-making risk. Scientific uncertainty refers to variations in the natural system, uncertainty of empirical information and errors in estimation, the validity of wildlife-habitat models, and the relevance of the questions that have been asked. Decision-making risk refers to the way uncertain information is used in formulating operational attitudes toward risk in making management decisions.

Conducting a risk analysis consists of estimating probabilities of random favorable and unpleasant events, estimating results from each possible sequence of decisions, and using the probabilities of chance events in calculating expected "payoffs" from each decision at any given point in time (Marcot

1986b). Management decision-makers use results of the risk analysis to help determine a risk attitude and to make the final decision. Risk analysis seems useful as a tool within adaptive management to assess management effectiveness.

In a risk assessment framework, adaptive management includes: (1) the validation of the results of the risk analysis and of its assumptions; and (2) the creation of new management objectives should the existing ones prove in some manner to be invalid. However, Marcot (1986b) cautions that even constant monitoring of a population may fail to reveal lag effects of potential extinction, and much of the risk to population viability consists of accumulated probabilities of extinction over time.

Risk analysis may be helpful in assessing scientific ignorance of natural processes, the occurrence of unexpected events, and institutional inability to actually conduct planned management activities. These factors might be analyzed by asking "what-if" questions involving error analysis or sensitivity analysis.

Summary.--We believe that developing a responsive monitoring program should include the following, in order of priority (Fig. IV-2):

- (1) conduct new research to construct management guilds (some of which could be used in ordinations to classify managed forest mosaics);
- (2) include land-type associations and traditional vegetation classification systems;
- (3) develop reliable, landscape-level, GIS-generated spatial statistics (such as those in Table III-1);
- (4) construct and validate wildlife-habitat models that predict habitat use (perhaps PATREC models, or cumulative effects models), as influenced by landtypes, habitat type, successional stage, and stand structural features;
- (5) Sensitive species listed in Table IV-3 probably can be categorized by life-history attributes that may confer sensitivity: migratory traits, vagility, habitat specialization, nesting behavior, etc. The suggested strategy is to develop information for conducting risk-analysis for uncommon or species sensitive to forest modification; and
- (6) consider development of expert system models.

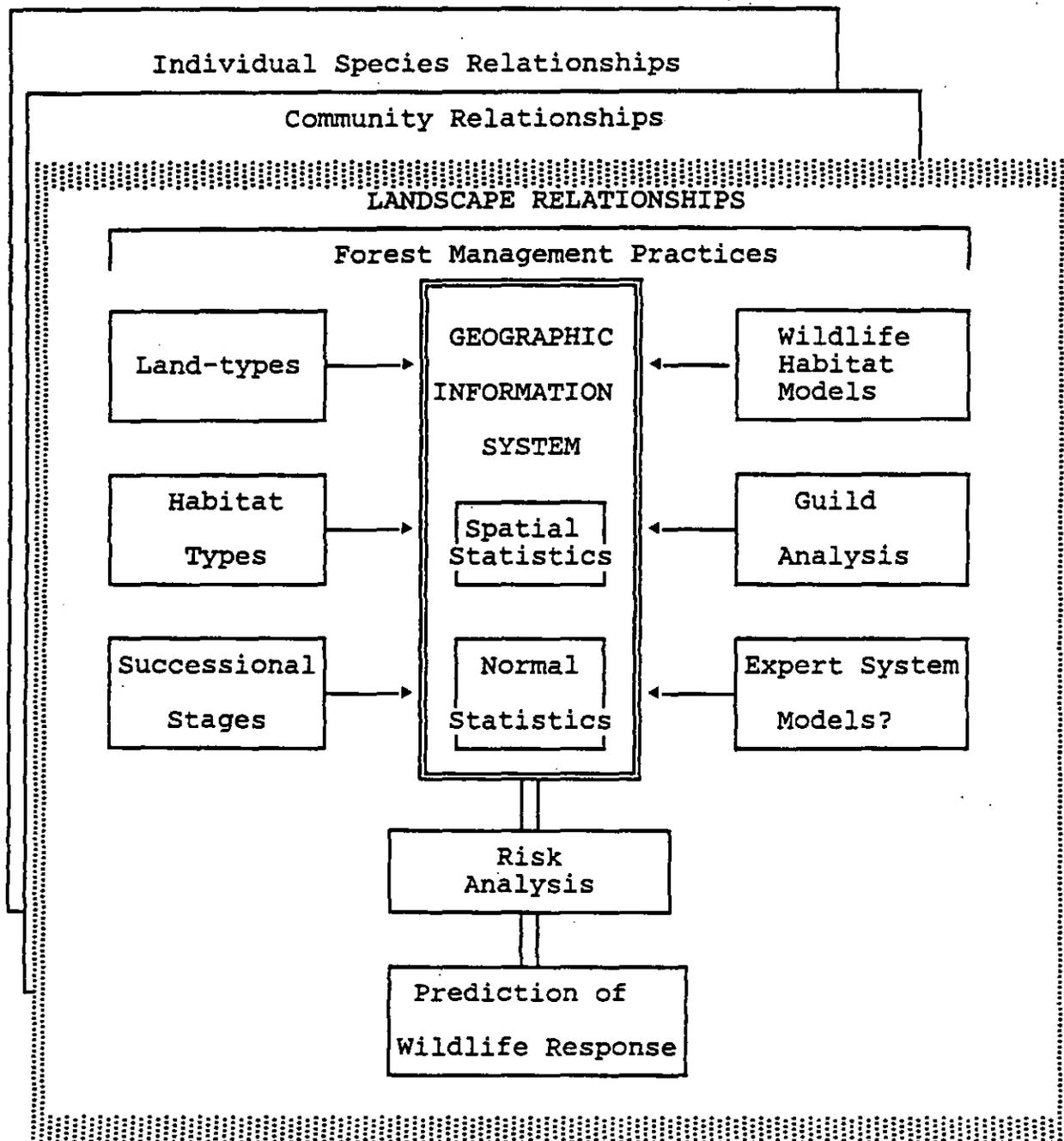


Figure IV-2. Conceptual model of information storage and analysis system for evaluating wildlife-diversity responses to forestry practices on state and private lands in Washington.

CHAPTER V. MANAGEMENT EXPERIMENTATION

Adaptive Concepts

Virtually all naturally-occurring and man-dominated ecosystems are mosaics of environmental conditions. Therefore, a systematic approach is essential for evaluating wildlife responses to habitat change and heterogeneity. With this chapter, we suggest opportunities for modified silviculture to promote greater variation among and within successional stages, validate wildlife-habitat models, and build new algorithms for evaluating wildlife responses in managed forests. In our view, a thorough quantification of natural and induced variation (as discussed in Chapter IV) is precisely the strategy which will permit adaptive management to occur.

Wildlife ecology/management is comprised of successive approximations--there is no final truth. The fact that responsive learning through management "experiments" may proceed much more quickly than through conservative management and basic research has been noticed by some managers for many years. Previously, researchers generally were isolated from the wealth of experimental opportunities afforded by whole-system manipulations conducted by managers. The TFW Agreement provides for a scientifically-based process in which management treatments can be designed to produce new scientific knowledge. In Chapter IV we described several topics for research and development related to monitoring (e.g., management guilds, GIS, landtype mapping). In this chapter, we describe a management-experimentation program designed to simultaneously evaluate management effectiveness and answer important ecological questions.

The 4 basic steps to consider (Walters 1986) include the following:

- a. Outlining management problems in terms of measurable objectives, both hidden and explicit;
- b. representing current understanding of managed systems in terms of more-explicit models that spell out assumptions and predictions clearly enough so that management errors can be detected and used as a basis for further learning;
- c. recognizing and announcing biological uncertainty and its propagation through time in relation to management actions, using statistical measures of risk and imaginative identification of alternative hypotheses (models) consistent with experience but pointing toward opportunities for improvement; and
- d. designing policies that provide for continuing resource production while simultaneously probing for better understanding and untested opportunity.

The TFW process recognizes each piece of forest (and indeed, each tree), whether in a managed or unmanaged state is wildlife habitat, and will continue to be wildlife habitat. For evaluating the quality of that habitat, the management-as-experiment process proposed for development can result in validated, predictive statements about how variations in forest treatments and conditions will influence wildlife use as well as reproduction and survival.

The state of science and consensus opinion at any time constrain the questions that are likely to be asked in adaptive environmental assessment and the range of answers that are considered reasonable. Recognition of limits and of deficiencies in knowledge should make us all more alert to evidence that does not fit into current versions of how forest systems work (Committee on Applic. of Ecol. Theory to Environ. Prob. 1986). Adaptive processes identify such evidence and exploit its potential to create new understanding.

The literature review indicated that much of the current scientific understanding for integrating wildlife and forestry resulted from somewhat simplified research designs that contained specific assumptions (e.g., clearcutting results in a specific successional sequence in a particular forest type). Others attempted to assess the short-term impacts of timber harvesting on wildlife and their habitats. Studies which did examine responses over a successional sequence purposely minimized variation within successional stages in order to maximize understanding across successional stages.

These studies suffer from an operational difficulty in that different populations are examined in different successional stages. Also, since it is recognized that a single climax forest association can have several successional pathways, clearly there is a need to examine wildlife responses to successional variation. We believe that quantifications of variation within successional stages and measures of temporal and spatial variation of habitat conditions among forest stands will lead toward greater understanding of ecological processes and fundamental wildlife diversity responses to managed forests (Figure V-1). Therefore, the most fertile questions for expanding management options can be answered through experimental designs that incorporate replicates of natural and human-induced variations in habitat conditions in managed stands and landscapes.

Forestry studies suggest that to reproduce the original tree species composition of northeastern hardwoods, it is necessary to use a mixture of even-aged and uneven-aged silviculture. The mixture of gap sizes is the same as the one that characterized the primeval forest. In Washington forests, silvicultural systems and modifications (e.g., upland management areas, riparian management zones, snags, dead and downed logs) might



Fig. V-1. Previous research designs considered succession as a homogeneous gradient from clearcutting to mature and old-growth. The TFW Program will benefit from research that quantifies variation available within each successional stage.

apply as a mixture to maintain structure and function in managed forests.

Research and Development Program

The foregoing discussions generated a series of broad, related questions which provide both a yardstick for success as well as new management options in an ongoing feedback process for research and evaluation. An underlying conceptual philosophy for the research/evaluation program is developed in the following paragraphs.

Heterogeneity is natural; management alters some processes, producing the operational difficulty of detecting, defining, and describing patches in a way that accords with patch perceptions of animals under study. We humans view structure on a scale different from most wildlife and also tend to emphasize those factors that accord with our own dominant senses, primarily vision. Animals probably perceive environmental mosaics in different ways.

Unless we adopt an animal-centered view of the environment, we are unlikely to discern or measure elements of forest system dynamics that are really important. Instead, we may well document apparent community "patterns", that are little more than artifacts, or products of our misperception of reality. Therefore, solutions must be founded on detailed knowledge of natural history of the animals, and must examine discontinuities in behavior of animals in space to reveal possible dimensions of environmental patch structure and condition as perceived by the animals.

Replicated management experiments, as outlined by Holling (1978), Romesburg (1981), McNab (1983, 1985), Walters (1986), and Eberhardt (1988) are the cornerstone of the program. The complexity of human and natural interactions probably has already resulted in numerous natural experiments. Natural experiments can be exploited to reveal forestry practices which already have resulted in suitable habitat for certain wildlife in managed forests. Moreover, we suggest options for creating stand and landscape configurations to achieve specific management-research objectives. We firmly believe that manipulating forest conditions for experiments is one of the greatest opportunities within the TFW Program (Figure V-3).

We encourage designing stand- and landscape-level evaluations of wildlife responses to manipulations that alter forest processes in a variety of ways: for example, small-patch cuts (< 20-ac.), shelterwood cuts without final entry; individual tree selection, fertilization effects; responses to pre-commercial and commercial thinning; determining variance in wood volume and debris volume relative to energy budgets of wildlife. Therefore, the suggested strategy employs combinations of natural experiments, experimental manipulations, short- and long-term

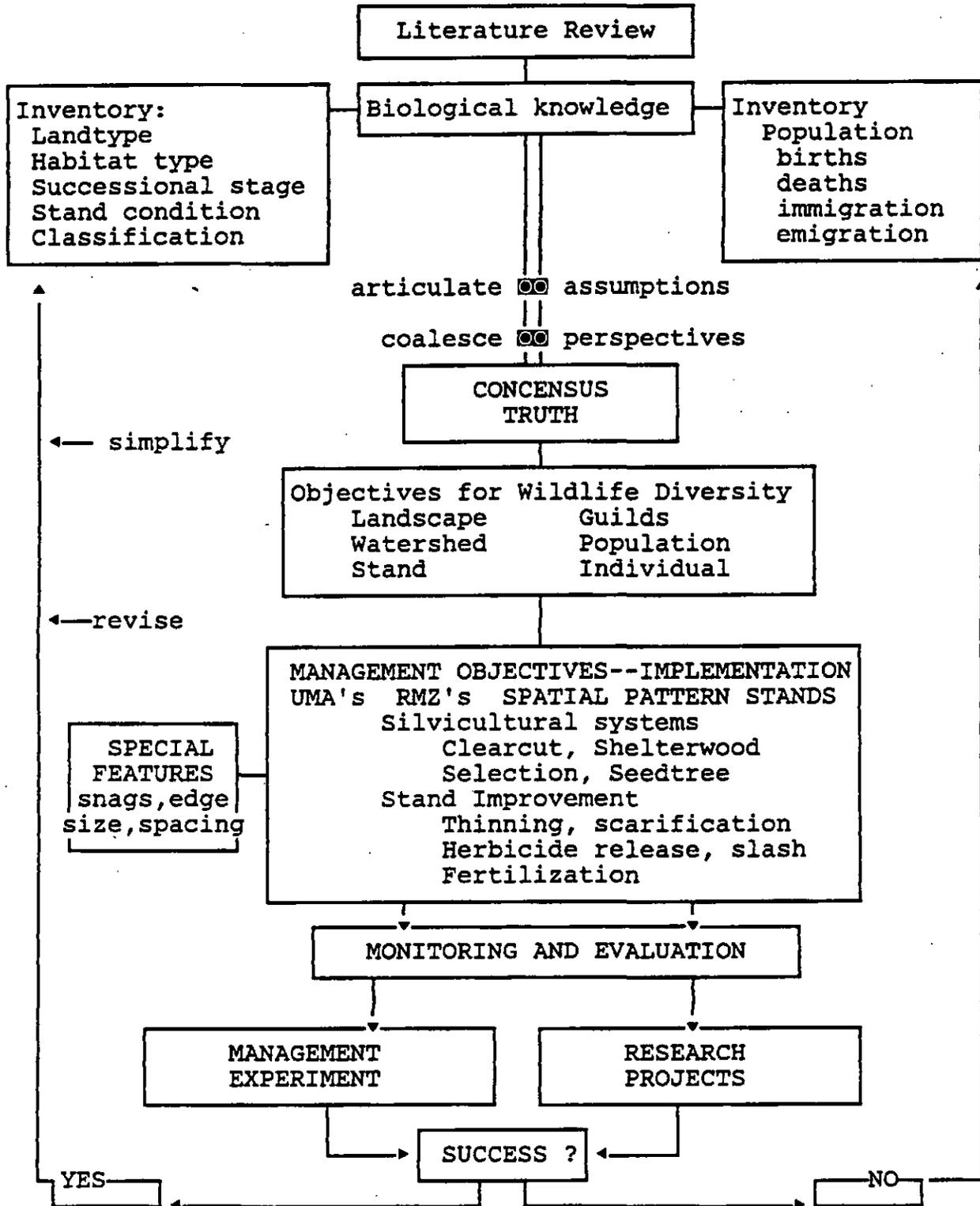


Figure V-2. Process of management experimentation and research with feedback loops for adapting forest practices.

studies, and extensive as well as intensive investigations. The process should be viewed as option-creating.

When one reflects upon the multi-dimensional matrix of possibilities, the number of management-experiment combinations seems endless. For example, landscape diversity has been influenced by land-forming processes, natural disturbance factors (e.g., wildfire, insects, diseases, windthrow), natural physical and biological features (e.g., species composition, vegetal succession, climate, soils), and the 4 silvicultural systems, modified by intensive procedures (e.g., fertilization, thinning, genetic controls). Further variation can be added by scale (e.g., size of harvest unit), arrangement, juxtaposition), and the manner in which logging is conducted (e.g., tractor, skyline, high-lead, balloon, helicopter).

Gaming Models--Development of gaming models can summarize existing information on specific topics (Grant 1986, Starfield and Beloch 1986). Such models make use of computer simulation to test "what-if" questions for management and to suggest new research. For example, Hobbs (1989) developed a gaming model for managers to predict winter mortality of mule deer in Colorado. Hobbs concluded that the most cost-effective management for mule deer would be to improve availability of high-quality forage. We suggest TFW cooperators consider contracting for modeling for sensitive species, using life-history traits, similar to Urban and Shugart (1986).

Topics for Research

The TFW CMER Committee might consider developing a series of management experiments designed to address questions of relationships between wildlife populations and habitat conditions, both at the stand level and at the drainage-landscape mosaic level. The general underpinnings of the research program include probing basic determinants of habitat selection and population regulation among vertebrate wildlife.

An operational GIS allows characterization of spectral signatures of certain habitat conditions which have been ground-truthed within managed forests. The GIS then can be used for inquiries about the distribution of those habitat conditions over the landscape. After management guilds have been developed (as suggested in Chapter IV), the GIS, in concert with preliminary wildlife inventories, landtype mapping and spatial parameters, can be used to delineate replicate landscapes of forest mosaics within each physiographic province in the State of Washington. The replicate landscape mosaics would provide the basis for a series of management experiments. In the sections below, we describe topics that appear important for research investigation; no order of priority is implied.

Landscape-level Topics.--Several questions and topics can be developed that address questions associated with landscape

ecology. Primarily, they include the process of forest fragmentation as it relates to wildlife, and the concept of a shifting mosaic of dynamic forest patches subject to relatively systematic, human-directed changes and random probability of natural disturbance:

- (a) What is the significance of the ratio of managed (disturbed) patch size to the average home range size of the affected wildlife?
- (b) What is the significance of the frequency of patch change (management) to the average lifespan of vertebrate wildlife?
- (c) Does species composition at sites where disturbance is concentrated in time (e.g., large fires) resemble species composition at sites where disturbance is concentrated in space?
- (d) Low abundance may contribute to a reduced frequency of forest-patch occupancy; what is the relationship between population size, rate of change in population size, and occupancy of a managed forest mosaic?
- (e) What landscape features can be used as a basis for predictions (or models) of seasonal habitat use (and forest-patch occupancy) in relation to patch size, isolation, and within-patch dynamics?
- (f) Franklin and Forman (1989) wonder if altering the size or spatial configuration of the timber-harvest regime might reduce fragmentation effects by minimizing the amount of forest-opening edges. They hypothesize that aggregating rather than dispersing cutting sites, may be more desirable. Larger, contiguous cutover areas might allow silvicultural prescriptions to create more heterogeneous conditions (e.g., with green trees, snags, and down logs). What are the likely relationships among vertebrate wildlife within a context of extensive managed forests that are adjacent to unmanaged forests?
- (g) What is the relative importance to wildlife of habitat that creates corridor-like connections between patches of primary habitat?
- (h) What is the optimal spatial relationship between Upland Management Areas and Riparian Management Zones for maintaining wildlife diversity?

Stand-level Topics.--Numerous topics and questions can be generated that relate to on-site or project relationships between wildlife and managed forest stands. These vary from modifications in logging procedures to developing better understanding of successional pathways in forests.

- (a) What are the relationships between wildlife populations and communities with nutrient enrichment via sludge/fertilizer amendments that promote forest growth?
- (b) Will slash-piling enhance winter use of clearcuts by mammals that normally are found in old-growth forests?
- (c) What physical and stand parameters can be used to predict the size and number of snags and/or downed logs that might be available for wildlife use in managed stands?
- (d) What silvicultural practices are needed to create snow-intercept cover in managed forests that are used as winter range by big game animals?
- (e) What is the functional relationship between variation in abundance of wildlife species (or management guilds) and structural characteristics of vegetation in managed stands?
- (f) What factors limit abundance of vertebrate wildlife in managed stands?
- (g) What are the successional pathways that occur in managed forests within major forest vegetation associations in Washington?

APPENDIX A. REVIEW OF LITERATURE ON PARAMETERS INFLUENCING WILDLIFE USE OF MANAGED FORESTS IN WASHINGTON

We present a synthesis of literature that addressed wildlife-habitat relationships primarily at watershed- and stand-level interactions with managed forests. The reader should quickly note that variable life-history traits among vertebrate wildlife cause any scale hierarchy to require some interpretation: the "landscape" for interactions among amphibian populations may only be part of the home range for a mammal the size of a grizzly.

REPTILES AND AMPHIBIANS

Landscape-Level Relationships

Scant information is available on parameters which influence responses of amphibians or reptiles to forest treatments at the landscape level in the Pacific Northwest. The following summary reveals some responses to the geography of treatments.

Rosenberg and Raphael (1986) examined responses of amphibians to forest fragmentation at the plot-, stand-, and 400-ac (1000 ha) level in northwestern California. They described their study area as a web of interconnected old-growth forest patches. Amphibian diversity measured in 0.1-acre plots decreased with increasing distance from clearcuts and was directly proportional to the amount of clear-cut edge. At the 400-ac level species richness was directly proportional to the length of clearcut edge present, but was not related to the percentage of clearcut area.

Individual species responded differently to fragmentation in Rosenberg and Raphael's (1986) work. The abundance of rough skin newts was correlated positively with the total amount of stand/clearcut edge within 400-ac blocks. But abundance of Ensatina in forested plots was not related with either the percentage of clear-cut areas within 400-acre blocks or the total clear-cut edge.

Rosenberg and Raphael (1986) found no correlations between reptile diversity and any of the parameters measured. At the 0.01-ac plot level, however, distance to clear-cut stands was related positively to abundance of sagebrush lizards. And presence of adjacent hardwoods in forested stands was correlated negatively with abundance of southern alligator lizards. Density of Ensatina in 0.1-ac forested plots was related positively with proximity and length of clear-cut edge in the stand (Rosenberg and Raphael 1986).

Stand-Level Relationships

Several researchers examined herptile-abundance response to forest seral stages, although the studies did not focus on forest practices. In general, herptile species diversity, numbers, and biomass are lower in clearcuts (Bury 1983) than other successional stages, and amphibians are more numerous in forested stands as compared to clearcuts (Raphael 1988). Diversity of salamanders and the abundance of 1 species (Ensatina) have been correlated positively with increasing forest-stand size.

Welsh and Lind (1988) observed that herptile species diversity and abundance were greater in old-aged stands than in younger stands in northwestern California. Similarly, Raphael (1984, 1988) observed that salamander density increased with stand age, but salamander species richness was equal among stand-age classes, and the average number of reptiles was lowest in older forests. Also, Bury and Corn (1988b) and Aubry et al. (1988) found that species richness did not vary with succession in western Washington.

On the other hand, a series of studies in northwestern California, southwestern Oregon and western Washington found that increasing stand age was related to increased abundance of some individual amphibian species. These included Del Norte salamanders (Raphael 1984, Raphael 1988) and tailed frogs (Bury and Corn 1988a), which were rare or absent in clearcuts. Larch Mountain Salamanders were found only in moderately wet old-growth forests in the Washington Cascades (Bury and Corn 1988b), and were present in forest plots but absent for 3 years from an adjacent clearcut on a talus slope in the Columbia River Gorge (Herrington and Larsen 1985). Pacific Giant Salamanders occurred in 50% of old-growth plots in northwestern California, but were absent in 6-10 year-old clearcuts (Bury 1983).

Stand age was not related to presence of other species, including northwest salamander (Raphael 1984, Bury and Corn 1988a), western redback salamander (Aubry et al. 1988), and rough skin newt (Raphael 1984, 1988, Bury and Corn 1988a, Welsh and Lind 1988). Some species declined with increasing stand age, including the western skink, sagebrush lizard, western fence lizard (Raphael 1984, Bury and Corn 1988a), and Pacific tree-frog (Bury 1983, Raphael 1984, Bury and Corn 1988a). However, Welsh and Lind (1988) speculated that observations of western tree frogs in younger forests may be influenced by proximity to breeding areas and higher visibility than in old-growth stands. And Bury and Corn (1988b) hypothesized that migration of adults to and from breeding sites obscures habitat relationships.

The studies listed above suggest that herptile relationships with successional stages may vary among geographic regions and among studies. For example, Ensatina were more abundant in older forest stands in 3 studies in northwestern California (Bury 1983,

Raphael 1988, Welsh and Lind 1988), but their abundance was not related to stand age in Oregon (Bury and Corn 1988b) or the Cascades of Washington (Aubry et al. 1988). Bury (1983) found clouded salamanders were more abundant in clear-cut stands than in old-growth in northwestern California, but Raphael (1984, 1988) and Welsh and Lind 1988 observed clouded salamanders increased with increasing stand age in northwestern California. Finally, abundance of black salamanders increased with stand age in 1 study (Raphael 1988) but not in another (Welsh and Lind 1988), and California slender salamanders were more abundant in young (< 150-yr) stands in 1 study (Welsh and Lind 1988) but more abundant in old-growth (vs. clearcuts) in another (Bury 1983). Possible reasons for these differences are not evident.

Other Parameters

Moisture gradients (dry-mesic-wet) influence species diversity of amphibians within old-growth stands. Fewer species are present in the wettest sites in western Washington (Bury and Corn 1988b), and fewer were found in dry vs. mesic forest stands in northwestern California and Southwestern Oregon (Welsh and Lind 1988).

Individual species respond separately to moisture gradients. The southwest salamander and black salamander are not influenced by moisture gradients. Clouded salamanders are more abundant in mesic sites than in dry sites (Welsh and Lind 1988). Ensatina is less abundant in the wettest sites (Bury and Corn 1988b), the Larch Mountain salamander is found only in moderately wet sites, and western redback salamanders are found only in moderately wet and wet sites in the western Cascades of Washington (Bury and Corn 1988b). Pacific tree frogs exhibit greater numbers in mesic forests than in other types (Welsh and Lind 1988).

Also, the amount and status of downed woody debris influences the presence and abundance of some, but not all, species of amphibians. Salamander density increased with total volume of downed woody debris in western Washington (Aubry et al. 1988) and northern California (Raphael 1984, 1988). For example, clouded salamanders used crevices and bark under downed logs in clearcuts in the redwood region of California. Clouded salamanders persist in clearcuts as long as adequate numbers of Class 2 logs are retained (Raphael 1988). California slender salamanders are favored by crevices and loose bark of downed woody debris in old-growth forest plots (Bury 1983). Bury and Corn (1988b) found Oregon slender salamanders were associated with coarse woody debris in old-growth forests of the Western Cascades of Oregon, and they correlated Ensatina abundance with density of Class 4 and 5 logs.

NONGAME BIRDS

Introduction

Fragmentation.--An extensive literature is available which quantifies forest-habitat responses of birds. Recent evidence, primarily from eastern deciduous forests, suggests that landscape fragmentation may cause dramatic changes in the abundance of regional avifauna (Forman et al. 1976, Galli et al. 1976, Whitcomb et al. 1977). This topic has been investigated in northern Europe (e.g., Haila 1986) and the Pacific Northwest (Rosenberg and Raphael 1986). However, research designed to evaluate responses to fragmentation requires further development (Verner 1986).

A clear distinction should be made between (1) changing spatial structure (fragmentation), and (2) decreasing total area (land conversion) of forests, as potential factors affecting bird population sizes (Haila 1986). In present forests, the total forested area is not expected to change significantly, although a relatively stable proportion may lack trees temporarily following timber removal. Fully managed forests should contain a continuum of stands in various seral stages in which mature and old-growth forest tracts, such as Upland Management Areas, are surrounded by habitat which is variably suitable to forest-interior birds.

Fragmentation is best evaluated from a landscape scale. Bird populations "isolated" in forest fragments surrounded by extensive tracts of young, regenerating stands, are part of a larger dynamic system of regional populations (Haila 1986).

Developing a better understanding of avian responses to fragmentation will require evaluation of processes at various levels of resolution. This is true partly because physical parameters that influence species distributions and populations are not understood completely. Responses to various edge and area conditions are topics that have been investigated relative to forest fragmentation.

Variable avian responses to fragmentation have been noted in Finland. Helle (1985) observed that sedentary hole-nesters strongly decreased even in an area where "old forest" (150 yrs) was adjacent to extensive "virgin" forests (no age given) in a national park. Virkkala (1987) found that fragmentation increased the densities of 3 species, but northern taiga-forest species declined. He also observed that densities of "old-forest" (no age given) species were nearly the same as in the 1940-50s, although total numbers in Finland declined since that time.

Parameters Influencing Nongame Birds at the Landscape Level

Area.--Following MacArthur and Wilson's (1967) theory of island biogeography, a vast ecological literature developed regarding the relationship between habitat-island size or area and the number of species present. This relationship is observed in forests that are divided into woodlots of various sizes (Galli et al. 1976). Now, it is more important to understand biological processes that drive observed patterns of bird abundance in different sized forest tracts than to define the relationship (Haila 1986). While the type of forest treatment can influence avian use of an area, in many cases the extent of the modification may be more important (Blake 1982).

The literature reveals the following about species-area relationships (total number of species as a function of habitat area): (1) the species-area relationship is non-linear and can take a variety of forms; (2) each species has a particular relationship; (3) the underlying factors that determine the relationship vary; and (4) chance is important in determining community relationships. For example, Moore and Hooper (1975) found that nearly all species in English woodlots increased in proportion to the area of woods and this increase varied with each species.

Raphael (1984) found a positive correlation between forest-tract size and bird species presence in northern California; the random combination of species in 25-ac plots could duplicate those present in larger blocks. This also was found to be true for various eastern deciduous forest studies (Galli et al. 1976, Forman et al. 1976, Whitcomb et al. 1977). However, Raphael (1984) cautioned that greater sampling effort in larger stands can explain observed increases, as Woolhouse (1983) observed in Britain. Nevertheless, species-diversity relationships within Washington's managed forests may be influenced, in part, by size of forest patches.

The availability or juxtaposition of minimum suitable habitat area required for successful breeding influences occupancy by birds. While some species may be able to use several smaller fragments, provided that the fragments are sufficiently close (Anderson and Robbins 1981), others likely would be absent from fragments smaller than the minimum required area (Moore and Hooper 1975, Galli et al. 1976, Raphael 1984). In Finnish forests, an increase of smaller birds concomitant with a decrease of larger birds indicated an association between territory size (associated with body size) and forest fragmentation (Helle 1985). This concept may relate to voluntary Upland Management Areas used in TFW.

Isolation of stands can influence species richness and diversity. In northwestern California, the number of species and their densities were influenced by the degree of stand isolation. The response of individual species varied, with 8 decreasing and 17 increasing with stand insularity (Rosenberg and Raphael 1986).

Presumably, increase in structural complexity and close mixing of habitats influences avian communities. Derleth et al. (1989) hypothesized that greater patchiness was associated with greater richness and diversity in Maine avian communities.

Edge.--The amount of forest edge has a strong influence on the density and diversity of forest birds. Forest edge, regardless of stand size, reduces the area available for species which use interior portions of forests (Helle 1985). Small stands (e.g., < 0.5 ac.) are essentially edge habitat, because all points are near the edge. Consequently, small stands are influenced more by edge than large stands (> 100 ac.). In Finland, the density of breeding birds was about 25% higher at the edge than in the interior of forests (Vickholm 1983, in Haila 1986). Similar findings in small New Jersey woodlots reflect the greater abundance of species not usually found in forest interiors (Forman et al. 1976). In northern California, avian species richness increased significantly in more fragmented stands and in plots containing more edge (Rosenberg and Raphael 1986). Species which appeared sensitive to fragmentation in Rosenberg and Raphael's (1986) work included the spotted owl and pileated woodpecker. The sharp-shinned hawk showed trends toward sensitivity to fragmentation.

Habitat fragmentation can change relationships between non-game birds and nest predators. With increasing fragmentation and reduction in forest-unit size, more nests may be exposed to potential predation. If nest predators tend to stay near the edge of woodlots, ground nests in the interiors of small forest stands are more accessible to them. In addition, interior-forest nests likely are placed in vegetation less dense than that near the woodlot edge (Chasko and Gates 1982). Artificial, open, nests placed in small woodlots had higher rates of predation than artificial nests placed in larger tracts (Wilcove 1985). However, Ratti and Reese (1988) were unable to demonstrate a nest-predation effect in small woodlots in northern Idaho.

Forest fragmentation may increase the risk of nest parasitism for some birds. Cowbirds, an obligate nest-brood parasite, may have increased in numbers in association with increases by ecological generalists, and edge and farmland species (Brittingham and Temple 1983). With decreasing woodlot size, a larger proportion of forest habitat becomes available to brown-headed cowbirds. Brittingham and Temple suggest that forest-interior birds in small woods declined as a result of increasing cowbird pressure and predict that if cowbirds continue to increase in fragmented eastern deciduous forest, many forest-dwelling songbirds will continue to decline.

The density and richness of bird communities were found to depend on the condition of edge in a Maine study (Small and Hunter 1989). Non-maintained edges held high densities of birds, while maintained edges had lower density and diversity. The maintenance of an abrupt boundary apparently prevented the

development of structural complexity capable of sustaining higher densities and numbers of species.

Stand-level Relationships

Clearcut.--Clearcut size and age-class frequently are discussed as major determinants of avian use, although variation occurs among species. Total densities of birds were not different among 5 age-classes and 3 size-classes of clearcuts in aspen forest in southwestern Colorado (Scott and Crouch 1988b). After treatment, total numbers of birds were lower in clearcuts than at edges, but there was no change in numbers in the control plots or the strip of aspen left between clearcuts. There were fewer cavity nesters on clear cuts (6% of birds), than on the control plots (33%), leave strips (34%), and edges (39%).

Clearcut age influences avian responses, and delays in revegetation influence conditions that attract certain birds to clearcuts. For example, bird populations in Utah lodgepole pine stands showed differences in preference for stands 17-37 years after clearcutting (Austin and Perry 1979). All species were present in stands that were logged, but only 6 species apparently were not influenced by the treatments. Several species were attracted to the treatment stands and several preferred them.

Height and cover of foliage influence density of bird populations in clearcuts. In western Oregon, increasing cover and height of deciduous trees accounted for 74.5% of the variation of habitat use in clearcuts (Mannan and Meslow 1984). The total density of birds decreased with increased height of conifers, but increased with increased cover of deciduous trees. Overall density increased when patches of deciduous trees formed breaks in plant communities dominated by shrubs and conifers.

The availability of standing and fallen dead wood material influences bird populations in forest clearcut areas. In northern California, the number of individual birds declined in a stand after clearcutting, but increased to a level greater than that found in the original forest by the 3rd year following harvest (Hagar 1960). The numbers of woodpeckers increased in association with the availability of dead and dying trees. Winter wrens were common in the weed/brush stage, and abundant when culled logs were present.

Partial Harvest.--The availability of snags influences presence of snag-using birds. McClelland (1980) examined responses in control, shelterwood (sawlog, intensive log, intensive tree, near complete cut), group selection, and clearcut treatment plots in Montana. Bird responses varied across the treatments. Woodpeckers were observed feeding most often, in descending order, in control, shelterwood, and uncut islands. The near-complete treatment area received little feeding use, in contrast to the other shelterwood treatment areas. There was little feeding in the clearcut area. Most nesting occurred in

the old-growth habitat, but birds foraged in adjacent cut areas.

The spacing of vegetation (openness) appears to be the most important proximate factor influencing densities of species that are more abundant in managed forests than in old-growth. For example, Mannan and Meslow (1984) compared use of an 85 year-old stand thinned in 1971 and stands greater than 200 years old in northeastern Oregon. The managed stand supported more breeding birds, largely due to the high numbers of dusky flycatchers, chipping sparrows and ruby-crowned kinglets, which use more-open areas.

Foraging substrate was found to influence densities of non-cavity-nesting, foliage-gleaning insectivores. In Arizona, higher breeding densities occurred in thinned stands than in stands never cut or cut in the 1920's (Brawn and Balda 1988). The authors noted that the amount of foliage, and hence availability of potential foraging substrate are, therefore, not reliable determinants of avian habitat quality on the plots surveyed.

Seral Stages.--Although individual species may be influenced by advancing stand age, the community as a whole may remain relatively unchanged. In northwestern California, densities of 17 species decreased, while the community showed no difference relative to stand age (Raphael 1984).

Martin (1988) demonstrated that species numbers in Arizona forests were correlated with variations in density of foraging and nesting substrates. Species numbers are more closely correlated with foliage for nesting than foraging. The results are consistent with the prediction that nest sites increase with density of foliage at nest height and that birds select habitat partly on availability of nest sites which minimize nest predation risk.

The abundance of snags accounted for the presence of most hole-nesting species in a study in northeastern Oregon (Mannan and Meslow 1984). The use of snags was greatest in older forests in western Oregon, since snags are greater in abundance in older forests (Mannan et al. 1980). In an Arizona natural area, Brawn and Balda (1988) found that high densities of most secondary cavity-nesters was probably due to a high density of ponderosa pine snags. Curiously, primary cavity-nesters were not most abundant on the natural area.

Size and condition of snags influence use by cavity-nesters. Zarnowitz and Manuwal (1985) observed the responses of 14 cavity-nesting species in 4 forest age-classes, each with and without snags, in the Olympic National Forest, Washington. Most cavity nesting species occurred in old-growth. The densities of obligate cavity-nesters increased with dbh and the density of Class II and III snags. Snag occupancy was associated with tree diameter and decomposition status in a New York study (Swallow et al. 1986).

The importance of snag size and condition also was demonstrated in a study in western Oregon (Mannan et al. 1980). The density of hole-nesting birds was correlated positively with the mean dbh of snags. In addition, hole-nesting bird density was correlated with snag condition, specifically the presence of broken tops, loss of bark, and interior decay.

Species richness, total abundance, and diversity were greater in plots with snags, compared to plots without snags, in a pine-hardwood forest in Texas (Dickson et al. 1983). The most obvious difference was the presence of cavity-nesting birds in plots with snags. However, other birds were more abundant in snag plots as well. Snags were used as perches, and as foraging sites. Four species were more abundant in the snagless plots. In Arizona, the total number of species/breeding pairs, and the percent of all breeding species/breeding pairs were lowest on a plot without snags (Balda 1975). Similarly, in western Washington, Zarnowitz and Manuwal (1985) found 13 cavity-nesting species in snag plots and 9 species in plots without snags. Snag use in New York was higher in forests with high snag densities (Swallow et al. 1986).

In another Arizona study, Scott (1979) observed similar responses to treatments in three stands (control, harvested but retaining snags, harvested and snags also cut), in ponderosa pine forest. Species composition increased 31% and bird species density increased 38% in the control stand, and 32% and 23%, respectively, in the stand where snags were left. Species composition decreased 27% and density decreased 51% in the stands where the snags were cut. However, when birds were considered on the basis of guilds, there was no difference in response.

Brawn and Balda (1988) investigated the importance of nest site availability for cavity-nesters in an experimental study in northern Arizona pine-oak forest. Sixty nest boxes were placed in each of 3 stand types (control, thinned and open). Only 3 of 6 species increased in response to the placement of nest boxes. Brawn and Balda concluded that secondary cavity-nesters can be limited by nest sites, but the magnitude of the limitation is mitigated by habitat structure: as the suitability of habitat decreases, the density of a species decreases to a point where not all suitable nest sites are utilized.

Successional changes influence species abundance, diversity, composition and density (Meyers and Johnson 1978) in loblolly and shortleaf pine stands. The numbers of breeding species increased at first, declined through middle successional stages, and then increased to new high levels about 45 years from the initial disturbance (Meyers and Johnson 1978). Shugart et al. (1978) report a similar pattern in southern forests. They also observed a tendency for diversity to increase through succession, although annual variation occurs in individual study stands. Stand age had no consistent effect on diversity of cavity-nesting bird

species in western Washington (Zarnowitz and Manuwal 1985).

Conner and Adkisson (1975) investigated avian diversity in clearcuts (1, 3, 7, and 12 years after cutting), pole stands (mean of 30 years after cutting), and mature stands (110-180 years) in the mixed oak woodlands of Virginia. Species diversity was lower in the one-year old stand, but there was no difference in species diversity among the other 5 stands. The number of birds observed in the 3, 7, and 12 year-old clearcut stands was higher than in the other stands. Most of the species present in the 1, 3, and 7 year old clear-cut sites were absent from the pole and mature stands.

Food availability may affect the densities of some birds. In Arizona, Brawn and Balda (1988) found that high densities of seeds from thick understory herbage, plus large populations of arthropods (responding to the lush herbaceous growth), were reflected by the high densities of several bird species on burned plots.

BIRDS OF PREY

Birds of prey (raptors) include birds that evolved primarily as predators or scavengers. They provide an example of convergent evolution, or unrelated animals growing to look like one another because they have the same way of life (Newton 1976, 1979, Mikkola 1983). For the purposes of this discussion, birds of prey include raptors (diurnal birds of prey), owls (nocturnal equivalents of raptors) and common raven, a predatory passerine.

The competitive exclusion principle predicts that in a given situation one species will be more efficient than another. By competing for the same limited environmental resources, an efficient species eventually will replace the other (Hardin 1960, Levin 1970). Related species often differ, either in size or habitat, thereby minimizing competitive effects (MacArthur and Levins 1964). These differences may relate to specialized habits in their use of resources, which in turn influence the number of co-existing species and the evolution of the community in general (Mikkola 1983).

Lack (1971) presented an extensive review of ecological isolation in birds. There is however, little known of the isolation mechanisms in birds of prey (Newton 1979, Mikkola 1983). Niche differences have been demonstrated for habitat selection (Sonerud 1986), nesting habitat (Titus and Mosher 1981, Moore and Henny 1983), hunting methods (Jaksic and Crothers 1985), and food (Hörnfeldt 1978, Phelan and Robertson 1978, Steenhof and Kochert 1988). Additionally, while not a niche

dimension, body features (morphology) serve indirectly as evidence of ecological segregation in food and feeding habits (Mikkola 1983). Sexual size dimorphism in hawks and owls is well known (Snyder and Wiley 1976). According to Gause's principle, (1934) related sympatric species, when forced to evade inter-specific competition, may drift apart in structural differences and adjust to their required ecological specialties. Size similarity between related species, and even species belonging to a different genus, is often enough to cause or make possible inter-specific competition among co-existing birds, (Mikkola 1983). Larger predators utilize food sizes unavailable to smaller predators, while the reverse is seldom true (Wilson 1975).

Birds of prey are usually monogamous (Mikkola 1983, Newton 1976). Normally, nest sites and food govern the distribution of breeding birds of prey. Where nest sites are widespread, many species nest solitarily in contiguous or overlapping home ranges, but where sites are concentrated in relation to feeding areas, nesting territories may be grouped. Given enough nest sites, breeding density is regulated by food availability: (a) species which live on fairly stable food sources show fairly stable densities; (b) species which live on fluctuating food supplies show fluctuating densities. Spacing behavior acts as an intermediate population-regulating mechanism, adjusting density to food supply (Schoener 1974, Newton 1976).

Most birds of prey that breed in forested habitat in North America are arboreal. All forest dwelling diurnal raptors, with the exception of falcons, build stick nests, primarily in trees. Forest owls use old raptor stick nests, other types of arboreal platforms (broken topped trees, duff piles on limbs, etc.) or cavities. Udvardy (1951), in an extensive review, found that competition for breeding places is most common among hole-nesting birds. Where nest trees or other nest sites are scarce, the presence of one species may influence the numbers and distribution of another (Newton 1976).

Predators may divide food resources simply by hunting in different ways, at different times (e.g., day vs night), and in different places (Mikkola 1983). Ecological, morphological, and bioenergetic correlates of hunting mode in hawks and owls were discussed by Jaksic and Carothers (1985).

Food resources further influence distribution and community relationships of birds of prey. Some species tend to be food generalists (they generally hunt, capture and eat the most available prey item(s), McArthur 1961), whereas others are specialists (they capture specific items, such as birds, fish, etc.). Food-niche characteristics have been documented for a number of birds of prey (Herrera and Hiraldo 1976, Phelan and Robertson 1978, Mikkola 1983, Steenhof and Kochert 1988).

There are indications that not all territories and nest

sites in a region are equally good, that birds detect these differences and that they compete more strongly and exhibit greater site tenacity at higher-quality sites (Newton 1976). In forest habitats, natural areas of varied structure and tree composition generally support more wildlife than do managed, uniform stands of conifers (Newton and Moss 1977). Habitat loss takes two forms, the reduction of a former widespread habitat to tiny fragments or the degradation of a former habitat by land use practices which lead to reductions in prey. In the 1st instance, the raptor population is restricted in distribution but, within the habitat fragments remaining, it may live at no less a density than before. In the 2nd instance, the population shows no restriction in distribution, but lives at much lower density than before.

In practice, raptor populations are affected by both forms of habitat loss. Two measures generally have been taken to counter these threats: (a) find the remaining areas of good habitat and preserve as many as possible; and (b) increase the carrying capacity of certain areas so that they will support more birds of prey than previously (Newton 1979). The TFW process would seem useful in management of some raptors via Upland Management Areas and Riparian Management zones, and by learning how to increase the carrying capacity through stand manipulations.

Documentation of relationships between silvicultural activities and birds of prey is largely nonexistent, so we examined empirical studies of habitat relationships in addition to the material presented on responses to management activities.

Landscape-level Relationships

Fragmentation.--Because raptors often occur at low densities, those in managed forest habitat seldom have been studied at the landscape level. Thiollay and Meyburg (1988) surveyed the effect of forest fragmentation on raptors on the Island of Java (> 50,000 mi²). They found that Java forest raptors showed a positive correlation between reserve (patch) size (2 - 200 mi²) and the abundance index of all species. Populations of 8-10 pairs of large hawk eagles with 8-12 mi² home ranges were considered to be at risk of extinction in forest patches smaller than 8-75 mi², apparently because of patch/population isolation. Reserves of less than 120 mi² could not support viable populations of all forest raptors. Of 10 species, only 3 survived in the 2 mi² reserve, 6 at 60 mi², 8 at 100 mi² and 9 (probably 10) at 200 mi². Similar data have not been collected in the Northwest or elsewhere.

Stand-level Relationships

Partial Harvest.--Foraging within a shelterwood harvested site by a radio-tagged spotted owl in California was reported by Solis (1983). After a prescribed burn, the owl was not observed

to forage in this area again until understory vegetation began to grow again. In southeastern Idaho, Franklin (1987) found that great gray owls preyed extensively on pocket gophers in clearcuts. Servos (1986) found that logging activity could enhance Manitoba great gray owl habitat by opening up dense stands.

Other Practices/Empirical Habitat Studies.--Anderson (1985) presented data for up to 14 years post-harvest for 90 nests/44 bald eagle territories in Oregon and Washington in pine/fir regions. By region, 61% to 77% of sites have had logging activity. Territory occupancy and fledging success were nearly identical between logged and unlogged areas. Anderson found: (a) activities were compatible when forest management activities were planned and conducted during non-critical periods; (b) nesting success is highly variable even in the absence of forest management; (c) consideration should be given to management of entire stands, not just nest trees; (d) manipulation of dbh, height, density and form through silviculture can enhance existing sites or provide new sites; and (e) the influences of weather and food supplies on nesting success are unknown, but appear important. Jensen (1988) offered an additional example of forestry/bald eagle compatibility in Montana.

Surveys of logged (24%) vs. unlogged (76%) beach front in southeast Alaska showed consistent use of available sites. This was attributed to a lack of alternate locations. Two types of impact were shown: 1) short term, where adults were forced to modify their behavior; and 2) long term, where nest sites were eliminated by windthrow. Of 136 nests 20% were lost to windthrow in one winter. The value of "beach fringe" (= buffer strip) nests was questioned because of the high loss of nests. Variable productivity observed the year after extensive windthrow was attributed to displaced pairs interfering with breeding pairs (Corr 1974).

Bald eagle surveys in British Columbia by Hodges et al. (1983) found active nests averaged 1.3/mi in "undisturbed" habitat; 49% in southern B.C.; 82% in northern B.C. "Disturbed" (logged) habitat with some remnant old-growth trees was used by eagles for nesting and perching. Areas without old-growth trees (s. B.C. = 21%, n. B.C. = 10%) were not used for nesting and were used less than their availability for perching. Age of the areas surveyed was not mentioned.

Modification of bald eagle nesting habitat (5 pairs) in northeast California by silvicultural manipulation (thinning, 20 year rotations and tree retention) was described by Burke (1983). These manipulations created a mosaic of "existing, regenerating and growing" bald eagle habitat. Unfortunately, no follow-up was made to determine if these sites remained active/productive.

In northwestern Washington, Hanson et al. (1980) indicated that clearcutting on or adjacent to winter roosts, or disturbance

via activity and/or noise (chainsaws) caused early departure or outright abandonment by bald eagles. Abandonment was attributed to direct disturbance, possible changes in micro-climate characteristics and total removal of habitat.

In western Washington, Anderson and Bruce (1980) found golden eagle nests (n=6) located on the edge of clearcuts, typically at or just below canopy height. Tree size varied from 4.9-8.5 feet dbh and 125-235 ft tall. Foraging occurred in adjacent clearcuts (<10 yrs old) for medium-sized mammals, primarily snowshoe hare and mountain beaver. Servheen (1978) observed similar foraging patterns for golden eagles in western Washington.

Melo (1979) described the effects of timber harvest modification on two pairs of osprey, one in California and one in Arizona. An active site cut in 1975 was subsequently inactive in 1976, but active in 1977. The second nest, active in 1972, was in a 40-ac management unit. Thirty percent of the basal area was removed as close as 200 feet from the nest tree via a haul road 100 feet from the nest. The site fledged two young and was "active" each year thru 1979.

Levenson and Koplín (1984) tested the effects of current human disturbance during three nesting seasons at 19 osprey nests. Three levels of disturbance were monitored: (a) minimal/low, (b) relatively constant, (c) none early to constant and intense later. There was a significant difference in productivity between levels 1 and 3, none between 1 and 2, 2 and 3 (category 1 = 1.27 fledglings, 3 = 0.40). The decline in productivity was attributed to a decline in the percent of occupied nests successfully producing young. The major form of intense activity was logging.

Garber et al. (1974) monitored the silvicultural manipulation of trees for osprey nests. Fifteen live trees were topped at 82-130 feet, where trunks were less than 2 feet. Ospreys used 1 of the 15 topped trees, but nested in 12 of 20 artificial sites.

Much has been written on spotted owls in the Pacific Northwest, but little work has been published on direct effects of forest management. Forsman et al. (1982) noted that when old-growth stands were harvested, some spotted owl pairs reacted by shifting their activities into adjacent areas that had not been cutover. The location of other pairs, apparently confronted with insufficient habitat, could not be determined. Similar examples for most forest-dwelling birds of prey are common in the literature. There appears, however, to have been no systematic effort to identify the level and type of activity or other reasons for these reactions (e.g., forest structure, prey availability).

The amount of afforestation was found to adversely affect

common ravens in England and Scotland (Marquiss et al. 1978, Newton et al. 1982). Land planted to trees offered much less carrion to ravens and after the canopy closed (7-10 years), prey species either disappeared or became unavailable. When canopy closure of > 70% within 1 km or > 50% within 2 miles occurred, clutch size reduction or desertion was noted. Levels of desertion varied, probably reflecting the overall quality of the original habitat and the alternative food sources available. Fleming and Speich (1988) hypothesized that raven numbers in western Washington apparently increased because of habitat created by forestry practices.

In Norway, raptor (8 species) use of a clearcut in winter relative to that of "older" forest (no age given), was lower than during the snow-free season (Sonerud 1986). This was attributed to lower relative availability of prey in the clearcut when the ground was snow-covered than when it was snow-free.

The sizes of raptor home ranges are quite variable and generally reflect a positive correlation between body size and territory size. Bald eagle winter range in Klamath Valley, Oregon was estimated to be 1000 mi² (Dellasala et al. 1988). The average home range of 8 spotted owls in Oregon was about 3000 ac (Forsman 1980). Average home range for 9 barred owls in Minnesota was 565 acres (Nicholls and Warner 1972). Nesting flammulated owls in Colorado had average home ranges of 40 ac (Reynolds and Linkhart 1987). Varying home range sizes in this context serve to illustrate the fact that the scale of land-use practices likely influences co-existing birds of prey in different ways.

Aspect of nest sites can influence habitat selection for some species. Goshawks at lower latitudes have been found to select nest sites in the nw-ne quadrant significantly more than other aspects, while goshawks in Alaska selected southern exposures (McGowan 1975, Reynolds et al. 1982, Moore and Henny 1983, Fleming 1987).

Elevation may limit the distribution of some birds of prey. Numerous authors described altitudinal limits for spotted owls (e.g., Gould 1977, Forsman et al. 1984). In Washington and Oregon the upper elevational limit for spotted owls varies, with higher limits in southern southern and drier forests (Forsman et al. 1984).

Distance to a permanent water source has been shown to be important for a number of birds of prey. Accipiter nests are located invariably in the vicinity of water. This may be related to micro-climate or need for drinking and bathing water (Shuster 1980, Reynolds et al. 1982, Moore and Henny 1983, Fleming 1987). In all cases, water was generally closer than 650 yd. Bald eagles and ospreys also select nest sites close to water, allowing access to preferred prey (fish). Grubb (1980) examined 218 bald eagle nests that averaged 282 feet from water. Fifty-

five percent of these nests were within 150 ft of a shoreline and 92% were within 600 ft. Barred Owls in New Jersey were found in close association with freshwater wetland forest habitat (Bosakowski et al. 1987) and similar findings were reported from other areas (Fuller 1979, Elody 1983). Swamps and marshes almost always were associated with a greater abundance and diversity of prey species.

Thermoregulation may play a role in nest site selection. In Oregon, northern goshawk nests probably received higher insolation during early hours of the day. Incubation with this species begins in April, with brooding in May. Insolation may help mitigate the effects of exposure (Moore and Henny 1983). Canopy coverage directly overhead (88%) provides shading during periods of higher temperature.

Tree species is discussed in birds-of-prey literature primarily in the context of a preferred nest, roost, perch or stand. Numerous examples exist for nest tree species selection. In western Washington, Fleming (1987) found 30 of 31 northern goshawk nests in Douglas-fir. Goshawks in South Dakota selected ponderosa pine almost exclusively (Bartelt 1974), while Cooper's hawks in Oregon exhibited similar preference (77% used/34% total) for Douglas-firs (Moore and Henny 1983). Of 218 bald eagle nests in western Washington, 70% were in Douglas-fir and 17% in Sitka spruce (Grubb 1976). Western Oregon bald eagle nest trees were Douglas-fir (74%) and Sitka spruce (23%) (Anthony and Isaacs 1981). Reynolds et al. (1987) found that virtually all reported nests of flammulated owls were in stands that contained at least some ponderosa pine. The majority of spotted owl nests in Oregon (Forsman et al. 1984) and northern California (LaHaye 1988) are in Douglas-firs. Selection or avoidance of a tree species for a nest tree is probably due to growth form and foliage patterns unique to each tree species.

Roosting and perching preference has been documented for most forest raptor species. Studies of spotted owl roost sites in California (Barrows and Barrows 1978, Barrows 1981) generally concluded that tree species was relatively unimportant. Hayward (1984) found boreal- and Saw-whet owls used dispersed roost sites, while screech-owls returned to preferred roost sites. Hayward (1983) concluded that roost sites were chosen to provide protection from predators more than thermal economy. At one bald eagle roost Douglas-fir was the only tree species selected in greater proportion than its availability, but selection was apparently related to its tendency to produce open crowns with heavy branches. Bald eagles selected trees with specific features (exposed lateral branches, large size, etc.) that were related to roosting behavior and size of bald eagles (Keister 1981, Dellasala et al. 1988). Daytime perches of bald eagles (and other birds of prey) are often dead trees, trees that emerge above the canopy, small groups of trees or other sites that overlook feeding sites (Steenhof 1976, Stalmaster and Newman 1979, etc.). Kenward (1982) found 75 of 79 observed goshawk

attacks originated directly from a perch. Nicholls and Warner (1972) found that barred owls avoided several habitat types (open fields, marshes) where perches were not present.

Stand age is a factor that influences the distribution and abundance of birds of prey. In Oregon over 90% of all spotted owl pairs located were in old-growth forest (> 200 years) and none were in forests younger than 36 years (Forsman et al. 1984). Dawson et al. (1987) felt that the quantity of old-growth in Spotted Owl home range was a determinant of home range size.

Accipiter species in Oregon occupy a single macrohabitat, but the vegetative structure associated with each successional stage creates a type of patchiness within the heterogenous macrohabitat. Sharp-shinned hawk nests occur in young, even-aged stands (25-50 years), Cooper's hawks in even-aged, 2nd growth stands (50-70 years) and goshawks in mature or old-growth conifers (150 years +) (Reynolds et al. 1982, Moore and Henny 1983). McGarigal and Fraser (1984) found great horned owls and barred owls were significantly more common in old forest (81-212 yrs) than young forest (12-72 yrs) in Virginia. They felt the reason for the difference was related to the influence of stand structure on hunting techniques and possibly nest structure requirements.

Structure of trees used for nesting, perching and roosting appears to be an important parameter that varies among birds of prey. Numerous studies on northern goshawks note that nests are usually placed in the bottom 1/3 of the canopy, most often on dead undercanopy limbs or in forks or on deformities (Dixon and Dixon 1938, Moore 1980). Fleming (1987) found that 10 of 11 goshawk nests placed in deformed/multiple-topped conifers were in stands classified as small sawtimber. He speculated that use of these deformities allowed goshawks to use small sized trees in stands that wouldn't otherwise be suitable for nesting because limb-size would be unable to physically support large goshawk nests. Cooper's hawks in Oregon commonly (64% of observed nesting) used mistletoe clumps for nest platforms (Moore and Henny 1983). Studies also document spotted owls use platform structures in trees (Forsman et al. 1984, LaHaye 1988).

Height and dbh of trees are important to some birds of prey. Goshawks generally require trees with big limbs to support their nests (but see above) and tend to use one of the larger trees on their nest sites (Bartelt 1974, McGowan 1975, Reynolds et al. 1982, Moore and Henny 1983). Fleming (1987) found that nest tree dbh at 31 goshawk nests in western Washington averaged 25-35% larger than the average stand dbh. Numerous bald eagle studies reference the use of dominant or co-dominant trees (e.g., Anthony and Isaacs 1981). These trees tend to be taller, of larger diameter and often extend above the canopy. For bald eagles, this may be important for access.

Tree density may effect foraging efficiency of some birds of

prey. Reynolds and Linkhart (1987) suggested that the specific insect-hawking foraging tactics used by flammulated owls required open areas provided by the open crowns and park-like spacing of old-growth trees. Conversely, spotted owls do not forage in shrub/sapling habitats (Forsman et al. 1984), probably because of their inability to hunt effectively in densely vegetated sites (Gutierrez 1985). This appears to be substantiated by the ability of the smaller male to forage in denser cover than females (Solis 1983, Sisco and Gutierrez 1984).

Canopy height was important in a study of nest site selection by 4 hawk species (Titus and Mosher (1981). No nests were found in areas with canopy heights < 50 ft. Canopy height was negatively correlated with the total number of overstory trees < 10 inches dbh and positively correlated with basal area.

Crown volume has been demonstrated as a habitat variable influencing nest site selection for some raptor species. Moore and Henny (1983) found that Cooper's hawks and sharp-shinned hawks chose similar sites of high crown volume (212-23000 cu ft) in younger successional stands, while northern goshawks chose older stands with lower crown volume (17,600 cu ft). Sites chosen by Cooper's hawks and sharp-shinned hawks for nesting may provide concealment from avian predators. Use of mistletoe growth and placement of nests within dense canopy support this idea (Moore and Henny 1983).

Accessibility to nests may be important and influence nest site selection. Moore and Henny (1983) found a correlation between Accipiter body size and spacing of stems and foliage. They found lower crown volumes and fewer and larger trees as body size increased between the 3 species. Newton (1986) made similar observations with two European Accipiter species. Red-tailed hawk nests were consistently located in the same relative height (77-81%) of different trees (Bohm 1978, Titus and Mosher 1981, Bednarz and Dinsmore 1982). Results support the idea that unobstructed access to the nest is important for this species (Orians and Kuhlman 1956, Mader 1978).

Structurally damaged trees can provide suitable habitat for some species. Of 47 spotted owl nests in Oregon, 30 were in cavities in damaged old-growth conifers (Forsman et al. 1984). Similarly, Accipiter hawks often choose structural deformities on which to place their nests. Fleming (1987) found 11 of 31 goshawk nests in western Washington were located on deformities, either a leader break or in the crotch of a double leader.

Snags are used for perching and/or nesting by virtually all forest dwelling birds of prey. Structure and accessibility seem to be more important than the fact that the trees are dead. Dimension of barred owl nest snags in Maryland was found to be at least 10 inches and those with cavities 30 ft or more above the ground were preferred (Devereaux and Mosher 1984). Spotted owl nest trees (n=47) in Oregon averaged 49 inches, with none

less than 29 inches. Nests were 39-180 feet high, averaging 75 ft (Forsman et al. 1984).

Cavities in living and dead trees provide nest and roost sites for many owl species and kestrels (Devereaux and Mosher 1981, Forsman et al. 1984). Forsman (1976) hypothesized that mortality of spotted owl nestlings was greater at platform nests than at cavity nests. Predation at cavity nests was found to be different for different species of European owls, primarily depending on size of the birds and entrance hole requirements. Nest predation was highest in Norwegian closed mature spruce forest where holes were > 2.1 inches. This resulted from greater accessibility of cavities by a major predator, the marten (Sonerud 1985).

Downed woody material has been shown to be important to nesting turkey vultures (Bent 1937). This species often nests on the ground, placing its nest next to a downed log or in a hollow log. Downed logs and stumps are frequently used for "plucking posts" by Accipiter hawks. These are areas where prey are plucked before being delivered to a nest (Beebe 1976, Reynolds et al. 1982).

Understory and shrub density have been shown to be characteristic of some raptor nest sites. Accipiter nest sites typically occur in stands with limited understory (Reynolds et al. 1982, Fleming 1987). Shuster (1980) found the heaviest understory at goshawk nests in Colorado was 1975 stems/ac, with an average height of 3 ft. Western Washington goshawk sites typically displayed little understory development, with 71% of Olympic Peninsula nests classified as "poor" (Fleming 1987). In Minnesota, Nicholls and Warner (1972) documented use of woodland areas free of dense understory by barred owls and felt this aided hunting by making it easy to observe, fly, and attack terrestrial prey.

Edge influences habitat selection by birds of prey. Red-tailed hawks in Ohio demonstrated high (90%) use of edges adjacent to open woods for nesting (Howell et al. 1978). Goshawks in England showed a clear preference for woodland compared with open country or deep woodland up to 650 ft from open country (Kenward 1982). This indicated that birds were selecting areas with small woods for their ranges, rather than extensive forest. Range size was significantly related to the proportion of woodland edge in each area.

While individual habitat parameters are important, it has been demonstrated that a combination of these factors operate together to define habitat selection in birds of prey. Titus and Mosher (1981) found that nests of 4 woodland hawk species could be successfully characterized using 29 quantitative variables. Their conclusion was that most trees are probably not suitable for nest placement, and nests trees may be a limiting factor in some otherwise suitable habitats.

CAVITY-NESTING DUCKS

Six species of ducks in the Pacific Northwest utilize forested habitat for breeding; all are secondary cavity users. These include wood duck, common and Barrow's goldeneyes, bufflehead, common and hooded mergansers. Baseline habitat data for most cavity-nesting ducks that occur in the Pacific Northwest are limited. With regard to timber management, the breeding season and use of cavities are of primary importance. Documentation of relationships between silvicultural activities and cavity-nesting ducks is virtually non-existent, so we examined empirical studies of habitat relationships.

The influence of stand size on occupancy has been investigated for wood ducks. Stand size used by wood ducks in Minnesota averaged 48 acres (range = 2.5-135 ac) in 60-75 year-old aspen to 62 acres (range = 5-100 ac) in 100-120 year-old northern hardwoods. An average of 38 trees/ac \geq 11 inches at nest sites was not significantly different from density at random sites (40/ac) (Gilmer et al. 1978). In Wisconsin, Soulliere (1988) found an average of 26 trees/ac $>$ 1 foot dbh at potential wood duck sites.

Canopy closure has been measured at wood duck nest sites but its influence on site occupancy is unknown. Canopy closure at wood duck nests in Minnesota northern hardwood stands averaged 63% (range = 50-80 %). Nest sites in mature aspen (60-75+ yrs) had less canopy closure, averaging 48% (range = 20-70%) (Gilmer et al. 1978).

Three studies evaluated the minimum diameter of trees needed to produce a suitable cavity for nesting by ducks. Eight inches dbh was concluded to be minimum for buffleheads (Erskine 1972). Minimum tree size for wood duck cavities in Wisconsin and Minnesota hardwood forests was felt to be between 11-12.2 inches (Gilmer et al. 1978, Soulliere 1988).

In Wisconsin, density of suitable wood duck cavities averaged 0.26/ac and were found primarily in 4 deciduous species (Soulliere 1988). In Minnesota, Gilmer et al. (1978) found an average density of 1.6 cavities/ac, but found no nests in coniferous forest.

Cavity dimension may influence relative reproductive success in cavity-nesting ducks. Eriksson (1979) found a positive correlation between clutch size and incubation efficiency of common goldeneyes in relation to nest-box size. There apparently has been no similar work on natural cavities.

Selection of nest trees by woodpeckers, the primary cavity excavators, probably influences the availability of cavities used by wood ducks and buffleheads (Erskine 1972, Gilmer et al. 1978). In Minnesota, potential cavities were clustered rather than distributed randomly, and a significant relationship existed

between orientation of the cavity entrance, the nearest canopy opening and distance to water (Gilmer et al. 1978).

Intraspecific competition may influence nest availability for cavity-nesting ducks (Erskine 1964, Boyer 1975, Savard 1988). Savard (1988) listed 7 species of wildlife encountered in nest-boxes intended for Barrow's goldeneyes in British Columbia. Boyer (1975) found 65% of natural cavities unoccupied and 62% of nest-boxes unused, suggesting low competition for sites. Erskine (1972) felt competition for potential bufflehead cavities was low, although cavities were occasionally rendered unusable by tree swallows or mountain bluebirds.

Tree species may vary in their ability to produce long-lasting nest cavities. Durability of tree cavities used by buffleheads in British Columbia was 33% for aspens and 50% for Douglas-fir after 15 years (Erskine 1977).

Lumsden et al. (1986) used nest-boxes in Ontario, Canada, to test entrance hole size, substrate preference and nest height influences on nest selection. They found that hole size did not significantly influence choice by common goldeneyes, but common mergansers selected larger holes (8.3 x 5.1 in), while hooded mergansers and wood ducks favored small holes (5.1 x 4.0 in). All ducks chose boxes with wood shaving substrates.

Cavity height may be important in nest selection for some cavity nesting ducks. In Minnesota, wood ducks used cavities as low as 13.1 feet, but the average for different types of trees was 30-35 feet. Historic common merganser nest records for the southeastern United States indicate that most were at tree heights between 10-18 feet, primarily in cottonwoods (Kiff 1989). In Ontario, common goldeneyes selected nest boxes placed high (14.8-19.7 ft) significantly more often when available, but boxes at 10 feet were used when other selections were unavailable (Lumsden et al. 1986). It is unlikely that buffleheads have any marked preference for cavity height. Height of shrubs at the base of the tree may, in part, influence nest height chosen by flickers, the primary excavator of cavities used by buffleheads (Erskine 1972).

Proximity to water may be the over-riding consideration for nest site selection by buffleheads. Virtually all bufflehead nests are located within 650 feet of water (Erskine 1972). Permanent water was located < 550 yards from 76% of wood duck nests in Minnesota; however, 24% were located farther than 0.6 mile (Gilmer et al. 1978). Ball (1973), also in Minnesota, found wood duck hens with broods as far as 2.4 miles from water.

Food resources used by cavity-nesting ducks relative to Washington's managed forests are unknown. Wood duck foraging for acorns in flooded Arkansas forest (Briggs 1978) and dispersion of breeding common merganser pairs relative to the availability of juvenile Pacific salmon in streams on Vancouver Island (Wood

1986) have been described. While a situation similar to Briggs' seems unlikely in the Pacific Northwest, forestry practices that affect fish conceivably could influence the distribution and abundance of mergansers.

Numerous papers address or describe snag-use for individual species of cavity-nesting ducks, but only 2 specifically refer to silvicultural relationships. In Minnesota, unmerchantable aspen left standing following commercial timber harvest provided cavities used by wood ducks (Gilmer et al. 1978). One-third of these stands were less than 20 acres and were situated in inaccessible areas that couldn't be efficiently logged.

Management implications for maintaining wood duck natural cavities in the north-central United States have been discussed. Retention of cavity trees and other mature/over-mature trees, some trees of different species in various age-classes, and a silvicultural approach that encourages development of natural cavities, in coordination with timber management, was felt to have distinct advantages over providing artificial cavities (Gilmer et al. 1978; Soulliere 1988).

FOREST GROUSE

Forest grouse (blue, spruce, ruffed) use forested habitat year-round (Aldrich 1963). Available literature suggests a number of habitat/habitat-use parameters exist in common for all 3 species that are relevant to forest management. Canopy closure, tree density, height of trees, etc. have rather consistent effects on grouse. Forest grouse clearly reach highest densities in forest-mosaic situations including younger successional forest and older forest stages. The optimal spatial and/or temporal composition of the mosaic would seem to be open for interpretation.

Stand type is important to all grouse, providing basic life requirements (Zwickel and Bendell 1985, Gullion 1988). Logging provides high quality ruffed grouse habitat through the control of forest age-class distribution, stand distribution and size of harvested parcels (Gullion 1988). The basic stand type varies among grouse species and, to some extent, within the same species. In some places, the preferred forms of vegetation are climax types for the region where they occur; in other places they are early seral stages (Aldrich 1963).

The 3 species exhibit specific habitat associations in North America. Blue grouse occur primarily in lowland Douglas-fir, western hemlock and western redcedar forest west of the Cascades crest in Washington, but also occur in montane, mixed coniferous forest (Donaldson and Bergerud 1974, Zwickel and Bendell 1985, Niederleitner 1986). Keppie (1987) noted that spruce grouse are distributed across North America in coniferous and mixed-wood boreal forest, typically at low densities. Generally, spruce grouse are found in lodgepole pine in western North America (Aldrich 1963, Johnsgard 1973). Gullion (1988) noted the strong correlation between aspen and ruffed grouse. Consequently, ruffed grouse enjoy the widest distribution of any resident game bird on the continent. Generally, ruffed grouse are found in deciduous, 2nd-growth habitats (Aldrich 1963).

Landscape-level Relationships

Blue grouse.--Donaldson and Bergerud (1974) found the greatest densities of blue grouse in heterogeneous vegetation consisting of a logged mosaic of several ages of Douglas-fir (≤ 40 years) in British Columbia, which apparently resulted in optimal food and cover.

Patchy openings in dense forest and increased edge in open habitats both tend to decrease the distance a bird must move to find cover or forage. Male territories (1.99 acres) in Montana contained an average 675 ft of edge (Martinka 1972). In both coastal Douglas-fir forest and interior ponderosa pine/Douglas-fir forest, preferred sites for territories tend to have maximum edge (Donaldson and Bergerud 1974). Discriminant analysis

determined that edge was the most constant variable on all territories in Montana (Martinka 1972).

Blue grouse may move several miles from breeding locations to wintering areas. Male blue grouse on Vancouver Island descend to lowlands in late March and early April and return to uplands by July (Fowle 1960). Use of higher elevations/forests is probably related to food and cover preference (Beer 1943, Zwickel and Bendell 1972, King and Bendell 1982, Stauffer and Peterson 1986, and others). Curiously, none of these studies mention distance from winter to summer range. In British Columbia, wintering sites were located closer to breeding areas than expected by chance (Hines 1987). In another British Columbia, King (1971) found adult males wintering in subalpine forest, but was unable to find adult females and juveniles. In Hines' study, males and females wintered at different elevations (above and below 1000 ft); there were no sex-related differences in habitat use by adults or juveniles. All successional stages were used, but a preference was shown for mature forest (≥ 250 years), while early successional stages (≤ 20 years) were used frequently and mid-successional stages (21-100 years) were used infrequently. Western hemlock provided 66% of the tree cover (94% total cover) at observed wintering locations in Hines' study.

Blue grouse in Idaho summered below 6900 ft and wintered at "high elevation" (≥ 7500 ft) in stands of 50% closed conifer (Stauffer and Peterson 1986). Hines (1987) postulated winter use of higher, poor-soil sites was a result of preference for higher elevations and mature forest, which were not available in lowland sites.

Ruffed grouse.--Using telemetry, Godfrey (1975) found that Minnesota ruffed grouse broods used lowlands with a diverse/profuse ground vegetation primarily for feeding 63.3% of the time. Uplands were used less often (13.4%) and 10-25 foot lowland conifers (spruce/larch) were used solely for roosting. Three studies observed that ruffed grouse avoided slopes exceeding 20° (Boag and Sumanik 1969, Porath and Vohs 1972, Salo 1978).

Stand-level Relationships

Blue grouse.--Zwickel and Bendell (1985) note that blue grouse may increase spectacularly (up to 225 adult males/mi²) in lowland Pacific Coast forest that has been clearcut. Similarly, Niederleitner (1986) found significantly higher blue grouse use in earlier successional stages (3-13 years) than mid-successional (59 years) or old-growth (251+ years). Zwickel and Bendell (1985) indicate that current logging practices can have both positive and negative effects on blue grouse. Recently-logged lowlands are colonized rapidly by "surplus" grouse (juveniles, non-territorial subadults, etc.) and populations persist until canopy closure approaches 75%. However, stand age at canopy closure is often 15-25 years. Zwickel and Bendell (1985) noted

that logging at high elevations and implications of logging winter range were unknown.

Canopy closure influences use by blue grouse in managed stands. In British Columbia the highest densities of blue grouse occurred in forests with an approximate 50% canopy closure; the lowest densities were in closed-canopy forests (Donaldson and Bergerud 1974). Another British Columbia study found heavy use of open and very open sites (4-15 year old clearcuts and burns) (Hines 1987). Study sites had between 560-1200 trees/ac, with low basal area and trees < 5 feet tall. Open-site characteristics also influenced use in southeastern Idaho: canopy closure of 32-45%, 54-410 saplings/ac, and 40-70 trees/ac (Stauffer and Peterson 1986). Optimal spring-fall blue grouse habitat, therefore, appears to contain the following: $\leq 50\%$ canopy closure, low basal area and low to moderate tree density.

Understory, ground cover and edge are functions of mosaic, canopy closure, tree density and tree height. Physical characteristics of the herbaceous layer influence use of managed forests by blue grouse broods (Mussehl 1965, Donaldson and Bergerud 1974, Hines 1977). Females with broods prefer 7-9 foot-tall herbaceous vegetation, approximately 70% ground cover, and location close to cover of shrubs and conifers. Clumps of small trees and shrubs may enhance brood habitat by providing nest sites and protection from predators. Woody cover increases in importance as chicks mature. These relationships vary with annual precipitation patterns: in dry years broods may be restricted to margins of coniferous forest or riparian zones where herbaceous plants occur (Mussehl 1963).

Openings in the canopy/shrub-layer and relative size of trees influence establishment of male blue grouse territories during the spring breeding season. In British Columbia, habitat that had the greatest density of "hooting" (= displaying) males consisted mostly of heterogeneous vegetation within a mosaic of all age classes of Douglas-fir. Irregular tree height and openings in the canopy afford better visibility for "hooting" males; visibility appeared important in detecting females that move into male territories. A moderate canopy (50%) may provide protection from weather and predators while facilitating courtship behavior (Donaldson and Bergerud 1974).

In areas with taller or more-homogeneous vegetation, blue grouse use elevated objects such as tree limbs or rocks as hooting platforms (Bendell and Elliot 1966). In Alaska, Doerr et al. (1983) found numbers of "hooting" males up to 45 times higher in old-growth forests than clearcuts (no age provided).

Tree species composition influences the selection of winter roosts. In Idaho, 95% of roosts were in Douglas-firs (57% of total conifers). Douglas-fir was also used as a food source. Large Douglas-firs with dense foliage provided food, and possibly

protection from predators and winter weather (Stauffer and Peterson 1986).

Results of nitrogen fertilization trials in younger successional forest (≤ 25 years) indicate that cover, biomass and fruit production of herbs respond significantly to fertilization. However, blue grouse breeding densities and reproductive success did not increase on fertilized areas. It was concluded that some grouse (yearlings and brood hens) were attracted to fertilized areas, perhaps because of the nutritious food and better herbaceous cover, but nitrogen levels did not seem to limit population densities (Ash 1979, Ash and Bendell 1979).

The availability of water may influence blue grouse habitat use. Several authors imply that free water is necessary for blue grouse occupancy, but do not offer evidence to support this claim (Beer 1943, Marshall 1946). The availability of water may be less important when berries and other succulent foods are available (Fowle 1960). Moisture appeared to influence distribution of hens in British Columbia. Females were most common in mesic pasture, but moisture did not seem to be a condition for brood occurrence; no broods were found in moist alder-sword fern stands (Donaldson and Bergerud 1974). In Montana, brood use of edge habitat varied with climatic conditions. During a drought year, brood distribution was restricted to margins of coniferous forest where adequate herbaceous habitat developed (Mussehl 1963).

Zwickel and Bendell (1985) summarized the dynamics of blue grouse and logging/wildfire in the Northwest: the most usual case following logging is a rapid increase in numbers, stabilization at a moderate density (38-75 adult males/mi²) until canopy closure ($\approx 75\%$), then a decline to a very low density or local extinction (Figure A-1). Observations indicate that blue grouse will not reoccupy such areas until the canopy is again opened by logging or some other agent such as wildfire.

Partial Harvest--Relationships of grouse to precommercial thinnings or logging on winter range have not been examined. Variations in slopes, aspects and altitudes will complicate interpretations. Most viable populations of blue grouse persist for approximately 25% (15-25 years) of a planned silvicultural period. Silvicultural treatments which might extend the productive period for blue grouse have been suggested: wide spacing of planted trees, not planting sites of low timber productivity, and intensive thinning throughout the forest rotation. Selective logging and small clearcuts (10-60 ac) might be beneficial in opening the canopy and allowing regeneration (Zwickel and Bendell 1985). With the curtailment of fire, logging is probably necessary to maintain blue grouse habitat. Where blue grouse breeding habitat is paramount silvicultural practices such as mistletoe control, terracing on clearcut areas and thinning probably should be discouraged (Martinka 1970).

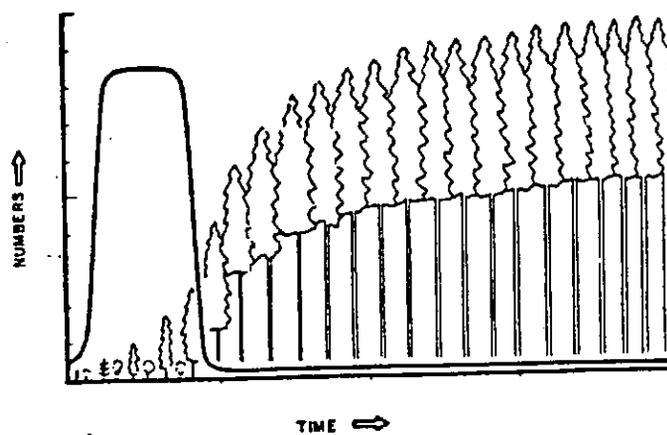


Fig. A-1. Changes in numbers of blue grouse during a clearcut logging rotation in coastal forests of British Columbia (from Zwickel and Bendell 1985).

Spruce grouse.--Spruce grouse appear to be associated with early successional, fire-induced disclimax forests (Pendergast and Boag 1971) or bogs (Aldrich 1963). Associations with managed forests are very limited in the literature (Haas 1974) and most studies have been conducted in areas of natural regeneration. While the spruce grouse's need for display areas, nesting habitat and cover from predators undoubtedly contributes to seasonal changes in habitat, the radical change from a summer to a winter diet (conifer needles) may be the primary factor determining the seasonal relationships of habitat use and forest management. Consideration of these habitat requirements may be important in understanding factors such as distribution, dispersal, migration and population regulation in this species (Allan 1985).

Spruce grouse habitat in Alberta was described by Herzog and Boag (1978) as dominated (72%) by lodgepole pine 35-48 years old, 25-50 ft tall, with 1534 trees/ac. In Minnesota, a general preference by spruce grouse for areas of stunted open-grown forest (no age given) was suggested, but reasons for the preference were not given (Haas 1974). Washington spruce grouse habitat was described as equally mixed stands of lodgepole pine and Englemann spruce. Basal area of pine and spruce represented 63% and 31% of the study area. Grouse observations were associated with sites having deadfall logs, a slope of 12-28%, shrub density of less than 10,000 stems/ha, and trees having the lowest live limbs at heights ranging from 5-14 ft (Ratti et al. 1984).

Tree density (645-915 trees/ac) and tree height (25-32 ft) influenced sites used for foraging by females and males, respectively, in Maine (Allan 1985). Similar tree densities (740-987/ac) were described in another Maine study (Hedberg 1980). Using tree vertical profiles, Maine winter habitat of both sexes had a greater density of vegetation in the middle and upper strata (80%) and less ground cover, while summer habitat had a more open canopy (70%) and greater density of ground vegetation (Allan 1985). In Washington "activity trees" (trees used for feeding and/or loafing-roosting) were found to be larger-than-average lodgepole pines with significantly different canopy heights (70% vs 35%) in the 45-70 ft range (Ratti et al. 1984).

In Alberta, spruce grouse nests with the best overhead and lateral concealment had better hatching success, but well-concealed nests also failed. Data suggested that nests reflected the general shelter of the surrounding site; success was not statistically related to stem density. Females, on average, responded mostly to vegetation at the immediate location of the nest. Protection from inclement weather was apparently not a factor in site selection, but concealment was and probably lessened the chances of predation (Keppie and Herzog 1978).

Dense sites with little undergrowth and ground vegetation, permit greater visibility of displaying spruce grouse males

(McDonald 1968). They also use dead trees at this time of the year, which may enhance visibility of the courtship display (McDonald 1968, Allan 1985). Dense conifer thickets (2.5-5.2 ac) with little understory and dead trees were used repeatedly as preferred display sites in Alberta (Herzog and Boag 1977).

Seasonal variations in the availability of food supply influence habitat use. Needles of various conifers are seasonally important in diets. Incubating Alberta females spent 74% of their time feeding on new leaders in mature spruce trees, even though these trees comprised only 4% of the study area. These were mainly in wet areas where past fire had not penetrated, but data were not presented to determine if the presence of the trees, moisture, or both, were influencing factors (Herzog and Boag 1977, Herzog 1978). Others have reported similar observations (Boag 1970, Pendergast and Boag 1971, McCourt et al. 1973). Female spruce grouse with broods tend to incorporate forest openings (size or type of opening not specified) for feeding largely because of the greater availability of berries, insects and ground vegetation found in those areas (Pendergast and Boag 1970, Allan 1985).

Transition from summer foods to use of western larch in fall apparently reflects a gradual adjustment to winter diet. Shedding of larch needles at the end of October is considered to be a direct stimulus for spruce grouse to move into winter habitats and shift feeding to other conifers (Jonkel and Greer 1963, Allan 1985). Winter food consists almost entirely of conifer needles (Crichton 1963, Pendergast and Boag 1970, Zwickel et al. 1974). Washington spruce grouse fed in lodgepole pine/Englemann spruce forests, but Englemann spruce needles were insufficient to sustain them (Hohf et al. 1987).

Spruce grouse use significantly denser stands in winter (Allan 1985). In north-central Washington roosting occurred in Englemann spruce, with similar activity reported for Colorado and Montana. Englemann spruce might provide greater thermal cover than the lodgepole pine used for feeding (Hohf et al. 1987). Selection of habitat by spruce grouse may represent a compromise between visibility for courtship and display, while encompassing suitable escape cover, feeding and roosting areas.

Ruffed grouse.--A dietary relationship involving herbaceous vegetation and deciduous trees influences habitat use by ruffed grouse. Various plants and insects are important in seasonal diets (Bump et al. 1947, Godfrey 1975, Rogers and Samuel 1984). Diets include hard and soft fruits in autumn, soft fruits and buds of deciduous trees and shrubs in winter, and leaves and floral parts of non-woody plants in spring and summer. The outstanding characteristics of ruffed grouse habitats are young successional status and diversity of fresh herbaceous growth (Bump et al. 1947, Porath and Vohs 1972, others). A result of growth form and partial influence of water flow in spring, patterning of herbaceous vegetation tends to be clumped rather

than uniform or random (Godfrey 1975). Most studies suggest clearcutting or wildfires produced the conditions preferred by ruffed grouse.

Optimal ruffed grouse habitat in clearcut aspen communities remains for 25-30 years, but some stand use occurs until age 60 or so (Table 1; see Gullion 1977 for review). In Pennsylvania, however, loss of ground-layer vegetation due to shading resulted in a decline of grouse brood use by the 7th year after clearcutting hardwood forest. After 10 years the areas were unusable as brood grounds, and the adult population declined because of inadequate reproductive replacement (Sharp 1963). Clearcutting Massachusetts pole- and saw-timber size oak and northern hardwoods produced suitable ruffed grouse brood cover (Healy et al. 1984). In western Washington, Salo (1978) found ruffed grouse aggregated in mixed stands 40-50 years old, primarily red alder/conifer/black cottonwood associations. Grouse also were associated with small (160 x 160 ft) forest openings, but as with other grouse studies, spatial-temporal relationships of the mosaic were not discussed.

Table 1. Ruffed grouse habitat development within an individual even-aged aspen stand (Guillion 1972, 1977, Gullion and Svoboda 1972, Healy et al. 1984).

Stand age (years)	No. aspen stems/ac	Habitat
1	12,000	Brood habitat until about 10 years of age.
10	8,000	Winter and breeding cover until stem density declines to < 5,000/ha.
25-60	2,000-1,000	Nesting cover; male flower buds are primary winter food resource.
> 60	< 1,000	Stand needs to be regenerated to maintain grouse habitat.

Aspen is not present in western Washington and the role of black cottonwood, found at most ruffed grouse sites, is unknown. Although tree height/density data were not noted, western Washington ruffed grouse were not found in stands (\leq 40-50 years) with basal areas over 430 ft²/ac unless deciduous trees were present (Salo 1978). Importance of a deciduous shrub stratum at acceptable ruffed grouse "drumming" (= courtship) sites is well documented (Dorney 1959, Gullion et al. 1962, Boag and Sumanik 1969).

Drumming logs, or display stages, are an important ruffed grouse habitat component. Drumming logs (logs lying on the ground) usually are located in deciduous areas, typically in shrubby cover with no overhead forest canopy, in an edge

situation, or in stands with less than 60% crown closure. Since leaves are not on deciduous trees in spring and fall drumming seasons, these are essentially open situations (Gullion et al. 1962).

Salo (1978) described a typical drumming site in western Washington as having 1.83 logs/territory; a main log 27.3 inches in diameter; a stage height of 24.8 inches; a log length of 34.4 ft; overhead cover of 37 inches; and a visibility radius \geq 50 ft. Important understory species of drumming sites were western hemlock and vine maple. Coverage averaged 52%, with vine maple contributing approximately 50% of that total. Forest crown and understory cover affect the development and species composition of the shrub stratum. The growth form of a well-developed vine maple understory gives overhead protection and shading reduces the growth of the shrub stratum (46% cover).

Drumming sites typically had high densities of woody stems and a sparse canopy of low shrubs and ground cover (Palmer 1963, Stoll et al. 1979, etc.). Vegetation density apparently isn't the only factor governing choice of drumming sites. A combination of cover factors such as juxtaposition of cover types, log placement in relation to edge and population density are probably involved (Palmer 1963). Areas acceptable as perennial drumming centers are not necessarily coincident with areas used by females, broods, or as wintering grounds (Gullion et al. 1962).

In Alberta, average canopy closure at drumming logs was lower for used sites (66%) than for non-used sites (109%). Average density of saplings at drumming logs was significantly less at logs than random plots (1,544/ac vs. 2,266/ac). Males were not found in dense stands (< 1" saplings). Saplings of 2-3" dbh were less common at logs than plots (24% vs. 61%). Tree size differed by < 4% between drumming logs and random plots, with only 8% being over 30 years old (Rusch and Keith 1971).

Visibility is apparently important to displaying ruffed grouse. The two dominant species of ground vegetation at western Washington drumming sites were sword fern and salmonberry. Drumming log height averaged 24 inches, taller than sword fern. In areas of sparse canopy, salmonberry becomes very dense, but during the April-June drumming season salmonberry has no leaves and visibility apparently remains acceptable. Red huckleberry (8 ft tall) characterized drumming sites lacking crown cover or large openings (Salo 1978).

Two ruffed grouse studies mention use of poorly drained, peat-organic lowlands or habitat along streams (Godfrey 1975, Salo 1978). While free water may be taken, food and cover development in moist areas may be just as important or more so. Context and importance for forest management are unknown.

Increasing conifer cover has been negatively correlated with

ruffed grouse. In western Washington, when the basal area of conifers increased above 215 ft²/ac, grouse numbers decreased (Salo 1978). In Minnesota, grouse broods used conifers only for roosting (Godfrey 1975). In Alberta, although 85% of all grouse observations were in aspen woods, approximately 38% of all mortalities occurred in spruce woods (Rusch and Keith 1971). Similarly, Gullion and Marshall (1968) concluded that conifers are an unessential or detrimental component of ruffed grouse habitat and may confer an advantage to predators. They reported significantly greater mortality of drumming male ruffed grouse when conifers were present.

Possible reasons for predation of grouse or grouse nests in managed forest conditions have been offered. Yahner and Scott (1988) evaluated predation of artificial ground and arboreal nests in mature, 60-65 year-old forest with 0%, 25% and 50% zones of adjacent clearcutting. Nest predation was highest (68%) in the 50% zone and lowest (9%) in the 0% zone. Crows and jays were the major predators. Ground nests were preyed upon less often (29%) than arboreal nests (71%).

Widen et al. (1987) suggested a "plant cycle hypothesis" to explain a female-skewed sex ratio among forest grouse killed by goshawks. An almost 1:1 negative correlation was found between the number of female grouse killed in spring and density of voles. Since goshawks do not eat voles in any quantity, it was felt that female grouse might be subjected to increased vulnerability in years of poor food abundance/availability (which resulted in low vole numbers). This was manifested by decreased incubation constancy, and poorer nutritional condition which necessitated more feeding trips which, in turn, exposed them to greater predation risk. Abundance and availability of forage are manageable items in a forest context.

Partial Harvest.--Management of ruffed grouse habitat in aspens has been well defined, but virtually nothing is known about silvicultural management for ruffed grouse in the Northwest. In Idaho, the effects of 25-30% thinning of pole-sized Douglas-fir lasted for 30-40 years. Thinnings were used by ruffed grouse, but discouraged deer and mice, an important point where reforestation is intensive (Hungerford 1969). Clearcutting adjacent 25-50 acre blocks every 10 years, on a 40 year rotation with commercial thinnings has been suggested as being suitable for ruffed grouse in western Washington (Salo 1978). This is, however, unproved and Salo pointed out the need to study ruffed grouse/black cottonwood/red alder associations in this area.

MAMMALS

Confirmation of density-dependent habitat selection at the scale of macro-habitat (Morris 1988) suggests that abundances of temperate zone mammals may result from macro-habitat processes. Population density responds to overall resource abundance, which in northern mammals should be correlated more with macro- than micro-habitat. Also, other selective forces (e.g., predation rates, physiological tolerances, and social interactions) are unlikely to depend mainly upon micro-habitat features. Daily foraging behavior by individuals may depend on micro-habitat conditions, but not population density. For TFW, this means that evaluations of forest management should be concerned with macro-habitat relationships at the stand or larger scale.

Most field investigators have evaluated proximate cues to habitat selection by small mammals because ultimate factors are too difficult to identify and measure. Proximate and ultimate factors include substrate moisture, substrate composition, microclimate, food availability, vegetation cover (links to food, thermal, and predator avoidance), nest-site availability, predation, and competition.

Although the influence of competition on habitat use and abundance in small mammal communities is unclear (Dueser and Porter 1986), forest management can be expected to influence small mammals by influencing relationships among competitors. For example, Grant (1969, 1971) found that deer mice and red-backed voles invaded open grasslands after meadow voles were removed. Also, the northern red-backed vole inhabits grasslands only on islands where competitors are absent (Cameron 1964). Other evidence, however, indicates that competitive interference may be a relatively unimportant influence (Morris 1984).

Although most authors did not quantify competitive interactions, they provide a quantitative assessment of habitat use under dynamic conditions. Many studies documented correlations between species abundance and habitat variables. For example, Miller and Getz (1977) correlated the abundance of red-backed voles with the percentage of ground covered by woody debris in every habitat type in a New England study. Morris (1984) found red-backed voles were abundant in transition forest in Alberta which had debris but not in one clearcut without debris. Other studies suggest soil moisture (Ramirez and Hornocker 1981) and competition (West et al. 1980) influenced abundance of red-backed voles.

Landscape-level Relationships

Limited information from landscape studies suggests that slash retention in clearcuts may result in stable abundance and diversity of small mammal populations. Scott et al. (1982) compared small mammal populations in 2 adjacent 100-ac forest drainages in central Colorado. Small mammals increased 94% in

the treated drainage (36% of the drainage was clearcut) and 50% on the uncut plot, although the differences were not statistically significant. Species diversity did not change and did not differ between cut and uncut drainages before or after treatment. The abundance of red-backed voles did not change following treatment, whereas least chipmunks increased, tending to be more numerous in clearcut sites.

Edges and variable stand conditions in mosaics of managed forests provide opportunities for occupancy by lagomorphs. Conroy et al. (1979) described snowshoe hare activity in a Michigan clearcut as "light to moderate", but "high" in ecotones between the clearcut and adjacent forest. Monthey (1986) used track counts in Maine to examine snowshoe hare activity and found that hares used a landscape predominated by clearcut habitat significantly greater than expected. Within this landscape, hares used uncut softwood patches more than expected.

Stand-level Relationships

Untreated Clearcuts.--Kirkland (1977) and Clough (1987) suggest that a well developed ground and shrub cover results in stable or increased small mammal species richness, evenness, and abundance in untreated clearcuts (those not burned or treated after harvest). Studies in forests outside the Pacific Northwest documented equal or higher levels of small mammal species diversity compared to early successional or uncut forest control stands (Kirkland 1977, Martell 1983b, Kirkland et al. 1985, Clough 1987). Moderate mammal species diversity levels in 2 studies (Kirkland et al. 1985, Clough 1987) were associated with moderate to high species evenness but low species richness, indicating that some species were excluded from untreated clearcuts.

In Maine, Martell (1983b) found that an increase in total abundance resulted almost entirely from changes in numbers of shrews. In his study, moss cover in clearcuts was much less than in uncut forest, but herb and shrub cover was similar. In addition, slash and litter were more abundant on the clearcut plot. Hooven (1973) found no change in small mammal abundance after clearcutting in western Oregon. During his study plant cover increased from 20% the first year to 70% the 3rd year after harvest. An increase in small mammal abundance on 3 unburned clearcuts in western Washington was associated with a six-fold increase in the amount of dead and downed woody material compared to the forest control stand (Gunther et al. 1983).

Although quantitative assessments rarely are made, the availability of herbaceous cover and downed wood material following clearcut harvest is often considered a significant factor influencing the abundance of small mammals. Miller and Getz (1977) found a positive correlation between deer mouse abundance and the cover of debris, but a negative correlation with low vegetation cover. The importance of slash and debris is

evident when considering that with very few exceptions (Harris 1968, Monthey and Soutiere 1985) the typical response of deer mice populations is to remain stable or increase in abundance following harvest (e.g. Kirkland 1977, Martell and Radvanyi 1977, Gunther et al. 1983, Martell 1983a, 1983b).

Increases in the abundance of red-backed voles following harvest were thought to result from abundant or even higher amounts of ground cover compared to uncut forest plots (Gunther et al. 1983, Monthey and Soutiere 1985). Moreover, decreases have been ascribed to reductions in ground cover (Hooven and Black 1976, West et al. 1980 [*C. rutilus* in Alaska], Ramirez and Hornocker 1981). Although none of these studies was specifically designed to quantitatively evaluate habitat effects, the results support an earlier hypothesis that log and ground cover is a limiting factor for this species (Tevis 1956).

Differences in cover values may influence survival rates of some species. Van Horne (1982) found that adult and juvenile deer mice occupying different microhabitats had differing overwinter survival rates. Individuals in "adult high-density high habitat" (high density of adults in habitat with high cover values for trees, shrubs, low canopy, and total canopy) had a higher survival rate than adult and juvenile occupants of "juvenile high-density low habitat" (high density of juveniles in habitat with low cover values).

Low conifer cover is an important habitat component of clearcuts for snowshoe hares. In New Brunswick, Parker (1986) found that cover and food provided by trees 3-10 ft tall were the most important factors influencing the distribution of this species during winter. Trees < 3 ft tall were not important because they were generally covered by snow during winter. Litvaitis et al. (1985) found that spring population density and overwinter survival were significantly correlated with understory density in Maine. Similar results were found in British Columbia where hare numbers decreased substantially after scarification removed about 85% of the low vegetation and cover (Sullivan and Moses 1986). Costa et al. (1976) thought that an increase in cottontail rabbit populations after clearcutting in a northern Arizona ponderosa pine forest resulted from an abundance of slash cover and a rapid increase in the production of herbs (an increase in herbs and shrubs of 504 lb/ac for the first 5 years following harvest).

Lagomorphs reoccupy clearcuts when reestablished understory vegetation provides suitable cover. Several studies documented the absence of snowshoe hares from clearcut study sites the 1st year after logging (Gashwiler 1959, Hooven 1969, Hooven and Black 1976). Hooven (1969) also noted the absence of hares on a 4 year-old regenerating clearcut. Gashwiler (1970) stated that hares began to occupy clearcuts after about 8 years and Hooven and Black (1976) thought the timing of this increase was associated with increased conifer and ground cover.

Overwinter survival by snowshoe hares is influenced by the availability of cover in clearcut stands. In young conifer plantations in British Columbia, Canada, Sullivan and Moses (1986) found a significantly higher proportion of immature/subadult animals in the treatment site compared to the control. Adults of either sex had a significantly higher survival rate on the control site, but there was no significant difference for juveniles. This seems surprising but may possibly result from higher rates of winter emigration from the treatment site by adults (also see Litvaitis et al. 1985, Parker 1986).

A number of authors have remarked on the importance of soil and plant moisture conditions (Hooven 1969, Miller and Getz 1977, Campbell and Clark 1980) for small mammals, but little is known about the response of animals to changes in moisture conditions following timber harvest. In Colorado, two species of shrews were consistently trapped near water in all habitats, suggesting that proximity to water or soil moisture was a more important habitat feature than physical structure (Spencer and Pettus 1966).

Changes in the abundance and availability of food supplies following clearcut harvesting often elicits a substantial response by mammals. A number of species, such as Douglas' squirrel and flying squirrel, face food shortages in clearcuts and occupy this habitat in very low densities (e.g. Gashwiler 1970, Hooven 1973). Trowbridge shrews occupy clearcuts if invertebrate prey is available in the litter layer (see Gunther et al. 1983) and are absent apparently when prey is lacking.

A number of species increase after harvest in response to a new abundance of food. Deer mice increase in response to an abundance of conifer seed available in reseeded clearcuts (Sullivan and Krebs 1981). Oregon voles (Sullivan 1980) and Columbian ground squirrels (Ramirez and Hornocker 1981) appear to increase in response to the availability of grasses and sedges in new clearcuts.

Female survivorship in deer mice is thought to be strongly influenced by availability of food resources during spring (Sadleir 1974). Van Horne (1981) hypothesized that male-biased sex ratios result primarily from low female survivorship. According to Van Horne (1981), such sex ratios should indicate "poor habitat", perhaps because males can better use their larger home ranges to find food. Additional support for this hypothesis is lacking and it may be difficult to distinguish the influences of food supply and ground cover (Van Horne 1982). Sex ratios of deer mice in clearcut and forest sites are typically skewed toward males (Petticrew and Sadlier 1974, Martell 1984), although equal sex ratios also have been documented in clearcuts and other seral stages (Campbell and Clark 1980).

Prescribed Burning.--Elimination of slash and vegetation in

burned clearcuts may result in temporarily-decreased abundance of small mammals (Spencer 1956, Gunther et al. 1983). In western Montana, the abundance of small mammals in 2 burned clearcuts decreased to below levels at the control stand immediately after the fires, increased to a point similar to control stands the 2nd year, and were slightly higher than control stand levels in 3 subsequent years (Halvorson 1982). Trends are expected to vary with burning intensity: deer mice increased with light burning in Halvorson's (1982) work, and voles may have been eliminated from the intensely-burned, south-facing site. The north-facing plot supported more deer mice and a greater species diversity compared to the south-facing plot.

The availability of herbaceous cover and wood material following prescribed burning elicits a response by small mammal species that is similar to that following clearcut harvest without treatment. However, because prescribed burning follows clearcut harvest it is often difficult to isolate responses unique to the burn. Most studies report stable or increasing populations of deer mice following the harvest and burn (e.g. Spencer 1956, Gashwiler 1959, Sims and Buckner 1973, Verme and Ozoga 1981, Halvorson 1982, Bock and Bock 1983, Martell 1984). Only Gunther et al. (1983) noticed a considerably lower number of mice on a burned site compared to an uncut forest site, although they did not use a pretreatment control to investigate the possibility of stochastic variation.

Shifts in mammal species composition may reflect changes in composition and amount of herbaceous and shrub cover. The abundance of small mammals in a clearcut 4 years after slash-burning was equivalent to that of a 125-year old Douglas-fir forest each year of a three-year study in Oregon (Hooven 1973). A number of species decreased in abundance on the slash-burn site, including Trowbridge shrew and vagrant shrew. These changes in abundance were balanced by substantial increases in the abundance of Oregon voles and deer mice on the burned plot. During the first year of Hooven's study ground cover was 40% and increased to 60% by the final year. The smaller slash was apparently largely eliminated by fire, leaving only larger cull material (Hooven and Black 1976:199).

Clough (1987) attributed a decrease in total abundance after prescribed burning to a sparse ground and shrub layer. Verme and Ozoga (1981) thought increased total abundance of small mammals resulted from rapid reestablishment of vegetation that compensated for the reduction of slash after prescribed burning. Gashwiler (1970) and Hooven and Black (1976) suggested that Oregon voles increased in abundance following fire although Gashwiler (1970) noted the increase only after vegetation cover had reached 30% (four years post-fire).

The importance of ground cover (slash, litter, vegetation) to successful occupancy of clearcuts by red-backed voles appears to be supported by the number of studies that report reduced

abundance following crown closure and a resulting shift from annual to perennial vegetation. Others also have reported abundances lower than those recorded in younger clearcuts (Kirkland 1977, Martell 1983b, Monthey and Soutiere 1985, Ramirez and Hornocker 1981); however, abundances comparable to uncut forests suggest that the typical population increases following harvest are fairly short-lived in some situations (Tevis 1956, Ramirez and Hornocker 1981).

Similarly, the availability of slash and litter appears to influence red-backed vole abundance in early successional phases. Martell (1983b) rarely found this species in early successional stands that were characterized by reduced litter and slash compared to young clearcuts. In Montana, Ramirez and Hornocker (1981) thought this species was uncommon in all young regenerating stands because of the absence of downed wood material. Although Kirkland (1977) and Monthey and Soutiere (1985) found red-backed voles more common in early successional stands neither could provide an explanation for the observed change in abundance.

The availability of nesting and roosting sites in logs and snags may influence the distribution of some species. For example, Thomas (1988) found that bat activity levels in Douglas-fir forests are higher in old-growth stands than in young (< 75 years old) or mature (100-165 years old) stands. Because bats forage very little in these forests the activity levels reported probably represent bats departing day roosts.

Partial Harvest Treatments.--The availability of slash and relatively low rate of disturbance to the vegetation in shelterwood cuts influences species abundance following treatment. In Ontario, shelterwood cuts that resulted in very minor loss of low vegetation cover continued to support stable populations of deer mice, red-backed voles, and shrews (Martell 1983b). Ramirez and Hornocker (1981) found similar results in Montana where deer mice and red-backed vole populations remained stable after a shelterwood cut. Populations of cottontail rabbits in northern Arizona ponderosa pine forests did not change in stripcuts, shelterwood cuts, patchcuts, or group selection harvesting, although a population increase in clearcut habitat was evident (Costa et al. 1976).

Monthey and Soutiere (1985) thought that significant increases in populations of red-backed voles, meadow voles, and shrews were related to greater ground cover following treatment. Swan et al. (1984) found higher numbers of deer mice on a 3-year cut with less shrub cover than in a 5-year cut with more shrub cover. This is surprising given the results above; however, the abundance of this species has been shown to be negatively correlated with vegetation cover (Miller and Getz 1977). Medin (1986) found that deer mice populations remained stable while red-backed voles were eliminated after treatment and prescribed burning that eliminated most slash and destroyed 80 percent of

voles to one dominated by deer mice. A shift from annual to perennial vegetation and a reduction of ground cover after canopy closure (5-7 yr) was thought to reduce small mammal abundance and biomass in a Georgia loblolly pine plantation initially treated by scarification (Atkeson and Johnson 1979).

The response of deer mouse populations to scarification is not consistent and very little information is available on this subject. Atkeson and Johnson (1979) noted a high population the 1st year following scarification, followed by a steady decrease. Conversely, deer mice were uncommon immediately after scarification in Ontario, Canada, and increased in abundance through the early regeneration phase (Martell 1983b).

Herbiciding.--Herbiciding changes the type of ground cover, which may influence small mammal diversity and abundance, although other physical factors may be involved. D'Anieri et al. (1987) found that most multiple captures occurred at trap sites with ground cover. In Oregon, species diversity and abundance on glyphosphate-treated sites increased the 1st year following treatment (Anthony and Morrison 1985), but decreased to pre-spray levels by the 2nd year.

The response of small mammals to the application of herbicides is variable, and most changes appear to result from changes in the availability or composition of vegetation. Anthony and Morrison (1985) noted increased abundance of red-backed voles on two sites following application of glyphosate. This increase in abundance was associated with increases in grass and forb cover on the treated site. Borrecco et al. (1979) found Oregon voles less common but noted no change in the abundance of Trowbridge's shrew on all sites after application of the herbicide 2,4-D. Application of this herbicide resulted in reduced plant species richness, and reductions in grass, forb, and total ground cover.

In Oregon, the relative abundance of the 5 most common species trapped at 3 study sites in 2 years provide equivocal results as to comparative responses to treatment by herbicides (Borrecco et al. 1979). Total abundances were similar in treated and untreated plots at 1 pair of study sites the 1st year and at 2 pairs the 2nd year. Abundances were much higher at untreated plots at 2 pairs of sites the 1st year and a single pair of sites the 2nd year. Analysis of vegetation community data indicates that control of grasses had no significant effect on shrub cover on treated sites. The untreated sites had a greater diversity of forbs and supported substantially higher numbers of Oregon voles, vagrant shrews, and Pacific jumping mouse. Deer mice and Trowbridge shrew numbers increased on treated sites in conjunction with an increase in shrubs and forbs.

Wildfire.--Although many small mammals survive prescribed burns or wildfires the immediate responses to fire often are substantial. In an intensely-burned aspen-fir-spruce and jack

pine forest in Minnesota small mammal abundance 1 week after the fire was much lower in burned sites compared to forest control stands (Buech et al. 1977). The study occurred shortly after the fire in autumn, so immigration and revegetation were largely precluded. Consequently, these immediate responses make it impractical to compare to studies conducted more than 1 year post-fire. Keith and Surrendi (1971) found that sites severely burned by wildfire in Alberta were abandoned by snowshoe hares but reoccupied after brushy cover was reestablished the 2nd summer following the fire.

The abundance of logs, snags, and debris following wildfire appears to influence the total abundance and composition of small mammals. Small mammal abundance and composition varied at 2 burned sites compared to a single unburned control in Minnesota (Krefting and Ahlgren 1974). Red-backed voles were more abundant on the unburned control and deer mice and other small mammals combined were more abundant on the burned sites. The decrease in total abundance occurred on a site where a 10 year-old jack pine stand had been completely consumed by fire. In comparison, the site with increased small mammal abundance had been a 70 year-old forest and many logs, snags and debris remained following the fire. This may have created greater habitat diversity for small mammals on that site.

Early Succession.--Although the association between habitat features and small mammal communities in early successional forests (8-39 years after harvest) have not been quantified, total abundance may be related to the amount or diversity of ground cover (Monthey and Soutiere 1985, Clough 1987). In Maine, relative abundance of small mammals in a group of sites 9-18 years after cutting was significantly higher than in 1-3 year old clearcuts and an uncut fir-spruce forest (Monthey and Soutiere 1985). These abundance patterns were thought to be related to greater ground cover and potential food available in the 9-18 year old sites.

In 6-15 and >25 year-old conifer stands in West Virginia, small mammal density was lower than values from stands < 5 yr old (Kirkland 1977). Granivore-omnivores were significantly less abundant in older stands compared to stands 6-15 years old, reflecting vegetation differences on the different-aged sites. Atkeson and Johnson (1979) found very low values of relative animal biomass 15 years after harvest compared to sites less than 5 years old in a Georgia pine plantation. They attributed this result to a shift from annual to perennial vegetation and a general reduction of ground cover after canopy closure at 5-7 years.

Decreases in the cover of slash and herbs, as well as in vegetative diversity, have been associated with decreases in deer mouse abundance in the early successional phase. Atkeson and Johnson (1979) rarely found this species in a loblolly pine plantation 15 years after planting. This represented a change in

abundance following crown closure and a resulting shift from annual to perennial vegetation. Others have also reported abundances lower than those recorded in younger clearcuts (Kirkland 1977, Martell 1983b, Monthey and Soutiere 1985, Ramirez and Hornocker 1981); however, abundances comparable to uncut forests suggest that the typical population increases following harvest are fairly short-lived in some situations (Ramirez and Hornocker 1981, Tevis 1956).

Similarly, the availability of slash and litter appears to influence red-backed vole abundance in early successional phases. Martell (1983b) rarely found this species in early successional stands that were characterized by reduced litter and slash compared to young clearcuts. In Montana, Ramirez and Hornocker (1981) thought this species was uncommon in all young regenerating stands because of the absence of downed wood material. Although Kirkland (1977) and Monthey and Soutiere (1985) found red-backed voles more common in early successional stands neither could provide an explanation for the observed change in abundance.

The availability of nesting and roosting sites in logs and snags may influence the distribution of some species. For example, Thomas (1988) found that bat activity levels in Douglas-fir forests are higher in old-growth stands than in young (< 75 years old) or mature (100-165 years old) stands. Because bats forage very little in these forests the activity levels reported probably represent bats departing day roosts.

Partial Harvest Treatments.--The availability of slash and relatively low rate of disturbance to the vegetation in shelterwood cuts influences species abundance following treatment. In Ontario, shelterwood cuts that resulted in very minor loss of low vegetation cover continued to support stable populations of deer mice, red-backed voles, and shrews (Martell 1983b). Ramirez and Hornocker (1981) found similar results in Montana where deer mice and red-backed vole populations remained stable after a shelterwood cut. Populations of cottontail rabbits in northern Arizona ponderosa pine forests did not change in stripcuts, shelterwood cuts, patchcuts, or group selection harvesting, although a population increase in clearcut habitat was evident (Costa et al. 1976).

Monthey and Soutiere (1985) thought that significant increases in populations of red-backed voles, meadow voles, and shrews was related to greater ground cover following treatment. Swan et al. (1984) found higher numbers of deer mice on a 3-year cut with less shrub cover than in a 5-year cut with more shrub cover. This is surprising given the results above; however, the abundance of this species has been shown to be negatively correlated with vegetation cover (Miller and Getz 1977). Medin (1986) found that deer mice populations remained stable while red-backed voles were eliminated after treatment and prescribed burning that eliminated most slash and destroyed 80 percent of

the litter.

As is true for other silvicultural practices, the availability of food resources following harvest influences occupancy. The density of red squirrels was reduced 65% in a spruce forest where the abundance (basal area not reported) of trees was reduced by 85% (Wolff and Zasada 1975). Sullivan and Moses (1986) compared the abundances of red squirrels in young thinned and unthinned lodgepole pine forest in British Columbia, Canada. They found that squirrels were more abundant in unthinned stands at two study areas during May-August (they did not sample at the thinned stands after this), but they were unable to explain this difference.

Apparent stability of small mammal abundance following selective harvest may be related to soil moisture conditions. In Wyoming, soils at selectively cut sites remained mesic following harvest whereas soils in clearcuts became xeric (Campbell and Clark 1980). However, an increase in soil moisture in clearcuts in Oregon (Hooven 1969, 1973) suggests that regional differences in climate and soil composition may elicit differing responses by small mammals.

CARNIVORES

Landscape Relationships

Wide-ranging carnivorous mammals encounter a variety of habitat types in managed or natural forest mosaics. Lindzey and Meslow (1977) found that radio-tagged black bears utilized vegetation types disproportionate to their availability in a forest mosaic on Long Island, Washington. Bears used 5-10 year-old regenerating clearcuts more than expected and regenerating stands ≥ 40 years old less than expected. Other vegetation types (15-20 year-old and mature stands) were used in proportion to their availability. The most important factor influencing the observed patterns appeared to be forage supplies.

Isolation of suitable habitat patches in a forest mosaic may influence the probability of site occupancy. In Ontario, Canada, deVos (1952) considered pine marten "common" and "abundant" on two large study plots in mature forest, but found none on a 3rd plot heavily logged and burned 30 years before his study. Moreover, the probability of encountering unsuitable patches should be higher for animals that move long distances. For example, lynx (Ward and Krebs 1985) and bobcat (Knick and Bailey 1986) are known to travel widely in response to declines in cyclic prey; the range of available and used habitat during these periods is unknown.

The available research indicates a positive relationship between pine marten home range size and the amount of the area that has been clearcut. In Maine, the proportion of regenerating clearcut habitat within the home range of 4 animals in summer

averaged 26% (range 16-47%) and for 3 animals in winter averaged 27% (range 20-36%)(Stevenson and Major 1982). Further examination of these data suggests a positive relationship between territory size and the proportion of the territory comprised of regenerating clearcut, but this premise is based on a small sample size (7). Soutiere (1979) found that home range size was largest for territories that included clearcuts. These findings do not imply use of this habitat, merely the proportion of each territory comprised of regenerating clearcuts.

The size of openings and proximity to cover appear to influence use of clearcuts. Pine marten are reported to generally avoid openings (e.g., Wynne and Sherburne 1984), although there is evidence they use this habitat (described below). Zager et al. (1977) found that 82% of the grizzly bear locations established in clearcuts were \leq 165 ft of cover. In addition, grizzly bears frequently use forested corridors when moving between adjacent harvest areas. Hugie (1982) found little use by black bears of clearcuts beyond 135 yards from forest cover. McCollum (1973) and Rogers et al. (1988) report little use beyond 200 yards from forest. Females with cubs during spring spend a majority of their time within 200 yards of large conifers that provide refugia for cubs in Minnesota (Rogers et al. 1988). In addition, black bears avoid large openings away from shade cover because they are easily heat-stressed (Rogers et al. 1988).

Based on 479 relocations for 9 mountain lions in northern Arizona ponderosa pine forest Van Dyke et al. (1986) determined that lions rarely visited active or inactive timber sales and used these areas significantly less than their availability. Mountain lions rarely moved through timber sales. In Utah, Van Dyke et al. (1986) found that all lions they radio-tracked had all or part of a timber sale within their home ranges; however, mountain lions there also used timber sales areas significantly less than their availability. In northern California, Raphael (1988) found that black bear, ringtail, fisher, and striped skunk were less common in brush/sapling stands compared to older seral stages but offered no explanation for this.

Despite the above information suggesting that certain carnivores rarely use openings, there is evidence to suggest they often use open habitat in proportion to its availability. In Montana, black bear use of clearcuts and burns was less than 10 percent from August through October (no use May-July), however this frequency of use was proportional to the availability of these habitats in the study area (Jonkel and Cowan 1971). In addition, because most food items of the black bear are shade intolerant, a large proportion of foraging occurs in openings (Rogers et al. 1988). Considerable damage to young trees by black bears in western Washington indicates use of young second growth forests (see Poelker and Hartwell 1973).

In Maine, nearly half of a male pine marten's territory was

clearcut and a female used clearcut habitat during summer in proportion to its availability (20%, Steventon and Major 1982). In the 2nd winter of a 2-year study in low boreal forest in Manitoba, marten used bogs in proportion to their availability (Raine 1983). Although openings such as roads, powerline rights-of-way and bogs were rarely used, marten in Newfoundland used conifer forests with low overstory density in proportion to its availability (Bateman 1986). A high proportion of ground squirrels in the summer/autumn diet of pine marten in Montana (Koehler and Hornocker 1977) and California (Zielinski et al. 1983) provides indirect evidence that they use open habitats to seek prey at some times. Despite a trend indicating infrequent use of clearcuts in Utah and northern Arizona, one male mountain lion frequently used timber sales areas but only 15 relocations were obtained for this animal before it was killed (Van Dyke 1986).

Stand-level Relationships

Very little information exists on the response of carnivores to the availability of prey in managed forests. A lower rate of occurrence was related to a reduction in the availability of suitable prey for pine marten (Major 1979, Douglass et al. 1983). In Utah and northern Arizona ponderosa pine forests Van Dyke et al. (1986) found that mountain lion activity near timber sale areas was often associated with the presence of food, and significantly more of these animals were transients (42% vs. 26% for residents).

Snags and fallen logs are important habitat features in forests that support pine marten. Snags (23%) and deadwood material (68%) were the most commonly used nesting sites in a California study (Martin and Barrett 1983). Hollow stumps, logs, and other woody material may be critical habitat features during winter when they provide shelter which facilitates adequate temperature regulation (Buskirk et al. 1988). Pine marten use the cover of downed wood material when crossing large openings (Koehler and Hornocker 1977). Blowdown, stumps, and snags also provide access to subniveal tunnels during winter (Bateman 1986).

Stands of various ages are often used proportionally less than their availability. Based on 196 detections of black bears in 6 seral stages in northern California, Raphael (1988) found a significant difference in frequency of occurrence across the range of seral stages. Bears were less common in brush/sapling and pole seral stages than in older stages. Soutiere (1979) found a lower density of pine marten in a clearcut forest than in partially harvested or uncut forest. Use of regenerating clearcuts was lower than expected compared to uncut softwoods and partially-cut mixed-woods (codifiers and hardwoods) stands in a study of eight marten equipped with radio tags (Steventon and Major 1982). In a telemetry study involving 5 marten in Maine, Wynne and Sherburne (1984) found that in summer males used clearcut openings significantly less than expected and females

avoided them entirely. The observations discussed above are in general agreement with the results of habitat use studies in Newfoundland (Snyder and Bissonette 1987) and California (Hargis and McCullough 1984). In winter, marten passed through clearcuts and other openings using the cover of fallen logs, but they did not hunt in these open areas (Koehler and Hornocker 1977).

BIG GAME ANIMALS

Most researchers believe that habitat selection links big game animals, or ungulates, with their environments in ways that influence population dynamics. Certainly, habitat selection influences population performance in managed forests because of relationships with factors that regulate population size: forage conditions, predation, hunting, and interactions with weather and climate. The habitat management problem is one of translating seasonal extremes of forage quality/quantity and the physical environment into predictions of ungulate response to management alternative for forests (Hudson et al. 1985).

Various solution-sets exist by which ungulates solve particular biological problems. Hudson et al. (1985) pointed out that habitat variation combines with significant behavioral, physical, and physiological adaptations to provide considerable options for big game animals to gather resources, yet buffer themselves from predators and daily and seasonal changes in weather. Thus, a big game animal's choice of habitat may be regarded as an optimization process, balancing costs and benefits among choices involving which habitat conditions to use.

The literature supports a conclusion that food acquisition is the basic determinant of ungulate habitat selection strategies (Marcum 1975, Schoen 1977, Franklin and Lieb 1979, Pedersen et al. 1979, Irwin and Peek 1983, Hanley 1984, Jenkins and Starkey 1984, Merrill et al. 1987). Certainly, forage density and nutrient quality are key factors influencing size and location of home ranges of deer and elk (Harper and Swanson 1970, Clary and Larson 1971, Franklin et al. 1975, Leslie 1983, Irwin and Peek 1983); ungulate populations are believed to be regulated by available food supplies (Caughley 1976); and there are correlations between vegetation biomass and ungulate biomass (Caughley 1976, Coe et al. 1976, Bobek 1977, Sinclair 1977, Skogland 1980).

In addition, the importance of forest vegetation and topography for escaping predators and hunters has been pointed

out (e.g., Black et al. 1976, Thomas 1979). Forest vegetation and topography also interact to determine the effect of snow on the amount of food available by influencing snow depths and deposition patterns. For example, forest canopies intercept falling snow, which may sublime or be re-distributed by wind before reaching the forest floor. Further, thermo-dynamic constraints modify basic habitat selection and activity patterns (Beall 1974, Black et al. 1976, Leckenby 1977, Thomas et al. 1979, Pedersen et al. 1979, Skovlin 1982, Parker and Robbins 1985, Zahn 1985).

No studies are available which relate community-level interactions between ungulate populations and other populations in managed forests. Roosevelt elk are considered to have a competitive advantage over black-tailed deer in areas where they coexist (Leslie 1983). Leslie et al. (1984) suggest that large downed logs in old-growth forest allow black-tailed deer to persist in areas occupied by Roosevelt elk on the Olympic Peninsula, because the deer can walk on the logs and use them as travel lanes. And Edge et al. (1986) suggest social relationships play a role in influencing interchange of elk herd groups from one drainage to the next in western Montana forests.

Landscape-level Relationships

Recent papers (Miller and Harris 1977, Picton 1979) suggest island biogeography concepts may apply in understanding long-term occupancy by ungulates of isolated mountain ranges. In Miller and Harris' (1977) work, the presence of large ungulate species in 13 East African reserves was not related to island size, but 1 reserve with 4 ungulate extirpations in recent times may support a case for isolation as a factor. However, Schwartz and Bleich (1986) doubt that even the normally sedentary bighorn sheep, which seem to possess relatively weak mechanisms for dispersal, are influenced significantly by isolation of habitats.

Studies show big game respond seasonally to various successional stages within managed forests. In general, the literature shows that fieldworkers have emphasized comparisons of use of relatively young clearcuts with use of uncut forests. Few studies have made adequate comparisons among the several successional stages (Irwin and Peek 1983). Elk in northern Idaho seasonally showed preference for every successional stage available, except late-seral, mature timber (Irwin and Peek 1983).

Landscape/Stand Relationships

Winter Relationships.--Increased food supplies in forest openings relative to dense forests, influence deer and elk use until temperatures decline to about 5° F (Arnold 1985). In colder weather, they choose dense vegetation, especially during windy conditions. The process of optimization explains why ungulates often select lee sides of slopes, boulders, ridges, or

forest stands during cold, sunny winter days, thus avoiding convective heat loss to wind yet remaining in full sunlight (Loveless, 1964, Beall 1974, Rudd 1985).

In winter, interactions between snow characteristics (depth, density, hardness), food supplies (quantity, quality, accessibility), and temperature and wind are strong determinants of ungulate distribution and habitat use. Densities of snow under forest canopies are greater and depth is less. And depth and duration of snow blankets have large influences on survival (Robinette et al. 1957, Picton 1984, Mech et al. 1987, White et al. 1987, Hobbs 1989).

Snow deeper than about 18 inches begins to constrain habitat use in elk, and snow deeper than about 28 inches precludes most elk (Beall 1976, Leege and Hickey 1977, Rudd et al. 1983). Conifer canopies influence these relationships by reducing windspeed and intercepting snow, such that relatively more forage is available under some forest canopies. Also, forest cover prevents extreme crusting of snow, at least until late winter. A 70% canopy closure can reduce energy costs of travel as much as 200% via limiting snow depths (Parker 1983). However, travel costs due to deep snow are not as important to elk energetics as reduction in energy intake caused by burial of food supplies (Wickstrom et al. 1984).

Elk select forest cover in winter in some areas that are subject to long, cold, snowy weather (Batchelor 1965, Janz 1980, Pedersen et al. 1980, Witmer and deCalesta 1983), but not in others (Knight 1970, Peek et al. 1982, Irwin and Peek 1983). Under extremely cold conditions with strong winds habitat selection in red deer and elk is associated with topography, low vegetation (Staines 1976), and timber stands. Roosevelt elk on the Olympic Peninsula eat Douglas-fir boughs on fallen trees, and litterfall is an important component of habitat selection (Leslie 1983). In northwestern Oregon Mereszczak et al. (1982) found that Roosevelt elk are influenced by the presence and abundance of nutritious herbaceous vegetation in seeded pastures.

Elk select winter bedding sites after lengthy wanderings through timber stands (Beall 1974). Topography plays a role in bed-site selection, by influencing speed and volume of air flow. In Beall's (1974) work large trees seemed to influence elk preference for bedding locations, which Beall hypothesized resulted in energy savings. Recent research, however, indicates that the costs of thermoregulation in winter are insignificant compared to the value of maintaining intake of digestible energy (Parker 1988, Hobbs 1989).

Cover is an important determinant of deer habitat selection during snowy winter periods when energy intake is low due to reduced availability of food of naturally low digestibility (Verme and Ozoga 1971, Moen 1973, Leckenby and Adams 1986). Closed canopies are preferred by white-tailed deer in winter in

northern Idaho (Owens 1981), and in periods of cold and snow in northwestern Montana (Mundinger 1979, Jenkins and Wright 1987). In southeastern Montana, patterns of habitat preference and survival over winter suggested that white-tailed deer operated under a strategy of habitat use that favored energy conservation (Dusek 1987). In the milder winter conditions of northwestern Oregon and southwestern Washington, Columbian white-tailed deer prefer Sitka spruce park forest (Suring and Vohs 1979). Isolated stands of conifers within otherwise relatively open mule deer winter range become important during untrafficable snow conditions (Hoover 1971).

Shrub clumps are effective in providing both thermal and forage benefits (Moen 1973), and influence elk winter habitat selection in some areas. Leege (1969) and Irwin and Peek (1983) observed that elk in northern Idaho used shrub clumps in seral brushfields both as winter cover and as a major food supply. In Irwin and Peek's (1983) work, elk avoided old-growth western hemlock and western red cedar stands, which were available within the winter range, except for a few weeks in early winter when green forage was available. On the east side of the Olympic Peninsula, Roosevelt elk use deciduous/coniferous forests < 150 years of age less than expected according to availability (Schrorer 1987). Shallow slopes along mixed coniferous/deciduous riverine areas are used. On the west side of the Peninsula the most-heavily used winter range areas include old-growth stands of mixed conifer and hardwoods in valley bottoms and lower slopes (Taber and Raedeke 1980).

Snow deeper than about 16 inches constrains deer use, and under such conditions, mule deer select for cover, often including forests (Loveless 1964, Jones 1974, Geist 1981, Leckenby and Adams 1986). This is particularly true for black-tailed deer on northern Vancouver Island (Jones 1974, Bunnell 1979, Taber and Hanley 1979). When herbs are buried by snow, digestibility of the diet of black-tailed deer decreased from 58% to 38% (Hanley and McKendrick 1985).

Food supplies and dense forest cover interact to affect deer habitat preferences in deep-snow areas (Barrett 1979). Mule deer use dense timber stands in winter in the Okanogan zone of northern Washington (Ziegler 1978), and forest stands with greater than 75% canopy in northern Idaho (Keay and Peek 1980). In these and other deep-snow areas, snow-intercept cover, lichen litterfall, and broken conifer branches are important as food sources (Rochelle 1980, Armleder et al. 1986).

Where deer must winter in forests containing large openings that accumulate deep snow that buries forage, food resources vary with snow depth in relation to stand volume or canopy cover (Bunnell 1979, Kirchoff and Schoen 1987). Snow-intercept cover, used by big game animals to find food during prolonged winters with deep snow, appears to provide significant nutritional benefits to ungulate populations in certain areas. However,

specific amounts, location, and structure in relation to food supplies and probability of lingering snow are yet to be determined by quantitative research. The capability for intercepting snow varies among areas and tree species (Schwab et al. 1987), so local information is required.

Harestad (1985) noted the importance of "critical" winter range for black-tailed deer (Salal/Douglas-fir old-growth) may be over-estimated. Failure to provide good forage supplies on "mild" winter range may be just as critical, because if only severe winter range is provided, it may be used all winter. Further, the other types allow deer to reduce the rate of loss in body condition or may even allow positive energy balance in mild years, allowing the population to grow (Mautz 1978).

Studies on the Selkirk caribou herd in northeastern Washington indicate winter habitat selection is predicated upon dense forest conditions which intercept snow and provide lichen and conifer-branch litterfall (Serveen and Lyon 1989).

Spring-fall Relationships.--On spring-fall ranges, one can expect ungulate use of managed forests to vary among successional stages across Washington: some seral stages will be used frequently in some areas but not in others, depending upon the choices available. Also, use of managed forests will depend partly upon the quantity and quality of forage present. In localities where the forest is comprised mostly of dense stands with little understory, big game are likely to make heavy use of clearcuts in spring and early summer, and perhaps less frequently, in fall (Harper 1971, Irwin and Peek 1983, Zahn 1985). Abundant food supplies provided the attraction to clearcuts seeded to grasses and legumes in the Wallowa Mountains of northeastern Oregon (Miller et al. 1981).

Young clearcuts in coniferous forests are used in spring and summer because of the abundant, digestible forage. As forage in open areas cures, elk use forested types with understory vegetation in delayed phenological stages (Harper 1971, Irwin and Peek 1983, Hanley 1984). During dry years, Rocky Mountain elk concentrate on moist sites, mostly at higher elevations, where succulent forage is still available (Collins et al. 1978, Marcum and Scott 1985). Wet meadows, dry meadows, clearcuts, and revegetated roads were preferred grazing sites in Utah (Collins et al. 1978).

Residual stands of conifers influence use of clearcuts and burned areas by both deer and elk. For example, Davis (1977) found that burned areas that contained standing dead timber were used more heavily by elk in Wyoming than sites without the dead trees. And advanced regeneration in western Cascades clearcuts influences Roosevelt elk use in summer (Zahn 1985). In Zahn's (1985) work, overall radiant energy fluxes within patches of advanced regeneration in 12- to 16-year old clearcuts were nearly as low as those measured in old-growth timber, suggesting such

patches are capable of providing microclimates similar to those in old-growth timber stands. These types were used repeatedly by elk for bedding, although they comprised only a small percentage of the total area available. Kowalsky (1964), Lyon and Jensen (1980), Irwin and Peek (1983), and Leckenby (1984) observed that the availability of at least some tree cover facilitates elk use of open areas. Harestad (1985) observed black-tailed deer in coastal Douglas-fir zones utilized small patches (< 2 ac) of residual conifers in clearcuts. It appeared that the residual patches facilitated access to the forage in clearcuts.

Topography also influences habitat use by elk, so the influence of forest management would be expected to vary in relation to slope steepness, benches, and aspect. For example, moderate slopes are favored by elk (Harper and Swanson 1970, Irwin and Peek 1983, Witmer and DeCalesta 1983). Edge et al. (1987) used discriminant comparisons of elk-selected and random habitat samples in western Montana. The most important variables were slope, foraging area within 650 feet, distance to open roads, and human disturbance. They believed summer habitat can be evaluated from maps and aerial photographs.

Closed canopy 2nd-growth forests, or mid-successional stages have little food value (Taber and Raedeke 1980). However, the cover value appears to determine elk habitat use in 2nd growth. Rocky Mountain elk in northeastern Oregon show preference for old-growth conifers during hot summer periods (Edgerton and McConnell 1976, Pedersen et al. 1980). Rocky Mountain elk and Rocky Mountain mule deer have significantly higher forage consumption rates in conifer understories than in grass communities of comparable biomass of palatable foods (Wickstrom et al. 1984).

Schoen (1977) working in the Cedar River Watershed in western Washington, found that throughout the year, un hunted elk preferred low-elevation 2nd-growth deciduous and mixed deciduous-coniferous forests, western hemlock zone clearcuts 5-15 years post-harvest, and riparian, wetland, and meadow communities. No specific preference for heavy cover was observed in fall, and all old-growth types generally were avoided.

Recent authors have emphasized the thermal benefits of overstory forest canopies to big game in summer. For example, Lyon (1979) wrote, "the behavior response [by elk] to hot, dry summer weather in two different years can be taken as further evidence of the importance of cool, moist habitat types to the overall integrity of elk summer ranges. Maintenance of body temperatures at some relatively constant level may be comparable to feeding as a daily preoccupation for elk." Thus, Black et al. (1976) defined thermal cover in terms of forest conditions: for Rocky Mountain elk it was taken to be a stand of coniferous trees 40 feet or more tall with average canopy cover > 70%. Optimum stand size was thought to be 30-60 acres, based upon pellet-group data (described in Thomas 1979) and radio-telemetry information

(Leckenby 1984). For mule deer, 2-5 acres is considered (i.e., hypothesized) optimal, including conifer stands a minimum of 300 feet in width.

Jenkins and Starkey (1984) hypothesized that the propensity of hunted Roosevelt elk to restrict use of open habitats to corridors near the forest edge implies forest cover facilitates elk use of extensively logged areas. Harper (1971) and Leckenby (1984) found that distance from standing timber cover is a factor in elk use of clearcuts. On the other hand, elk in extensively forested areas are influenced heavily by forage availability in hardwood communities (Leslie et al. 1984). So, management prescriptions have suggested that there is an optimal arrangement of forage- and cover-producing habitats (Black et al. 1976, Thomas 1979, Thomas et al. 1988). In the absence of human disturbance deer and elk use larger foraging areas (Willms 1971, Hanley 1984, Merrill et al. 1987), again depending upon plant phenology and topography.

Canopy closure > 75% influenced use of bedding sites by elk in Douglas-fir forests of western Montana, but they preferred to feed in areas with < 25% canopy cover (Marcum 1975). Nelson and Burnell (1975) also found the highest use occurred in forests with canopies > 75% in central Washington. Forage condition was the primary determinant of elk distribution and habitat selection in Montana, whereas thermal cover, which was abundant, was secondary (Marcum and Scott 1985). Merrill et al. (1987) wrote that estimated heat losses and observations suggest Roosevelt elk could cope physiologically or behaviorally with high heat gains in summer in the Mt. St. Helens blast zone of Washington. McCorquodale et al. (1988) made similar observations for Rocky Mountain elk in the Arid Lands Ecology (ALE) Reserve in central Washington.

Parker and Robbins (1984) felt that elk are much less stressed by high operative temperatures than deer, because elk possess an extensive sweat-gland system; in fact, elk often lay in the sun even when shade is available. Schrorer (1987) found that Roosevelt elk on the Olympic Peninsula did not seek forest shade in summer; elk were often observed in mesic meadows from mid-morning to late afternoon. However, these elk made increased use of timbered areas in fall, which Schrorer (1987) attributed to phenological development of forage.

Irwin and Peek (1983) found elk selected dense second-growth mixed hemlock and pine forests on north and east slopes in late summer and fall in northern Idaho. They hypothesized that phenological conditions of forage supplies were strong influences on elk habitat use. In Irwin's (1978) radio-telemetry study, cow elk with young calves used seral brushfields that were created as a result of clearcutting. During mid-day bedding periods, the elk were most frequently found near single conifers (<60 feet in height).

In northwestern Washington Hanley (1984) found that Roosevelt elk and black-tailed deer habitat use patterns varied within the spring and summer periods. Both species preferred 14-19 year-old seral stage clearcuts in May and June, but in July and August elk did not demonstrate a clear preference for any particular seral stage, although they avoided 2nd growth with closed canopies. From September to October elk shifted toward 8-13 year-old clearcuts and 450-550 year-old stands, showing no distinct preference or avoidance for any particular successional stage. Habitat use patterns by black-tailed deer were highly correlated with forage availability during all 3 bimonthly periods, but values of habitat preference by elk were not correlated with forage availability during any of the bimonthly periods.

On most elk ranges the most important determinant of habitat selection is presence of people. Lyon (1983) concluded human disturbance is an important factor regulating use of managed areas is human disturbance. In fact, almost every research effort involving forest roads and big game has found decreased use for at least $\frac{1}{4}$ mile on each side of traveled forest roads (Irwin and Peek 1979).

Rost and Bailey (1979) examined responses to roads for mule deer and elk east vs. west of the continental divide in Colorado. Road avoidance was greater: (a) east, rather than west, of the continental divide; (b) along more heavily traveled roads; (c) by deer, compared to elk; and (d) for deer in shrub habitats when compared to forested (pine and juniper) habitats. More pronounced avoidance east of the divide may have resulted from a greater availability of winter habitat away from roads than in west side zones.

In recognition of the effects of traffic and increased human access, road closures have been implemented in many western big game ranges. Irwin and Peek (1979) gathered evidence which suggested that road closures affect the minimum size of forest patch in which elk will remain for lengthy periods during hunting seasons. Big game responses to roads and road closures vary with vegetation density adjacent to the road (Burbridge and Neff 1976, Coggins 1976, Marcum 1975, Perry and Overly 1976, Ward 1976, Irwin and Peek 1979).

The literature revealed little information listing parameters which influence mule deer use of managed forests on summer range. Pac et al. (1984) provided preliminary information on mule deer behavior in relation to forest management in southwestern Montana. In their work only 3 of 149 combined summer activity centers of mule deer occurred in recently logged areas, despite significant clearcutting activity. Summer deer densities were highest in multi-aged stands of Douglas-fir and/or subalpine fir on moist north and east aspects. Lyon and Jensen (1980) showed that mule deer preferred Montana clearcuts with cover in the opening except where such cover inhibited forage

growth, and deer preferred openings in which logging slash was not a barrier to movement. Vegetation height in the clearcut was correlated positively with deer use of clearcuts.

Ecotonal situations are known to influence use of managed sites by big game. For example, Clark and Gilbert (1982) found that white-tailed deer pellet groups were significantly higher for edge vs. non-edge habitats in Ontario. Their data suggest that deer habitat can be evaluated using a quantifiable measure of edge as a habitat variable. They suggested a 4-mi² grid as the scale of resolution of habitat quality based upon edge.

In general, the premise that edge habitats influence summer use is confirmed by the literature. Indeed, Thomas (1979) and Thomas et al. (1979) indicated edge confers benefits to mule deer, especially where edge occurs between relatively open and cover areas. Kirchoff et al. (1983) counted black-tailed deer pellet groups to test the hypothesis of higher winter deer use within approximately 100-foot ecotone between old-growth forest and clearcuts (ave. age = 5.4 yrs) in southeastern Alaska. They found no evidence of increased deer use near old-growth/clearcut edge. Willms (1971) documented slightly higher winter black-tailed deer use along edges between logged and mature forest on northern Vancouver Island, British Columbia. Kirchoff et al. (1983) concluded the value to deer of edge in managed forests cannot be assumed without careful local evaluation.

Summary.--We observed in the big game literature a series of descriptive or observational studies which attempted to infer the reasons for the observed patterns. General agreement appears to exist on which variables should be included in models of elk-habitat relationships (Thomas et al. 1988). However, less consensus and understanding exist on how the variables interact to exert an overall influence on big game use of managed forests. Clearly, new research is needed that experimentally probes basic determinants of habitat selection or predicts seasonal habitat selection for ungulates in managed forests. Even after decades of work, we still need a reliable algorithm for predicting ungulate population responses to habitat change.

**APPENDIX B. MATRIX OF HABITAT VARIABLES INFLUENCING WILDLIFE
HABITAT USE**

The following lists include matrices of habitat factors that influence wildlife use in managed forests. This information was developed directly from the details on habitat relationships presented in Appendix A. Species groups are presented in the same order as in Appendix A. These groups are listed according to both landscape- and stand-level relationships.

For many cases, published information could not be found that described studies conducted in managed forests, so we inferred managed-habitat relationships from studies of habitat selection and use. We emphasized studies of physical and biological factors influencing habitat use in forests.

MONITORING VARIABLES-Reptiles and Amphibians

Level	variable	unit	wildlife output	topic species/guild	geographic location	References	comments			
Stand	stand age	age in years	abundance	reptiles	n.w. CA	Raphael 1984, 1988	greater in pole and early successional stages			
				northwest salamander	v. OR	Bury and Corn 1988b	not related to stand age			
				n.w. CA	Raphael 1984					
				clouded salamander	n.w. CA	Raphael 1988, Welsh and Lind 1988	greater abundance in >250 year-old stands			
				black salamander	n.w. CA	Raphael 1988	fewer observed in young stands			
				n.w. CA/s.w. OR	Welsh and Lind 1988	not correlated with stand age				
				Ensatina	n.w. CA	Bury 1983	less common in 6-10 year-old stands than in old-growth			
						Welsh and Lind 1988	more abundant in old-growth than young			
					s.w. OR	Bury and Corn 1988b	no correlation with age			
					w. WA	Aubry et al. 1988	no age-related changes in abundance			
				Del Norte salamander	n.w. CA	Raphael 1984	not present in stands < 250 years old			
						Raphael 1988	positive correlation with stand age			
					n.w. CA/s.w. OR	Welsh and Lind 1988	positive correlation with stand age			
					w. WA	Bury and Corn 1988b	detected only in moderately wet old-growth			
					w. WA	Aubry et al. 1988	present in all stands 55-750 years old			
						Bury and Corn 1988b				
					n.w. CA	Raphael 1984, 1988				
					n.w. CA/s.w. OR	Welsh and Lind 1988	no correlation with stand age			
					n.w. CA	Bury 1980	found only in 6-10 year-old clearcuts			
					OR/WA	Bury and Corn 1988b	relatively few in 30-650 year-old forests			
					n.w. CA	Raphael 1984	negative correlation with stand age			
					n.w. CA/s.w. OR	Bury and Corn 1988b	rare or absent in clearcut; common in closed canopy			
					OR/WA	Bury and Corn 1988b				
					n.w. CA	Raphael 1984	negative correlation with stand age			
					n.w. CA	Raphael 1984	negative correlation with stand age			
					n.w. CA	Raphael 1984	negative correlation with stand age			
					n.w. CA	Bury 1983	lower richness, abundance, and biomass in clearcuts			
					w. WA	Aubry et al. 1988	2 of 6 species with sufficient data			
					n.w. CA/s.w. OR	Welsh and Lind 1988	greatest in old-growth			
					abundance	Ensatina	n.w. CA	Rosenburg and Raphael 1986		
				length/proximity of edge	ft	abundance	rough-skinned newt	n.w. CA	Rosenburg and Raphael 1986	positive correlation with amount of edge
				stand edge	ft	abundance	sagebrush lizard	n.w. CA	Rosenburg and Raphael 1986	positive correlation
				distance to clearcut	ft	abundance	sagebrush lizard	n.w. CA	Rosenburg and Raphael 1986	negative correlation
presence of hardwood	presence	abundance	sagebrush lizard	n.w. CA	Rosenburg and Raphael 1986	negative correlation				
ambient temperature	degrees	abundance	reptiles	OR/WA	Bury and Corn 1988b	some in < 30 year-old forest				
moisture gradient	moisture index	abundance	amphibians	w. WA	Bury and Corn 1988b	fewer present in wetter sites				
				n.w. CA	Welsh and Lind 1988	fewer in dry than mesic sites				
			northwest salamander	OR/WA	Bury and Corn 1988b	no relationship				
			black salamander	n.w. CA/s.w. OR	Welsh and Lind 1988	no relationship				
			clouded salamander	n.w. CA/s.w. OR	Welsh and Lind 1988	greater abundance in mesic				
			Ensatina	w. WA	Bury and Corn 1988b	less abundant in wetter sites				
					Aubry et al. 1988	were abundant with increased moisture				
			v. red-backed salamander	w. OR	Bury and Corn 1988b	found only in mesic and moist sites				
			Pacific treefrog	n.w. CA/s.w. OR	Welsh and Lind 1988	greater numbers in mesic sites				
downed woody material	presence	presence	salamanders	w. WA	Aubry et al. 1988	important to 2 of 6 species				
		density	salamanders	n.w. CA	Raphael 1988	density increased				
		abundance	clouded salamander	n.w. CA	Bury 1983	greater numbers in clearcuts than old-growth thought to be related to downed material				
			Calif. slender salamander	n.w. CA	Bury 1983	less common in clearcuts than old-growth stands; associated with woody debris				

Landscape	class II logs	volume/abundance	abundance	clouded salamander	n.v. CA	Raphael 1988	positive correlation with downed wood mass present in clearcuts with class II logs
	class IV and V logs	no./area	presence/absence	clouded salamander	n.v. CA	Raphael 1988	
			abundance	Hesatina	v. WA	Bory and Corn 1988	
	stand area	total area	abundance	3 salamanders	n.v. CA	Raphael 1984	not correlated with stand area
			species richness	Hesatina	n.v. CA	Raphael 1984	positive correlation with stand area
				reptiles and amphibians	n.v. CA	Raphael 1984	positive correlation with stand area

MONITORING VARIABLES-non-game birds

Level	variable	units	wildlife output	topic species/guild	geographic location	references	comments
Stand	stand age (succession)	age in years	diversity	community	WA	Comer and Adkisson 1975	lower in 1-yr plot than other ages to 12 yrs no difference among 5 age classes of cuts initial increase, followed by decline and later increase as succession continues bird populations had varying responses abundance 3rd yr following harvest was greater than in vocut forest diversity generally increases during succession (review)
			density	community	Colorado	Scott and Crouch 1988b	
			richness	community	s.e. USA	Keyers and Johnson 1978	
			richness, density abundance	community community	Utah n. CA	Austin and Perry 1979 Hagar 1960	
	diversity	community	s. USA	Shogart et al. 1978	diversity generally increases during succession (review)		
	diversity density	cavity users community	w WA n.w. CA	Zarnowitz and Manuval 1985 Raphael 1984	species diversity not related to stand age density of 17 species positively correlated with stand age; no differences at community level vs stand age		
	canopy cover	% cover	density	community	w. OR	Morrison and Meslow 1983a ['84b]?	density correlated with increased deciduous cover
	openness	% cover	abundance	community	n.e. OR	Nannan and Meslow 1984	the species more common in managed stands were thought to be influenced by openness
			density	community	AZ	Szaro and Balda 1979	aerial foragers increased in irregular strip- shelterwood stand; open-area birds increased, species of dense vegetation eliminated after treatments opened stands
	foliage density	% cover	nest predation, forage availability	breeding birds	AZ	Martin 1988	greater foliage density correlated with minimized nest predation; number of species correlated with foraging and nesting substrate
	tree height	height		community	w. OR	Morrison and Meslow 1984b	height of deciduous trees accounted for much of variation of habitat use in clearcuts
	vegetation cover	% cover	richness	community	AZ	Blake 1982	lower richness in burned area probably related to removal of vegetation cover
	foraging substrate	amount	density	cavity user	AZ	Brown and Balda 1988	foraging substrate or other factors are thought to be limiting if nest sites common
	food availability	availability	diversity	community	AZ	Szaro and Balda 1979	all except ground-feeding birds eliminated from stand following clearcut
	snag availability	#/area	density	flycatchers	CO	Scott et al. 1982	Olive-sided Flycatcher increased after treatment
			density	woodpeckers Lewis' Woodpecker	n. CA n.e. CA	Hager 1960 Block and Brennan 1987	woodpeckers used dead and dying trees more common on burned stand where snags smaller but more abundant
density, diversity			cavity nesters	w. WA	Zarnowitz and Manuval 1985	densities of obligates correlated with snag abundance	
			snag users	n.e. OR	Nannan and Meslow 1984	abundance of large snags accounts for presence of nest hole-nesting species in older forests	
				w. OR	Nannan et al. 1980	use of snags concentrated in older forests; this primarily related to greater availability there	
		density, diversity	community	TX	Dickson et al. 1983	higher density and diversity at site with snags	

		use		Rocky Mtn region	McClelland 1980	control, shelterwood, and uncut islands used more than other treatments
				AZ	Scott 1979	species composition and density decreased on plot without snags; no change at guild level
snag condition	decay class		snag users	NY	Swallow et al. 1986	snag use related to size (DBH) and decomposition status
		density		v. OR	Hannan et al. 1980	density correlated with snag age (via snag age); snags mostly > 60 cm dbh, 15 m tall, and with broken-tops, few branches, < 100% bark cover, interior decay
				v. WA	Zarnowitz and Hancock 1985	density increased with increase in availability of large snags
snag size	dbh	density	snag users	v. OR	Hannan et al. 1980	positive correlation between density of hole-nesters and average dbh of snags
				NY	Swallow et al.	snag use related to dbh
nest sites	availability			AZ	Brawn and Balda 1980	secondary cavity nesters limited by nest availability (experimentally)
downed woody material patch diversity	presence/absence heterogeneity	density density	Winter Wren community	D. CA	Hager 1960	1's higher when logs on ground
				ND	Chasko and Gates 1982	patches provide nesting opportunities
				v. OR	Morrison and Heslow 1983	density increased where patches of deciduous trees present in clearcuts
		richness, density	community	ND	Niel and Hanowski 1984	results in high species richness and density in early successional stages
		richness, diversity	community	Maine	Derleth, McAuley, and Dwyer 1989	greater patchiness hypothesized to promote greater richness and diversity
habitat structure	vegetation diversity	density	snag users	NY	Swallow et al. 1986	snag use increases in forest with high diversity and density of trees, and high basal area of snags
		foliage ht diversity	community	many	MacArthur and Levins 1964, others	increased species diversity associated with foliage height diversity
Landscape						
stand area	unit area	density richness richness density, richness	community community community community	CO Great Britain NJ n.w. CA	Scott and Crouch 1988b Moore and Hooper 1975 Galli et al. 1976 Raphael 1984	no difference among 3 size classes of cuts rate of occupancy correlated with area smaller patches support fewer species 5 of 46 species had positive correlation with area; species richness correlated with stand size
		species composition	community	n CA	Rosenberg and Raphael 1986	composition similar in all age classes of forest-patch size
		species composition	community	Finland	Helle 1985	the increase of smaller birds and decrease of larger birds may be related to territory size requirements
		nest predation	community	ND and TN	Wilcove 1985	open nests in small woodlots had higher predation rate than those in large tracts; predation may influence species loss from small tracts
		nest predation	ground-nesting birds	ID	Ratti and Reese 1987	no correlation between nest predation and woodlot size
		diversity	community	NJ	Ferman et al. 1976	species that require large patches are lost sooner than species that require smaller patches
		richness	community	AZ	Blake 1982	extent of habitat modification may be more important than type of modification
edge		density density, diversity	breeding birds breeding birds	Finland Finland	Vickholm 1982 Helle 1985	densities at edge 25% higher than interior edge reduces area available for interior

		diversity richness, density	community community	NJ Maine	Forman et al. 1976 Swall and Hunter 1989	non-forest species add to richness at edge maintained edges (e.g. along rights-of-ways) had lower diversity and density because habitat not diverse; non-maintained edges had high densities
fragmentation		density, diversity diversity	community	CO CO	Scott and Crouch 1988b Scott and Crouch 1987	higher diversity and density at edges increased diversity after harvest of 3 20-ac units
		density	breeding birds breeding birds	Finland Finland	Helle 1985 Virkkala 1987	some species increased; hole-nesters decreased some species increased, others declined
		nest parasitism	breeding birds	USA	Brittloghan and Temple 1983	authors hypothesize that increased fragmenta- tion will promote greater nest parasitism by cowbirds
isolation	juxtaposition	richness, density	community	n.v. CA	Rosenburg and Raphael 1986	8 species decreased in insular stands; 17 species had greater abundance there

MONITORING VARIABLES-Raptors

Level	variable	units	wildlife output	topic species/guild	geographic location	references	comments	
Stand	human disturbance	presence/absence	presence/absence	Bald Eagle	n.w. WA	Hanson et al. 1980	early departure or abandonment of roost associated with activity at nearby clearcut	
			productivity	Osprey	n.w. CA	Levenson and Koplin 1984		significant decline in productivity associated with subjective assessment of logging intensity
	nest exposure	buffer width	use	Osprey	s.e. AK	Corr 1974	20% of known nests lost to windthrow 1 winter	
			stand age	age in years	use	Bald Eagle		British Columbia, Canada
	nesting				Spotted Owl	OR	Forsman et al. 1985	97% of pairs in stands > 200 years old
					Great Gray Owl	OR	Bryan and Forsman 1987	45 of 60 nests in mature/old-growth
					Accipiter species	OR	Reynolds et al. 1982, Moore and Henny 1983	3 species occupy stands of different ages
	abundance				Great Horned Owl, Barred Owl	VA	McGarigal and Fraser 1984	more common in old than young forest
					Red-tailed Hawk	VT	Orians and Kuhlman 1956	nest at 77-81% of tree height
	tree size	height, dbh	nest support		Bald Eagle	OR	Anthony and Isaacs 1981	size may be important for access
					Northern Goshawk	OR	Moore and Henny 1983	used larger trees
	snag/tree size	dbh	nest site		Spotted Owl	OR	Forsman et al. 1984	all 47 nest trees > 29 in
					Barred Owl	ND	Devereaux and Mosher 1980	nest trees > 25 cm
	tree deformity	availability, type	nest site		Northern Goshawk	WA	Fleming 1987	11 of 31 nests in deformed conifers
					Cooper's Hawk	OR	Dixon and Dixon 1938	
	nest availability	availability, type	nest site		Accipiter species	OR	Moore and Henny 1983	64% of nests on mistletoe clumps
					Spotted Owl	OR	Moore 1980	varied for 3 species
	nest availability	availability, type	nest site		Northern Goshawk	WA	Forsman et al. 1984	30 of 47 nests in damaged conifers
					Accipiter species	OR	Irwin et al. 1989	used mistletoe clumps (38%)
	nest availability	availability, type	nest site		Spotted Owl	CA	Dixon and Dixon 1938	understory limbs
					Spotted Owl	OR	Moore 1980	varied for 3 species
	nest availability	availability, type	nest site		Spotted Owl	OR	Forsman et al. 1984	used platform structures
					Spotted Owl	CA	LaHaye 1980	used platforms and old hawk nests
nest availability	availability, type	nest site		Spotted Owl	WA	Irwin et al. 1989	used old Accipiter nests (31%)	

Landscape	cavity availability	availability	nest site	Osprey	CA	Garber et al. 1974	silvicultural manipulation of nest trees				
				Common Raven	WA	Fleming and Speich 1988					
				Barred Owl	ND	Devereaux and Mosher 1984					
	cavity height	height	nest location	Barred Owl	ND	Forsman et al. 1984, others	lower mortality rate than birds at platform nests				
				hole size	size	use, nest location		3 owl species	Norway	Sonerud 1985	> 9 m
				perch availability	1/area	hunting use		Goshawk	England	Kenward 1982	75 of 79 attacks initiated from perch
	tree species	availability	nesting	Barred Owl	MI	Nicholls and Warner 1972	not found in habitat which lacked perch sites				
				Northern Goshawk	WA	Fleming 1987	30 of 31 nests in Douglas-fir				
				Cooper's Hawk	SD	Bartelt 1977	nested almost exclusively in ponderosa pine				
			roosting	Bald Eagle	OR	Henny and Moore 1983	77% of nests were in Douglas-fir				
				Bald Eagle	WA	Grubb 1976	78% Douglas-fir, 17% Sitka spruce				
				Spotted Owl	OR, WA	Forsman et al. 1984, Irwin et al. 1989	most nests in Douglas-firs				
downed wood material	availability	nest site	Bald Eagle	OR	Anthony and Isaacs 1981	74% Douglas-fir, 23% Sitka spruce					
			Spotted Owl	OR, CA	Dellusola et al. circa 1986	tree species generally not important					
			3 small owls	CA	Keister 1981	tree species generally not important					
understorey cover	stems/ac; height	foraging	Barrows and Barrows 1978,	CA	Barrows 1981	tree species generally not important					
			Barrows 1981	ID	Rayward 1983	roost sites chosen for predator protection more than thermal economy					
			Turkey Vulture	USA	Bent 1937	nest in or adjacent to					
distance to water	distance	plucking post	Accipiter species	OR	Reynolds et al. 1982	typical at sites					
			Barred Owl	MI	Nicholls and Warner 1972	lack of understorey thought to aid hunting					
			Bald Eagle	w. WA	Grubb 1980	average distance of 218 nests was 232 ft					
edge	linear measure	use	Barred Owl	NJ	Bosakowski et al. 1987	close association with water (swamps)					
			Red-tailed Hawk	MI	Fuller 1979						
			Goshawk	NY	Blody 1983						
fragmentation	amount	density, diversity	raptor community	Ohio	Novell et al. 1982	90% use of edge					
			breeding	Bald Eagle	England	Kenward 1982	distribution significantly related to edge				
			Osprey	Java	Thiollay and Keyburg 1988	density and diversity related to patch size					
mosaic	presence, amount	presence/absence	Bald Eagle	B.C., Canada	Hodges et al. 1983	lower densities in logged beachfront areas					
			use/productivity	Spotted Owl	CA, AZ	Melo 1979	variable responses to logging activities near nest				
			Bald Eagle	OR	Forsman et al. 1984	moved/disappeared from highly fragmented areas					
afforestation	1 cover	foraging	raptor guild	B.C., Canada	Hodges et al. 1983	logged habitat used < than availability; unlogged had more nests/km					
			foraging, nesting	Golden Eagle	Norway	Sonerud 1986	snow depth in clearcuts during winter influences use of that habitat				
			density, productivity	Golden Eagle	WA	Servheen 1978	foraged in clearcut				
			Golden Eagle	WA	Anderson and Bruce 1980	nest adjacent to clearcuts; foraged in cut areas					
			Raven	England/Scotland, England	Marquiss et al. 1978, Newton et al. 1982	1 afforestation influenced occupancy, prey availability, and productivity					

MONITORING VARIABLES-Cavity-nesting ducks

Level	variable	units	wildlife output	topic species/guild	geographic location	references	comments
Stand	stand size	acres	presence/absence; density	Wood Duck	MI	Gilmer et al. 1978	used telemetry
	distance to water	linear distance			MI (both)	Ball 1973, Gilmer et al. 1978	associated with food availability
	food availability		survival, growth, mortality	Bufflehead Wood Duck	British Columbia, Canada Arkansas	Erskine 1972 Briggs 1978	all nests < 650 ft forage for nest in flooded forest
	tree density	trees/area	presence/absence	Common Merganser	B.C., Canada	Wood 1986	distribution of breeding pairs influenced by availability of salmonids
	tree size; height	height, dbh	presence/absence	Wood Duck	MI	Gilmer et al. 1978	11 in dbh minimum
	canopy closure	(%)	presence/absence	Bufflehead	B.C., Canada	Erskine 1972	8 in dbh minimum
	tree condition	alive/dead	presence/absence	Wood duck	MI	Gilmer et al. 1978	48% aspen, 63% hardwoods
	cavity presence	#/area	presence/absence	Bufflehead	MI	Gilmer et al. 1978	unmerchantable stands < 20 ac
	cavity height	height	use/predation	Wood Duck Common Goldeneye	MI Ontario, Canada	Gilmer et al. 1978 Lonsden et al. 1986	cavities created primarily by flickers; 33% aspens, 50% conifers after 15 yrs 1.6 cavities/ac Nest success related to cavity height; below 4.5 m predated more often.
	hole size	size of opening	use	Common Merganser	s.e. USA	Kiff 1989	nests at 10-18 ft
	cavity size		use/competition	Wood Duck	MI	Gilmer et al. 1978	average nest height 30-35 ft
			use/competition; brooding success	several species Common Goldeneye	Ontario, Canada Europe	Lonsden et al. 1986 Eriksson 1979	influences use for some larger species positive correlation between clutch size and nest box size
	cavity permanence	age in years	presence/absence	Bufflehead	B.C., Canada	Erskine 1977	cavities created primarily by flickers; 33% aspens, 50% conifers after 15 yrs

MONITORING VARIABLES-Forest Grouse

Level	variable	units	wildlife output	topic species/guild	geographic location	references	comments	
Stand	stand age	age in years	density	Blue Grouse	British Columbia, Canada	Niederleitner 1986	higher use in stands 3-13 yr old compared to mid-successional (59 yr) or old-growth sites forests < 10 yrs used as brood habitat densities higher (7.2/100ha) in old-growth forest than in clearcuts (0.2/100ha) mid-successional stages used less frequently during winter	
				Ruffed Grouse		Gullion 1977		
				Blue Grouse	s.e. AK	Doerr et al. 1983		
				density	Blue Grouse	B.C., Canada		Hines 1987
	edge	linear ft./territoryuse		Blue Grouse	NY	Martinka 1972		most consistent habitat characteristic
	mid-story cover ground vegetation	% cover	presence	Ruffed Grouse	B.C., Canada	Donaldson and Bergerud 1974		most consistent habitat characteristic
					w. WA	Salo 1978		vine maple cover = 50%; total = 52%
			use	Spruce Grouse	PA	Sharp 1963		reduction in ground cover associated with reduced brood production
			use	Spruce Grouse	Maine	Allan 1985		in winter used sites with less cover than in summer
	canopy closure	% cover	nest site selection use by males density	Spruce Grouse	Alberta, Canada	Kepple and Herzog 1978		nest sites correlated with vegetation cover
				Spruce Grouse	Alberta, Canada	Kepple and Herzog 1978		used dense sites with little ground vegetation
				Ruffed Grouse	w. WA	Salo 1978		"aggregated" in stands with 62% closure
				Blue Grouse	B.C., Canada	Donaldson and Bergerud 1976		highest density in forests with 50% canopy closure; low density in closed canopy
			use	Spruce Grouse	Maine	Allan 1985		winter cover 80%; summer 70%
			use	Ruffed Grouse	Alberta, Canada	Rusch and Keith 1971		closure at down logs significantly lower (66%) than random logs (109%)
			use (cover)	Blue Grouse	B.C., Canada	Donaldson and Bergerud 1974		**** shelter/courtship value (hypothesis)
		% conifer cover	density	Ruffed Grouse	s.e. ID	Stauffer and Peterson 1986		45% cover in fall-winter
					w. WA	Salo 1978		negative correlation between conifer cover and grouse abundance
	site heterogeneity		presence/absence density	Ruffed Grouse	w. WA	Salo 1978		grouse associated with small forest openings
				Blue Grouse	B.C., Canada	Donaldson and Bergerud 1974		highest density of displaying males was in mosaic
		use	Spruce Grouse	NY	Martinka	thickets; 675 ft of edge		
				Alberta	Pendergast and Boag 1978	females with broods feed in openings		
stem density	trees/ac	density	Blue Grouse	Maine	Allan 1985	females with broods feed in openings		
				B.C., Canada	Hines 1987	heavily used 4-15 yr-old sites had 560-1200 trees/ac		
			Spruce Grouse	Maine	Allan 1985	density (1611-2290 trees/ha) influenced use by males		
	basal area trees/ac	presence/absence	Ruffed Grouse	Maine	Hedberg 1980	grouse used area with 1050-2460 trees/ha		
			w. WA	Salo 1978	not present in stands with BA > 48n /ha			
			Ruffed Grouse	Alberta, Canada	Rusch and Keith 1971	sapling density at logs significantly less than random log sites		
	presence of clumps	use	Blue Grouse	NY	Russehl 1963	clumps of saplings and shrubs may provide brood cover		
dead trees (snags)			Spruce Grouse	Alberta, Canada; Maine	McDonald 1968, Allan 1985	use of dead trees may enhance visibility of courtship display		
downed logs		courtship	Ruffed Grouse	Alberta, Canada	Herzog and Boag 1977	courtship display		
				w. WA	Salo 1978	1.83 logs/ territory for drumming		
				MI, Mich., Alberta, Ohio	Gullion et al. 1962, Palmer 1963, Boag	logs present at all sites		
					Sumanik 1969, Stell et al. 1979			

	moisture conditions	presence	Blue Grouse	B.C., Canada	Donaldson and Bergerud 1974	moisture appeared to influence distribution of females
			Ruffed Grouse	MI	Godfrey 1975	use poorly-drained, peat-organic lowlands
				w. WA	Salo 1978	use habitat along streams
Landscape	hardwood presence 0/area	presence	Ruffed Grouse	w. WA	Salo 1978	black cottonwood present at most sites
	mosaic		Ruffed Grouse	MI	Godfrey 1975	used profuse lowland ground vegetation; roosted in upland conifers

MONITORING VARIABLES-Non-game mammals

Level	variable	units	wildlife output	topic species/guild	geographic location	References	comments	
Stand	litter cover; depth downed woody material	1/2 cover; depth measure presence/absence	prey availability	small mammals	w. WA	Gutther et al. 1983	clearcuts lacked downed woody material and considered too xeric for this species	
			presence/absence	red-backed vole	NY	Ramirez and Hornocker 1981		
			density	red-backed vole	Ontario, Canada	Hartell 1983a		site scarification reduced debris cover; this species absent within 2 years of treatment
	herbaceous cover	1/2 cover	diversity	density	deer mouse	Ontario, Canada	Hartell 1983a	numbers increase after scarification
				density	long-tailed vole	s.e. AK	Van Horne 1982b	density positively correlated with log cover
				density	small mammals	Maine	Clough 1987	
	understory vegetation cover	1/2 cover	diversity	density	n. red-backed vole	AK	West et al. 1988	hypothesize that lack of cover leads to unequal distribution of overwintering females and subsequent distribution
				density	deer mouse	s.e. AK	Van Horne 1982a	no correlation between adult density and herbaceous cover
				density	long-tailed vole	s.e. AK	Van Horne 1982b	density and shrub cover positively correlated
				density	snowshoe hare	Maine	Litvaitis et al. 1985	
	shrub density	1/2 cover	density	density	cottontail rabbit	s. AZ	Costa et al. 1976	population increase after treatment associated with increased herb production
				overwinter survival	small mammals	Maine	Clough 1987	
	sapling cover and height	1/2 cover; foliage volume	density	density	deer mouse	New England	Miller and Getz 1977	negative correlation between density and increasing vegetation cover
				density	snowshoe hare	Alberta, Canada	Keith and Surrendi 1971	reoccupation occurs following brush regeneration second year after wildfire
tree stem density	stems/ac; canopy cover	density; biomass	density	snowshoe hare	British Columbia	Sullivan and Moses 1986	numbers decreased after scarification removed 85% of low cover	
			density	snowshoe hare	British Columbia	Sullivan and Moses 1986	adults had lower survival rate on scarified plot	
soil moisture	wet, moist, dry	density	density	snowshoe hare	British Columbia	Sullivan and Moses 1986	conifers 3-18 ft tall provide accessible food and thermal cover in winter	
			density	snowshoe hare	New Brunswick	Parker 1986		
distance to water	linear distance	density	density	several species	Georgia	Atkeson and Johnson 1979	shift from annual to perennial vegetation after canopy closure at 5-7 years	
			density	red-backed vole	CT	Carey et al. 1981	density correlated with increasing moisture	
food availability	abundance	density	density	shrews	CO	Spencer and Pettus 1966	Sorex cinereus, S. vagrans caught near water	
			density	shrews	CO	Spencer and Pettus 1966	Sorex nanus most common shrew in clearcut; had greatest average trap distance from water	
stand age	age in years	abundance, diversity	abundance	red-backed vole	OR	Ure and Naser 1982	hypothesized that loss of hypogeous fungi from clearcuts may influence occupancy	
			diversity	shrews	w. WA	Gutther et al. 1983	hypothesized that insect prey more available after clearcut harvest	
vegetation nitrogen content	amount/area	abundance	abundance	small mammals	Maine	Clough 1987		
			growth, survival	small mammals	Ontario, Canada	Hartell 1983b	community response varied by treatment type	
ecotone interspersions	edge	density	density	small mammals	West Virginia	Kirkland 1977	higher diversity in clearcut stage	
			density	long-tailed vole	Maine	Monthey and Soutiere 1985	abundance increased following harvest	
Landscape	edge	density	density	long-tailed vole	s.e. AK	Van Horne 1982b		
			density	snowshoe hare	Michigan	Conroy et al. 1979	use of clearcut "light to moderate"; high use of ecotone between clearcut and forest	
			density	snowshoe hare	Maine	Monthey 1986	greater than expected use of landscape dominated by clearcuts; softwood patches	

MONITORING VARIABLES-Carnivores

Level	variable	units	wildlife outputs	topic species/guild	geographic location	References	comments
Stand	canopy cover	% canopy cover	home range	pine marten	ID	Koehler and Hornocker 1977	highest activity during normal winter snow depths was in stands with > 30% canopy cover
					CA Newfoundland	Bargis and McCollough 1984 Bateman 1986	prefer areas with 100% cover used coniferous forests with low overstory density in proportion to avail
	basal area	basal area	home range	pine marten	Soviet Union	Grakov 1972	caught fewer animals where harvest exceeded 65-75% of forest
	distance to cover	feet	presence/absence	black bear	NH	Rogers et al. 1988	most use of clearcuts < 200 m from forest cover
				grizzly bear	MT	Zager et al. 1977	82% of 35 locations in clearcuts were within 50 m of cover
	cover availability	amount, quality	presence/absence	black bear	NH	Rogers et al. 1988	shade cover in openings important to minimize heat stress; females with cubs forage within 200 m of rough-barked trees easily climbed by cubs
	forage availability	foliage volume	home range	black bear	NH	Rogers et al. 1988	availability of berries
	prey availability	density	home range	pine marten	Maine Northwest Territories	Majer 1979 Douglass et al. 1983	presence related to prey availability suggest that martens select areas based partly on prey abundance
					ID	Koehler and Hornocker 1977	summer food habits from scat analysis indicates use of openings
					CA	Zielinski et al. 1983	food habits from scat analysis indicates use of openings
	snag availability	basal area/ac	nest sites roost sites	pine marten	CA NY	Martin and Barrett 1983 Buskirk et al. 1988	23% of the nest sites found cover facilitates thermal regulation
	downed wood material	presence/absence; tons/ac presence/absence	nest sites home range	pine marten	CA ID Newfoundland	Martin and Barrett 1983 Koehler and Hornocker 1977 Bateman 1986	68% of the nest sites found used when crossing openings
			abundance	carnivores	n.w. CA	Raphael 1988	provides subnival access during winter no species more common in younger seral stages compared to older
		density	pine martens	Newfoundland	Snyder and Bissonette 1987	capture rates higher in residual stands than in clearcut	
Landscape	opening size	feet; area	home range	pine marten	CA	Bargis and McCollough 1984	largest opening crossed in winter was 135 m; meadows > 50 m crossed using cover of scattered trees
				black bear	MT	Jonkel and Cowan 1971	used clearcuts in proportion to availability
	fragmentation	total size; proportion treated	hunting area home range	pine marten pine marten	ID Maine	Koehler and Hornocker 1977 Soutiere 1979, Steventon and Majer 1982	travel in clearcuts; don't hunt there territory size appears to increase in proportion to amount of clearcut in home range
	stand age	age in years	home range	black bear	s.w. WA	Lindzey and Meslow 1977	5-10 year-old stands used in greater proportion than availability; stands > 40 years used less than availability

MONITORING VARIABLES - Big Game

Level	variable	units	wildlife output	topic species/guild	geographic location	references	comments	
Stand	canopy closure	percent	winter use	elk	s.e. AK PNW n. ID	Batchelor 1965 Peek et al. 1982 Irwin and Peek 1983	forest cover important during snowy weather	
				white-tailed deer	n. ID n.w. NY n.w. NY	Owens 1981 Kundtlinger 1979 Jenkins and Wright 1987		
				mule deer	CO	Loveless 1964 Jones 1974 Gelst 1981		
				black-tailed deer	n.e. OR Vancouver Is. Vancouver Is. Vancouver Is.	Leckenby and Adams 1986 Jones 1974 Bunnell 1979 Taber and Hanley 1979		
				mule deer	n.-c. WA n. ID n. Vancouver Is.	Ziegler 1978 Keay and Peek 1980 Rochelle 1980		
				caribou	n. ID	Johnson and Peek 1984 Serreen and Lyon 1989		
				mountain goat	PNW	Arnett and Irwin 1989		
				elk, deer	PNW	Black et al. 1976		
				elk	v. NY	Marcus 1974		
				low vegetation cover	presence/absence			
	elk	n. ID	Koen 1973 Leege 1969					
	mule deer	s.e. AK	Irwin and Peek 1983 Barrett 1979					
	caribou	n. ID	Ziegler 1978 Keay and Peek 1980 Rochelle 1980					
	food supplies				elk	n. ID Olympic Pen., WA	Johnson and Peek 1984 Serreen and Lyon 1989 Taber and Raedeke 1980	little food value in 2nd growth/mid succession forest
					elk	v. NY n. ID	Marcus 1974 Irwin and Peek 1983	prefer to forage in areas with < 25% canopy cover hypothesize that phenology of forage supply influences habitat use
					elk	n. ID v. WA	Irwin and Peek 1983 Schrorer 1987	generally avoided old-growth forests in winter use deciduous/coolifer forests < 150 yrs old less than expected
					elk	v. WA v. WA	Kahn 1985 Schoen 1977	summer clearcut use influenced by advanced regeneration; old-growth used on clear, hot days used areas 5-15 yrs post-harvest; generally avoided old-growth in summer
	stand age	age in years			elk	n. ID	Irwin and Peek 1983	select second-growth in late summer/fall seasonal patterns of habitat use
					elk	v. WA	Schrorer 1987	
	stand size	area (ac)			elk and deer	n. ID n.w. WA	Irwin and Peek 1983 Hanley 1984	optimal stand size based upon habitat use data optimal size 30-60 ac
elk					n.e. OR, s.e. WA n.e. OR	Thomas 1979 Leckenby 1984		

edge effect
human disturbance
cover/forage ratio

elk
elk
elk,deer

n.w. WA
Mt
n.e. OR, s.e. WA

Jenkins and Starkey 1984 hypothesized ecotone facilitates use of clearcuts
Lyon 1983 major effect of open roads
Black et al. 1976

Landscape

integration of cover,
roads, and forage

habitat effectiveness
index

elk

w. OR
n.e. OR, s.e. WA

Windon et al. 1986 habitat effectiveness model
Thomas et al. 1986 habitat effectiveness model
Thomas et al. 1988 habitat effectiveness model; winter range

APPENDIX C. GLOSSARY OF TECHNICAL TERMS USED IN TEXT

- adaptive management** process that modifies management policy as a result of experience gained in implementing management decisions that are designed as scientific experiments.
- algorithm** a procedure for solving a mathematical problem in a finite number of steps that frequently involves repetition of an operation (Webster).
- alluvial lands** lands characterized by clay, silt, sand, gravel, or similar detrital material deposited by running water (Webster).
- area model** a class of mathematical models which are designed to assess relationships over a broad area.
- balloon logging** a tree-removal procedure that utilizes balloons to move trees to landings.
- Baye's theorem** a statistical theory in which probabilities are associated with individual events, and not merely with sequences of events (Webster).
- biomass** the quantity of any living organism per unit area; usually measured in terms of weight/area.
- central tendency** the description or measurement of average or median values.
- continuous variable** a variable that is capable of being grouped into classes.
- cover type** unit of habitat classification involving structural resources that enhance wildlife reproduction and/or survival, usually a descriptive term for the current vegetation conditions on a site.
- density dependent** effect or a factor is increasingly expressed with increasing population density.
- deterministic assumptions** assumptions based on the pretext that occurrences are causally determined by preceding events or natural laws (Webster).
- dispersal** the act of an animal leaving its living place and seeking another suitable home site.
- disturbance regime** collective set of factors which can cause ecologically significant change to forests or habitats.
- diversity** the relative degree of abundance of wildlife species, plant species, communities, habitats, or habitat features per unit of area (Thomas 1979).

edge effect the increased richness of flora and fauna occurring in the transition zone where two plant communities or successional stages meet and mix (Brown 1985).

equilibrium state of adjustment between opposing or divergent influences or elements (Webster).

expert system a computer-based consultation program which uses built-in rules to help classify, diagnose, or plan.

faunal assemblage collective list of species occurring within an environment.

fragmentation process of reducing size and connectivity of stands which comprise a forest.

fluvial lands lands influenced or produced by stream action (Webster).

guild a group of plants or animals that have ecological interrelationship and a similar mode of life; e.g. species which use tree boles for nesting (Brown 1985).

habitat selection behavioral process by which animals perceive their environments and make decisions about which habitat conditions to use.

habitat suitability index specific mathematical equation or rule-set for evaluating the condition of a habitat.

habitat type the aggregate of all areas that support, or can support, the same primary vegetation association; a classification of environmental settings characterized by a single plant association; the expression through the plants present of the sum of the environmental factors that influence the nature of the climax (Daubenmire 1976).

HEP, or habitat evaluation procedures a set of rules for conducting quantified analysis of wildlife habitat quality.

helicopter logging a timber-harvest procedure utilizing helicopters to move logs to landings.

high-lead a cable yarding system utilizing a spar or tower in order to provide lift to one end of the logs as they are dragged across the ground to a landing; suitable for yarding at distances of up to 1200 feet (Brown 1985).

independence the outcome of a trial, or sample from a population does not depend in any way on what happens in other trials.

interspersion the intermixing of plant species and plant communities that provide habitat for animals in a defined

area (Thomas 1979).

juxtaposition the act of arranging stands in space (Thomas 1979).

landscape ecology the study of ecological relationships involving aggregate landforms in a region.

landtype association a particular unit with characteristic soils and landforms, which are permanent elements of ecosystems that have predictable patterns.

managed forest human-directed shifting mosaic of dynamic forest patches.

management guild an ecological collection of wildlife species that respond in a similar way to a variety of habitat changes.

management indicator species wildlife species whose population changes are hypothesized to reflect similar responses to management by other species which use the same habitats.

metapopulation population comprised of more-or-less disjunct sub-populations.

model a formalized expression of a theory or of the causal situation that generated observed data.

model validation process of developing an acceptable level of confidence that a specific model adequately represents the actual biological process and field situation being simulated.

mosaic a descriptive term for a mixture of vegetational successional stages and habitat types in an area.

niche the peculiar arrangement of food, cover, and water that meets the requirements of a particular species (Hanson 1962 from Thomas 1979).

ordination a quantitative method for classifying a collection of units into categories that contain similar items.

patch a part or area distinct from that about it (Webster).

patch diversity relative degree of abundance of distinct vegetational types or cover types in an area.

pole stand a forest stand of trees generally between 4-9 inches diameter at breast height. Trees enter this stage when lower branches begin to die and remain until crown growth slows and crown expansion is noticeable (adapted from Brown 1985).

population sink environment in which a species' population survives but does not reproduce successfully.

productivity the rate at which a population increases or the rate at which harvestable surpluses of animals are produced.

proximate factors items or forces which act directly to influence an animal's behavior.

recruitment production of sexually mature animals.

richness a measure of the relative degree or number of plant or wildlife species or both associated with particular habitat conditions (Thomas 1979).

risk analysis process of quantifying probabilities of chance pleasant and unfavorable events associated with uncertainty during a decision-making exercise.

seral stage the relative transitory aggregation of plants and animals within a sere; a preclimax stage of succession (Brown 1985).

skyline a cable yarding system utilizing a spar or tower to provide lift to one end of the logs as they are dragged across the ground to a landing; such systems are capable of yarding for distances up to 2600 feet (Brown 1985).

spatial statistics a set of analytic procedures for processing mapped data (Berry 1987).

stochastic events random or unpredictable occurrences.

stratification division of a population into sub-populations or groups.

successional stage a stage or recognizable condition of a plant community which occurs during its development from bare ground to climax (Brown 1985).

thermal cover structural conditions within a site, usually vegetational, which ameliorate effects on wildlife from inclement weather.

tractor yarding a method of moving logs across the ground from the point of felling to a central location or landing through the use of tracked or wheeled vehicles (Brown 1985).

ultimate factor environmental factor which influenced evolution of genetic adaptations of a species by affecting reproduction or survival.

windthrow a tree or trees uprooted or felled by the wind (Ford-Robertson 1971 from Brown 1985).

APPENDIX D. WILDLIFE/FORESTRY RESEARCH PROGRAMS IN THE PACIFIC NORTHWEST

I. U.S. Forest and Range Experiment Station, LaGrande, Oregon

Elk/cattle/timber harvest interactions are being examined in the Blue Mountains, at the U.S. Forest Service's Starkey Experimental Forest and Range, 30 miles southwest of LaGrande, Oregon. The work involves controlled conditions within large, game-proof fenced pastures.

The scientific community is unable to answer specific questions regarding intensive forest management and deer/elk habitat management. Some wildlife managers and many hunters view intensive forest management as incompatible with deer/elk production. Conversely, some forest managers view deer/elk habitat management as a constraint to their ability to produce maximum timber volumes. The issue of "preference versus requirement" is used as a pivot point by both sides to argue the pros and cons of ungulate management.

If the timber manager is required to meet certain habitat standards for elk/deer habitat that are not biological requirements for optimal performance by the elk and deer, then opportunity and management costs are being incurred needlessly. Conversely, if the present habitat standards and guidelines for deer/elk reflect "requirements" rather than species "preference", game managers and managers of other resources must alter their management strategies to meet those "requirements" to obtain their overall goals.

In addition, numerous unanswered questions include management of road traffic and intensive timber management and harvest, opportunity costs for road-related recreation, and hunting effects on big game population composition.

Objectives of the Starkey Program include:

- a. To develop consumptive forage equivalency rates for mule deer, elk, and domestic livestock that can be used throughout the intermountain west;
- b. To determine the effects of intensive forest management practices upon the physiological, biological, and behavioral traits of mule deer, elk, and domestic livestock;
- c. To determine the effect of elk bull/cow ratios and male age upon the breeding coefficient of the cow cohort; and
- d. to determine the relationship between motorized vehicle traffic on forest roads and the habitat effectiveness of adjacent forest vegetative types.

II. U.S. Forest Service, Pacific Northwest Research Station

The Forest Service has developed a Spotted Owl Research, Development & Application Program (RD&A), which conducts research and monitoring of northern spotted owls in Washington, Oregon, and California. The program includes coordinated monitoring among the National Forests in Region 6 and Region 5. Research and monitoring topics are coordinated among several Forest Service work units. Those conducted under the auspices of the Pacific Northwest Research Station include:

- a. Adult spotted owl monitoring on the Eugene and Medford BLM Districts (conducted by Cooperative Wildlife Research Unit, OSU).
- b. Spotted owl habitat use: Oregon Coast Range
- c. Demographic characteristics of spotted owl populations in the Oregon Coast Range and Olympic Peninsula of Washington.
- d. Use and home range characteristics of spotted owls on the Olympic Peninsula, Washington.
- e. Northern spotted owl and northern barred owl habitat use in northern Washington (in cooperation with Washington Department of Wildlife).
- f. Patterns of flying squirrel abundance: Coast Range and Olympic Peninsula prey studies.
- g. The ecology of the spotted owl on the Willamette National Forest: Prey studies (in cooperation with Cooperative Wildlife Research Unit, OSU).
- h. The ecology of spotted owls on the Willamette National Forest: habitat use and demography (in cooperation with Cooperative Wildlife Research Unit, OSU).

Other studies conducted by the Pacific Northwest Station include the following:

- a. Pilot study: technique evaluation and development for studies of the ecology of marten in the Pacific Northwest.
- b. Northern goshawk habitat associations: a pilot study on the Olympic Peninsula and western Cascades of Washington.
- c. Landscape-level analysis of vertebrate community data.
- d. Long-term monitoring of avian populations and environmental conditions in different-aged Douglas-fir forests.

III. USDA Forest Service, Forestry Sciences Laboratory, Juneau, Alaska.

For the past 8 years this lab has focused on the nutritional ecology of black-tailed deer (Hanley, pers. comm.). Black-tailed deer are considered a "management indicator species" because of their relatively large home ranges, need for a diversity of habitats within relatively close proximity to one another, and their need for old-growth forest during winter. Deer are probably the most difficult to accommodate in logging plans in southeast Alaska, primarily in relation to huntable populations. Station scientists believe an understanding of the nutritional interactions between deer and their environment is paramount to understanding how forest management affects the productive capacity of habitat for deer.

Thus, research centers on overstory-understory relationships; the chemical ecology of understory plants; overstory-snow relationships; energy costs; deer diet composition, quality, and intake rate; net foraging efficiency relative to forest conditions; and the role of tannins and other phenolic plant compounds in protein and energy digestion relative to the diet selection process in deer.

In the future, the lab will decrease emphasis on deer and shift to working with riparian habitats and ecology of birds, with questions on frugivory and the role of birds in dispersal of understory seeds.

IV. Oregon Department of Fish and Wildlife (ODF&W).

The Department conducts or supports research on a number of projects involving wildlife and forestry. For example, ODF&W supports the Starkey big game research project described above, and cooperates in research conducted at Oregon State University. The Department recently produced a second-draft of a Wildlife Research Plan (Bright and Marshall 1989), which recognizes the need for development as well as verification of wildlife-habitat relationships models. The research plan directs its emphasis mostly to those species which are of special concern, including threatened, endangered or those which could become threatened or endangered, the most popular game species, and those which cause damage to commodity uses or game species.

The Department also contributes to the following studies:

- a. Response of vertebrates to habitat conditions along riparian zones in the Oregon Coast Range.
- b. Determine nest locations and habitat of marbled murrelet.
- c. Black bear ecology/damage study.

- d. Determine white-headed woodpecker habitat requirements.
- e. Feasibility of using habitat indices for inventory of black-tailed deer.

Literature Cited:

Bright, L. and D.B. Marshall. 1989. Wildlife Research Plan--
Second Review Draft. Oregon Dep. Fish & Wildlife,
Portland.

V. Washington Department of Wildlife (WDW).

WDW conducts a host of monitoring efforts to gather data such as population inventories and surveys for routine management programs. Also, WDW conducts research to support management. Research topics which relate to forestry and wildlife are described below:

- a. The North Cascades grizzly bear project includes identification of bear presence, distribution, and population dynamics as well as Landsat mapping of habitat.
- b. Projects on the ecology of big game species involve a series of information-gathering efforts on mule deer (e.g., habitat use, productivity, habitat suitability modelling); black-tailed deer; Rocky Mountain elk (productivity, habitat use, mortality habitat modelling); Roosevelt elk (productivity, census techniques, mortality rates, habitat suitability modelling).
- c. Remote sensing research involves gathering up-to-date cover type mapping data and exploring new areas of spatial data analysis, including interfacing with WDW's geographic information system. This project includes an expansion of mapping of old-growth and forest stand conditions to the Olympic Peninsula.
- d. WDW is developing habitat models for more sophisticated analysis of geographic data to supply habitat quality measures. This work aims to investigate the need for and develop new methods, and evaluate the Landsat cellular system relative to the GIS vector system as data processing media.

VI. University of British Columbia Department of Forest Sciences, Vancouver.

Scientists in the Department of Forest Sciences at the University of British Columbia (UBC) conduct forestry/wildlife research, most of which is coordinated with the forest industry, British Columbia Fish and Wildlife Branch, and the B.C. Forest Service (F.Bunnell, pers. comm.). The Integrated Wildlife Intensive Forestry Research program (IWIFR) has been developing various models and handbooks, following 4 years of coordinated

research. The work relates to the TFW Program because they examined and subsequently rejected HSI models and explored various indices (specifically for black-tailed deer and Roosevelt elk). Currently, they use a hierarchical key to focus quickly on potential problem areas and potential solutions.

Related to IWIFR, the UBC researchers have been examining a program to treat landscape patterns of cutblocks on coastal Douglas-fir and cedar-hemlock. In addition funding was acquired recently to manage for wildlife diversity in coastal forests, in a new program.

Present coordinated research includes the following:

- a. Managed stands for deer winter range. This work is cooperative with B.C. Forest Service, Canadian Forest Service, and the forest industry. This large project will provide tools for designing silvicultural prescriptions to create black-tailed deer winter range in managed stands.
- b. Locating stands for deer winter range. This relatively large effort involves some cooperation with the U.S. Forest Service in Juneau, Alaska. Major cooperators include the forest industry, B.C. Forest Service, and B.C. Fish and Wildlife. The goal is to expand the previous project (listed above) to a watershed or landscape basis in coastal forests. Assuming that cooperators can create winter range in a stand, researchers are asking where the stands should be located, how many should be involved, etc.
- c. Edge, interspersion, and black-tailed deer. Scientists are examining the concept of edge-effects as they may relate to black-tailed deer. The work is developing techniques which are useful in integrative management.
- d. Influence of forestry practices on bald eagles. Scientists are exploring the feasibility to develop approaches to integrated management that are not unduly restrictive around bald eagle nests.

VII. Center for Streamside Studies in Forestry, Fisheries, and Wildlife, University of Washington, Seattle.

A major element of the Center for Streamside Studies includes research on interactions of forestry, fisheries, and wildlife (Naiman and Raedeke, pers. comm.). Three topics have been identified for research focus: interactions, landscape patterns, and disturbance. Some of the important research topics are listed below. The scientists are deeply interested in addressing management aspects associated with the TFW Agreement.

- a. Role of large animals in influencing the structure and dynamics of aquatic and terrestrial ecosystems.

- b. Role of patch boundaries in regulating the flow of materials and information between terrestrial and aquatic ecosystems.
- c. Long-term landscape modification by beaver and moose.
- d. Population dynamics of managed deer populations on commercial forest lands.
- e. Elk forage enhancement through grass-seeding on forest plantations in western Washington.
- f. Wildlife and landscape patterns in a managed forest (part of a GIS study of inter-relationships of all resources at the watershed level).
- g. Status of riparian wetlands in the Cedar River Watershed, and their importance as wildlife habitat, particularly as elk winter range.
- h. Impacts of timber sales on deer winter range habitat use and movement patterns in the southern Cascades of Washington.
- i. Relationships between wildlife and habitats in old-growth forests.
- j. Wildlife response to nitrogen fertilization and suppression of tree regeneration on a right-of-way.
- k. Effects on wildlife of sludge amendment to commercial forests.
- l. Small mammal damage to forest plantations.
- m. Life history of the forest deer mouse.
- n. Habitat use patterns of great horned owls in a mosaic of old-growth and managed forests.
- o. Bird populations in managed forests, with special reference to forest thinning.

VIII. Oregon State University Departments of Fish and Wildlife and Forest Science, Corvallis, Oregon.

Several coordinated research programs conduct examinations of forestry/wildlife relationships at Oregon State University. These are conducted primarily through the two Departments and include the Cooperative Wildlife Research Unit and the Coastal Oregon Productivity Enhancement Program (COPE). Research topics run the range of scale from stand-related studies to landscape relationships. Several important topics are listed below.

- a. Response of small mammals and amphibians to site preparation adjacent to riparian zones in the central Coast Range.

- b. Spotted owl nest site characterization.
- c. Differential habitat use by male and female mule deer.
- d. Spotted owl and great-horned owl densities in relation to forest fragmentation.
- f. Determine spotted owl home range and habitat use (specific studies listed above, in cooperation with Forest Service).
- g. Determine beaver habitat relationships.
- h. Determine response of Douglas-fir seedlings to simulated deer damage.
- i. Winter and summer roost site selection by white-breasted nuthatches.
- j. Effects of forage improvements on Roosevelt elk in the Coast Range.
- k. Landscape studies using GIS and wildlife habitat models.

IX. NCASI Wildlife Biology Program, Corvallis, Oregon

The National Council of the Paper Industry for Air and Stream Improvement conducts cooperative research on forestry/wildlife relationships with the goal of understanding interactions in managed forests. Currently, the program cooperates with the forest products industry, Bureau of Land Management, U.S. Forest Service, U.S. Fish and Wildlife Foundation, U.S. Fish and Wildlife Service, Oregon Department of Fish and Wildlife, and Washington Department of Wildlife. Program topics for investigation are described below.

- a. Spotted owl nest site characterization in a managed forest, Wenatchee National Forest, Washington.
- b. Spotted owl home range and habitat use in a managed forest mosaic, Eugene, Oregon.
- c. Demography of spotted owls in a reserved area and managed forest, Wenatchee National Forest, Washington.
- d. Relationship of spotted owls and forest practices in managed forests, Washington and northern California.
- e. Radio-telemetry applications for habitat relationships of marbled murrelets.
- f. Relationships between forest cover, forage conditions, snow characteristics, and big game energetics, northeastern Oregon and southeastern Washington.

g. Marbled murrelet distributions and habitat association along Oregon coast (contribution to OSU research listed above).

APPENDIX E. LIST OF SCIENTIFIC NAMES ASSOCIATED WITH COMMON NAMES USED IN TEXT

Common Name	Scientific Name
A. Amphibians	
Ensatina tailed frog	Ensatina escholtzii Ascaphui truei
red-legged frog	Rana aurora
Pacific tree frog	Hyla regilla
spotted frog	Rana pretiosa
rough-skinned newt	Taricha granulosa.
clouded salamander	Aneides ferreus
Del Norte salamander	Plethodon elongatus
Dunn's salamander	Plethodon dunni
Cope's giant samamander	Cicamptodon copei
Pacific giant salamander	Dicamptodon ensatu
Larch Mountain salamander	Plethodon larselli
northwestern salamander	Ambystoma gracile
Van Dyke's salamander	Plethodon vandykei
Olympic salamander	Rhyacotriton olympicus
Oregon slender salamander	Batrachoseps wrighti
slender salamander	Batrachoseps attenuatus
Western redback salamander	Plethodon vehiculum
B. Reptiles	
alligator lizard	Gerrhonotus coeruleus
sagebrush lizard	Sceloporus graciosus
western fence lizard	Sceloporus occidentalis
western skink	Eumeces skiltonianus
western pond turtle	Clenmys marmorata
C. Birds	
mountain bluebird	Sialia currucoides
western bluebird	Sialia mexicana
bufflehead	Bucephala albeola
brown-headed cowbird	Molothrus ater
yellow-biilled cuckoo	Coccyzus americanus
wood duck	Aix sponsa
bald eagle	Halaieetus leucocephalus
peregrine falcon	Falco peregrinus
dusky flycatcher	Empidonax oberholseri
Barrow's goldeneye	Bucephala islandica
common goldeneye	Bucephala clangula
northern goshawk	Accipitier gentilis
blue grouse	Dendragopus obscurus
ruffed grouse	Bonasa umbellus
spruce grouse	Dendragopus canadensis
Cooper's hawk	Accipiter cooperi

red-tailed hawk	<i>Buteo jamaicensis</i>
sharp-shinned hawk	<i>Accipiter striatus</i>
great blue heron	<i>Ardea herodias</i>
black-crowned night heron	<i>Mycticorax mycticorax</i>
ruby-crowned kinglet	<i>Regulus calendula</i>
common loon	<i>Gavia immer</i>
purple martin	<i>Progne subis</i>
common merganser	<i>Mergus merganser</i>
hooded merganser	<i>Lophodytes cucullatus</i>
merlin	<i>Falco columbarius</i>
marbled murrelet	<i>Brachyramphus marmoratus</i>
osprey	<i>Pandion haliaetus</i>
barred owl	<i>Strix varia</i>
boreal owl	<i>Aegolius funereus</i>
flammulated owl	<i>Otus flammeolus</i>
great gray owl	<i>Strix nebulosa</i>
great horned owl	<i>Bubo virginianus</i>
saw-whet owl	<i>Aegolius acadicus</i>
screech-owl	<i>Otus kennicotti</i>
spotted owl	<i>Strix occidentalis</i>
chipping sparrow	<i>Spizella passerina</i>
tree swallow	<i>Tachycineta bicolor</i>
black swift	<i>Cypseloides niger</i>
Vaux's swift	<i>Chaetura vauxi</i>
turkey vulture	<i>Cathartes aura</i>
black-backed woodpecker	<i>Picoides arcticus</i>
Lewis' woodpecker	<i>Melanerpes lewis</i>
pileated woodpecker	<i>Dryocopus pileatus</i>
three-toed woodpecker	<i>Picoides tridactylus</i>
white-headed woodpecker	<i>Picoides albolarvatus</i>
winter wren	<i>Troglodytes troglodytes</i>

D. Mammals

pallid bat	<i>Antrozous pallidus</i>
Townsend's big-eared bat	<i>Plecotus townsendii</i>
black bear	<i>Ursus americanus</i>
grizzly bear	<i>Ursus arctos</i>
mountain beaver	<i>Aplodontia rufa</i>
bobcat	<i>Lynx rufus</i>
caribou	<i>Rangifer tarandus</i>
least chipmunk	<i>Eutamias minimus</i>
red-tailed chipmunk	<i>Tamias ruficaudis</i>
black-tailed deer	<i>Odocoileus hemionus columbianus</i>
mule deer	<i>Odocoileus hemionus hemionus</i>
white-tailed deer	<i>Odocoileus virginianus</i>
Columbian white-tailed deer	<i>Odocoileus columbianus leucurus</i>
red deer	<i>Cervus elaphus</i>
Rocky Mountain elk	<i>Cervus elaphus nelsoni</i>
Roosevelt elk	<i>Cervus elaphus roosevelti</i>
fisher	<i>Martes pennanti</i>
snowshoe hare	<i>Lepus americanus</i>
northern bog lemming	<i>Synaptomys borealis</i>
mountain lion	<i>Felis concolor</i>

Canada lynx	Lynx canadensis
pine marten	Martes americana
deer mouse	Peromyscus maniculatus
Pacific jumping mouse	Zapus trinotatus
fringed myotis	Myotis thysonodes
Keens's myotis	Myotis keenii
long-eared myotis	Myotis evotis
long-legged myotis	Myotis volans
pronghorn	Antilocapra americana
cottontail rabbit	Sylvilagus nuttalli
ringtail	Bassariscus astutus
bighorn sheep	Ovis canadensis
Trowbridge shrew	Sorex trowbridgii
vagrant shrew	Sorex vagrans
striped skunk	Mephitis mephitis
western gray squirrel	Sciurus griseus
California ground squirrel	Spermophilus beecheyi
flying squirrel	Glaucomys sabrinus
Douglas squirrel	Tamiasciurus douglasii
red squirrel	Tamiasciurus hudsonicus
meadow vole	Microtus pennsylvanicus
Oregon vole	Microtus oregoni
red-backed vole	Clethrionomys californicus
gray wolf	Canis lupus

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