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WILDLIFE USE OF MANAGED FORESTS: A LANDSCAPE PERSPECTIVE

Volume 1

Executive Summaries
and
Introduction and Technical Approach

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TIMBER FISH
&WILDLIFE

December 19, 1997

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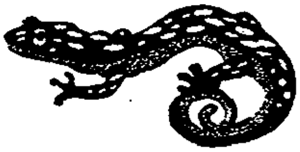

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PREFACE

The TFW Landscape Study was a collective endeavor by hundreds of people in both private industry and state and federal agencies. Without their cooperation, hard work, and commitment to the TFW research process, this study would not have been possible. We would especially like to thank the many members of the TFW Wildlife Steering Committee, who strongly supported this project throughout the many challenges we faced during its implementation, especially Lenny Young, Tim Cullinen, Pete Haug, Tom Hoppensteadt, Bob Anderson, Jim Rochelle, and Neal Wilkins. We would also like to thank Brian Gilbert and Jack Ward of Champion International, Bob Anderson of Weyerhaueser, and all of the foresters and wildlife biologists at Champion's Kapowsin and Buckley Tree Farms and Weyerhaueser's Vail Tree Farm for their help and cooperation. Jim McGowan at the Colville National Forest assisted with establishing a Memorandum of Understanding for work on the Forest. Personnel throughout the Colville National Forest system, including their GIS staff, assisted us with mapping and selection of study sites. Bob Anderson and Dick Dunton of the Department of Natural Resources, Dave Whitwill of Boise Cascade, and Dwight Opp and Lorin Hicks of Plum Creek Timber Company provided information on watershed management and gave us permission to work on lands they manage. We thank John Young, John Lehmkuhl, William Heise, and Beth Galleher of the USDA Forest Service Research Lab in Olympia for their many contributions to site selection and landscape-scale analyses through interpretation of satellite imagery and subsequent GIS analyses. The Gap Analysis Program at the University of Washington provided Landsat data through a cooperative agreement. Dave Wherry helped with the transfer of data and Doug Call, Kelly Cassidy, Lisa Nordstrom, and Lori Campbell assisted in its classification for the east-side studies. Kim Clarkin at the Colville National Forest provided the watershed boundary map. Many graduate students, including Janet Erickson, Matt Frazier, Scott Pearson, Angela Stringer, and Levon Yengoyan at the University of Washington; Jennifer Ballard, Deborah Beutler, Doug Call, Dan Friesz, Steve Mech, and Kevin Pullen at Washington State University; and Bob Griffith at Eastern Washington University, were instrumental in the successful implementation of this study. We would also like to acknowledge the hard work and dedication of the hundreds of biological technicians who collected data in the field, often in wet and rugged conditions. The project benefitted from comments provided at an early planning workshop by Rich Everett, Luke George, Bruce Marcot, Sandra Martin, Barry Noon, James Peek, Martin Raphael, and Gary White. Nancy Sturhan and Linda Chiles of the Department of Natural Resources handled administration of many contracts with patience and skill. Janet Jones of the Forest Service Research Lab in Olympia compiled and desktop-published all 3 volumes of this final report, which were copied and bound by Tim Mullen, Sharon Aubry, and Julie Ridgway.

WILDLIFE USE OF MANAGED FORESTS: A LANDSCAPE PERSPECTIVE

OVERALL EXECUTIVE SUMMARY

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PURPOSE AND APPROACH

During this century, a substantial part of the forested area of the State of Washington has been committed to timber production. Until very recently the few well-replicated descriptions of habitat use by wildlife have been conducted in unmanaged forests. To assist managers in their task of meeting the wildlife habitat objectives of the Timber, Fish and Wildlife Agreement (1987), we designed a survey to describe the patterns of wildlife response to landscapes managed primarily for timber production on both sides of the Cascade crest. Our charge in this project was twofold:

- to describe the species composition and abundance levels of wildlife and plant communities occurring in forest stands of varying structural stages and landscape configurations in watersheds managed primarily for timber production, and

- to develop methods for analyzing wildlife responses to landscape-scale habitat conditions in managed watersheds.

Our approach was to sample terrestrial vertebrate groups in forest stands occurring over a gradient of silvicultural manipulation. We wished to sample forest stands that represented the predominant size and structural classes found in watersheds that ranged from moderate to high levels of forest fragmentation. This approach allowed statistical analysis of wildlife habitat relationships primarily at the stand, but also at the landscape scale. To accomplish this we needed to stratify landscapes by landscape-use patterns. This was done by using satellite imagery and developing GIS methodologies that enabled us to classify landscapes by age class, subdivide the landscape by watershed, calculate the pattern or character of the landscape, and examine areas for their management intensity and potential as study areas. Due primarily to different harvest histories on lands west and east of the Cascade crest, landscape patterns differed sharply between the 2 regions. Responding to these differences, we designed the studies for the conditions of each region. We were interested in examining the effects of stand size, stand structure, and landscape context of stands on wildlife community patterns.

West of the crest, where forests have been harvested in uniformly large blocks, we had to forego the examination of the effects of stand size. Because pre-commercial and commercial thinning is a common practice on intensively managed stands we decided to examine its effects instead. After an extensive GIS analysis of the southern Olympic peninsula and the southern slopes of the Cascade Range, we located 20 study areas in southern Pierce County on the Buckley and Kapowsin tree farms owned by Champion International and another 4 study areas on the Vail tree farm owned by Weyerhaeuser. The sites are broadly representative of major timber producing areas in western Washington. The study areas varied from 30-80 ha on sites less than 900 m in elevation, were dominated by Douglas-fir, and located away from riparian areas. The study areas consisted of 6 replicates of 4 structural types: a clearcut stage resulting from harvest within 2-3 yrs with adjacent stands at least 5 yrs older, a pre-canopy stage about 12-20 yrs after harvest that had been pre-commercially thinned with adjacent stands recently clearcut or with fully developed canopy, a closed-canopy stage about 30-40 yrs after harvest that had not been commercially thinned with adjacent stands in any stage, and a harvest stage about 50-70 yrs after harvest that had been commercially thinned with adjacent stands in any stage.

East of the crest, we selected 2 sizes of stands for study: (1) 12-15 ha, which represented the smallest harvest units typically observed, and also approximated the mean size of Upland Management Areas (UMAs) created in this region, and (2) >34 ha, which represented the largest harvest units. We examined 3 structural stages of forest: a clearcut stage resulting from harvest <6 yrs with adjacent stands >15 yrs older; a pre-canopy stage about 16-20 yrs after harvest with adjacent stands recently clearcut or with fully developed canopy; and a closed-canopy stage >60 yrs after harvest that had not been commercially thinned with adjacent stands in any stage. Six stands of each size and forest type were selected for a total of 36 stands. These study areas were located in Stevens and Pend Oreille counties on lands owned or administered by the Colville National Forest and Plum Creek Timber.

In both regions we conducted an extensive series of spatially nested vegetation surveys on each study area to acquire information on stand composition and structure. These data were used in our search for consistent patterns of association between habitat structure and wildlife species composition and relative abundance. On the west side, landscape variables were incorporated into the study using metrics calculated for actual watershed boundaries as well as metrics calculated from 100-ha, 1,000-ha, 5,000-ha, 10,000-ha, and 25,000-ha concentric circles centered on the study areas. We used the FRAGSTATS program to calculate the landscape metrics. On the east side, we examined a number of landscape metrics based on polygon representations of forest stands. Because of intercorrelations of many variables, we used a reduced set to describe condition within watersheds and within 1-km radius circles centered on study stands or trapping grids. Variables included number, perimeter, and area of forest stands. Reflective of their different natural histories, we used various metrics and scales when analyzing landscape effects for different wildlife taxa.

TAXA OF THE GENERAL SURVEY

Species and species groups chosen for study were those that could be sampled well at stand-level scales. We endeavored to include as many species as possible within this constraint as befitting one of the first attempts to characterize wildlife use of managed forests in the Pacific Northwest. All taxa were sampled for 3 consecutive years (west-side: fall 1992 through spring 1995; east-side: spring through fall 1993-1995). We sampled forest-floor amphibians and small mammals, diurnal breeding birds, and in an exception to the constraint of stand-level scale, we also included bats. This was done to add some information to a poorly understood group. On the west side, we used echolocation call detectors to assess bat habitat use and on the east side, we used radio-telemetry to determine roost use by the silver-haired bat.

Additionally, we conducted studies on species or species groups that were expected to show a response to landscape pattern. On the west side we studied the roost-site characteristics of the long-legged myotis bat, the movement patterns of the northwestern salamander, and the movement patterns of bobcats. On the east side, we investigated the dispersal patterns of amphibians on a fragmented landscape using molecular markers as an alternative to mark-recapture methods, the use of forested strips as corridors for small mammals, the factors determining the composition of shrew assemblages, and differences in the rates of predation on artificial nests as a function of habitat and stand size. Summaries of these studies are provided in the executive summaries for the west and east side regions that follow this section. Results from these supplemental studies are provided in their entirety in the appendices of Volumes 2 and 3.

AMPHIBIANS AND REPTILES

On the west side, we conducted pitfall trapping using 6 × 6 pitfall arrays with 15-m spacing in all stands for 28 days after the onset of fall rains during 1992-1994 and captured 842 individuals of 11 species, including 6 salamanders and 5 frogs and toads. Seven pond-breeding species, 2 headwater stream-breeding species, and 2 woodland species were caught. The northwestern salamander, western redback salamander, and ensatina dominated the terrestrial amphibian communities in all 4 structural classes.

Only 2 additional species, the roughskin newt and red-legged frog, had sufficient captures for statistical analyses. All species previously found in naturally regenerated, unmanaged forests in the south-central Cascade Range in Washington also occurred in these intensively managed plantations. Among structure classes, species richness was highest in pre-canopy stands and lowest in closed-canopy stands. As has been reported elsewhere, there was no evidence that species richness increased with stand age. Physiographic influences play an important role in determining amphibian diversity within forested landscapes. Four of the 5 species analyzed showed a negative association with elevation. Harvest-age stands had the highest overall amphibian abundance and relatively high abundance values for many species. The harvest-age structure class was also high in mean species richness, and was the only class in which all 11 species were found. Amphibian community structure in harvest-age stands closely resembled that found in unmanaged forests. There were few indications that clearcut areas represent substantially lower quality habitat for terrestrial amphibians than other structure classes. Clearcut areas contained the highest abundances of western redback salamanders and roughskin newts, and had a higher expected species richness than closed-canopy forests. However, clearcut areas may represent relatively hostile environments for most stream-breeding amphibians. The 2 thinned stand conditions had the highest species richness, and harvest-age stands had almost twice the total captures of any other structure class. In contrast, unthinned closed-canopy forest had the lowest species richness, and was the only structure class in which roughskin newts and red-legged frogs were not captured. Both pre-commercial and commercial thinning appear to improve overall habitat conditions for terrestrial amphibians in forest plantations.

Physiographic conditions and stand structure were more important influences on amphibian communities in managed forests than amounts of dead woody material. Although levels of woody debris do not currently appear to limit populations of terrestrial amphibians in these landscapes, managed stands in this region still contain many large residual stumps and logs in advanced stages of decay from the original, unmanaged stand. Providing for the replacement of large logs in future stands may be important for the persistence of amphibians occurring at low population levels in managed landscapes. Amphibian diversity and abundance in intensively managed forests was more strongly influenced by stand structure and physiographic conditions than by landscape characteristics. Because harvest-age stands had much higher amphibian abundances and species richness than either closed-canopy forests or clearcut areas, silvicultural strategies that maintain a higher percentage of harvest-age forest conditions within managed landscapes might provide substantial benefits to terrestrial amphibians.

On the east side, amphibians and reptiles were captured during the pitfall surveys for small mammals. The number of captures was low and included a total of 81 individuals of 3 amphibians and 1 reptilian species. The western toad was the most common species. This species was captured more frequently in regenerating and clearcut forests than in the mature forests. It was more common in small stands than in large stands. There was considerable yearly variation in the numbers of toads observed. Western toads were captured more frequently at some sites than at others. Long-toed salamanders were captured more often in the clearcut and mature forests than in the regenerating forests. More individuals were captured in small stands than in large stands. This species also exhibited inter-year changes in capture numbers. Captures of this salamander were more evenly distributed among sites than were those of the western toad. The Columbia spotted frog and the northern alligator lizard were rarely captured. These results indicate that pitfall trapping does not effectively sample amphibian and reptile communities in northeastern Washington. The distribution of the long-toed salamander is interesting, because it suggests that the presence of coarse woody debris in clearcut areas can provide appropriate microhabitat for this species.

SMALL MAMMALS

On the west side, we sampled small mammals concurrently with terrestrial amphibians each fall from 1992-1994. We caught 3,720 individuals of 18 species. Reflecting the capture method, most of these were shrews and moles (2,564). The remaining captures consisted of rodents (1,134) and ermine (22). Statistically significant differences among the 4 structural classes were found for 9 species: vagrant shrew, montane shrew, marsh shrew, Trowbridge's shrew, shrew-mole, coast mole, creeping vole, forest deer mouse, and ermine. We found strong trends for 2 others, the southern red-backed vole and the deer mouse. Statistically significant differences among years were found for 6 species: marsh shrew, shrew-mole, creeping vole, long-tailed vole, Townsend's vole, and ermine. The most common pattern was for a given species to favor either early or mature classes. Only the forest deer mouse favored the closed-canopy class. Differences among years were more pronounced in the rodents, which reached highest abundance in 1994 and showed a disproportionate preference for the clearcut areas. Captures of 5 other species were too low for statistical analyses (water shrew, masked shrew, Townsend's mole, Townsend's chipmunk, and northern flying squirrel), but low abundance was attributable to causes other than forest management. Species richness for all small mammals was consistent across the structural classes in each year, despite a considerable increase in 1994. The mean number of species per site was about the same between clearcut sites and those in harvest-age forest, however, the species composition of these most dissimilar habitats was very different. Predictably, species richness was highest in the pre-canopy class. This is most likely due to a simple mixing of species from either end of the habitat gradient. Species richness was lowest in the closed-canopy class in each year. The high population abundance of rodents in 1994 drove the richness pattern between years. Of the 15 variables constituting the ground cover data set, 13 showed statistically significant correlations with small mammal species abundance.

Landscape-level effects on individual species were limited to positive associations between the percent area and patch sizes of favored habitat within landscapes. Correlations at the local landscape scale were generally weak at both the species and community levels and offered little help in understanding habitat occupancy patterns. The primary determinants of small mammal assemblages in these forests were at the stand level. Most small mammal community change in these forests takes place early in forest succession. At canopy closure, species turnover affects more than half of the species in the system. Although landscape-level planning probably will be needed for species such as larger mammals and birds, it is less a concern for small mammals. Most focus can be on stand-level characteristics in forests where habitat blocks are in excess of 30 or 40 ha.

The pitfall capture records of this study have assembled about the same species list as that resulting from the U.S. Forest Service's old-growth forest studies. All species found in unmanaged forests in the old-growth study are present in these managed forests. The closed-canopy stage does not support a unique mammalian community, but rather is characterized by the loss of several species that are unable to survive the unproductive ground-level environment caused by a closed and very dense canopy. This structural class should be moved toward the structure of a more mature class to enhance the productivity of the small mammal community. This might best be accomplished with a program of vigorous thinning. Management must focus on elements that contribute to compositional and structural diversity within stands. Even in these intensively managed forests there are old snags, large stumps from initial harvests, and very old downed woody material that plays an important role in enhancing the habitat quality of these younger stands. The replenishment of these elements must be planned.

On the east side, we sampled small mammal populations using pitfall trapping grids (6 × 6 arrays and 10-m spacing). Grids were placed in areas considered representative of the stands and were buffered from riparian areas. Trapping was conducted for 2 weeks at each site during late May, June, or early July of each year. Small mammal communities are relatively diverse in northeastern Washington. Of 18 species observed (3,739 captures), we examined the habitat and landscape relationships of 11 species for which there were sufficient data. The total number of captures varied substantially between years, increasing from 743 in 1993 to 2,248 in 1994 and then declining to 718 in 1995. Total number of captures was similar in the 3 forest types in 1993 and 1995, but was greater in clearcuts in 1994. Red-backed voles, deer mice, long-tailed voles, montane voles, masked shrews, montane shrews, and vagrant shrews were all captured more frequently in 1994 than in other years.

Statistically significant differences in use of the 3 forest types were found for 9 species. Red-backed voles and masked shrews were more abundant in closed-canopy forest than in clearcut or regenerating stands. Deer mice, pocket gophers, jumping mice, and long-tailed voles were most abundant in clearcuts. No species was most abundant in regenerating stands, but vagrant shrews, heather voles, and montane voles were largely absent from closed-canopy forest and were equally abundant in clearcuts and regenerating stands. Pygmy shrews and montane shrews were less abundant than the other shrew species and showed no association with any forest type. There were minimal effects of stand size on distribution. Over all years, species richness was highest in clearcuts (7 species) followed by regenerating (5.1) and closed-canopy forests (4.5), and did not differ with stand size. The general pattern of species richness held for all years, despite the addition of some species in 1994.

For each of the 11 species, we examined their habitat and landscape relationships using stepwise regression analysis. The best predictors of small mammal abundance and distribution were variables describing specific components of vegetative structure (e.g., woody debris, shrub area; most R^2-values between 50-80%). Variables describing the landscape surrounding trapping grids provided much less predictive power. Where landscape variables were important, they were associated with the forest type predominantly used by a species. For example, red-backed voles were positively associated with habitat features that are reflective of closed-canopy forest such as canopy cover, medium to large d.b.h. trees, low shrubs, and large, well-decayed woody debris. Of the landscape variables, red-back voles were positively associated with the total area of closed-canopy forest and negatively associated with increasing perimeter of closed-canopy stands (R^2 = 13-24%). A resident of clearcuts, the deer mouse, was associated positively with recent woody debris and negatively associated with medium-high shrubs and regenerating conifers. In the landscape regression, this species was positively related to the total area of clearcut within a 1 km-radius circle of the grid. Captures of the pygmy shrew, a species that showed no association with any of the 3 forest types, were associated positively with woody debris and area of medium-tall shrubs.

The composition and structure of small-mammal communities in the 3 forest types is similar to those of comparable forest stands in the region. Forest type is more important that stand size in explaining differences in small-mammal distribution. The level of forest fragmentation has probably not reached a level that prevents dispersal and recolonization of forest stands. The increases in populations of all species in 1994 resulted in wider distribution of many species among the 3 forest types. Such pulses in population numbers probably result in substantial mortality for many individual animals, but increase the probability that populations that might go locally extinct are restarted. For a closed-canopy forest specialist, the red-backed vole, we found evidence that strips of forest habitat become occupied when populations increase and may act as corridors. For shrew communities, we found no evidence that the distributions of individual species were the result of competitive interactions.

BATS

On the west side, we conducted stand-level surveys of bat activity using echolocation detectors. Each site was monitored for bat activity on 6 nights throughout each field season. We monitored over 2500 hrs and recorded a total of 967 echolocation calls. The number of detections was not significantly different between years, but was different among seral stages. Overall, activity levels were low with 46% of the nights having no detections. The highest detection rates were recorded in clearcut stands while closed-canopy stands had no detections. The high detection rate, presence of feeding buzzes, and low abundance of typical roosting structures (e.g. large snags and trees) within clearcut stands suggests these sites were used for foraging. The absence of activity in closed-canopy sites suggests that these stands were unsuitable habitat for forest-dwelling bats. Although a high density of snags were present in the closed canopy sites, the snags were typically small-diameter Douglas-fir whose lack of crevices and hollows make them unlikely roost sites. Harvest-age sites had the 2nd highest detection rates and were the only seral stage to have a binomial activity pattern with an initial peak of activity early in the evening and a secondary peak during the early morning hours, a pattern indicative of roosting. Large trees (50-100 cm d.b.h) and snags (>50 cm d.b.h. and > 15 m tall), which are "roost-type" trees for certain species in Washington state, were most abundant in these stands. Suitable roosting conditions may not be present for all forest-dwelling species, however, as suggested by the absence of calls for the big brown bat and silver-haired bat within the harvest-age stands. The landscape composition surrounding each site was also found to affect bat activity. It appears that bats may use landscapes more heavily if they are heterogeneous rather than composed of 1 patch type. Two distinct clusters of stands were produced based on the amount of edge produced by the juxtaposing clearcut and forested areas within the surrounding landscape. Those containing greater amounts of edge had high detection rates, while those consisting entirely of continuous forest had no activity. The suitability of a landscape for bats may therefore depend on the frequency and spatial distribution of different patch types, because bats require different habitats for foraging and roosting.

On the east side, we examined the roosting habits of the silver-haired bat, which preferentially uses older forests in other parts of its range in the Pacific Northwest. Bats were captured using mist nets and harp traps at stream, pond, and road sites in 1992 and 1993. Fifteen of 30 bats captured were equipped with radio-transmitters. Altogether, 15 roost sites used by 13 individuals were found. All roosts were <3.5 km from the initial capture site and were in dead or dying trees with exfoliating bark, extensive vertical cracks, or cavities. Fourteen roosts were in trees with diameter at breast height >30 cm. In many cases, the species of roost tree was a legacy of earlier forest species composition. Roost trees were significantly taller than neighboring trees, which may facilitate the location of roosts by bats. Analysis of a 15-m radius plot around each roost tree revealed that roost sites had significantly less overstory canopy closure, less understory, and shorter understory vegetation than comparable random plots. Reduced overstory canopy and understory vegetation provide a less cluttered environment for bat flight, which may be particularly important for newly volant young. Recruitment and retention of large snags (>30 cm d.b.h.) and the maintenance of structural complexity in forest patches in upland, as well as riparian areas, are important for the conservation of bat species in managed forests.

BIRDS

On the west side, during the springs of 1993-1995 we detected over 20,000 birds of 73 species within 50 m of point count stations. Bird species composition changed with forest development. Species such as the willow flycatcher, common yellowthroat, MacGillivray's warbler, white-crowned sparrow, dark-eyed junco, song sparrow, rufous-sided towhee and American goldfinch were characteristic of clearcut areas, whereas the chestnut-backed chickadee, golden-crowned kinglet, and winter wren were most abundant in forests. Species richness was lowest in closed-canopy and mature forest (37 species), highest in pre-canopy stands (57 species) and intermediate in clearcuts (47 species). The following species, listed in descending order of abundance, comprised over 75% of all bird detections: winter wren, Pacific-slope flycatcher, golden-crowned kinglet, chestnut-backed chickadee, Wilsons warbler, hermit-Townsend's warbler, Swainson's thrush, and Hutton's vireo. Bird species richness did not vary significantly among years. Total avian abundance was significantly different among age classes in 1994 and 1995 but not in 1993. Lowest abundance was consistently found in the closed-canopy class in all years. We tested the relationship between abundance and each habitat variable for each of 20 bird species for which we had an adequate sample size. Specific habitat variables that were positively correlated with most bird species abundance included: (1) counts of dominant Douglas-fir or true fir; (2) counts of sub-dominant conifers; (3) counts of sub-dominant coniferous trees 10-50 cm dbh and >3 m tall; and (4) counts of dominant or sub-dominant deciduous trees. Landscape effects on birds were examined at the individual species and community levels. Nine species had positive and 8 species had negative relationships with the amount of area clearcut harvested near the study sites. Virtually all species with a positive response are known to be associated with open, shrubby habitats. On the other hand, species with negative responses, such as the winter wren, golden-crowned kinglet, and chestnut-backed chickadee, are associated with canopied forests, so their responses also were predictable. Nine species were positively correlated with total area of mature forest. Those with positive responses included cavity-nesting species and species associated with multi-layered forest canopies. Five species had a negative association with area of mature forest. These included species that are most common in edge habitat.

At the community level, no significant relationships were found between bird species richness and total abundance and any of the landscape variables. The primary determinants of bird species richness and abundance appear to be at the stand level. The majority of birds associated with the managed forest landscape are small songbirds that have relatively small territories and home ranges. Less than 50% of the bird species showed any relationship with landscape variables. Birds with large territories and home ranges were simply not well represented in this data set. Forest management should be directed at maintaining high species richness for native species, increasing the number of cavity-nesting bird species, and increasing the number of raptors using managed forests. Potential silvicultural options include maximizing the number of large-diameter coniferous trees through green-tree retention, providing a constant supply of medium- and large-diameter dying trees and snags, allowing the growth of deciduous trees in clearcut areas, and allowing the development of a shrubby and herbaceous understory in forest stands.

On the east side, we used a circular point-count method for surveying bird populations which involved estimating the distance to birds detected within a 50-m radius. We established 3-4 and 12 point count stations on the small and large patches, respectively. The area covered by these stations is about 40% of the total stand area. We buffered the point count stations from the stand edge and from riparian areas or gaps in the stand. We completed 4 entries of each stand in 1993 and 6 entries per stand in 1994 and 1995. There were >20,000 detections of 110 species during the 3 years for all sites combined. Species richness was higher in clearcut (16.7 species/stand) and regenerating forests (16) than in closed-canopy forests (13.7). Species richness was greater on large stands (19 species/stand) than on small stands (11.9). Species richness was lower in 1993 (14.4 species/stand) than in 1994 (15.8) or 1995 (16.2), but this partly reflects lower sampling effort in 1993. The number of birds detected per point was higher in regenerating (4.0 counts/station) than in either closed-canopy (3.2) or clearcut stands (3.1). Small stands had more birds per point (3.7 counts/station) than did large stands (3.2). Species diversity and evenness were similar across all forest types. Species diversity and evenness were both greater in the larger stands.

Stand-level associations were examined for the 42 most common species. Fifteen species were detected more frequently in the closed-canopy forests than in either of the other 2 forest types. These species were pileated woodpecker, Cassin's vireo, winter wren, chestnut-backed chickadee, brown creeper, golden-crowned kinglet, Swainson's thrush, hermit thrush, varied thrush, evening grosbeak, Townsend's warbler, red crossbill, gray jay, and red-breasted nuthatch. More species were restricted to closed-canopy forest than to the other forest types. Eight species were detected more frequently in the regenerating forests and only 5 species were detected more frequently in clearcut areas. Several species were more common in 2 habitat types: 1 in the mature and regenerating forests; 5 in the clearcut and regenerating forests, and 2 in the closed-canopy and clearcut forests. Three species were detected equally across all 3 forest types. Few species differed in abundance between the 2 stand sizes. Four species were detected more commonly in larger stands and 9 in smaller stands.

Of 42 common species, regressions of the habitat variables on the detections for each species yielded significant models for 15 species. The predictive power of these models varied from 10-95%, but was generally low. Most species associated with closed-canopy forest were associated positively with canopy closure and number of large diameter trees, variables representative of this forest type. Additionally, Swainson's thrush was associated positively with a dense understory of shrubs and regenerating trees, and red-breasted nuthatch was associated positively with medium to large snags. Only 1 clearcut associate could be examined: the hairy woodpecker was related positively to number of snags and to open canopy cover. Warbling vireo and dusky flycatcher were most abundant in regenerating forests and were associated with number of regenerating trees, number of stumps, and shrub cover. Five other species that occurred primarily in open-canopy stands were associated negatively with either canopy cover or number of coniferous trees ($R^2 < 50\%$).

In contrast to these results, the predictive power of the regression models incorporating the landscape-level associations was consistently higher. As one would expect, closed-canopy species were all associated positively with the total area of closed-canopy forest within a 1-km radius of the transect. However, several of these species also were associated negatively with increasing perimeter of closed-canopy forest. This indicates that larger tracts of closed-canopy forest are important to these species. Abundance of the Hairy Woodpecker, a species of clearcuts, was related positively to area of clearcut, but also to perimeter of closed-canopy forest. This suggests that the juxtaposition of these habitats is significant. For 3 species, local habitat factors provided very little information on habitat relationships compared to regressions with the landscape variables. For example, western tanagers were negatively associated with perimeter of regenerating forest surrounding the transect. Another species, the yellow-rumped warbler, was not clearly associated with any forest type, but was associated positively with the area of closed-canopy forest and negatively with number and area of regenerating, clearcut, or disturbed patches.

GENERAL FINDINGS AND MANAGEMENT RECOMMENDATIONS

The managed forest landscape of western Washington is diverse and reflects patterns of ownership. The plantation environment that was studied on the west side is representative of management with a strict blocked rotation. Consequently the sizes and ages of stands are fairly uniform. In contrast, second-growth forests in northeastern Washington have been fragmented moderately by forest harvest. Over much of this region, stands of younger forest are usually embedded within a matrix of closed-canopy forest and there is a broader range of stand sizes. The potential vertebrate communities differ for biogeographic reasons between western and eastern Washington. Despite these differences, many of our results were very similar. Perhaps the most obvious finding is that our study areas within intensively managed forests on both sides of the Cascade crest still retain a diverse vertebrate fauna. For the west side, the species lists generated in this study and the lists from studies in unmanaged forests using similar censusing techniques were essentially the same. Although we do not have a similar benchmark from older unmanaged forests on the east side, most of the species one would anticipate to occur in these forests were observed.

Differences for most taxa were in the apportionment of individuals among species rather than differences in species composition. This means that the long-term retention of native species in intensively managed forests is a much more attainable goal than would be the case if species were missing. The actual long-term viability of many species is unknown, however, because we do not have information on reproductive success or survival. This problem is compounded for some species because of our inability to detect them or because of fluctuations in their numbers. We believe that the retention of most species should be attainable and that forest management can be used to provide critical habitat for species that have specialized requirements. The concern can shift from species introductions and faunal reconstruction to providing critical habitat elements for taxa in managed forests that seem underrepresented in the censuses— species that apparently are not doing as well as we would like to see in reference to unmanaged stands. Taxa worth looking at more closely in this regard for amphibians might include the tailed frog and Columbia spotted frog and for birds several cavity-nesting species, including the red-breasted nuthatch, brown creeper, hairy woodpecker, pileated woodpecker, red-breasted sapsucker, and Vaux's swift. No small mammal species appear to fall into this category at present. We can make fewer recommendations for bats because we require a better understanding of the structures that they require for roosting. Our studies of roost use by the silver-haired and long-legged Myotis bats, for example, points to the need to recruit and retain large snags on the landscape.

Responses to habitat fragmentation and to landscape structure vary with taxonomic group. The strength of relationships between wildlife occurrence and abundance at the stand and landscape scales was related to the details of their natural history. In general, species with limited vagility such as amphibians and small mammals showed strong use of particular seral stages and strong correlations with specific habitat elements within stands. For these species, correlations with landscape metrics were generally weak. Studies on the east side suggest that maintaining landscape elements such as corridors may be useful for retaining species such as the southern red-backed vole, which are restricted to closed-canopy forest. This species also was most abundant in large stands of closed-canopy forest. We believe that pulses in the abundance of many small mammal species, as observed in 1994, are important in driving habitat occupancy patterns. They allow both the recolonization of forest stands by forest-associated species and the colonization of clearcut areas by species favoring early successional habitat. For highly mobile species such as the bats and birds, landscape-scale variables were more useful. Bats, in particular, require attention at landscape scales because they commonly use different habitats for roosting and feeding.

Managed forest landscapes on the west side consist of 2 faunal assemblages: one characteristic of pre-canopy conditions and another of well-developed forest. At canopy closure, many common species are surpassed numerically by species of continuous forest. This changeover is particularly noticeable for small mammals and birds. Before canopy closure, bats of several genera use the airspace over clearcut areas and their associated edges. After canopy closure, the forest belongs almost exclusively to the genus *Myotis*.

In terms of vertebrate community metrics, species richness was highest on the west side in the pre-canopy class for all taxa except bats. We interpret this pattern as the co-occurrence of taxa adapted to either pre- or post-canopy conditions. Species richness could not be calculated for the bats, but the clearcut areas showed the highest richness for genera. While clearcut and harvest-age classes had roughly equal species richness for many taxa (though of different species), the closed-canopy class had the lowest species richness for all taxa. This class typically had low abundance levels for all taxa as well. We recommend that this class receive special management attention. A program of vigorous thinning would increase ground- and shrub-level structural diversity and productivity and open flight space for birds and bats. Variation in species richness was pronounced for small mammals, a pattern driven by high rodent abundance in 1994, but not significant for birds. Abundance of small mammals was high in 1994 and also significantly different in 2 of the 3 years for birds. On the east side, the patterns of species richness also point to differences in the structural condition of the managed forest. Small mammals reach their highest diversity and abundance in clearcut areas. This corresponds to the greater woody debris and understory structure of these stands. Avian species richness is highest in regenerating stands. Clearcut areas have not retained a sufficient number of trees for nesting birds and the closed-canopy stands have a reduced understory or shrub component. Additionally, the composition of regenerating stands draws from species that also utilize the other forest types. As an adaptive management strategy, some thinning of closed-canopy stands might increase their suitability by increasing shrub level diversity. On the east side, the retention of large (>34 ha) closed-canopy stands will help to maintain some avian species.

We are concerned that structural diversity be maintained in managed forests. On the west side, present levels of downed wood appear sufficient to allow persistence of most amphibians and small mammals. These stands still retain stumps and humus material from the original forest that are important elements for the forest-floor fauna.

Because these structures have not been replaced over the past several decades, they are being lost to the system. On the east side, the degree of downed wood is sufficient in clearcuts, but less available in closed-canopy forests. Thinning of closed-canopy stands could provide additional woody debris. Similarly, the scarcity of large-diameter snags is a concern throughout intensively managed forests. Reduced species richness and abundance is already observable for several cavity-nesting birds on the west side. Although bats have high mobility, the preference of some species for large snags and trees with cracks suggests that current silvicultural practices may result in the future loss of species as these relictual elements disappear. The retention of leave trees in harvest units is an important first step, but does not satisfy the need for large-diameter trees. Continuous replenishment of these structural elements is essential if managed forests are to support the full complement of native species.

EXECUTIVE SUMMARY
WEST-SIDE STUDIES

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PURPOSE AND APPROACH

During this century, a substantial area of the State of Washington has been committed to timber production. Until very recently the few well-replicated descriptions of habitat use by wildlife have been conducted in unmanaged forests. To assist managers in their task of meeting the wildlife habitat objectives of the Timber, Fish and Wildlife Agreement, we designed a survey to describe the patterns of wildlife response to intensively managed forest landscapes. Our charge in this project was twofold:

- to describe the species composition and abundance levels of wildlife and plant communities occurring in forest stands of varying structural stages and landscape configurations in watersheds managed primarily for timber production, and
- to develop methods for analyzing wildlife responses to landscape-scale habitat conditions in managed watersheds.

Our approach was to sample terrestrial vertebrate groups in forest stands occurring over a gradient of landscape conditions. This approach allowed statistical analysis of wildlife habitat relationships primarily at the stand, but also at the landscape scale. To accomplish this, we needed to stratify landscapes by landscape history. This was done by using satellite imagery and developing a GIS methodology that enabled us to classify landscapes by age class, subdivide the landscape by watershed, calculate the pattern or character of the landscape, and examine areas for their management intensity and potential as study areas. We located 20 study areas in southern Pierce County on the Buckley and Kapowsin tree farms owned by Champion International and another 4 study areas on the Vail tree farm owned by Weyerhaeuser. The study areas varied from 30 to 80 ha, were dominated by Douglas-fir, and located away from riparian areas on sites less than 900 m in elevation. The study areas consisted of 6 replicates of 4 structural types: a clearcut stage resulting from harvest within 2-3 yrs with adjacent stands at least 5 yrs older, a pre-canopy stage about 12-20 yrs after harvest that had been pre-commercially thinned with adjacent stands recently clearcut or with fully developed canopy, a closed-canopy stage about 30-40 yrs after harvest that had not been commercially thinned with adjacent stands in any stage, and a harvest stage about 50-70 yrs after harvest that had been commercially thinned with adjacent stands in any stage.

We conducted an extensive series of spatially nested vegetation surveys on each study area to acquire information on stand composition and structure. These data were used extensively in our search for consistent patterns of association between habitat structure and wildlife species composition and relative abundance. Landscape variables were incorporated into the study using metrics calculated for actual watershed boundaries as well as metrics calculated from 100-ha, 1,000-ha, 5,000-ha, 10,000-ha, and 25,000-ha concentric circles centered on the study areas. We used the FRAGSTATS program to calculate the landscape metrics. Reflective of their different natural histories, we used various metrics and scales when analyzing landscape effects for different wildlife taxa.

Species and species groups chosen for study were those that could be sampled well at stand-level scales. We endeavored to include as many species as possible within this constraint as befitting one of the first attempts to characterize wildlife use of managed forests in the Pacific Northwest. All taxa were sampled for 3 consecutive years (fall 1992 through spring 1995). We sampled forest-floor amphibians and small mammals, diurnal breeding birds, and in an exception to the constraint of stand-level scale, we also included bats. This was done to add some information to a poorly understood group. Additionally, we conducted 3 studies on species or species groups that were expected to show a response to landscape pattern. We studied the roost site characteristics of the long-legged myotis bat, the movement patterns of the northwestern salamander, and the movement patterns of bobcats. Results of these supplemental studies are summarized below and provided in their entirety in the Appendices for Volume 2.

SUPPLEMENTAL STUDIES

We conducted the bat study in the Teanaway River drainage near Cle Elum to obtain a reliable source of bats. We sought to compile a species list for the area, to compare echolocation calls from the Teanaway with western Washington echolocation call libraries, and to describe the roost site characteristics of the long-legged myotis. Bats were trapped using harp traps in July and August 1995. Echolocation calls were recorded from individuals of known species and 17 long-legged myotis bats were fitted with radiotransmitters. These individuals were tracked to successive roost sites, which were described by variables at the roost and by characteristics of the stands in which they occurred. Of the 124 individuals captured, 8 species were represented and 6 of these were in the genus *Myotis*. No myotis species could be reliably separated from other myotis species by echolocation call structure, and all myotis species had to be pooled for analysis. Altogether 28 roosts were found, some of which were used repeatedly for a total of 63 roost nights. Of the 28 roosts, 25 were in snags and 3 were in live trees. No roosts were found in rock crevices or buildings. Roost trees were larger in diameter and taller than nearby trees. There was no selection for tree species and canopy cover, and the roost was not different from stand averages. Management should plan for a continuous source of large snags for roosts.

The fall amphibian surveys indicated that the distribution of the northwestern salamander is not related strongly to stand age. To interpret these results, we investigated northwestern salamander use of forest stands at the Kapowsin tree farm and at Pack Demonstration Forest near Eatonville by trapping and individually marking migrating and dispersing salamander populations at breeding ponds, and by following individuals using radio-telemetry. Salamander movements were strongly correlated with mild wet weather. Salamanders tended to use forested habitat much more than open habitat when migrating to and dispersing from breeding ponds. These results suggest that the salamanders are sensitive to microclimate and show high use in areas with wet and mild conditions such as those found in forested habitat. Northwestern salamanders are not excluded by open habitat, however, as shown by their presence in clearcut areas and young forest stands.

Habitat utilization and home range size of bobcats were investigated at the Kapowsin tree farm from May 1993 through September 1994 with the use of radio-telemetry and Geographic Information Systems (GIS). A total of 751 independent locations were obtained for the 6 male and 4 female bobcats captured. The 95% Adaptive Kernel method indicated a mean annual home range size of 14.4 km^2 for males and 8.4 km^2 for females. Home range size and movement indices differed between sexes with males moving further between successive locations and having larger home range sizes. Home range size and movement indices did not differ between seasons; winter vs. summer and breeding vs. non-breeding. A density estimate of 0.22 cats/km^2 was calculated based on mean home range size and a mean home range overlap of 27%. Home range and density results from this study were consistent with those from a 1978 study conducted on the same study area and indicate similar population dynamics. Habitat analysis based on availability within the study area showed that differences in use versus availability were not the same for all habitat types. Young stands were the most consistently preferred. Clearcut areas (2-5 yrs old) and harvest-age stands were mostly avoided. Preferred habitat appeared to be related to abundant understory vegetative cover and prey availability. Thick understory vegetation provides bobcats with cover for stalking their prey as well as security when ledges or rock outcroppings are not available. Logging has returned forests to early successional stages that provide good cover and abundant prey for bobcats. Bobcats appear to be adaptable enough to survive in the marginal habitats created by timber harvest, while taking advantage of the preferred habitat types with high prey abundance. They appear to benefit from the patchwork mosaic of seral stages found in managed forest of western Washington.

TAXA OF THE GENERAL SURVEY

The following comments are specific to the taxonomic groups selected for study in the basic survey.

FOREST-FLOOR AMPHIBIANS

We conducted pitfall trapping in all stands in 6 × 6 pitfall arrays with 15-m spacing for 28 days after the onset of fall rains during 1992-1994 and captured 842 individuals of 11 species, including 6 salamanders and 5 frogs and toads. Seven pond-breeding species, 2 headwater stream-breeding species, and 2 woodland species were caught. The northwestern salamander, western redback salamander, and ensatina dominated the terrestrial amphibian communities in all 4 structural classes. Only 2 additional species, the roughskin newt and red-legged frog, had sufficient captures for statistical analyses.

All species previously found in naturally regenerated, unmanaged forests in the south-central Cascade Range in Washington also occurred in these intensively managed plantations, even though old-growth forests have been virtually absent from the study area for several decades. At present, differences between amphibian communities in these 2 forest conditions lie in their structure, not their species composition. However, differences in community structure suggest that intensively managed forest landscapes may have a lower overall amphibian habitat diversity than landscapes dominated by unmanaged forests. Consequently, the probability of this amphibian community persisting intact over the long term may be lower than in unmanaged landscapes. In addition, detrimental cumulative effects of intensive forest management on amphibian populations may only become apparent over very long time frames or additional rotations. Among structure classes, species richness was highest in pre-canopy stands and lowest in closed-canopy stands. As has been reported elsewhere, there was no evidence that species richness increased with stand age. Physiographic influences play an important role in determining amphibian diversity within forested landscapes. Four of the 5 species analyzed showed a negative association with elevation, and neither ensatinas nor western redback salamanders were captured in stands occurring above 610 m in elevation. This probably reflects the lowered ability of most amphibians to survive and/or flourish as climatic conditions become progressively colder.

The similarity of amphibian community composition within WRIAs indicates that topographical, zoogeographical, or ecological influences operating at the scale of river basins (60,000-81,000 ha) are more important in the structuring of amphibian communities than those occurring at the smaller scale of WAUs (sub-basins 4,000-12,000 ha in size). These findings may have implications for conducting amphibian surveys and for managing habitat at the landscape scale. Because communities within WRIAs are likely to be more similar in community composition than those occurring in different WRIAs, surveys will be most informative if field efforts are spread across as many WRIAs as possible. In addition, although prescriptions for landscape-scale management of habitat to maintain or enhance amphibian diversity and abundance do not yet exist, my results suggest that they should be developed and evaluated within relatively large watersheds, such as major river basins, not within smaller order watersheds. Amphibian community structure in harvest-age stands was unique. Harvest-age stands had the highest overall amphibian abundance and relatively high abundance values for many species. The harvest-age structure class was also high in expected species richness, and was the only class in which all 11 species were found. In addition, unlike other structure classes sampled, amphibian community structure in harvest-age stands closely resembled that found in unmanaged forests.

There were few indications that clearcut areas represent substantially lower quality habitat for terrestrial amphibians than other structure classes. Clearcut areas contained the highest abundances of western redback salamanders and roughskin newts, and had a higher expected species richness than closed-canopy forests. However, clearcut areas may represent relatively hostile environments for most stream-breeding amphibians, especially tailed frogs. Only a single tailed frog was captured in a clearcut area and this structure class was the only one in which Pacific giant salamanders were not captured.

No previous studies comparing amphibian communities in thinned vs. unthinned stands have been conducted in the Pacific Northwest. In this study, all pre-canopy stands were pre-commercially thinned and all harvest-age stands had been commercially thinned. The two thinned stand conditions had the highest species richness, and harvest-age stands had almost twice the total captures of any other structure class. In contrast, unthinned closed-canopy stands had the lowest species richness, and was the only structure class in which roughskin newts and red-legged frogs were not captured. Thus, both pre-commercial and commercial thinning appear to improve overall habitat conditions for terrestrial amphibians in forest plantations.

There was little evidence that varying levels of coarse woody debris had a significant influence on amphibian abundances. Physiographic conditions and stand structure were more important influences on amphibian communities in managed forests than amounts of dead woody material. There is strong evidence, however, that small (10-30 cm diameter) moderately decayed logs are important surface cover objects for woodland salamanders in both managed and unmanaged forests in western Washington. Although levels of woody debris do not currently appear to limit populations of terrestrial amphibians in these landscapes, managed stands in this region still contain many large residual stumps and logs in advanced stages of decay from the original, unmanaged stand. Eventually, these features will disappear from the landscape. Because they are still an integral part of the humus and soil layers in these stands, their presence may help to explain why few associations were found between terrestrial amphibians and coarse woody debris. Providing for the replacement of large logs in future stands may be important for the persistence of amphibians occurring at low population levels in managed landscapes.

Amphibian diversity and abundance in intensively managed forests was more strongly influenced by stand structure and physiographic conditions than by landscape characteristics. Because harvest-age stands had much higher amphibian abundances and species richness than either closed-canopy forests or clearcut areas, silvicultural strategies that maintain a higher percentage of harvest-age forest conditions within managed landscapes might provide substantial benefits to terrestrial amphibians. One could thin closed-canopy stands to reach vegetative and structural conditions found in harvest-age stands more quickly. Alternatively, one could extend rotation ages so that harvest-age conditions would constitute a higher proportion of the landscape. Extended rotations would increase the amount of high-quality habitat in managed landscapes, reduce ground-disturbing activities, and reduce the frequency of entries and of slash burning. In addition, extended rotations would improve habitat connectivity among mature forest stands, which may reduce the susceptibility of some amphibians to local extirpations.

FOREST-FLOOR SMALL MAMMALS

We sampled small mammals concurrently with terrestrial amphibians each fall during 1992-1994. Pitfall traps capture insectivores and non-jumping rodents well, but are less effective at capturing deer mice, chipmunks, and jumping mice. We chose this method because we wanted to emphasize those species that were not sampled well in previous studies. Many of the more common species, while not sampled effectively with pitfall traps are generally caught in sufficient numbers for statistical analyses. We caught 3,720 individuals of 18 species. Reflecting the capture method, most of these were insectivores (2,564). The remaining captures consisted of rodents (1,134) and ermine (22). Statistically significant differences among the 4 structural classes were found for 9 species: vagrant shrew montane shrew, marsh shrew, Trowbridge's shrew, shrew-mole, coast mole, creeping vole, forest deer mouse, and ermine. We found strong trends for 2 others, the southern red-backed vole and the deer mouse. Statistically significant differences among years were found for 6 species: marsh shrew, shrew-mole, creeping vole, long-tailed vole, Townsend's vole, and ermine. The most common pattern was for a given species to favor either early or mature classes. Only the forest deer mouse favored the closed- canopy class. Differences among years were more pronounced in the rodents, which reached highest abundance in 1994 and showed a disproportionate preference for the clearcut class.

Four of the 6 shrews were captured in sufficient numbers for statistical treatment. The vagrant shrew and the montane shrew favored early pre-canopy/edge environments and Trowbridge's shrew and the marsh shrew favored fully developed forest. Abundance of the marsh shrew was not well correlated with the presence of permanent water. It is possible that these forests are sufficiently wet to allow the marsh shrew to survive in a range of structural classes, although it does not do well in clearcut areas. The coast mole appears to be primarily an edge species with few captures in clearcut areas and harvest-age sites. The shrew-mole was caught in greater abundance and showed a clear preference for mature forest over all other classes. The shrew-mole has been identified as a species associated with old-growth forest. While the statistical association may be real, its abundance in the mature forest classes of this study argues against a strong biological dependency. It is clear, however, that this species does best in fully developed forest.

The deer mouse and the forest deer mouse show reciprocal patterns of habitat occupancy with the deer mouse most abundant in early stages of forest succession and the forest deer mouse in later stages. The forest deer mouse was the only small mammal to show its highest abundance in the closed-canopy stage. Its ability to use these forests, with their sparse understories, probably relates to their tree-climbing habit, which allows them access to resources in the tree canopy. Southern red-backed voles were found more often in mature forest than any other structural class. The distribution of this species was very patchy across all structural classes. Like the shrew-mole, this species has been thought to be associated with old-growth forests, and while its association with well-developed forests seems fairly certain, its association with old-growth forest remains unclear. The Townsend's vole and the long-tailed vole were not caught in very large numbers, but still showed significant differences between years, a feature showed by all 3 *Microtus* species. Townsend's vole was caught most frequently in the first 2 age classes. The long-tailed vole was essentially absent from the unthinned forest that supported a very poor understory plant community. Its distribution remains somewhat enigmatic, although the TFW Riparian project is finding most captures of this species within riparian zones. The highest and most consistent captures of the long-tailed vole in this study were from a site associated with a permanent wetland. The creeping vole showed a strong pattern favoring clearcut sites in all years. We caught 22 ermine in a pattern that was distinctive and consistent enough to be statistically significant both for class and year. Ermine are specialized predators of field voles and showed highest abundance in the first 2 structural classes, just as their prey. The difference between years was due to low abundance in 1993. Captures of 5 other species were too low for statistical analyses (water shrew, masked shrew, Townsend's mole, Townsend's chipmunk, and northern flying squirrel), but low abundance was attributable to causes other than forest management.

Species richness for all small mammals was consistent across the structural classes in each year, despite a considerable increase in captures in 1994. The mean number of species per site was about the same between clearcut sites and those in harvest-age forest, however, the species composition of these most dissimilar habitats was very different. Predictably, species richness was highest in the pre-canopy class. This is most likely due to a simple mixing of species from either end of the habitat gradient. Species richness was lowest in the closed-canopy class in each year. The high population abundances of rodents in 1994 drove the richness pattern between years.

Elevation of the study sites was not an important variable determining species richness, either for rodents or insectivores. We focused on stand mean values for ground cover variables recorded on the 3 × 3-m vegetation sampling plots. Of the 15 variables constituting this data set, 13 variables showed statistically significant correlations with small mammal species abundance. Multiple correlation coefficients (R^2) were rather high for the rodents, ranging from 0.54 to 0.92, but lower for the insectivores, from 0.22 to 0.70. Partial correlations for individual vegetation variables were low for both groups.

Several landscape metrics calculated for a 100-ha circular area were used to investigate correlations between landscape descriptors and small mammal species and community-level variables. The radius of this area is just over 564 m, a good distance for investigating the effects of nearby, contrasting habitat types on the structure of small mammal communities. Landscape-level effects on individual species were limited to positive associations between the percent area and patch sizes of favored habitat within landscapes. Correlations were generally weak at both the species and community levels and offered little help in understanding habitat occupancy patterns. The primary determinants of small mammal assemblages in these forests were at the stand level.

Most small mammal community change in these forests takes place early in forest succession. At canopy closure, species turnover affects more than half of the species in the system. The pre-canopy period is characterized by high species richness and by high rodent abundance. The abundance of insectivores tends to be higher in the period after canopy closure, although a large measure of this is due to the abundance pattern of Trowbridge's shrew, which tends to overwhelm the other species numerically. The pre-canopy period is also characterized by large differences in abundance between years, mostly but not entirely driven by the differences in microtine rodent abundance.

With the clear patterns of species associations among structural classes, creating habitat for desired small mammal communities seems possible. While landscape-level planning probably will be needed for species such as larger mammals and birds, it is less a concern for small mammals. Most focus can be on stand-level characteristics in forests where habitat blocks are in excess of 30 or 40 ha. Given a mix of stand ages, this area seems sufficient to allow breeding populations and species persistence. The pitfall capture records of this study have assembled about the same species list as that resulting from the U.S. Forest Service's old-growth forest studies. All species found in the old-growth forest study are present in these managed forests. Small mammal species not well sampled by pitfall traps, of course, will need information gathered for them using other techniques. The closed-canopy stage does not support a unique mammalian community, but rather is characterized by the loss of several species that are unable to survive the unproductive ground-level environment caused by a closed and very dense canopy. This structural class should be moved toward the structure of a more mature class to enhance the productivity of the small mammal community. This might best be accomplished with a program of vigorous thinning. Management must focus on elements that contribute to compositional and structural diversity within stands. Even in these intensively managed forests there are old snags, large stumps from initial harvests, and very old downed woody material that plays an important role in enhancing the habitat quality of these younger stands. The replenishment of these elements must be planned. Their maintenance will enhance not only the closed-canopy structural classes, but their retention during pre-canopy classes will enrich the early successional classes as well.

BATS

During the summers of 1993 and 1994, we conducted stand-level surveys of bat activity using Anabat II bat detectors. Each site was monitored for bat activity on 6 nights throughout each field season. We monitored over 2,500 hrs and recorded a total of 967 echolocation calls. The number of detections was not significantly different between years, but was different among seral stages. Overall, activity levels were low with 46% of the nights having no detections. The highest detection rates were recorded in clearcut stands while closed-canopy stands had no detections. The high detection rate, presence of feeding buzzes, and low abundance of typical roosting structures (i.e., large snags and trees) within clearcut stands suggests these sites were used for foraging. The absence of activity in closed-canopy sites suggests that these stands were unsuitable habitat for forest-dwelling bats. Although a high density of snags were present in the closed-canopy sites, the snags were typically small-diameter Douglas-fir whose lack of crevices and hollows make them unlikely roost sites.

The harvest-age structure class had the 2nd highest detection rates and was the only seral stage to have a binomial activity pattern with an initial peak of activity early in the evening and a secondary peak during the early morning hours. This pattern of activity coupled with low foraging rates suggests bats were dispersing away from roosts and commuting elsewhere to feed. If this interpretation of the observed activity pattern is correct, it would appear that some species of forest-dwelling bat could roost in mature second-growth forest. Large trees (50-100 cm d.b.h) and snags (>50 cm d.b.h. and >15 m tall), which are "roost-type" trees for certain species in Washington State, were most abundant in these stands. Suitable roosting conditions may not be present for all forest-dwelling species, however, as suggested by the absence of calls for the big brown bat and silver-haired bat within the harvest-age stands.

The landscape composition surrounding each site was also found to affect bat activity. Within the 100-ha landscapes, activity of the *Myotis* species group was positively influenced by 'Mean Nearest Neighbor Distance', a variable measuring the distance from a patch to the nearest neighboring patch of the same type. Within the 1000-ha landscapes non-*Myotis* activity was negatively influenced by the amount of mature forest. We think the bats may use landscapes more heavily if they are heterogeneous rather than composed of 1 patch type. Principal component and cluster analyses yielded the same pattern. Two distinct clusters of stands were produced based on the amount of edge produced by the juxtaposing clearcut and forested areas within the surrounding landscape. Those containing greater amounts of edge had high detection rates while those consisting entirely of continuous forest had no activity. The suitability of a landscape for bats may therefore depend on the frequency and spatial distribution of different patch types, because bats require different habitats for foraging and roosting.

BIRDS

This analysis is based on over 20,000 birds of 73 species detected within 50 m of point count stations. Bird species composition changed with forest development. Several species such as the willow flycatcher, common yellowthroat, MacGillivray's warbler, white-crowned sparrow, dark-eyed junco, song sparrow, rufous-sided towhee and American goldfinch were typical of clearcuts. In young forests, there were remnant species (e.g., MacGillivray's warbler, song sparrow, rufous-sided towhee) from the clearcut stage present as well as true forest species such as the chestnut-backed chickadee, golden-crowned kinglet, and winter wren. Species richness was lowest in closed-canopy and mature forest (37 species), highest in pre-canopy (57 species) and intermediate in clearcut class (47 species). Mean stand-level species richness for 1993-95 was highest in the pre-canopy and lowest in the closed-canopy class (20.6 and 14.2 species/stand). The following species were recorded in every stand type during each year: American robin, Swainson's thrush, winter wren, Wilson's warbler, western tanager, dark-eyed junco, and rufous-sided towhee. The following species, listed in descending order of abundance, comprised over 75% of all bird detections: winter wren, Pacific-slope flycatcher, golden-crowned kinglet, chestnut-backed chickadee, Wilson's warbler, hermit-Townsend's warbler, Swainson's thrush, and Hutton's vireo.

Bird species richness did not vary significantly among years. Species richness in 3 forested classes decreased significantly with elevation in 1994 and 1995 but not in 1993. Over 50% of the birds detected were neotropical migrants such as warblers, tanagers and flycatchers. Highest percentage of neotropical migrants occurred in the pre-canopy class. Most cavity-nesting birds were found in the harvest-age class. Highest mean species turnover occurred from the pre-canopy to the closed-canopy class, when 12 species were lost. No exotic bird species were detected in this study.

Total avian abundance varied significantly among years for all forest age classes. In 2 of the 3 years, avian abundance did not vary significantly with elevation. Total avian abundance was significantly different among age classes in 1994 and 1995 but not in 1993. Lowest abundance was consistently found in the closed-canopy class in all years.

At the species level, abundance of only 8 species did not vary significantly among years. We tested whether abundance of these 8 species was different among the 4 forest age classes for 1993-95 combined: golden-crowned kinglet, white-crowned sparrow, hermit-Townsend's warbler, common yellowthroat, MacGillvray's warbler, Steller's jay, rufous hummingbird, and American goldfinch. For other species, we tested responses to forest age classes for each year separately.

We tested the relationship between abundance and each habitat variable for each of 20 bird species for which we had an adequate sample size. Specific habitat variables that were positively correlated with most bird species abundance included: 1) counts of dominant Douglas-fir or true fir; 2) counts of sub-dominant conifers; 3) counts of sub-dominant coniferous trees 10-50 cm d.b.h. and >3 m tall; and 4) counts of dominant or sub-dominant deciduous trees.

Landscape effects on birds were examined at the individual species and community levels. We conducted the analysis on 28 species for which we had 30 or more detections. Significant relationships existed between species abundance for 6 of the 9 FRAGSTATS indices. Nine species had positive and 8 species had negative relationships with the amount of area clearcut harvested near the study sites. Virtually all species with a positive response are known to be associated with open, shrubby habitats. On the other hand, species with negative responses, such as the winter wren, golden-crowned kinglet, and chestnut-backed chickadee, are associated with canopied forests, so their responses also were predictable. Nine species were positively correlated with total area of mature forest. Those with positive responses included cavity-nesting species and species associated with multi-layered forest canopies. Five species had a negative association with area of mature forest. These included species that are most common in edge habitat. At the community level, no significant relationships were found between bird species richness and total abundance and any of the landscape variables. The primary determinants of bird species richness and abundance appear to be at the stand level. The majority of birds associated with the managed forest landscape are small songbirds that have relatively small territories and home ranges. Less than 50% of the bird species showed any relationship with landscape variables. Birds with large territories and home ranges were simply not well represented in this data set. Landscape-level habitat attributes may be more strongly associated with the distribution and abundance patterns of such species.

Although no threatened or endangered species were detected, forest management should be directed at 1) maintaining high species richness for native species; 2) increasing the number of cavity-nesting bird species; and 3) possibly increasing the number of raptors using managed forests. Potential silvicultural options include 1) maximizing the number of large-diameter coniferous trees through green-tree retention; 2) providing a constant supply of medium- and large-diameter dying trees and snags; 3) allowing the growth of deciduous trees (willow, maple, alder) in a majority of clearcuts; and 4) allowing the development of a shrubby and herbaceous understory in all forest stands. Variable-spaced thinning might help accomplish the last objective. The use of herbicides for deciduous tree and shrub control should be discouraged. When use is necessary, it should occur in August or September after the nesting season so as to avoid causing adult bird mortality and reproductive failures.

GENERAL FINDINGS

Perhaps the most obvious finding in the study is that these study areas within intensively managed forests still retain the species compositions of unmanaged forests in this region. The species lists generated in this study and the lists from studies in unmanaged forests using similar censusing techniques were essentially the same. Differences for most taxa were in the apportionment of individuals among species rather than differences in species composition. This means that the long-term retention of native species in intensively managed forests is a much more attainable goal than would be the case if species were missing. The concern can shift from species introductions and faunal reconstruction to providing critical habitat elements for taxa in managed forests that seem underrepresented in the censuses—species that apparently are not doing as well as we would like to see in reference to unmanaged stands. Taxa worth looking at more closely in this regard for amphibians might include the tailed frog and for birds several cavity-nesting species, including the red-breasted nuthatch, brown creeper, hairy woodpecker, pileated woodpecker, red-breasted sapsucker, and the Vaux's swift. No small mammal species appear to fall into this category. We are unable to make recommendations for bats because an extensive direct capture effort will be required to identify individuals to species.

The strength of relationships between wildlife occurrence and abundance at the stand and landscape scales was related to the details of their natural history. In general, species with limited vagility such as amphibians and small mammals showed strong use of particular seral stages and strong correlations with specific habitat elements within stands. For these species, correlations with landscape metrics were generally weak. For highly mobile species such as the bats and birds, landscape-level variables were more useful. Bats in particular require attention at landscape scales because they commonly use different habitats for roosting and feeding.

Managed forest landscapes consist of two faunal assemblages: one characteristic of pre-canopy conditions and another of well-developed forest. At canopy closure many common species are surpassed numerically by species of continuous forest. This changeover is particularly noticeable for small mammals and birds. Before canopy closure, bats of several genera use the airspace over clearcut areas and their associated edges. After canopy closure the forest belongs almost exclusively to the genus *Myotis*.

In terms of vertebrate community metrics, species richness was highest in the pre-canopy class for all taxa except bats. We interpret this pattern as the co-occurrence of taxa adapted to either pre- or post-canopy conditions. Species richness could not be calculated for the bats, but the clearcut areas showed the highest richness for genera. While clearcut and harvest-age classes had roughly equal species richness for many taxa (though of different species), the closed-canopy class had the lowest species richness for all taxa. This class typically had low levels of abundance for all taxa as well. We recommend that this class receive special management attention. A program of vigorous thinning would increase ground- and shrub-level structural diversity and productivity and open flight space for birds and bats. Variation in species richness was pronounced for small mammals, a pattern driven by high rodent abundance in 1994, but not significant for birds. Abundance of small mammals was high in 1994 and also significantly different in 2 of the 3 years for birds.

We are concerned that structural diversity be maintained in these forests. At present, levels of downed wood appear sufficient to allow persistence of most amphibians and small mammals. These stands still retain stumps and humus material from the original forest that are important elements for the forest-floor fauna. Because these structures have not been replaced over the past several decades, they are being lost to the system. Similarly, the scarcity of large-diameter snags is a concern. Reduced species richness and abundance is already observable for several cavity-nesting birds. Bats also require large snags and trees with cracks and crevices, although there are no data to allow an assessment in their trends of abundance. Continuous replenishment of these structural elements is essential if these managed forests are to support the full complement of native species.

EXECUTIVE SUMMARY

EAST-SIDE STUDIES

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PURPOSE AND APPROACH

This study was designed to examine wildlife responses to varying landscape conditions in managed forests of Washington to assist resource managers in meeting the wildlife habitat objectives of the Timber Fish and Wildlife Agreement (1987). Old-growth coniferous forests have been replaced by second-growth "managed" forests over large areas of the Pacific Northwest. In northeastern Washington, subsequent harvest has created a mosaic of forest patches of varying sizes and ages within a matrix of closed-canopy forest (<90 years). These forests differ in species composition from pre-settlement conditions, and lack some structural features of old-growth forests. Extensive studies of old-growth forests have pointed to the need to conduct replicated studies of the wildlife use of managed forests. Our objectives were:

- to describe the species composition and abundance levels of wildlife and plant communities occurring in forest stands of varying structural stages and landscape configurations in watersheds managed primarily for timber production, and
- to develop methods for analyzing wildlife responses to landscape-scale habitat conditions in managed watersheds.

Our approach was to sample terrestrial vertebrate groups in forest stands that represented the predominant age and size classes on the landscape and that were located in watersheds that were moderately to highly fragmented. This approach allowed replicated statistical analysis of wildlife habitat relationships at the stand level. Additionally, we were able to examine the effects of landscape level factors on wildlife by considering the spatial context of each stand.

Prior to selection of stands, we conducted a landscape analysis of the managed forests of northeastern Washington. This was done by using satellite imagery and developing a GIS methodology that enabled us to classify forest stands by age class, subdivide the landscape by watershed, calculate landscape metrics describing each watershed, and examine watersheds for their management intensity and potential as study areas.

LANDSCAPE ANALYSIS

The analysis of the disturbance gradient and context of stands for the watersheds was an important part of watershed and study site selection. To evaluate the consequences of forest conversion on species diversity and population dynamics, the pattern and extent of habitat fragmentation on such landscapes had to be understood. In 1993, we addressed this problem by using geographical information systems to map and analyze fragmentation of coniferous forests in northeastern Washington (Stevens and Pend Oreille counties). Landsat imagery (1991) was classified into 4 broad habitat types: cleared in previous 10 years; deciduous and disturbed canopy from harvesting operations; regenerating forest of <30 years; and mature, closed-canopy of >30 years. For 20 watersheds (≈2,000-7,000 ha) representing a range of harvest intensities, we created polygon representations of forest patches and a database containing the habitat type, size, and perimeter for each patch in a watershed. We determined the number of patches and the proportion in each habitat type, and calculated dominance, contagion, and 2 fractal indices for each watershed. We also examined a line-transect sampling method to estimate the probabilities of encountering changes in habitat types.

Principal components analysis was used to examine intercorrelations among variables and to describe patterns of fragmentation. The first component characterized the general gradient in fragmentation. Fragmentation increased with increases in the number of patches, areal proportion of disturbed and clearcut habitats, and number of transitions per transect. There were concordant reductions in dominance, areal proportion of mature forest, and the proportion of transects without transitions. The second component reflected differences in the extent of regeneration forest and thus in the length of time over which fragmentation has taken place. The third component highlighted differences in the way that fragmentation has taken place. Decreasing values corresponded to decreasing numbers of regenerating and clearcut patches, but to increasing size of clearcut patches. Constancy in perimeter-area fractals indicated that the processes determining stand shape, primarily forest harvest, were similar across all watersheds.

STUDY SITES

Managed forests in northeastern Washington consist primarily of second-growth mixed coniferous trees. Composition of these forests varies with local conditions, but the principal tree species are grand fir (*Abies grandis*), western larch (*Larix occidentalis*), lodgepole pine (*Pinus contorta*), Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and western redcedar (*Thuja plicata*). Forest harvest over the past 30-40 years has created a mosaic of stands that vary in age and size within a matrix of closed-canopy forest. Up to 50% of some watersheds have been converted from closed-canopy forest (>60 years) to younger aged stands. We selected 7 watersheds that ranged from moderate to high degrees of disturbance (30-50% conversion from closed-canopy to clearcut or regenerating forest).

Based on our GIS analysis we selected 2 sizes of stands for study: (1) 12-15 ha, which represented the smallest harvest units typically observed, and also approximated the mean size of Upland Management Areas (UMAs) created in this region, and (2) >34 ha, which represented the largest harvest units. We examined 3 structural stages of forest: a clearcut stage resulting from harvest <6 years with adjacent stands >15 years older; a pre-canopy stage about 16-20 years after harvest with adjacent stands recently clearcut or with fully developed canopy; and a closed-canopy stage >60 years after harvest that had not been commercially thinned with adjacent stands in any stage. These structural stages represent most of the managed forestlands in this region.

For the 7 watersheds, we mapped all stands that met our size and structural stage criteria. Six stands of each size and forest type were selected for a total of 36 stands. These study areas were located in Stevens and Pend Oreille counties on lands owned or administered by the Colville National Forest and Plum Creek Timber.

VEGETATION ANALYSIS

We conducted an extensive series of vegetation surveys on each study area to acquire information on stand composition and structure. These data were used in our examination of patterns of association between habitat structure and wildlife species composition and relative abundance. We used 2 protocols to consider features of likely importance to birds and to small mammals, respectively. On transects monitored for birds, we established 24- by 24-m plots (576 m^2) at each point-count station. We determined the species, size, and status of all trees; percentage of canopy cover; species, area, and height class of all shrubs; measures of woody debris; number, height, and type of regenerating trees; and ground cover. On grids trapped for small mammals, we measured similar variables, but with a broader array of variables describing woody debris.

Statistical analysis indicated that significant differences existed in the 3 forest types, and that similarities within forest types were greater than between forest types. Vegetation differed slightly between the 2 stand sizes. The 3 forest types differed in percentage canopy cover and in size and height distributions of trees. Although canopy cover in clearcuts was reduced relative to the other forest types, tree height was relatively large because of the retention of some live trees during harvest. The shrub canopy was developed to a greater extent in regeneration and clearcut stands than in closed-canopy forest.

TAXA OF THE GENERAL SURVEY

Species and species groups chosen for study were those that could be sampled well at stand-level scales. All taxa were sampled for 3 consecutive years (1993-1995). We conducted surveys of small mammals and birds on the 36 stands to evaluate the effects of local and landscape habitat conditions and stand size on distribution. We used radio-telemetry to determine roost use by the silver-haired bat (*Lasionycteris noctivagans*). Additionally, we conducted studies to examine (1) the dispersal patterns of amphibians on a fragmented landscape using molecular markers as an alternative to mark-recapture methods, (2) the use of forested strips as corridors for small mammals, (3) the factors determining the composition of shrew assemblages, and (4) differences in the rates of predation on artificial nests as a function of habitat and stand size. Results of these supplemental studies are summarized below and provided in their entirety in the Appendices for Volume 3.

SMALL MAMMALS

Small mammals were sampled using pitfall traps arranged in a 6 × 6 station grid (10-m spacing) on each stand. Pitfall traps capture insectivores and non-jumping rodents well, but are less effective at capturing deer mice, chipmunks, and jumping mice. We chose this method because we wanted to emphasize those species that were not sampled well in previous studies. Many of the more common species, although not sampled efficiently with pitfall traps, are generally caught in sufficient numbers for statistical analyses. Grids were placed in areas considered representative of the stands and were buffered from riparian areas. Trapping was conducted for 2 weeks at each site during late May, June, or early July of 1993, 1994, and 1995.

Small mammal communities are relatively diverse in northeastern Washington, and we caught a total of 3,739 individuals of 18 species. Eleven species had sufficient captures to allow statistical analysis. The total number of captures varied substantially between years, increasing from 743 in 1993 to 2,248 in 1994 and then declining to 718 in 1995. Total number of captures was similar in the 3 forest types in 1993 and 1995, but was greater in clearcuts in 1994. Red-backed voles (*Clethrionomys gapperi*), deer mice (*Peromyscus maniculatus*), long-tailed voles (*Microtus longicaudus*), montane voles (*Microtus montanus*), masked shrews (*Sorex cinereus*), montane shrews (*Sorex monticolus*), and vagrant shrews (*Sorex vagrans*) were all captured more frequently in 1994 than in other years.

Statistically significant differences in use of the 3 forest types were found for 9 species. Red-backed voles and masked shrews were more abundant in closed-canopy forest than in clearcut or regenerating stands. Deer mice, pocket gophers (*Thomomys talpoides*), jumping mice (*Zapus princeps*), and long-tailed voles were most abundant in clearcuts. No species was most abundant in regenerating stands, but vagrant shrews, heather voles (*Phenacomys intermedius*), and montane voles were largely absent from closed-canopy forest and were equally abundant in clearcuts and regenerating stands. Pygmy shrews (*Sorex hoyi*) and montane shrews were less abundant than the other shrew species and showed no association with any forest type. There were minimal effects of stand size on distribution. Over all years, species richness was highest in clearcuts (7 species) followed by regenerating (5.1) and closed-canopy forests (4.5), and did not differ with stand size. The general pattern of species richness held for all years, despite the addition of some species in 1994.

For each of the 11 species, we examined their habitat and landscape relationships using stepwise regression analysis. Habitat characteristics of each trapping grid were based on our vegetation analysis and included descriptors of trees, shrubs, regeneration, woody debris, and ground cover. We examined the landscape context of each grid by considering habitat patches within a 1 km-radius circle. Because many landscape descriptors are intercorrelated, we considered only the number of patches, total perimeter, and proportion of area in each forest type. Because of the temporal variability in relative abundance, we examined habitat use for each year that the number of captures was >20.

The best predictors of small mammal abundance and distribution were variables describing specific components of vegetative structure (e.g., woody debris, shrub area; most R^2-values between 50-80%). Variables describing the landscape surrounding trapping grids provided much less predictive power (R^2-values <20%). Where landscape variables were important, they were associated with the forest type predominantly used by a species. For example, red-backed voles were positively associated with habitat features that are reflective of closed-canopy forest such as canopy cover, medium to large d.b.h. trees, low shrubs, and large, well-decayed woody debris ($R^2 = 60-75\%$). Of the landscape variables, red-back voles were positively associated with the total area of closed-canopy forest and negatively associated with increasing perimeter of closed-canopy stands (R^2 = 13-24%). A resident of clearcuts, the deer mouse, was associated positively with recent woody debris and negatively associated with medium-high shrubs and regenerating conifers (R^2 = 48-60%). In the landscape regression, this species was positively related to the total area of clearcut within a 1 km-radius circle of the grid (R^2 = 10-20%). Captures of the pygmy shrew, a species that showed no association with any of the 3 forest types, were associated positively with woody debris and area of medium-tall shrubs.

The composition and structure of small-mammal communities in the 3 forest types is similar to those of comparable forest stands in the region. Forest type is more important that stand size in explaining differences in small-mammal distribution. The level of forest fragmentation has probably not reached a level that prevents dispersal and recolonization of forest stands. The increases in populations of all species in 1994 resulted in wider distribution of many species among the 3 forest types. Such pulses in population numbers probably result in substantial mortality for many individual animals, but increase the probability that populations that might go locally extinct are restarted.

AMPHIBIANS AND REPTILES

Amphibians and reptiles were captured during the pitfall surveys for small mammals. The numbers of captures was low and included a total of 81 individuals of 3 amphibian and 1 reptilian species during 1993-1995. The western toad (*Bufo boreas*) was the most common species (48 captures). This species was more common in the regenerating (N = 31) and clearcut forests (N = 15) than in the mature (N = 2) forests. It was more common in small stands (35 captures) than in large stands (13). We captured western toads more often in 1995 (37 captures) than in 1994 (8 captures) or 1993 (3 captures). Western toads were captured more frequently at specific sites than at others. For example, in 1994, 4 of 8 captures were at a single site and, in 1995, 13 of 37 captures were at 1 site and an additional 10 were at another site.

A total of 27 long-toed salamanders (*Ambystoma macrodactylum*) was captured. This salamander was found more often in the clearcut (13 captures) and mature (10 captures) forests than in the regenerating forests (4 captures). More individuals were captured in small stands (22 captures) than in large stands (5 captures). Long-toed salamanders were captured more frequently in 1994 (14 captures) and in 1995 (11 captures) than in 1993 (2 captures). Captures of this salamander were more evenly distributed among sites than were those of the western toad. For example, in 1994 no more than 3 of the 14 total captures were at a single site and, in 1995, only 4 of the 11 captures were at a single site.

The 2 captures of *Rana luteiventris* were in 1993 on 1 small regenerating stand. The northern alligator lizard (*Elgaria coerulea*) was found only at 1 large clearcut stand. There were 3 captures of this lizard in 1994 and 1 capture in 1995.

These results indicate that pitfall trapping does not effectively sample amphibian and reptile populations in northeastern Washington. The distribution of the long-toed salamander is of interest because it suggests that the presence of coarse woody debris in clearcuts provides appropriate microhabitat for this species.

BIRDS

We used a circular point-count method for surveying bird populations which involved estimating the distance to birds detected within 50 m. We established 3-4 and 12 point count stations on the small and large patches, respectively. The point-count stations were 50 m-radius circles. The area covered by these stations was about 40% of the total stand area. We buffered the point count stations from the stand edge and from riparian areas or gaps in the stand. We completed 4 entries of each stand in 1993 and 6 entries per stand in 1994 and 1995. We recorded >22,000 observations of 110 species during the 3 years over all sites combined.

Species richness was higher in clearcut (16.7 species/stand) and regenerating forests (16) than in closed-canopy forests (13.7). Species richness was greater on large stands (19 species/stand) than on small stands (11.9). Species richness was lower in 1993 (14.4 species/stand) than in 1994 (15.8) or 1995 (16.2), but this partly reflects lower sampling effort in 1993. The number of birds detected per point was higher in regenerating (4.0 counts/station) than in either closed-canopy (3.2) or clearcut stands (3.1). Small stands had more birds per point (3.7 counts/station) than did large stands (3.2). Species diversity and evenness were similar across all forest types. Species diversity and evenness were both greater in the larger stands.

Stand-level associations were examined for the 42 most common species. Fifteen species were detected more frequently in the closed-canopy forests than in either of the other 2 forest types. These species were pileated woodpecker, Cassin's vireo, winter wren, chestnut-backed chickadee, brown creeper, golden-crowned kinglet, Swainson's thrush, hermit thrush, varied thrush, evening grosbeak, Townsend's warbler, red crossbill, gray jay, and red-breasted nuthatch. More species were restricted to closed-canopy forest than to the other forest types. Eight species were detected more frequently in the regenerating forests (dusky flycatcher, warbling vireo, Nashville warbler, orange-crowned warbler, Wilson's warbler, black-headed grosbeak, chipping sparrow, and brown-headed cowbird) and only 5 species were detected more frequently in the clearcut forests (red-naped sapsucker, hairy woodpecker, Clark's nutcracker, Townsend's solitaire, and pine siskin). Several species were more common in 2 habitat types: northern flicker, olive-sided flycatcher, American robin, MacGillivray's warbler, and dark-eyed junco in the clearcut and regenerating forests, and mountain chickadee and western tanager in closed-canopy and clearcut stands.

Few species differed in relative abundance between the 2 stand sizes. Clark's nutcracker, Townsend's solitaire, Nashville warbler, and yellow-rumped warbler were detected more commonly in larger stands. Northern flicker, brown creeper, golden-crowned kinglet, Cassin's vireo, orange-crowned warbler, Townsend's warbler, chipping sparrow, dark-eyed junco, and pine siskin were found in smaller stands.

Of 42 common species, regressions of the habitat variables on the detections for each species yielded significant models for 15 species. The predictive power of these models varied from 10-95%, but was generally low. Most species associated with closed-canopy forest were associated positively with canopy closure and number of large diameter trees, variables representative of this forest type. Additionally, Swainson's thrush was associated positively to a dense understory of shrubs and regenerating trees, and red-breasted nuthatch was associated positively with medium to large snags. Only 1 clearcut associate could be examined. The hairy woodpecker was related positively to number of snags and to open canopy cover. Warbling vireo and dusky flycatcher were most abundant in regenerating forests and were associated with number of regenerating trees, number of stumps, and shrub cover. Five other species that occurred primarily in open canopy stands were associated negatively with either canopy cover or number of coniferous trees ($R^2 < 50\%$).

In contrast to these results, the predictive power of the regression models incorporating the landscape-level associations was consistently higher. As one would expect, closed-canopy species were all associated positively with the total area of closed-canopy forest within a 1 km-radius of the transect. However, several of these species also were associated negatively with increasing perimeter of closed-canopy forest. This indicates that larger tracts of closed-canopy forest are important to these species. Abundance of the hairy woodpecker, a species of clearcuts, was related positively to area of clearcut, but also to perimeter of closed-canopy forest. This suggests that the juxtaposition of these habitats is significant. For 3 species, local habitat factors provided very little information on habitat relationships compared to regressions with the landscape variables. For example, western tanagers were negatively associated with perimeter of regenerating forest surrounding the transect. Another species, the yellow-rumped warbler, was not clearly associated with any forest type, but was associated positively with the area of closed-canopy forest and negatively with number and area of regenerating, clearcut, or disturbed patches.

ROOST USE BY BATS

We examined the roosting habits of the silver-haired bat (*Lasionycteris noctivagans*), which preferentially uses older forests in other parts of its range in the Pacific Northwest. Bats were captured using mist nets and harp traps at stream, pond, and road sites in 1992 and 1993. Fifteen of 30 *L. noctivagans* captured were equipped with radiotransmitters. A total of 15 roost sites used by 13 individuals was found. All roosts were <3.5 km from the initial capture site and were in dead or dying trees with exfoliating bark, extensive vertical cracks, or cavities. Fourteen roosts were in trees with diameter at breast height >30 cm. In many cases, the species of roost tree was a legacy of earlier forest species composition. Roost trees were significantly taller than neighboring trees, which may facilitate the location of roosts by bats. Analysis of a 15-

m radius plot around each roost tree revealed that roost sites had significantly less overstory canopy closure, less understory, and shorter understory vegetation than comparable random plots. Reduced overstory canopy and understory vegetation provide a less cluttered environment for bat flight, which may be particularly important for newly volant young. Recruitment and retention of large snags (>30 cm d.b.h.) and the maintenance of structural complexity in forest patches in upland, as well as riparian areas, are important for the conservation of bat species in managed forests.

SUPPLEMENTAL STUDIES

AMPHIBIAN MOVEMENTS

The question of how the movement patterns of amphibians are affected by forest harvesting is important to understanding the population dynamics and probabilities of local extirpation for these species. Because population numbers fluctuate from year to year, recolonization of suitable habitats is essential. We recognized that direct methods of examining this question (e.g., mark-recapture) were not logistically feasible in northeastern Washington and would provide insufficient data. Consequently, we began examining molecular methods for analyzing the genetic "tags" of individuals. We hoped to be able to sample populations across the landscape and infer the movement of individuals by the presence or absence of these tags within the sample populations. As work progressed, we realized that our initial question had to be reframed by asking whether local populations had become genetically differentiated as a result of fragmentation. We found insufficient variation from DNA sequencing of the mitochondrial genome. Consequently, we developed 7 microsatellite markers and studied their frequency distributions in 8 *Rana luteiventris* (3 loci) and 3 *Hyla regilla* (4 loci) populations.

The use of microsatellite loci is relatively recent and several estimates of population structure and genetic distance have been developed for use with these markers. We found a broad scale pattern of isolation-by-distance that was consistent with earlier findings based on protein electrophoresis. Isolation-by-distance regression models suggested neighborhood sizes of 19 to 274 frogs for *R. luteiventris* populations. Four recently proposed estimates of genetic distance for microsatellites failed to recover the correct geographic relationships for *R. luteiventris* populations, although an estimator derived from an infinite alleles model was successful. More research is needed to determine the general applicability of estimators derived specifically for microsatellite markers. Estimators derived from an infinite alleles model may be adequate for descriptions of genetic diversity in natural populations.

We also investigated a multilocus detection method for measuring microsatellite variation simultaneously at many loci. This technique reveals a series of alleles within a profile of DNA fragments created by restriction enzymes. Because of potential measurement errors, comparisons of different individuals require grouping alleles of similar size into bins. We found that a measure of similarity between individuals (mean band sharing) increased continuously with increasing bin width. By using replicate gels we calibrated a binning algorithm and found significant differentiation between *R. luteiventris* (N = 5) and *H.* regilla (N = 2) populations ($P \leq 0.008$). These results were consistent with results from single locus markers. Consequently we now have 2 techniques for potentially examining the genetic divergence of amphibian populations following habitat modification.

USE OF CORRIDORS BY SMALL MAMMALS

Corridors have been advocated as a means of ameliorating the effects of habitat fragmentation on wildlife populations and communities. Their usefulness, however, depends on several factors, including degree of fragmentation, corridor size, habitat quality, and especially the habitat requirements and mobility of the particular species. Distributions of small mammal species were analyzed across clearcut, regenerating, and closed-canopy forest habitats in northeastern Washington by live-trapping in 1993 and 1994. Red-backed voles were identified as a forest-restricted species, able to perceive corridors in the landscape. Use of corridors by red-backed voles was found to be highly variable due to interyear variation in population density. When populations were high, corridors provided additional habitat for colonization. Corridor stands, however, contained lower quality habitat than forest stands, lacking habitat characteristics preferred by red-backed voles. At lower population levels, the corridors only provided a dispersal route between the larger, more preferred forest stands.

Variation in habitat quality and corridor size appeared to influence vole abundance within corridors. Size of forest patches did not affect vole presence or abundance due to differences in habitat surrounding forest patches. Thus, the relative importance of corridors to red-backed vole distribution and persistence remains uncertain. However, these connective strips do serve a function as additional habitat in periods of high population density and are capable of facilitating movement between forest patches for forest-interior small-mammal species.

INTERSPECIFIC AND TROPHIC RELATIONSHIPS OF SHREWS

Five species of shrew occur in northeastern Washington: *Sorex hoyi*, *S. cinereus*, *S. vagrans*, *S. monticolus*, and *S. palustris*. We examined the patterns of community composition for these species to determine the extent that interspecific competition might influence their distribution. Shrew populations were sampled on the 36 landscape sites and on an additional 36 riparian and adjacent upland sites. The sites were sampled from 1993-1995 using pitfall and snap-traps. We considered 3 functional groups of species based on body mass, jaw, and cranial measurements. If interspecific competition is important, an assembly rule predicts that only 1 species should be present in each functional group until each has a species present. Assemblages that do not follow this rule are termed *unfavored* states. Seven, 11, and 10 of 17 possible assemblages of small, medium, and large species were observed in 1993, 1994, and 1995, respectively. We compared frequency distributions of the actual shrew assemblages to those generated by a random model. The shrew assemblages appeared to follow the assembly rule in 1993 when 37 favored states were observed compared to 24 expected from random. Unfavored states were observed 54 times compared to only 18 favored states in 1994, indicating species did not follow the assembly rule in that year. The assembly rule also failed in 1995, when there was no significant difference between observed and expected assemblages. We also examined food habits of shrews based on stomach contents. *S. hoyi*, the smallest species, consumed the longest prey. Prey used by the other 4 species were similar in length. Intra- and interspecific diet similarity changed depending on shrew assemblage. The results of this study suggest that the patterns we observed are due to fluctuations in abundance between years and to the specific microhabitat requirements of each species.

NEST PREDATION

Nest predation is considered a major influence on the population dynamics and community relationships of many bird species. We conducted an experiment to examine how rates of predation on artificial nests are influenced by habitat and patch size, and to determine the relative importance of the major groups of predators. In 1994, we placed 20 ground nests on each of 5 replicate patches of 2 size classes (12-15, >34 ha) and 3 forest types (clearcut, regeneration, and closed-canopy) for a total of 30 stands. Nests were buffered from the edge of the patch. Predation rates did not differ among patch sizes. Rates of predation increased over time suggesting a functional response by predators. Mammals and birds accounted for about 74% and 20% of all predation events, respectively. Predation increased with increasing forest structure (clearcut < regeneration < closed canopy). The significantly greater predation in closed-canopy forest than in regeneration or clearcut patches ran counter to our expectations. This result is explained largely by the greater importance of mammalian predators and particularly the distribution of the red squirrel (*Tamiasciurus hudsonicus*). Indeed the relative occurrence of red squirrels paralleled the distribution of depredated nests in the first trial. Relative abundance of red squirrels was the best predictor of degree of predation across patches. Populations of red squirrels were much higher in 1994 then in 1993 or 1995. Consequently, the results of studies such as ours might vary depending on when they are conducted.

GENERAL CONCLUSIONS

The second-growth forests in northeastern Washington have been fragmented moderately by forest harvest. Over much of this region, stands of younger forest are usually embedded within a matrix of closed-canopy forest. At current levels of fragmentation, the diversity of terrestrial vertebrates appears to be relatively high. Although we do not have a benchmark from unmanaged forests of greater age for this region, most of the species one would anticipate to occur in these forests were observed. The actual long-term viability of many species is unknown, however, because of our inability to detect them or because of fluctuations in their numbers. We believe that the retention of most species should be attainable and that forest management can be used to provide critical habitat for species that have specialized requirements.

Responses to habitat fragmentation and to landscape structure vary with taxonomic group. Small-mammal assemblages are diverse in northeastern Washington. Their composition is primarily a function of microhabitat selection by and variability in abundance of individual species. All small-mammal species had their greatest abundance in 1 of the 3 forest types and showed typically strong associations with variables describing microhabitat conditions. Correlations with landscape variables were very weak. Maintaining landscape structures such as corridors may be useful for retaining species such as the red-backed vole, which are restricted to closed-canopy forest. We believe that pulses in the abundance of many small-mammal species, as observed in 1994, allow recolonization of forest stands. Such pulses may be linked to climatic conditions or to variation in predation pressure. Management actions that affect woody debris (e.g., broadcast burning) and the shrub layer are the most likely to alter species composition of mammalian assemblages. Changes in the distribution and abundance of small mammals may affect the population dynamics of avian species through nest predation.

Many avian species also were associated with particular habitat types, but these species were less likely to be tied to specific microhabitat variables. However, our analysis indicates that retention of specific structural elements such as snags is particularly important for some species (e.g., hairy woodpecker, pileated woodpecker, red-breasted nuthatch). Landscape-level factors with the exception of stand size provided better predictors of relative abundance for most species. This result corresponds to the greater mobility and broader range of these species. The retention of large (>34 ha) closed-canopy stands will help to maintain some species.

The patterns of species richness also point to differences in the structural condition of the managed forest. Small mammals reach their highest diversity and abundance in clearcuts. This corresponds to the greater woody debris and understory structure of these stands. Avian species richness is highest in regenerating stands. Clearcuts have not retained a sufficient number of trees for nesting and the closed-canopy stands have a reduced understory or shrub component. Additionally, the composition of regenerating stands draws from species that also utilize the other forest types. As an adaptive management strategy, some thinning of closed-canopy stands might increase their suitability by increasing shrub level diversity.

There is a clear need to maintain structural diversity within the managed forests of northeastern Washington. This was particularly evidenced by the patterns of roost use of the silver-haired bat. Although bats also have high mobility, the preference of this species for large diameter snags suggests that current silvicultural practices may in future result in the loss of species as these relictual elements disappear. The retention of leave trees in harvest units is an important first step, but does not satisfy the need for large d.b.h. trees. The production and retention of snags is also of importance to primary and secondary cavity nesting birds. The degree of downed wood is sufficient in clearcuts, but less available in closed-canopy forests. Thinning of closed-canopy stands could provide additional woody debris.

INTRODUCTION AND TECHNICAL APPROACH

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INTRODUCTION

The managed forests of Washington State encompass approximately 17,305,000 acres (7,003,333 ha) of which about 63% are on State and private lands (Card and others 1985). The Timber Fish and Wildlife (TFW) Agreement (1987) introduced both a framework for management practices on State and private forests, and a mechanism to evaluate and modify management practices to achieve stated resource goals. A critical question facing TFW resource managers is how to balance the TFW wildlife goal to "provide the greatest diversity of habitats (particularly riparian, wetlands and old growth), and to assure the greatest diversity of species within those habitats for the survival and reproduction of enough individuals to maintain the native wildlife of Washington forest lands" with the timber resource goal of "...continued growth and development of the State's forest products industry..." (Timber Fish and Wildlife 1987).

Franklin and Forman (1987) have proposed that the number, size, and arrangement of stands in a managed forest landscape could be modified to achieve different wildlife objectives. However, we must first be able to analyze and predict wildlife responses to varying landscape conditions. The response of wildlife species to local stand conditions has been hypothesized for certain species (see Thomas 1979, Brown 1985, Irwin and others 1989), but these responses have so far only been evaluated in the field in unmanaged forests (Ruggiero and others 1991); no comprehensive research on wildlife communities in managed forests has been conducted in the Pacific Northwest. Even less is known of the response of wildlife populations and communities at the landscape scale.

Analysis of wildlife habitat relationships can be approached from a hierarchical perspective (e.g., Urban and others 1987, Irwin and others 1989). Irwin and others (1989) identified 3 spatial scales in managed forests - landscape, stand, and gap - with 3 corresponding wildlife habitat analysis levels. At the stand scale, we are concerned with habitat features such as stand size, shape, and seral stage. When we view managed forests from a landscape perspective, however, we are concerned with the spatial and temporal patterns occurring among stands. At the landscape scale we consider, for example, distances between stands, connectivity of stands, and conditions surrounding stands. The response of wildlife populations at stand and landscape scales will depend upon the particular life history characteristics of a species, the environment (habitat and other species present), and the species' population structure, which reflects the interaction of life history and the environment (Gilpin and Soulé 1986).

Primary concerns in this study of wildlife in managed forest landscapes are the responses of wildlife species to harvest regimes that affect the composition, size, and juxtaposition of habitat stands. The interaction of stand size and type, characteristics of adjacent stands, isolation of stands, and mobility of species will influence species composition in a stand despite potentially high habitat suitability (e.g., Lehmkuhl and Ruggiero 1991). For example, individuals of species with small home range requirements and limited dispersal capabilities might be restricted to a single forest stand. If a stand is too small it might be unable to support a viable population (i.e., births < deaths) over time and the limited mobility of the species might preclude recolonization. Individuals of species with better dispersal capabilities might inhabit the stand intermittently by immigrating from nearby "source" habitats, but be incapable of reproducing there (i.e., a "sink" habitat). This is the "source-sink" effect (Van Horne 1983, Pulliam 1988). Wide-ranging species can move between stands to exploit preferred resources, but the spatial distribution of stands can affect their use of habitat as well (e.g., Milne and others 1989). In such a case, isolated stands of preferred habitat might not be used by these animals, whereas stands of less preferred habitat might be used if they are in close proximity to more suitable stands. Given the potentially varied and complex responses of wildlife to the distribution of stands in a managed forest landscape, we believe that it is necessary to examine wildlife habitat relationships at the stand scale in addition to analyzing effects resulting from the landscape context of the stand. To this end, we developed both a database of information on wildlife populations and methods to evaluate wildlife responses to changes in forest habitat conditions as a result of timber harvest at both stand and landscape scales. This information will assist resource managers in accomplishing the wildlife habitat objectives of the Timber Fish and Wildlife Agreement (1987).

The objectives of the project were:

- To describe the species composition and abundance levels of wildlife and plant communities occurring in forest stands of varying structural stages and landscape configurations in watersheds managed primarily for timber production; and

- To develop methods for analyzing wildlife responses to landscape-scale habitat conditions in managed watersheds.

TECHNICAL APPROACH

DEVELOPMENT OF THE STUDY DESIGN

Milne and others (1989) review some of the difficulties in predicting the occupation or use of habitat at the landscape scale. These problems include: (1) at some times the landscape may be "unsaturated" with organisms and thus many suitable areas may be vacant. This possibility reduces our ability to accurately describe the habitat requirements of wildlife species; (2) humans may recognize different landscape variables to which other species may perceive no difference; (3) models based on the analysis of correlations between animal abundance and landscape variables which lack explicit spatial information make it difficult to translate correlational relationships to maps; and (4) if populations are monitored infrequently, our ability to describe spatial variation in habitat use will be reduced. Thus the success of habitat modeling is contingent on animal density, human perceptions, spatial information, and temporal variation in wildlife populations and habitat needs (Milne and others 1989).

The study design we adopted had to meet these challenges and still be logistically feasible. In evaluating the types of approaches that one might develop for studying wildlife habitat relationships at the watershed scale, we considered the following 2 approaches:

(1) Wildlife-habitat relationships based on sampling conducted at the landscape scale. The variables in this analysis describe the spatial composition of many (20-30) watersheds as the primary sample units. Variables might include habitat diversity, diversity at any point in the landscape (i.e., contagion), and the shape of habitat patches (i.e., fractal dimension). These types of variables can be calculated easily from map data entered into a GIS. Wildlife sampling is conducted throughout each watershed on a very broad scale (e.g., by running transects). Vertebrate sampling conducted at the scale of entire watersheds would, by necessity, result in data with relatively low predictive power, such as presence-absence data. The landscape variables can then be used, for example, as independent variables in a regression model to predict such wildlife community characteristics as species richness or patterns of abundance among species.

Although this approach seems appealing, it has several critical problems. First, obtaining an adequate number of samples (i.e., watersheds) for statistical analysis is difficult due to the necessity of sampling terrestrial vertebrates throughout entire watersheds. Second, by sampling different sets of watersheds in each year to increase sample size, one loses the ability to examine or account for temporal variation in species abundance. Third, the extensive sampling across landscapes that is needed for this approach precludes collection of precise stand-scale data that are largely unknown for managed forest stands. Finally, we might find that many wildlife species do not respond to the landscape variables that we perceive or measure, making model development unachievable. Initial work in the Pacific Northwest thus far indicates that the predictive power of landscape variables is weak (Lehmkuhl and others 1991).

(2) Wildlife-habitat relationships based on sampling conducted at the stand scale. In this approach, terrestrial vertebrate groups are sampled in stands occurring along a gradient of landscape conditions. Selected stands vary both in structural stage and landscape context (i.e., spatial relationship of a stand to others in the watershed). This approach allows statistical analyses of wildlife habitat relationships at both the stand and landscape scales. For example, one could attempt to predict components of wildlife community composition within stands according to variables of type and context and then integrate these stand-scale results into a landscape-scale model. We selected this approach because it provides a sufficient sample of stands for analysis, a high resolution of important stand-scale habitat relationships, and incorporates most of the spatial variation of the watersheds. Temporal variation in habitat use was examined by sampling each stand for 3 consecutive years.

Originally, our objective was to design vertebrate community studies that could be applied uniformly in both East-side and West-side study areas. As we began surveying potential study areas and consulting with public and private land managers, however, it soon became apparent that such an approach would not be possible. Differences in forest composition and structure and, in particular, past timber management practices, result in stand conditions and landscape configurations that differ dramatically east and west of the Cascade Crest. Consequently, although we made a concerted effort to standardize both vertebrate and vegetation sampling methodologies, East-side and West-side studies differ in many ways from each other, especially in terms of the stand types sampled and the approaches taken for landscape-scale analyses. This section describes the development of the landscape-scale GIS databases for the West-side and the East-side studies, the process of selection and location of study sites, and the general procedures used in wildlife and vegetation sampling. More detailed descriptions of the sampling procedures accompany appropriate sections. In addition, this section summarizes selected directed studies conducted in association with the broader landscape project.

DEVELOPMENT OF LANDSCAPE-SCALE GIS DATABASES AND SELECTION OF STUDY SITES

WEST-SIDE STUDIES

Development of a GIS database--The first step in implementing our sampling design was to develop a means of discriminating between managed landscapes according to the age, size, and pattern of forest stands, and the intensity and manner in which these landscapes have been logged. We began by stratifying landscapes according to management history. Target landscapes were those consisting primarily of second-growth Douglas-fir dominated forest in southwestern Washington. We used Landsat Thematic Mapper (TM) imagery to create a surrogate structural-stage classification based on species, canopy structure, and stand age. With these data, we developed a

GIS methodology that enabled us to classify landscapes by age class, subdivide the landscape by watershed, calculate the pattern or character of the landscape, and examine areas for their management intensity and potential as study areas (see Young and others 1993 [Appendix] for a more detailed description of our analytical approach).

We classified each 25-m TM pixel into 1 of 5 habitat classes (4 forest age classes and 1 "other" category) using both unsupervised and supervised classification procedures. Age classes identified were 3-8 yr, 10-20 yr, 50-80 yr, and >80 yr; forests in the 20-50 yr age range could not be confidently separated from other forest types with the data available to us. To facilitate the processing of this very large and spatially extensive data set, we resampled the original classification to 100- x 100-m (1 ha) pixels using a nearest-neighbor resampling algorithm. We quantified the range of landscape patterns present in our data set by subdividing the age-class map into major watersheds (60,000-81,000 ha) using boundaries provided by the Washington State Water Resource Inventory Area (WRIA) classification. This divided the area encompassed by the satellite imagery into 68 whole or partial watersheds (basins). To ensure that basins considered for sampling in this study were similar in ecological and physiographic characteristics, so that major differences among landscapes would result primarily from management history, we selected the subset of 19 basins located west of the Cascade crest and east of the Puget Trough Physiographic Province (Franklin and Dyrness 1973) for further analysis (Fig. 1).

We assessed variation in landscape conditions in each of these 19 basins by calculating standard landscape indices (diversity, dominance, and fragmentation) and found that the range of index values among watersheds was narrow, i.e., most of the variation in landscape pattern was occurring within basins. To provide a more useful unit of landscape analysis, we subdivided each basin into sub-basins (3rd-order watersheds) ranging in size from 4,000-12,000 ha according to guidelines provided by the Washington State Department of Natural Resources (DNR) for delineating a Watershed Administrative Unit (WAU). This procedure resulted in 119 sub-basins mapped by age class (Fig. 2). To further refine this data set for our study objectives,

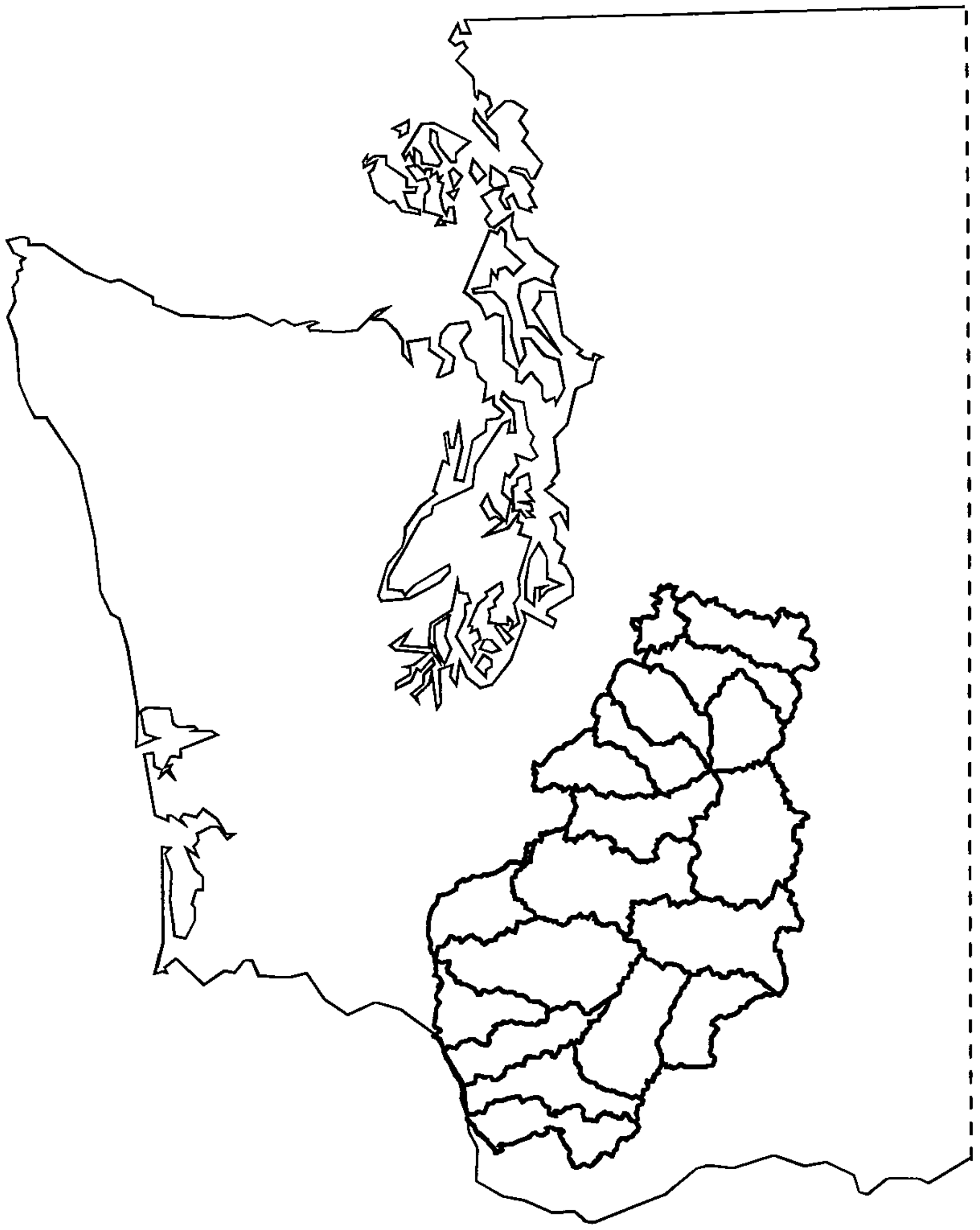

Figure 1. Map showing the west-side GIS analysis area subdivided into 19 WRIAs (Water Resource Inventory Areas).

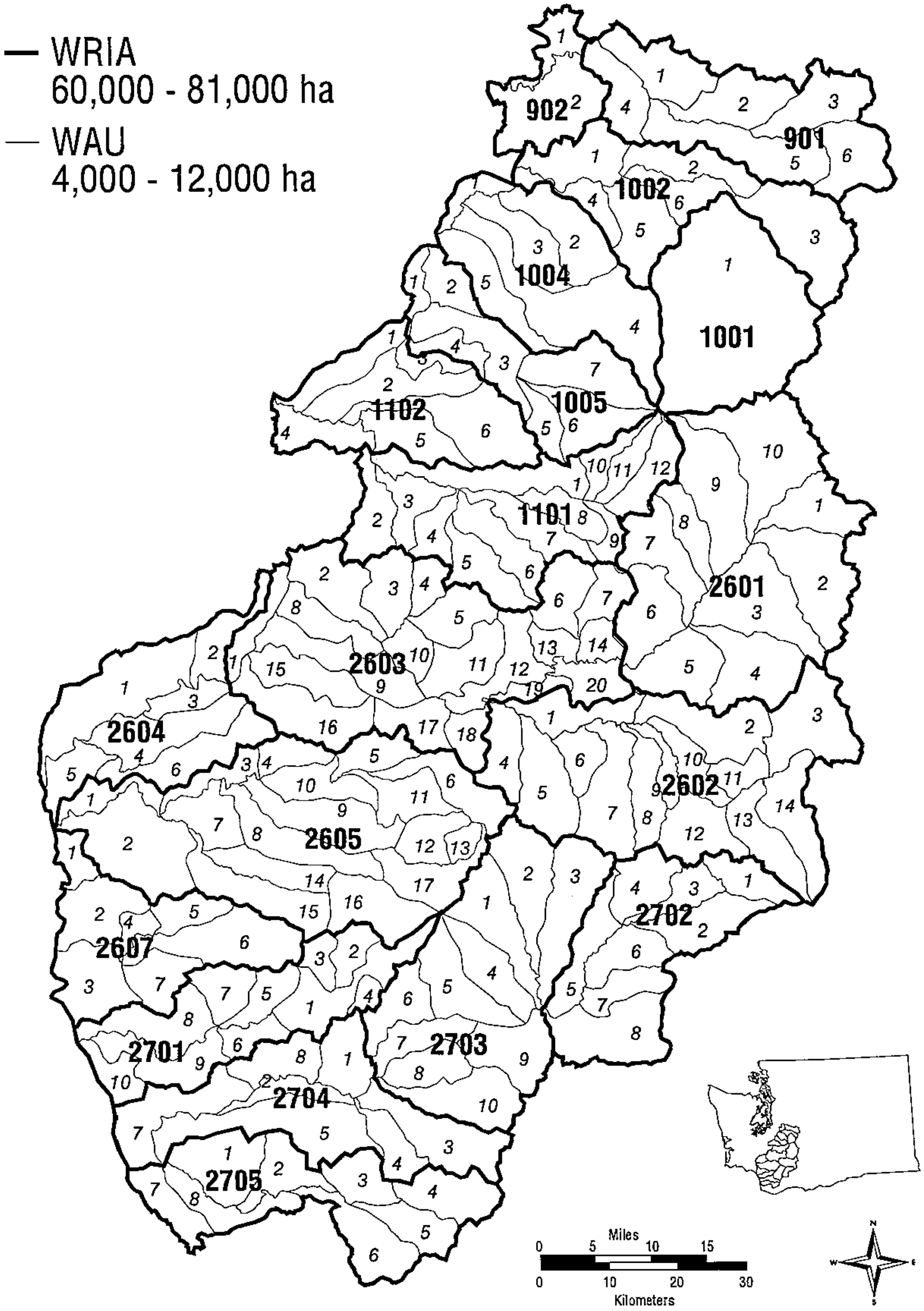

Figure 2. Map showing the west-side GIS analysis area subdivided into 119 WAUs (Watershed Administrative Units).

we screened out those sub-basins that were over 5,000 ft in elevation; located in Wilderness Areas, National Parks, National Monuments, or Wildlife Refuges; or that had been strongly influenced by the 1980 eruption of Mount St. Helens. We also eliminated sub-basins that had >20% old-growth forest or less than 20% in the 3-8 yr age class, as these landscapes would not be representative of intensively managed landscapes. This resulted in a total of 79 sub-basins to use in describing the range of landscape conditions occurring in intensively managed forest landscapes in the Cascade Range of southwestern Washington (Fig. 3).

Areal amount of each forest age class, the distribution of stand sizes, and 3 indices of landscape pattern (dominance, contagion, and fractal dimension) were calculated and evaluated to describe the range of available stand conditions in these 79 sub-basins. To establish a landscape pattern gradient, we used principal components analysis (PCA) to reduce information in the original variables to 4 independent components, and then grouped the sub-basins using cluster analysis on the principal components loadings. We then used discriminant function analysis to test the strength of our classification.

PCA explained about 82% of the total variance in the data set. Loadings on Component 1 (35.4% of variation explained) showed that sub-basins differed primarily along an urban/patch complexity gradient characterized by the amount of "other" land-type, a lack of old growth, and patch shape complexity (fractal dimension). Component 2 (20.5%) further differentiated sub-basins along a clearcut gradient based on the dominance of stands 3-8 yr of age. Component 3 (13.5%) represented a patch-clumpiness gradient based on the contagion of forest types, or the spatial complexity of the juxtaposition of types. The 4th component (12.2%) of landscape pattern differentiated sub-basins according to the absence of stands in the 10-20 yr age class.

We used a k-means cluster analysis algorithm to assign landscapes to groups based on loadings for the 4 principal components described above. Discriminant analysis showed that the resulting 5 groups were significantly separable; all observations but 1 were classified correctly. The 5 groups represent different landscape configurations and establish a gradient of landscape pattern resulting from differing intensities of fragmentation and varying natural and cultural influences. In

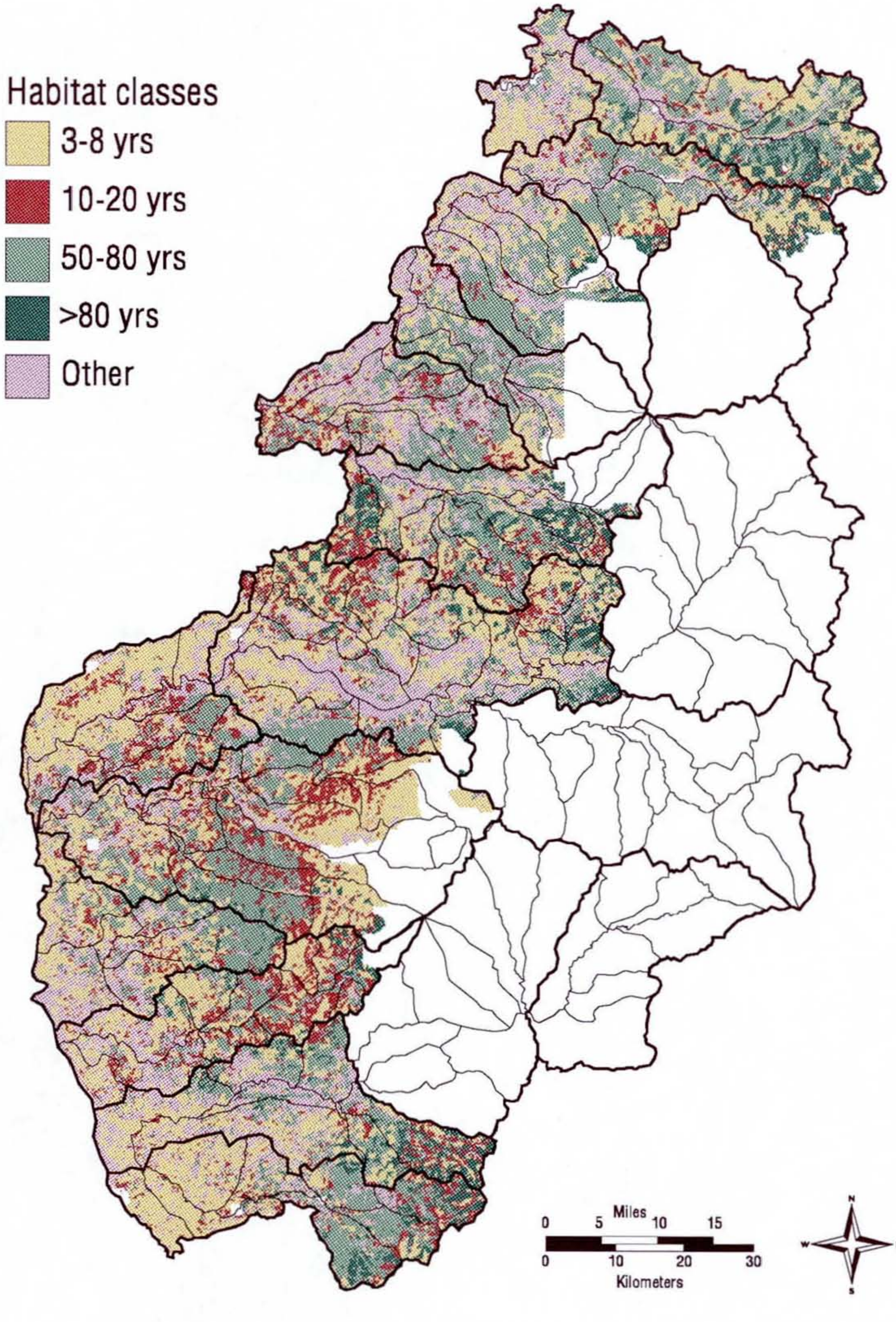

Figure 3. Map showing the west-side GIS analysis area classified into 100 x 100 m pixels in 5 habitat classes. Reserves, high-elevation areas, and watersheds dominated by old-growth forests have been screened out.

other words, these groups represent the range of landscape structure types occurring in intensively managed forest landscapes at low to mid-elevations in the Douglas-fir/western hemlock forest zone of western Washington. We visited 2-3 sub-basins in each group to field check the structure types as determined from satellite imagery and multivariate analysis; these visits confirmed the existence of the described landscape structure types.

Group 4 was composed of landscapes dominated by younger seral stages and high contagion or clumpiness. This landscape type was typical of managed forests in varying stages of regrowth, having approximately equal proportions of patches in the clearcut, young, and mature seral stages with little, if any, old growth. This suggests a managed landscape harvested in rotations. We focused our study-site selection efforts at landscapes within this group to ensure that the stands we selected for wildlife sampling would be located in intensively managed forest landscapes. These landscapes provided similar seral stages, an intensive management history, and a variety of patch sizes and configurations to choose from. With this stratification process, we were able to select study areas for sampling wildlife populations that were located in landscapes having similar management histories, thereby reducing the amount of landscape-scale variation on wildlife populations that is unrelated to forest management.

Selection of study areas--We then used our reclassified satellite imagery, topographic maps, orthophotos, and ground reconnaissance to locate our primary study areas within sub-basins classified in Group 4 in the area encompassed by the 19 WRIAs. We initially focused our efforts on Weyerhaeuser and DNR lands in the southern Cascades near the Columbia River. We excluded this area from further consideration, however, because we typically encountered only extensive areas containing very large patches dominated by only 1 or sometimes 2 of our target age classes. In addition, we found that many areas to the south were not suitable due to confounding environmental influences from the Yacolt burn, the 1980 eruption of Mount St. Helens, or high amounts of residual old growth.

After several months of reconnaissance, we decided to locate 20/24 study sites on the Kapowsin and Buckley tree farms on land owned and managed by Champion International in southern Pierce County. Among these 20 study sites, 5 are in Township 17N Range 5E, 1 is in T17N R6E, 7 are in T18N R5E, 4 are in T18N R6E, and 3 are in T19N R7E. The remaining 4 stands are located in T16N R5E of the Vail Tree Farm on land owned and managed by Weyerhaeuser (Fig. 4). We chose these study areas for a number of reasons: the landowners were extremely cooperative and helpful, and were clearly interested in participating in the study; the area had been entirely cut over in the last 70 yrs or so, and is virtually devoid of residual old growth; there is a great deal of heterogeneity in stand composition and juxtaposition on these tree farms; all of the target age-classes are represented; much of the area is fragmented into many relatively small patches; and, lastly, it is only about 1 hour's drive from both Seattle and Olympia, which kept logistical constraints and travel costs to a minimum. We sampled a total of 24 stands: 6 replicates in each of 4 structure classes in stands ranging in size from 80-200 ac.

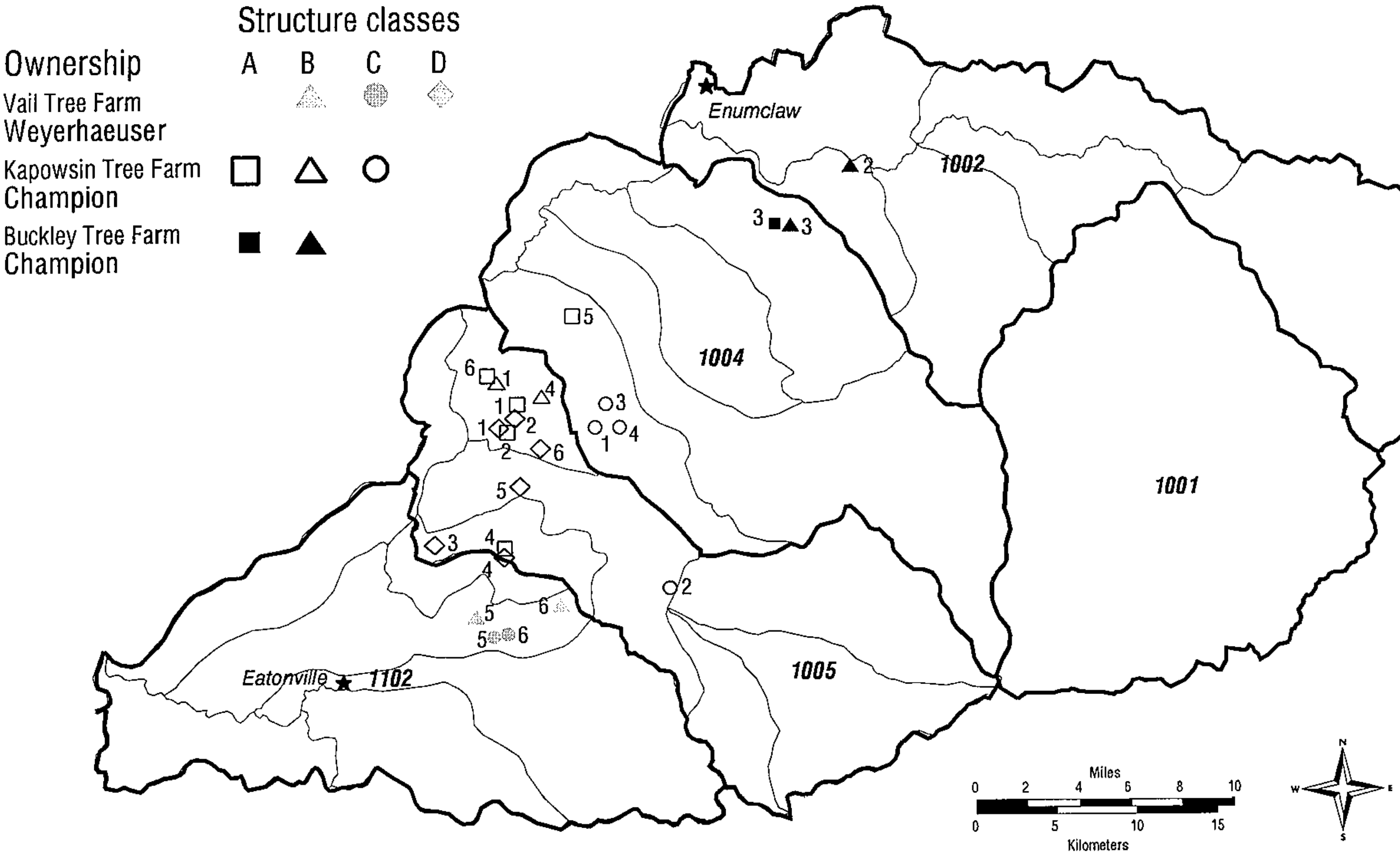

Figure 4. Map showing the location of the 24 study sites, their structure classes, and the WRIA and WAU in which each was located. Structure class A = clearcut, B = pre-canopy, C = closed-canopy, and D = harvest stage.

DESCRIPTION OF THE 4 STRUCTURE CLASSES

General Selection Criteria

No entry for 3 years (thru December 1995)

Within ½ mile of road; year-round access to sites

Avoid stands with riparian zones larger than headwater creeks (i.e., no large riparian zones with distinctive vegetation)

Must have a 75- x 75-m area where target structure is present for pitfall grid

Stands should contain Douglas-fir as the dominant species

Bogs/marshes OK if minor component of stand

Steep slopes OK as long as site is not excessively drained or too steep for effective pitfall trapping

Roads OK as long as contiguous area is present for pitfall grid

Elevation <3,000 ft

Structure class 1 - Clearcut Stage

Site characteristics at the initiation of sampling include the following:

Age: 2-3 yr since cutting

Tree Height: Herb stratum; seedlings generally <3 ft tall

Reproduction: Planted, or natural regeneration in progress

Other Species Present: Weedy invaders, red alder

Landscape Context: Adjacent stands at least 5 yr older

Target Stand Structure: Regeneration unit with only herb and shrub layers present (Fig. 5a).

Structure class 2 - Pre-canopy Stage

Age: 12-20 yr

Tree Height: Mid-canopy approx. 20-30 ft tall

Silvicultural Treatment: Pre-commercial thinning has occurred in stand within the last 5 yr

Other Species Present: Red alder, big-leaf maple, shade tolerant conifers; herbs and shrubs present in lower strata

Landscape Context: Adjacent stands recently clearcut or with fully developed tree canopy

Target Stand Structure: Canopy closure is mid-way between clearcut and a fully closed canopy. Lower branches on conifers dead or dying. Light interception at ground level is high; light reaches ground between trees. Low to moderate amounts of slash resulting from thinning operation may be present (Fig. 5b).

Structure class 3 - Closed-canopy Stage

Age: 30-40 yr

Tree Height: Full canopy height

Silvicultural Treatment: Not yet commercially thinned, history of pre-commercial thinning preferable, but not required

Other Species Present: May be some residual alders or maples in patches. Little or no herbs or shrubs present

Landscape Context: Adjacent stands in any other stand structure

Target Stand Structure: Stand is densely stocked with a wide range of stem diameters. Light interception within stand is low. Small snags and forest-floor woody debris common in stand (Fig 6a).

Structure class 4 - Harvest Stage

Age: 50-70 yr

Tree Height: Full canopy height

Silvicultural Treatment: Stand has been commercially thinned, history of pre-commercial thinning preferable, but not required

Other Species Present: Herb and shrub layers re-established, salal, Oregon grape, and Vaccinium typically present; vine maples occur in openings, sword fern in moist sites

Landscape Context: Adjacent stands in any other stand structure

Target Stand Structure: Uniform stem diameters; trees widely spaced with a stocking level of approx. 100-150 trees per acre. Canopy closed, but moderate amounts of light are filtering into stand (Fig 6b).

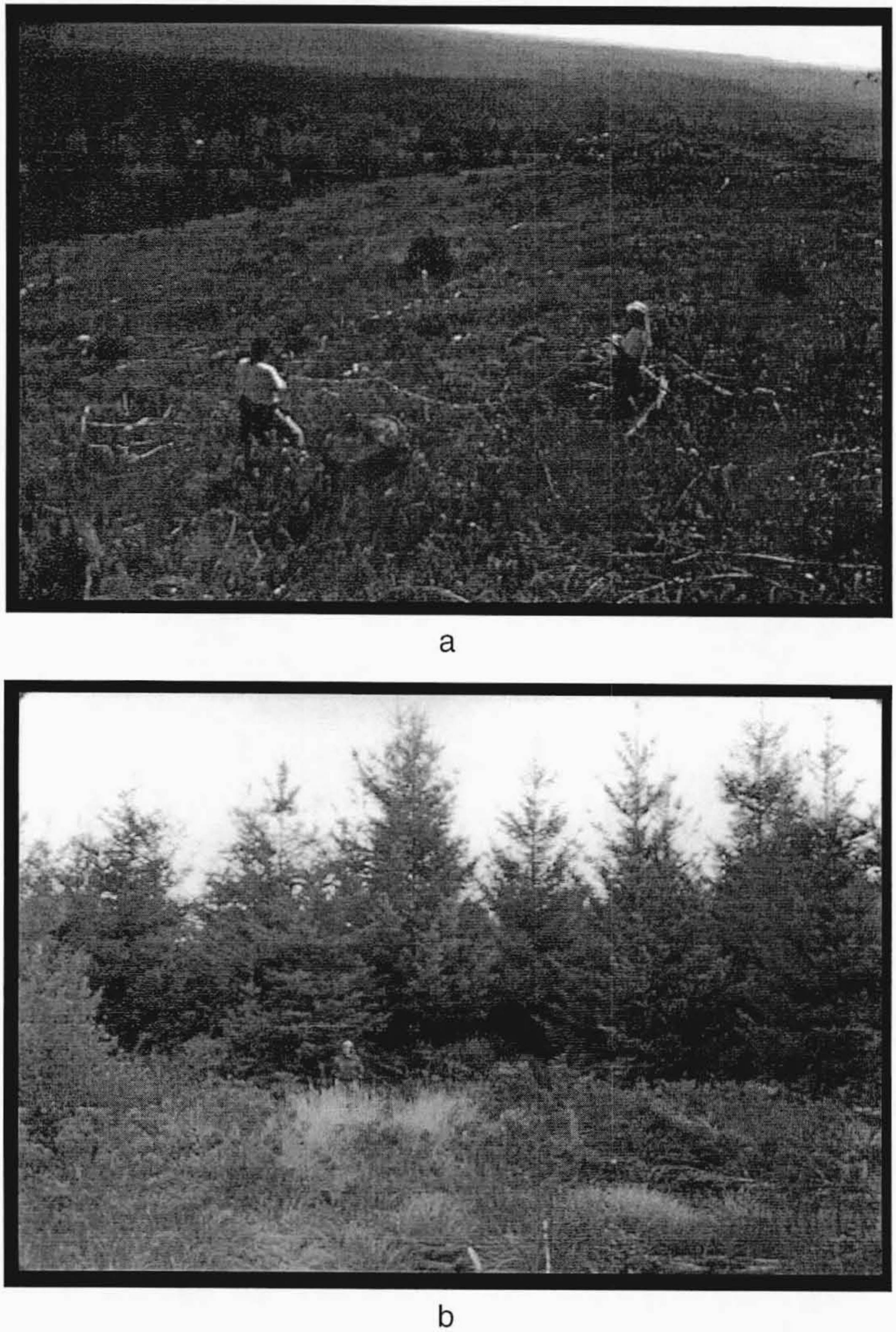

Figure 5. Photographs of representative stands in each structure class: (a) clearcut and (b) pre-canopy.

Figure 6. Photographs of representative stands in each structure class: (a) closed-canopy and (b) harvest.

EAST-SIDE STUDIES

Development of GIS database--Old-growth coniferous forests have been replaced by second-growth, "managed" forests over large areas of the Pacific Northwest. Subsequent harvest has created a mosaic of forest patches of varying size and age within a matrix of closed-canopy forest (<90 years). To evaluate the consequences of these changes on species diversity and population dynamics, the pattern and extent of habitat fragmentation on such landscapes must be understood. The analysis of the disturbance gradient and context of stands for the watersheds was an important part of watershed and study site selection. We wished to work in watersheds that ranged from moderate to high degrees of disturbance for the region, and select stands that were representative of current practices of forest management.

We selected the managed forests of northeastern Washington (Stevens and Pend Oreille counties) for study. The composition of these second-growth coniferous forests varies with local conditions, but the principal tree species are grand fir (*Abies grandis*), western larch (*Larix occidentalis*), lodgepole pine (*Pinus contorta*), Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and western redcedar (*Thuja plicata*). We acquired Landsat TM data for the region taken on 2 August 1991 (Fig. 7). The data had a pixel resolution of 25 x 25 m on a Universal Transverse Mercator projection. We used the 3 visible bands (1-3) and 3 infrared bands (4, 5, and 7) in a hybrid supervised-unsupervised classification using VICAR (Jet Propulsion Laboratory, Pasadena, California, USA) and ERDAS (Earth Resources Data Analysis Systems, Atlanta, Georgia, USA) image processing software on a mainframe computer. First, we selected 25 areas that represented a range of habitat conditions by viewing bands 3, 4, and 5 with ERDAS. An 80 x 100 pixel window of relatively homogeneous habitat was sampled from each area. VICAR was then used to construct spectral signatures for each window using all 6 bands. The signatures were then combined for the 25 areas, and any that were redundant or ambiguous were removed based on a similarity index and on the magnitude of values in the variance-covariance matrix associated with each signature. Next, we randomly selected 10 areas (100 x 100 pixels) from the entire scene and classified them using the set of signatures and a maximum likelihood classification algorithm in VICAR. This algorithm examines the

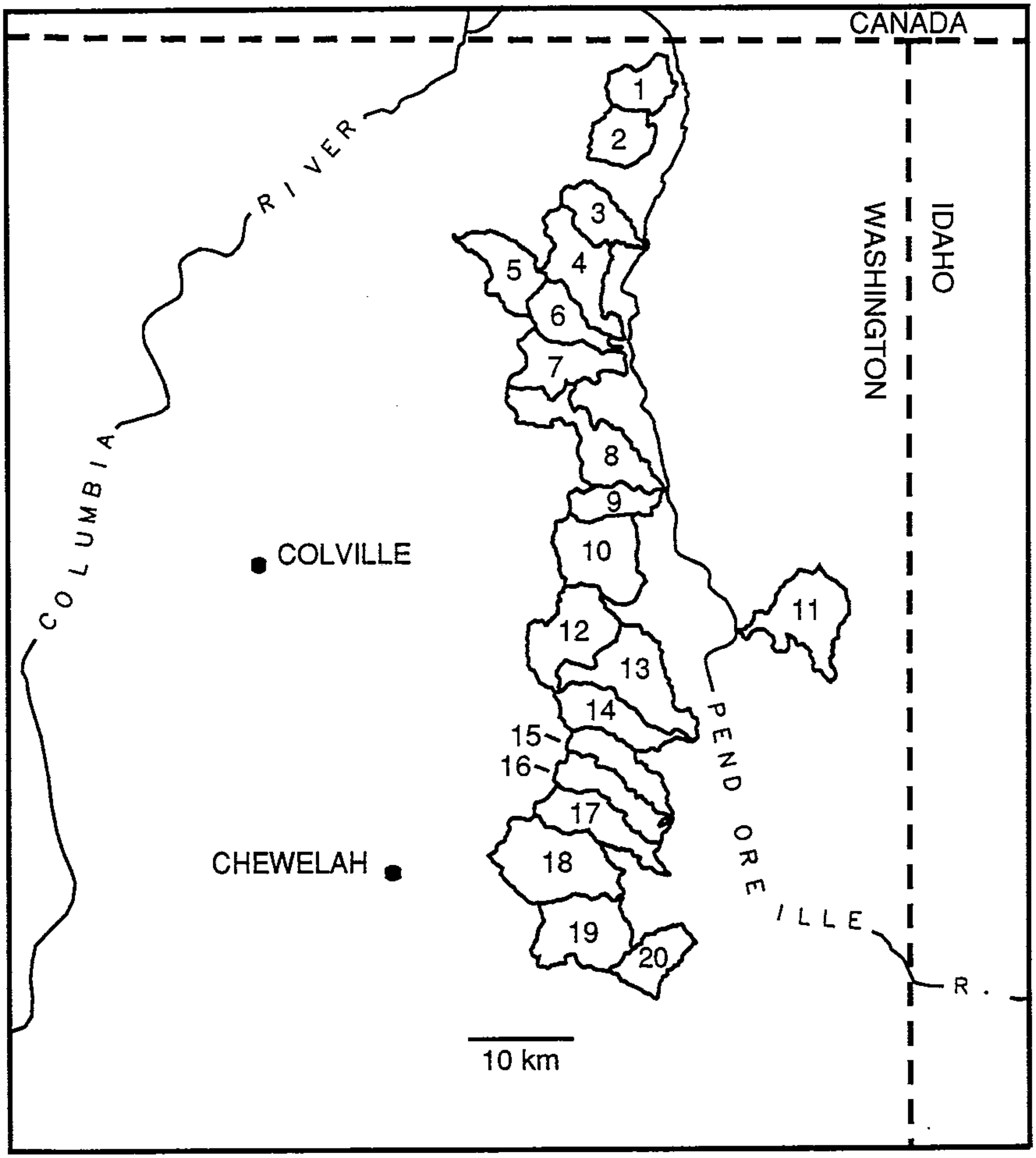

Figure 7. Location of the 20 watersheds in the Selkirk Mountain range of northeastern Washington. Numbers correspond to the listing of watersheds in Table 1. The figure borders indicate the boundaries of the Landsat TM scene.

reflectance values for each pixel (using all 6 bands) and classifies the pixel based on its similarity to the available spectral signatures. We examined the resulting classification and constructed new signatures for areas that could not be classified because they deviated significantly from the initial signatures. This process of classifying random areas was repeated 3 more times until a maximum of 1% of the sampled pixels were unclassified or until none of the unclassified pixels were in meaningful clumps that could be further sampled. This final set of spectral signatures was used to classify the entire scene with a maximum likelihood classifier.

The final classification had 137 classes. We transferred the classified data to a personal computer and used IDRISI (Clark University, Worcester, Massachusetts, USA) to group similar classes into broad habitat categories. This grouping eliminated most of the natural spatial heterogeneity within patches. By ground-truthing and comparisons with orthophotos, we found that we could classify the following habitats (we discuss misclassification below):

1. cleared areas (clearcuts, basal area retention cuts, roads, and agriculture)
2. disturbed, open-canopy forests (e.g., commercial thinning)
3. regenerating, closed-canopy forests (<30 yr old)
4. mature, closed-canopy forests (>30 yr old)

Water, clouds, and developed areas also were identifiable, but were ignored in the analysis. Deciduous forest comprised a very small percentage of the total area. Because this habitat was generally associated with regenerating forest, it was lumped with that class. Cleared areas were primarily clearcuts and basal area retention cuts for the region we examined.

We obtained a map delineating the watersheds which contain 1st order streams in the region (Kim Clarkin, USDA Forest Service, unpubl. data). We selected 20 watersheds that were managed primarily for timber production and that varied in their degree of forest-harvesting activity (Fig. 7). Although some natural fragmentation occurred in these watersheds, most habitat disturbance was from timber harvest-related activities. In some cases, the watersheds were very large and we used sub-watersheds for analysis. The watersheds were between ~2,000 and 7,000 ha (Table 1). The watershed is an appropriate unit for studying anthropogenic fragmentation because

Table 1. Characteristics of the 20 watersheds in northeastern Washington.

Watershed		Area (ha)	Perimeter (km)	R[a]	M	D	C	R	M	D	C	Dominance	Contagion
1.	Fence	2638	22.2	18	4	9	16	10.3	75.5	2.1	12.1	0.60	10.35
2.	Flume	2836	22.0	24	1	3	15	7.3	78.1	0.6	13.9	0.70	10.40
3.	Sweet	2866	25.9	35	8	0	26	23.0	62.2	0.0	14.8	0.47	9.91
4.	Jim Cedar	4665	41.2	43	16	15	38	25.1	59.0	2.8	13.1	0.36	9.70
5.	Smackout	3418	30.2	32	11	29	23	9.4	59.4	11.3	19.9	0.29	9.66
6.	Little Muddy	2937	27.3	28	21	32	39	12.6	49.9	13.9	23.5	0.16	9.46
7.	Muddy	3534	34.8	38	15	42	37	9.8	66.6	10.5	13.2	0.38	9.72
8.	Lost	5741	52.4	59	13	44	33	10.4	69.8	7.0	12.7	0.45	9.90
9.	S Fork Lost	2383	24.9	37	6	7	14	16.6	73.3	1.3	8.7	0.59	10.02
10.	Ruby	5739	33.0	85	5	21	42	9.8	72.2	3.9	14.0	0.52	9.94
11.	Mill	5456	42.3	34	9	15	27	4.0	63.6	2.2	30.2	0.52	10.02
12.	Upper Tacoma	5402	35.8	55	7	6	28	10.0	71.3	0.5	18.2	0.58	10.07
13.	Tacoma	6275	45.5	95	6	26	52	11.7	74.9	3.0	10.4	0.58	10.00
14.	S Fork Tacoma	3958	35.2	56	3	14	20	8.0	80.4	2.6	9.0	0.70	10.17
15.	E Fork Small	3175	32.1	47	5	10	30	13.3	65.5	2.9	18.2	0.43	9.84
16.	Small	3544	38.3	33	10	10	23	7.1	54.3	1.3	37.4	0.44	9.89
17.	Winchester	4201	38.5	46	10	30	26	16.1	49.3	5.5	29.1	0.22	9.60
18.	N Fork Calispell	7160	38.5	85	26	44	45	11.8	57.9	5.9	24.3	0.31	9.78
19.	M Fork Calispell	5588	34.0	55	9	9	25	8.8	69.9	1.6	19.7	0.54	9.96
20.	S Fork Calispell	2982	25.3	26	3	9	17	13.8	71.6	2.6	12.0	0.53	10.00

[a] R = regeneration; M = mature, closed-canopy; D = disturbed canopy; C = clearcut.

timber management typically has been planned on this scale. We refer to each watershed by the name of its primary stream. For each watershed, we used its boundary to extract its pixels from the classified Landsat scene. We used a raster-to-vector conversion program (POLYVEC) in IDRISI to create habitat polygons for each of our classes. The individual class coverages were then combined using ARC/INFO (ESRI, Redlands, California, USA) with a ~2 ha (32 pixels) minimum polygon size. Errors in the alignment of polygon boundaries from different habitat classes resulted in slight changes in the area and perimeter of some polygons. An ARC/INFO coverage of the road network (Washington Department of Natural Resources) was overlaid on the habitat maps. Primary roads, which occurred in only 2 watersheds, were buffered by 50 m and classified as clearcut. High elevation areas occurring on the periphery of 2 watersheds were eliminated from the maps. A small section of agricultural and developed land on the edge of 1 watershed was also removed from consideration.

The conversion from raster to vector formats provided 3 advantages. First, the storage requirements for the vector data were substantially less and made it possible for us to conduct our analyses on personal computers. Second, the ARC/INFO coverages automatically provided a complete database of the forest patches including area, perimeter, and habitat type. Third, our interest is in the fragmentation resulting from forest harvest, and the conversion provided a simple way to reduce the naturally occurring spatial heterogeneity within patches. For example, a few pixels representing gaps in an otherwise closed-canopy coniferous forest would be included with the latter.

We determined the misclassification rate by field verifying the habitat class for 136 stands in 7 watersheds. A total of 12 (8.8%) stands were misclassified and most errors (83%) occurred between adjacent age classes (7 errors between regenerating and cleared, 2 between regenerating and closed-canopy, 2 between cleared and closed-canopy, 1 between disturbed and cleared). The location of stands on steep, north-facing slopes accounted for most errors. As an additional check, we conducted a discriminant function analysis to determine if the 3 predominant habitat classes (closed-canopy, cleared, and regenerating) could be correctly classified with 4 variables. We selected 12 stands of each habitat and counted the number of trees in 4 d.b.h. classes (4-10, 11-25, 26-50, and >50 cm) on several 576-m^2 plots in each stand. The number of plots varied with the size of the stand, so we calculated the mean number of trees per plot for each size class. The only classification errors were for 2 regenerating stands that were classified as cleared (6% error).

Watershed analysis--For each watershed, the number of patches and the proportion of total watershed area in each habitat class were determined. To examine the degree to which each watershed was dominated by 1 habitat type, we calculated a diversity index of dominance, D, using the proportion of area in each habitat class (P_i) (O'Neill and others 1988), as follows:

$$D = \ln \mathrm{N} + \Sigma P_i \ln P_i. \qquad (1)$$

This equation was summed over all N habitat classes (N = 4 for all watersheds). Higher values of D indicate greater dominance by a habitat class.

The amount of edge habitat produced by timber harvest could significantly affect vertebrate biodiversity. Previous studies have examined the relationship between the area and perimeter of forest patches by estimating fractal values (e.g., Krummel and others 1987; Ripple and others 1991) using the following regression equation:

$$\ln \mathrm{P} = \beta_0 + \beta_1(\ln \mathrm{A}^{½}) + \epsilon, \qquad (2)$$

where P and A are the perimeter and area of each patch, respectively; β_0 is the intercept; β_1 is the PA-fractal value; and ϵ is the residual error term (De Cola 1989). Theoretically the values of β_1 must be ≥ 1 and <2. These extreme values represent minimal and maximal amounts of perimeter per unit area, respectively. We estimated PA-fractal values for each watershed and habitat using Eq. 2. Then to determine if the fractal values differed among watersheds, we constructed 2 additional models for each habitat with the data from all watersheds. The first model added dummy variates (1 for a given watershed and 0 otherwise) to estimate the intercept for each watershed. The second model added cross-products terms (dummy variates × ln $\mathrm{A}^{½}$) to the previous model to estimate the slopes for each watershed. A significant partial F-test of the improvement provided by the 2nd equation over the 1st would indicate that the PA-fractal values differ among watersheds. For each regression equation, we examined the residuals and calculated Cooks' D as a measure of influence at a = 0.5 (Neter and others 1990).

Milne (1991) proposed another fractal index that should be more sensitive to patch dispersion. The index is based on a probability-density function of the number of pixels (m) of a particular habitat type in square sampling windows of increasing length (L) centered on each pixel of that habitat. This analysis is best conducted on raster maps and we converted our polygon maps back to this form. We calculated the frequency distributions for each habitat type in each watershed using window lengths of 3, 5, 7, 9, 11, 17, 23, and 25 pixels (i.e., ca. 150 to 625 m). Following Milne (1991), we calculated and logarithmically transformed the 1st moment of each distribution, $p(m,L)$, and then regressed these values against the logarithm of L. The $p(m,L)$-fractal is the slope of the regression line. The intercept, k, is expected to relate the predominance of a pattern to the map extent in which it occurs, and we examined the relationship between k and $p(m,L)$-fractal values.

The re-rasterized map also provided a convenient way to estimate contagion for each watershed (O'Neill and others 1988). This diversity index measures the degree to which habitat types are clumped on a map. Large values reflect large, contiguous patches, whereas small values correspond to a map divided into many small patches. We used equation 2 in O'Neill and others (1988) to calculate contagion.

We developed an additional method of assessing fragmentation based on a line-transect. We considered 2 questions from the perspective of an animal traveling in a straight line across a watershed. How many times will the animal cross a habitat boundary? What are the probabilities of making a transition from 1 habitat type to any other? To address these questions, we developed a computer program for randomly placing 1-km line-transects onto our habitat maps. The 1-km length might correspond to the dispersal distance of an amphibian or small mammal. For each watershed, we generated 200 random lines in a fixed sampling frame (15,400 x 12,500 m) that could accommodate the largest watershed. We overlaid the lines onto the habitat map with ARC/INFO and removed any that were outside or crossed the watershed boundary. Using a database manager (dBASE IV, Borland, Scotts Valley, California, USA) and a custom program, we tallied the number of transitions per line and the frequency of each type of transition (e.g., between habitat1 and habitat 2). Preliminary trials indicated that the mean values of these variables stabilized after about 20 lines were sampled. Consequently, we repeated the procedure if <20 lines fell within a watershed. This approach allowed for increased sampling with watershed area (N = 20 for the smallest watershed to N = 62 for the largest).

Several of the variables that we examined were intercorrelated. Consequently, we conducted principal components analysis (PCA) on the correlation matrix to describe relationships among the descriptive variables and patterns of variation across watersheds.

Characterization of watersheds-The watersheds varied in size, perimeter (Table 1), and shape (Fig. 7). From ~20-50% of the area in each watershed has been converted from mature, closed-canopy forest to other forest types. The number of patches and proportions of each category differed considerably among the watersheds indicating differences in the extent of fragmentation. Dominance was highly correlated with contagion ($r = 0.94$, $P < 0.0001$) and the proportion of total area in mature, closed-canopy forest ($r = 0.91$, $P < 0.0001$). As expected with such diversity measures, different proportions of the 4 habitat categories provided similar measures of dominance (e.g., Ruby and Mill, Table 1). Contagion also was correlated with proportion of total area in mature, closed-canopy forest ($r = 0.88$, $P < 0.0001$).

Among 80 possible regressions, 78 could be conducted to estimate PA-fractal values for each watershed and habitat type. Of these only 2 were nonsignificant ($0.1 > P > 0.05$) due to small sample size (N = 3 in both cases). The R^2 values were generally high and averaged between 87-98% for the 4 habitats. Cook's D statistic indicated that 16 of 19 (84%) regressions for the closed-canopy forest had observations that strongly influenced the regression line. These influential points were always the largest patches in the watershed and were usually substantially larger than the next smaller patch. The 59 regressions for the other habitats were not as affected by influential data points (largest patches were influential 9 times and smallest patches were 5 times). The estimates of PA-fractal values for the 20 watersheds showed considerable variation and overlap for the 4 habitats (Table 2).

Table 2. Summaries of PA-fractal estimates from regressions and p(m,L)-fractal values for each watershed by habitat.

	PA-fractal values					p(m,L)-fractal values				
Habitat	$\overline{X}$	SE	N	min	max	$\overline{X}$	SE	N	min	max
Regeneration	1.32	0.02	20	1.11	1.44	1.59	0.02	20	1.35	1.80
Mature, closed-canopy	1.46	0.02	19	1.34	1.60	1.91	0.01	20	1.81	1.96
Disturbed	1.42	0.04	19	1.15	1.78	1.33	0.03	19	1.13	1.64
Clearcut	1.28	0.03	20	1.14	1.51	1.67	0.03	20	1.35	1.85

Further analyses showed few differences in PA-fractal estimates between watersheds, but significant differences between habitat types. Tests of the hypothesis that the slopes were the same for all watersheds could not be rejected for closed-canopy (F = 1.1, df = 18, 151, P = 0.33), clearcut (F = 1.44, df = 18, 564, P = 0.11), and disturbed (F = 1.25, df = 17, 345, P = 0.22) habitats. For these habitats, the fractal values (± 1 SE) estimated across all watersheds were 1.45 ± 0.02 for closed canopy (R^2 = 0.98, df = 1, 188, F = 8158.6, P < 0.0001), 1.26 ± 0.02 for clearcut (R^2 = 0.92, df = 1, 583, F = 6753.3, P < 0.0001), and 1.37 ± 0.03 for disturbed (R^2 = 0.86, df = 1, 380, F = 2354.4, P < 0.0001). All assumptions concerning the residuals were met and no influential observations were detected. Additional partial F-tests indicated that the PA-fractal estimates differed significantly (P < 0.0005) among the 3 habitats. We did not examine differences among watersheds for regeneration patches because of the lack of variation in the PA-fractal values for the other 3 habitats.

The $p(m,L)$-fractals were somewhat less variable among watersheds than the PA-fractals (Table 2). Values for closed-canopy patches, in particular, fell within a very restricted range. Ranking of the mean values for the 4 habitats corresponded to the mean proportion of total watershed area in each habitat type. The $p(m,L)$-fractal values for each habitat were highly correlated with the proportion of that habitat in a watershed (closed-canopy: r = 0.89, P = 0.0001; clearcut: r = 0.59, P = 0.006; regeneration: r = 0.85, P = 0.0001; disturbed: r = 0.76, P = 0.0002). The p(m,L)-fractal values were correlated negatively with the intercept term k for all habitat types (closed-canopy: r = -0.83, P = 0.0001; clearcut: r = -0.98, P < 0.0001; regeneration: r = -0.73, P = 0.0003; disturbed: r = -0.65, P = 0.002).

The line-transect analysis indicated differences in the relative fragmentation of the watersheds. The mean number of transitions per transect, the proportion of transects without transitions, dominance, and contagion were all highly intercorrelated ($P < 0.001$). Transition probabilities between habitats were highly variable reflecting differences in the frequency of occurrence and spatial distribution of habitat types among watersheds (Table 3). Most transitions were between closed-canopy and regeneration patches and between closed-canopy and clearcut patches (46.0 ± 4.2 and 27.9 ± 3.6% [$\overline{X}$ ± 1 SE], respectively). The transition between mature and disturbed patches accounted for another 10.1 ± 2.2%. The 3 other types of transition accounted for a total of about 16%.

We conducted an exploratory PCA with all variables except the PA-fractals. Although this model was overspecified, we used it to determine redundant variables that could be eliminated, and then we conducted a 2nd analysis. Although additional variables could have been dropped, we present the analysis for 19 variables to demonstrate the similarities of some variables. The 1st 3 components explained 71% of the variance and were the most informative in describing fragmentation in these watersheds (Table 4). The remaining components each explained <9% of the variance.

The watersheds were well distributed on the 1st principal component (PC 1; Fig 8). PC 1 corresponds to a gradient of increasing proportion of area in clearcut, $p(m,L)$-fractals for clearcuts, and number of mature, disturbed, and clearcut patches, and decreasing proportion of area in closed-canopy forest (Table 4). These changes in area are mirrored by a reduction in dominance and contagion, an increasing number of transitions per transect, and a reduction in the probability of transects without transitions. The types of transitions also vary along this gradient with decreasing probability of changing between regeneration and closed-canopy forest, and increasing probability of transitions between disturbed and closed-canopy, and disturbed and clearcut patches. Two watersheds (Flume and Little Muddy, 2 and 6 in Fig. 8 and Fig. 9A, D, respectively) on opposite sides of PC 1 show the changes in number of patches, areal proportion of each type, and dominance, which correspond to the increasing probability of transitions on a transect and the increased number of transitions per

Table 3. Results of the line-transect analysis for the 20 watersheds in northeastern Washington. Probability of transition between habitats is the proportion of the total number of transitions occurring between 2 habitat types.

		Transitions per transect			Transects without transitions (%)	Probability of transition between habitats					
Watershed			SE	N		$R^a \leftrightarrow M$	$R \leftrightarrow D$	$R \leftrightarrow C$	$M \leftrightarrow D$	$M \leftrightarrow C$	$D \leftrightarrow C$
1.	Fence	1.6	0.6	20	55.0	53.1	0.0	21.9	0.0	25.0	0.0
2.	Flume	1.6	0.4	29	51.8	78.3	0.0	8.7	8.7	4.3	0.0
3.	Sweet	2.7	0.5	22	18.2	65.0	0.0	18.3	0.0	16.7	0.0
4.	Jim Cedar	3.0	0.4	42	19.1	59.7	2.4	20.2	8.9	7.3	1.6
5.	Smackout	3.9	0.5	29	10.3	19.5	1.8	7.1	5.3	50.4	15.9
6.	Little Muddy	4.8	0.7	23	13.0	27.3	1.8	15.5	16.4	19.1	20.0
7.	Muddy	3.6	0.4	30	13.3	28.0	2.8	1.9	41.1	13.1	13.1
8.	Lost	4.2	0.5	43	21.0	44.4	1.1	3.9	22.8	17.2	10.6
9.	S Fork Lost	3.1	0.5	24	16.7	66.2	0.0	1.4	9.5	23.0	0.0
10.	Ruby	2.8	0.4	44	22.8	52.8	0.0	3.2	17.6	24.8	1.6
11.	Mill	2.8	0.4	42	19.0	35.3	0.0	0.9	3.4	58.6	1.7
12.	Upper Tacoma	1.7	0.4	39	51.3	55.9	0.0	11.8	0.0	29.4	2.9
13.	Tacoma	3.9	0.4	41	12.2	57.0	1.3	3.8	13.9	21.5	2.5
14.	S Fork Tacoma	2.4	0.4	38	34.2	73.1	0.0	0.0	8.6	17.2	1.1
15.	E Fork Small	4.0	0.5	29	6.9	56.9	0.0	12.9	0.0	30.2	0.0
16.	Small	2.8	0.4	34	11.7	14.9	0.0	8.5	3.2	63.8	9.6
17.	Winchester	3.7	0.5	30	10.0	23.4	0.0	17.1	16.2	32.4	10.8
18.	N Fork Calispell	3.6	0.3	62	11.3	30.2	1.4	12.6	15.3	28.4	12.2
19.	M Fork Calispell	2.4	0.4	44	24.9	32.7	0.0	7.5	3.7	45.8	10.3
20.	S Fork Calispell	2.7	0.6	22	18.2	45.8	5.1	5.1	6.8	28.8	8.5

[a] R = regeneration; M = mature, closed-canopy; D = disturbed canopy; C = clearcut.

Table 4. Eigenvalues, variance explained, and correlations of the first 3 principal components (PC) for 19 watershed variables.

Variable	PC 1	PC 2	PC 3
Eigenvalue	7.0	3.4	3.0
Variance explained (%)	36.8	17.9	16.0
Number of patches			
Regeneration (R)	0.20	-0.17	0.50
Mature, closed-canopy (M)	0.82	0.19	0.05
Disturbed (D)	0.79	-0.10	0.50
Clearcut (C)	0.57	0.11	0.47
Proportion of total area			
Regeneration	0.05	0.92	-0.06
Mature, closed-canopy	-0.84	-0.22	0.47
Clearcut	0.63	-0.30	-0.65
Dominance or contagion	-0.90	-0.30	0.19
p(m,L)-fractal values			
Regeneration	0.05	0.93	-0.11
Disturbed	0.35	-0.43	0.42
Clearcut	0.49	-0.32	-0.27
Transitions per transect ($\bar{X}$)	0.81	0.15	0.23
Transects without transitions (%)	-0.76	0.10	0.04
Probability of transition between habitats			
R ↔ M	-0.82	0.29	0.33
R ↔ D	0.37	0.27	0.33
R ↔ C	0.04	0.71	-0.43
M ↔ D	0.50	-0.03	0.04
M ↔ C	0.27	-0.61	-0.40
D ↔ C	0.83	-0.09	-0.21

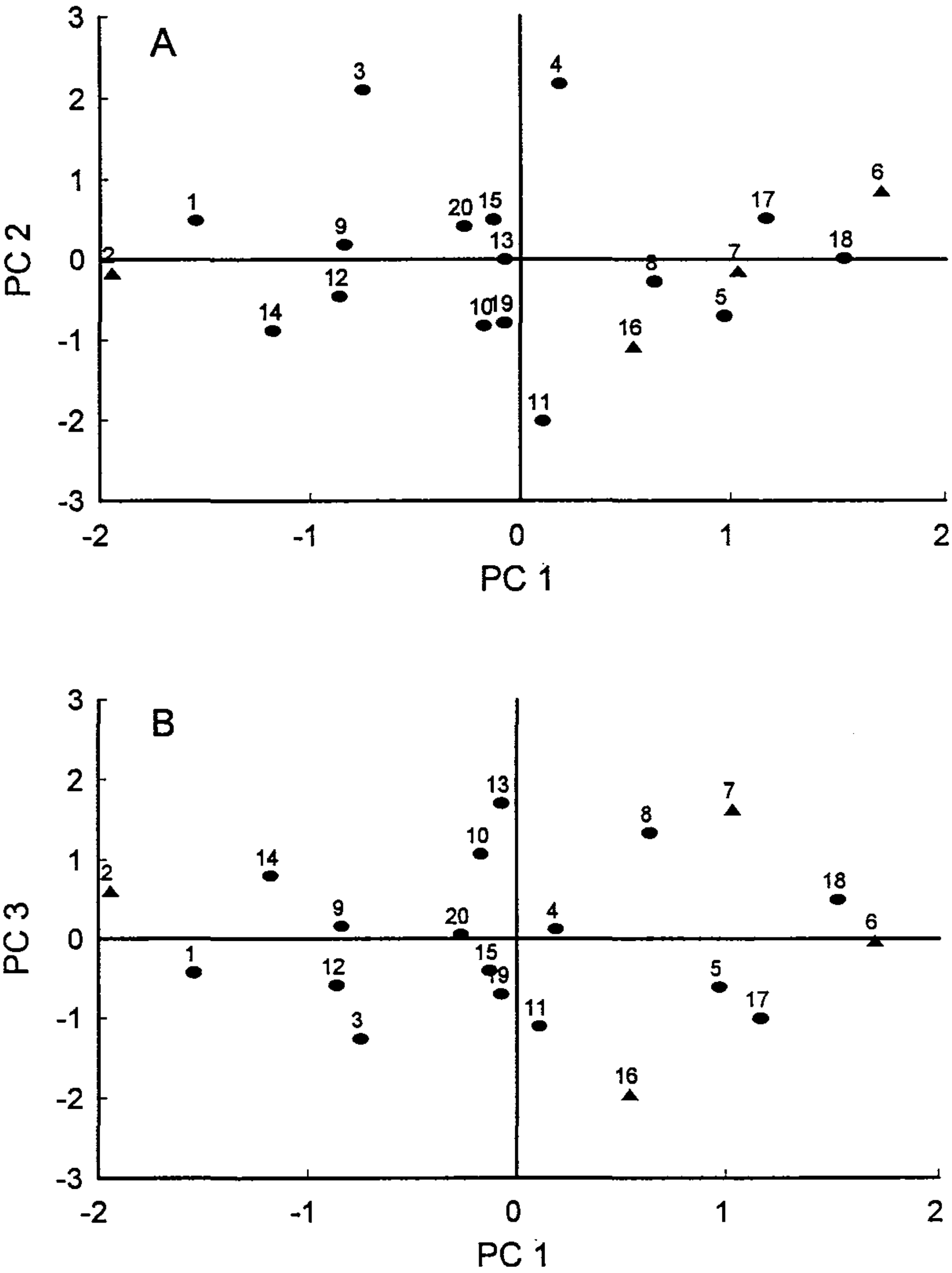

Figure 8. Positions of the 20 watersheds on A) the 1st 2 principal components and B) the 1st and 3rd principal components. PC1 reflects increasing intensity of fragmentation, PC2 corresponds to differences in the extent of regeneration forest, and PC3 corresponds to proportion and number of clearcut patches. Numbers correspond to the listing of watersheds in Table 1. The positions of the 4 watersheds illustrated in Fig. 9 are denoted by ▲.

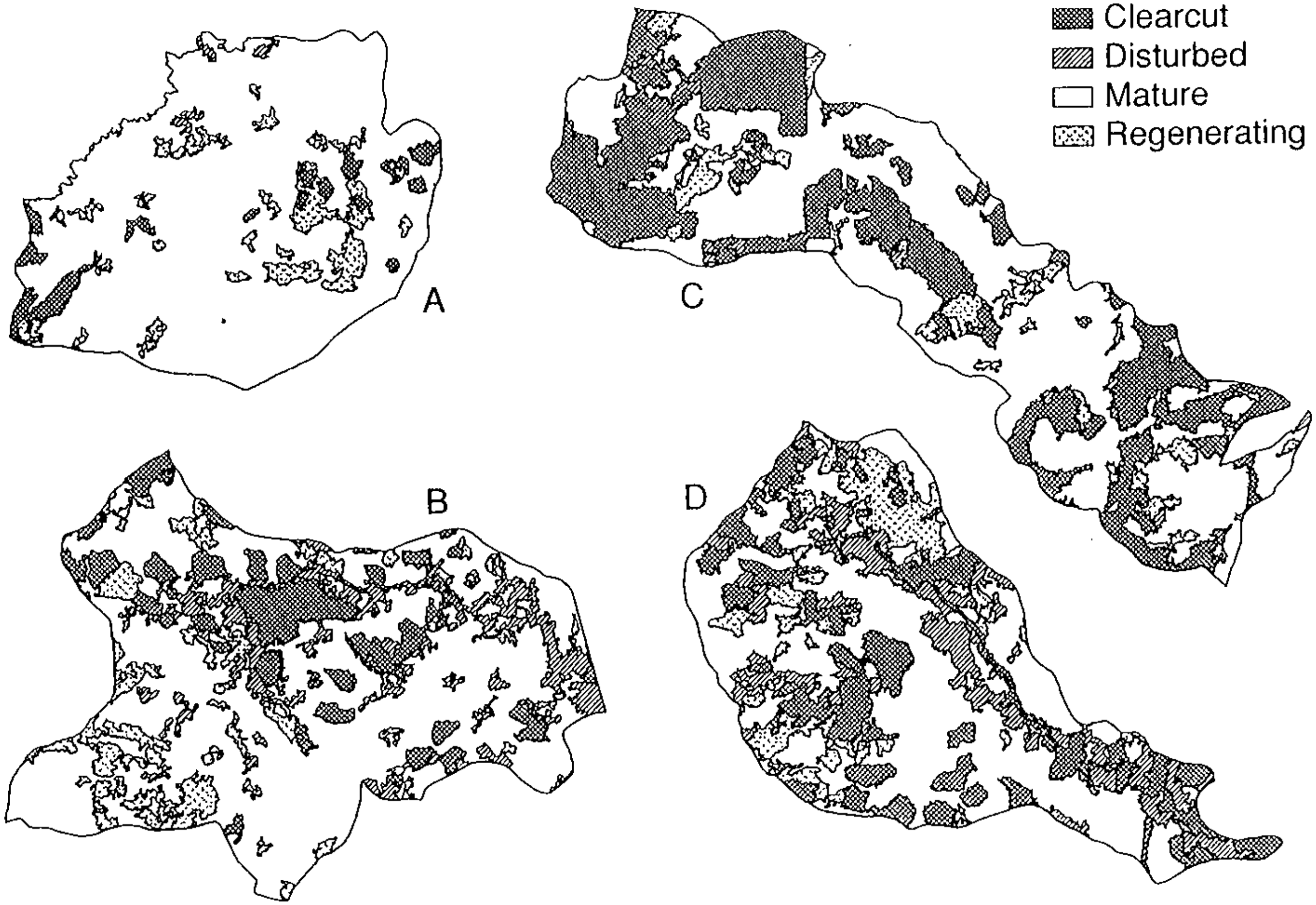

Figure 9. Representative watersheds illustrating the degree and type of fragmentation observed. A = Flume (2), B = Muddy (7), C = Small (16), D = Little Muddy (6). Numbers in parentheses correspond to labels in Figs. 7 and 8.

transect. A large proportion of Flume is in contiguous forest whereas the closed-canopy forest of Little Muddy is fragmented into patches and accounts for a much lower proportion of total area.

Increases on PC 2 correspond to increased area and $p(m,L)$-fractals for regeneration, increased probability of transition between regeneration and clearcut patches, and decreased probability of transition between closed-canopy and clearcut patches. Increasing values on PC 3 are associated with increasing numbers of regeneration, disturbed, and clearcut patches, but a reduction in the proportion of area in clearcut. The proportion of transitions between regeneration and closed-canopy and between closed-canopy and disturbed both increase on this component, but the proportion between closed-canopy and clearcut decreases. Two watersheds (Small and Muddy) are positioned on opposite sides of PC 2 (16 and 7 in Fig. 8 and Fig. 9B, C, respectively). Small has fewer clearcuts than Muddy, but they are larger in size and cover proportionately more area. As a consequence, the proportion of transitions between closed-canopy and clearcut is lower in Small than in Muddy. These differences are important because they point to differences in the way that fragmentation has taken place.

Selection of study sites--In 1993, based on the above analysis to characterize the size, shape, and context of stands of different forest types in watersheds in northeastern Washington (Stevens and Pend Oreille Counties), we selected 7 watersheds that ranged from moderate to high degrees of disturbance. For the 7 watersheds, we mapped all stands that met our size (i.e., 12-15 ha and >36 ha) and forest type (i.e., clearcut, regeneration, and mature closed canopy) criteria. Six stands of each size and forest type were selected for a total of 36 stands (Table 5).

The vegetative characteristics of each forest type are detailed in the section entitled Analyzing Landscape-Scale Influences on Wildlife Habitat Relationships. The mature, closed-canopy forest was about 60-85 years old. Clearcuts were <6 years since harvest. We selected clearcuts that had some standing live or dead trees (i.e., basal area retention cuts) because this was a typical forest management practice when the sites were selected.

Table 5. Name, stand type, size, and legal description for 36 study sites. CUT = clearcut; REG = regeneration; MAT = mature, closed-canopy forest.

Landscape site	Size class	Legal description	Ownership
		E Fork Small	
CUT 12W	30-50	T33N R42E S½ of NW¼ of Sec 12	FS-Newport
MAT 6	90+	T33N R43E E½ of Sec 6	FS-Newport
MAT 7N	30-50	T33N R43E N½ of SE¼ of Sec 7	FS-Newport
REG 7S	30-50	T33N R43E S½ of SE¼ of Sec 7	FS-Newport
REG 12E	90+	T33N R42E NE¼ of Sec 12	FS-Newport
		T33N R43E NW¼ of Sec 7	
		Little Muddy	
CUT 29W	30-50	T38N R42E NW¼ of Sec 29	FS-Sullivan
MAT 27	90+	T38N R42E W½ of SW¼ of Sec 27	FS-Sullivan
		T38N R42E N½ of NW¼ of Sec 34	
MAT 32	30-50	T38N R42E SE¼ of NW¼ of Sec 32	FS-Sullivan
		T38N R42E NW¼ of NE¼ of Sec 32	
REG 3	30-50	T37N R42E NW¼ of Sec 3	FS-Sullivan
REG 29E	30-50	T38N R42E NE¼ of Sec 29	FS-Sullivan
REG 33	30-50	T38N R42E SE¼ of SW¼ of Sec 33	FS-Sullivan
		T38N R42E SW¼ of SE¼ of Sec 33	
		Middle Fork Calispell	
CUT 27	90+	T32N R42E N½ of Sec 27	Plum Creek
MAT 31	90+	T32N R43E SW¼ of Sec 31	FS-Newport
REG 4	90+	T31N R42E SE¼ of Sec 4	Plum Creek
		Muddy	
CUT 5	30-50	T37N R42E S½ of NW¼ of Sec 5	FS - Sullivan
		T37N R42E N½ of SW¼ of Sec 5	
CUT 5E	30-50	T37N R42E S½ of SW¼ of NE¼ of Sec 5	FS - Sullivan
CUT 6	30-50	T37N R42E S½ of SW¼ of NE¼ of Sec 6	FS - Sullivan
		T37N R42E N½ of NW¼ of SE¼ of Sec 6	
MAT 3	30-50	T37N R42E NW¼ of SE¼ of Sec 3	FS - Sullivan
		T37N R42E NE¼ of SW¼ of Sec 3	
REG 4S	90+	T37N R42E S½ of Sec 4	FS - Sullivan
		T37N R42E N½ of NW¼ of Sec 9	
		North Fork Calispell	
MAT 1	30-50	T32N R42E E½ of SW¼ of Sec 1	FS-Newport
CUT 9	90+	T32N R42E S½ of Sec 9	Plum Creek
MAT 14	90+	T32N R42E E½ of Sec 14	FS-Newport
CUT 31	90+	T33N R42E Sec 31	Plum Creek
MAT 5	90+	T32N R42E Sec 5	Plum Creek
MAT 18	30-50	T32N R43E NW¼ of Sec 18	FS-Newport
REG 13	30-50	T32N R42E NE¼ of Sec 13	FS-Newport
REG 23	90+	T32N R42E E½ of NW¼ of Sec 23	Plum Creek
		Ruby	
REG 3	90+	T35N R42E Sec 3	FS-Newport
CUT 18	30-50	T35N R43E S½ of SW¼ of Sec 7	FS-Newport
		T35N R43E N½ of NW¼ of Sec 8	
MAT 12	90+	T35N R42E SE¼ of Sec 12	FS-Newport
REG 5	30-50	T35N R43E E½ of SW¼ of Sec 5	FS-Newport
		Winchester	
CUT 23	90+	T33N R42E Sec 23	Plum Creek
CUT 25	90+	T33N R42E N½ of Sec 25	Plum Creek
REG 26	90+	T33N R42E S½ of Sec 26	Plum Creek
		T33N R42E N½ of Sec 35	
CUT 36N	90+	T33N R42E NE¼ of Sec 36	Plum Creek
MAT 36	30-50	T33N R42E S½ of SW¼ of Sec 36	Plum Creek

We note that evaluation of upland management areas (UMA) was included in proposed design for this study. Although we did not examine stands that had been created specifically as upland management areas, our small stand size is representative of the sizes of UMAs that have been set aside in this region (N. Sturhan, Washington Department of Natural Resources, personal communication).

SAMPLING OF WILDLIFE POPULATIONS - GENERAL APPROACHES

SELECTION OF WILDLIFE SPECIES

The general design of the study was to survey vertebrate communities using techniques that provided information on the occurrence or abundances of species at the stand scale. We sampled a variety of taxa for which there were reliable sampling methodologies, including forest-floor amphibians and small mammals, diurnal breeding birds, and bats. All taxa were sampled for 3 consecutive years (West-side studies: fall 1992 - spring 1995; East-side studies: 1993-1995) to provide an adequate index of temporal variation in wildlife communities occurring within intensively managed landscapes. In addition, we conducted several directed studies. General descriptions of each vertebrate survey or directed study are given below. Detailed descriptions of the sampling methodologies and protocols employed for each are presented in the individual final reports that follow this introductory section.

TERRESTRIAL AMPHIBIANS AND SMALL MAMMALS

Based on extensive experience with amphibian and small mammal surveys gained during previous studies (Aubry and others 1988, Aubry and others 1991, Aubry and Hall 1991, Bury 1988, Bury and Corn 1987, 1988ab, Corn and others 1988, West 1991) we sampled terrestrial amphibians and small mammals with pitfall traps. Pitfall traps effectively capture surface-active amphibians and most small mammals, resulting in good estimates of relative abundance in forested habitats for both groups (Aubry and Hall 1991, West 1991). In addition, by capturing large numbers of individuals, this technique enables us to assess the demographic structure of populations through analyses of body-size classes for amphibians, and age-classes for small mammals.

BREEDING BIRDS

We used a modified point count method for surveying diurnal breeding bird populations at 100-m intervals along transects established in each stand. Point counts are discussed by Verner (1985) and have been used in several recent studies (e.g., Huff and Raley 1991, Huff and others 1991, Hutto and others 1986, Manuwal 1991, Manuwal and Carey 1991, Manuwal and Huff 1987, Verner and Ritter 1985). The point count method is superior to other methods for sampling forest birds due to relatively poor visibility in forested habitats, and the rugged topography typical of Washington mountains. Other methods, particularly strip transects, are inefficient at determining either the species richness of stands, or at estimating the relative abundance of each species (Manuwal and Huff 1984, Verner 1985).

BATS

It has been shown recently that bats inhabiting forests west of the Cascade Crest in Oregon and Washington roost statistically more often in old than young forest (Thomas 1988, Thomas and West 1991). Bats appear to be using areas with old trees for day-roosting, but leaving these sites to forage over water sources elsewhere, where the abundance of appropriately sized insects is higher than in the forest (Thomas 1988). Although the characteristics of natural roost sites have not been identified adequately for any bat species in Pacific Northwest forests, it is likely that as the average age of forests declines, so will opportunities for bats to roost in natural habitat. We sampled bats by using ANABAT II automated divide-by-N ultrasonic detectors. These devices yield a frequency count of bat passes per unit time by automatically recording bat echolocation calls on cassette tape after they have been electronically transformed into frequencies audible to humans. Because echolocation calls in some cases differ by species, or by groups of closely related species, they can be identified (Thomas and West 1989). The detectors do not require capture, do not affect bat behavior, can distinguish feeding calls from travel echolocation calls, and are capable of accumulating large sample sizes for statistical analysis.

SAMPLING OF HABITAT - GENERAL APPROACHES

WEST-SIDE STUDIES

Stand vegetation sampling--We measured structural and vegetational components of stands to (1) describe wildlife habitats at the stand scale, (2) correlate habitat features at the stand scale with wildlife population parameters, and (3) identify stand components altered by harvest that affect wildlife species. We sampled vegetation at 3 scales. At each bird sampling point and within each pitfall grid, we sampled herbs, low shrubs, and ground cover in 3- x 3-m square plots (9 m^2); tall shrubs, small to medium-sized trees and snags ($\leq$50 cm d.b.h.), and coarse woody debris in 15- x 15-m square plots (225 m^2); and large trees, snags, and stumps (>50 cm d.b.h.) in 45- x 45-m square plots (2,025 m^2).

EAST-SIDE STUDIES

The forested land of northeastern Washington is managed primarily by the U.S. Forest Service, the state Department of Natural Resources, private timber companies, and private landowners. Timber management practices during the last 100 years have included high-grading, clearcutting, post-harvest burning with replanting, fire suppression, selective cutting, and uneven-aged management. These management practices, coupled with natural variation in forest structure due to slope, aspect, edaphic characteristics, and fire, have resulted in a mosaic of forest stands of varying structure and spatial configurations. Based on our GIS analysis, the 3 most common forest types (82% of the total area) are mature, closed-canopy forest; regenerating forest; and clearcuts, including basal-area retention cuts; we selected these 3 forest types for our examination of wildlife use of managed forests. Although the forest stands in this region vary considerably in size, 2 size classes were both representative and sufficiently different to have biological significance: 12-15 ha and >35 ha.

We examined forest stand structure of the 36 study sites on 2 scales. First, we measured habitat features of plots established on the 300-400 m and 1,200 m point-count transects of the 13-ha and >34-ha stands, respectively. This sampling covered about 40% of the total stand area. Second, we measured habitat features at each of the

36 pitfall trapping grids (60 x 60 m). The point-count transects and the pitfall trapping grids never overlapped.

ANALYZING LANDSCAPE-SCALE INFLUENCES ON WILDLIFE HABITAT RELATIONSHIPS

WEST-SIDE STUDIES

To assess potential influences of landscape composition and configuration on wildlife communities occurring in our study stands, we derived landscape indices for each stand at a variety of spatial scales. Selection of the scales used in landscape analyses were based on the life history, home range characteristics, and dispersal capabilities of the taxa being analyzed. The scales at which these metrics were calculated included both ones based on actual watershed boundaries and those for which the area analyzed was independent of physiographic boundaries. Watershed metrics were derived for each of the 4 WRIAs in which our study stands occurred (60,000-81,000 ha) and for each of the 7 WAUs in which we sampled (4,000-12,000 ha). Landscape indices were calculated in concentric circles of 100 ha, 1,000 ha, 5,000 ha, 10,000 ha, and 25,000 ha using filtered 100-m pixel data (Fig. 10). For finer-scale analyses, we also derived landscape metrics for 100-ha and 1,000-ha circles using unfiltered 25-m pixel data. For analyses of amphibian and small mammal data, the landscape center was located on the pitfall grid, whereas for analysis of bird and bat data, the landscape center was located in the geometric center of the stand sampled (Fig. 11). We used the FRAGSTATS computer program (McGarigal and Marks 1995) to quantify landscape structure. This program derives a variety of indices including area metrics; patch density, size, and variability metrics; edge metrics; shape metrics; core area metrics; diversity metrics; and contagion and interspersion metrics (Table 6).

It is important to realize that there were substantial limitations in our ability to analyze landscape-scale influences on vertebrate communities. Our study was designed primarily to generate new information on wildlife habitat relationships occurring at the stand scale. Although we attempted to build a landscape-scale framework into the study design, meeting the constraints of our basic design often necessitated giving stand-scale considerations precedence over those occurring at the

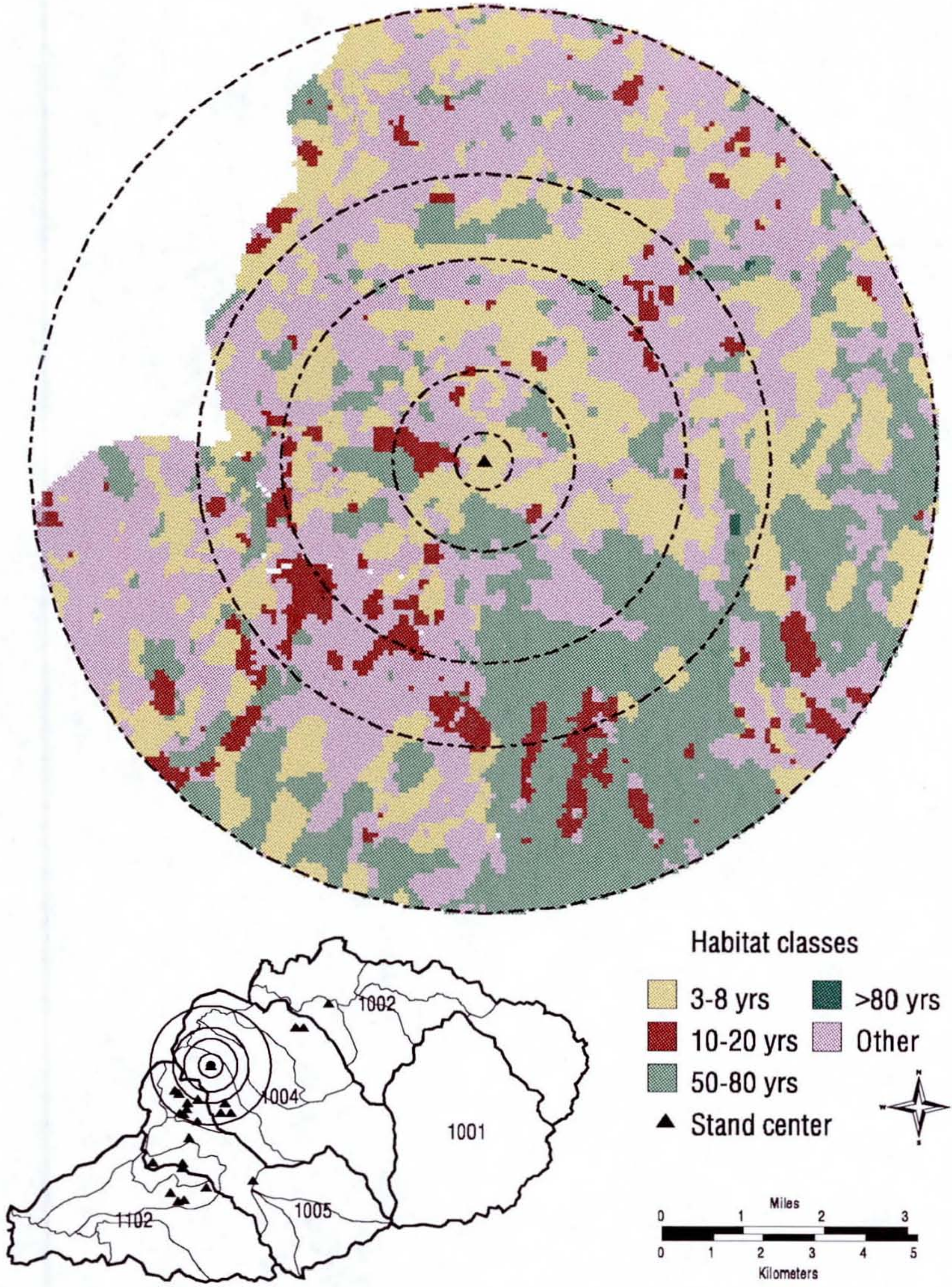

Figure 10. Map showing the 5 circular landscapes analyzed for each stand using 100 m pixel data; example shown is for clearcut stand 5. The smallest circle is 100 ha and progressively larger circles are 1,000, 5,000, 10,000, and 25,000 ha.

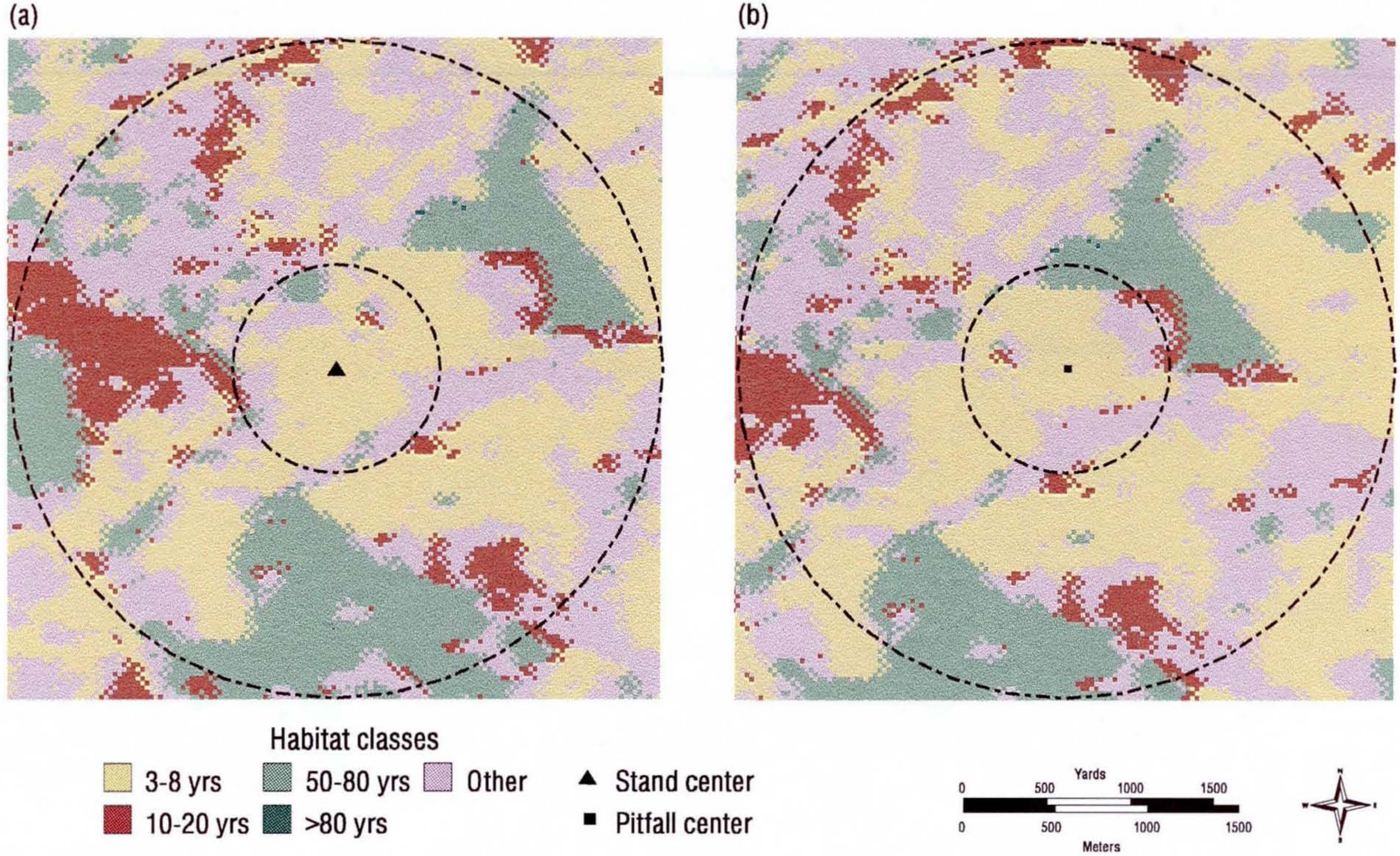

Figure 11. Maps showing the smaller 2 circular landscapes (100 ha and 1,000 ha) analyzed for each stand using 25-m pixel data for finer-scale analyses; example shown is for clearcut stand 5. In (a) the center is located in the geometric center of the stand for bird and bat analyses, and in (b) the center is located on the pitfall grid for amphibian and small mammal analyses.

Table 6. Metrics computed in FRAGSTATS, grouped by subject area (from McGarigal and Marks 1995; p. 14-15).

Scale	Acronym	Metric (units)
		Area metrics
Patch	AREA	Area (ha)
Patch	LSIM	Landscape similarity index (percent
Class	CA	Class area (ha)
Class	%LAND	Percentage of landscape
Class/landscape	TA	Total landscape area (ha)
Class/landscape	LPI	Largest patch index (percent)
		Patch density, size, and variability metrics
Class/landscape	NP	Number of patches
Class/landscape	PD	Patch density (number/100 ha)
Class/landscape	MPS	Mean patch size (ha)
Class/landscape	PSSD	Patch size standard deviation (ha)
Class/landscape	PSCV	Patch size coefficient of variation (percent)
		Edge metrics
Patch	PERIM	Perimeter (m)
Patch	EDCON	Edge contrast index (percent)
Class/landscape	TE	Total edge (m)
Class/landscape	ED	Edge density (m/ha)
Class/landscape	CWED	Contrast-weighted edge density (m/ha)
Class/landscape	TECI	Total edge contrast index (percent)
Class/landscape	MECI	Mean edge contrast index (percent)
Class/landscape	AWMECI	Area-weighted mean edge contrast index (percent)
		Shape metrics
Patch	SHAPE	Shape index
Patch	FRACT	Fractal dimension
Class/landscape	LSI	Landscape shape index
Class/landscape	MSI	Mean shape index
Class/landscape	AWMSI	Area-weighted mean shape index
Class/landscape	DLFD	Double log fractal dimension
Class/landscape	MPFD	Mean patch fractal dimension
Class/landscape	AWMPFD	Area-weighted mean patch fractal dimension
		Core area metrics
Patch	CORE	Core area (ha)
Patch	NCORE	Number of core areas
Patch	CAI	Core area index (percent)
Class	C%LAND	Core area percentage of landscape
Class/landscape	TCA	Total core area (ha)
Class/landscape	NCA	Number of core areas
Class/landscape	CAD	Core area density (number/100 ha)
Class/landscape	MCA1	Mean core area per patch (ha)
Class/landscape	CASD1	Patch core area standard deviation (ha)
Class/landscape	CACV1	Patch core area coefficient of variation (percent)

Table 6. Continued.

Scale	Acronym	Metric (units)
Class/landscape	MCA2	Mean area per disjunct core (ha)
Class/landscape	CASD2	Disjunct core area standard deviation (ha)
Class/landscape	CACV2	Disjunct core area coefficient of variation (percent)
Class/landscape	TCAI	Total core area index (percent)
Class/landscape	MCAI	Mean core area index (percent)
		Nearest neighbor metrics
Patch	NEAR	Nearest neighbor distance (m)
Patch	PROXIM	Proximity index
Class/landscape	MNN	Mean nearest neighbor distance (m)
Class/landscape	NNSD	Nearest neighbor standard deviation (m)
Class/landscape	NNCV	Nearest neighbor coefficient of variation (percent)
Class/landscape	MPI	Mean proximity index
		Diversity metrics
Landscape	SHDI	Shannon's diversity index
Landscape	SIDI	Simpson's diversity index
Landscape	MSIDI	Modified Simpson's diversity index
Landscape	PR	Patch richness (number)
Landscape	PRD	Patch richness density (number/100 ha)
Landscape	RPR	Relative patch richness (percent)
Landscape	SHEI	Shannon's evenness index
Landscape	SIEI	Simpson's evenness index
Landscape	MSIEI	Modified Simpson's evenness index
		Contagion and interspersion metrics
Class/landscape	IJI	Interspersion and Juxtaposition index (percent)
Landscape	CONTAG	Contagion index (percent)

landscape scale. In addition, it was not possible to classify the satellite imagery into habitat classes that fit precisely into the 4 structure classes in which we sampled wildlife populations. The satellite imagery was classified into the following 5 habitat classes: clearcut (3-8 yr), young (10-20 yr), mature (50-80 yr), old-growth (>80 yr), and other (including both forested and non-forested habitats). Thus, the closed-canopy structure class was not represented in the database we used for analyses of landscape-scale habitat relationships, nor did the mature habitat class include only harvest-age stands. To the extent possible, however, we used these landscape metrics to search for large-scale influences on the vertebrate communities we studied that could provide insights on the structure and composition of vertebrate communities in managed landscapes. This information will provide the basis for formulating new hypotheses that can be tested in future landscape-scale research.

EAST-SIDE STUDIES

To assess potential influences of landscape composition and configuration on wildlife communities occurring in our study stands, we derived landscape indices for each stand at 2 spatial scales. We considered areas within 1 and 3-km radius circles. For analyses of small mammal data, the landscape center was located on the pitfall grid, whereas for analysis of bird data, the landscape center was located in the geometric center of the stand sampled. For each circle, we calculated the number of patches, total perimeter, and percentage of total area of each forest type. There are a number of additional landscape metrics that one might examine including various measures based on fractals. We found in our analysis of landscape structure that many variables are intercorrelated. Consequently we chose to reduce our analysis to variables that would be readily interpretable.

DIRECTED STUDIES

WEST-SIDE STUDIES

CHARACTERISTICS OF BAT ROOST SITES

Several bat species in the Pacific Northwest are closely associated with old-growth Douglas-fir forests for roosting (Thomas and West 1991); thus, the recent and rapid reduction of old-growth forest on the west coast has undoubtedly had detrimental effects on these bat populations. Until the essential characteristics of movement patterns and roost sites are known, it will be very difficult to manage forests for the persistence of native bat populations. Recent advances in the miniaturization of radio-telemetry components has now made radio-tracking of bats feasible, and hold promise for gaining information on movements and roost characteristics.

Efforts to locate the bat roost study on or near the stands used for most other investigations in the project failed due to extremely low captures of bats. Rather than forego this directed study we moved it just east of the Cascade crest into the Teanaway River drainage north and northwest of Cle Elum, Washington. Captures of bats were sufficient there to obtain a good sample of radio-telemetry locations for the long-legged myotis (*Myotis volans*). Additionally, physiographic and vegetational features of the

Teanaway study area had been entered into a functional geographical information system by the Boise Cascade Corporation and made available to us for this project. This allowed us to conduct much more thorough site and stand-level analyses of the roost sites than would have been possible at the primary west-side sites.

To locate roost sites we captured bats with Tuttle traps (Tuttle 1974) and mist nets. Transmitters were placed on both male and female bats of sufficient size to carry the added transmitter weight well. Roost sites were located during the day using a GPS system and characterized with respect to the roost site, stand, and selected landscape variables. This work was done in partial fulfillment of the requirements for the Master's degree in Wildlife Science at the University of Washington by Matthew W. Frazier. The report for this component of the TFW Landscape Project is included as Appendix A of the West-side Study results found in Volume 2 of this report.

MOVEMENT PATTERNS AND REPRODUCTIVE ECOLOGY OF NORTHWESTERN SALAMANDERS

Pond-breeding salamanders, including the northwestern salamander (*Ambystoma gracile*), long-toed salamander (*A. macrodactylum*), tiger salamander (*A. tigrinum*), and roughskin newt (*Taricha granulosa*) in Washington, all migrate in the spring from overwintering habitats to breeding ponds, where they typically congregate in large numbers to mate and lay eggs (Nussbaum and others 1983). The adults return to terrestrial habitats soon after the breeding season is over. Another migration away from the ponds occurs in the late summer and early fall when metamorphosing larvae leave the ponds to seek overwintering habitats. We predicted that pond-breeding salamanders would be directly affected by landscape fragmentation because their life cycles involve yearly movements among stands, not simply within them. Because timber harvesting negatively affects habitat suitability for aquatic amphibians, due in part to the drier conditions that result from removal of the canopy (Bury and Corn 1988b), migratory salamanders may be unable to cross young plantations on their way to breeding ponds. Consequently, both the sizes, arrangements, and environmental conditions of patches (stands), and the availability of suitable breeding ponds within forested landscapes will strongly influence the reproductive ecology of migratory salamanders. The primary objective of this study was to investigate the movement patterns and reproductive ecology of northwestern salamanders in forested landscapes managed primarily for timber production. We used 2 approaches: trapping and individually marking migrating and dispersing populations at breeding ponds, and following selected large individuals by using radio-telemetry. The final report for this study is included as Appendix B in the West-side Study results volume; the Ph.D. dissertation that will result from this study will be forwarded to DNR upon completion, which is anticipated to occur sometime in 1998.

HABITAT UTILIZATION AND HOME RANGE SIZE OF THE BOBCAT IN MANAGED FORESTS OF WESTERN WASHINGTON

Habitat utilization and home range size of bobcats (*Felis rufus*) in managed forests of western Washington were investigated from May 1993 through September 1994 with the use of radio-telemetry and Geographic Information Systems (GIS). A total of 751 independent locations for 6 male and 4 female bobcats indicated a mean annual home range size of 14.4 km^2 for males and 8.4 km^2 for females. Home range size and movement indexes differed between sexes ($P < 0.05$) but not between seasons (winter vs. summer and breeding vs. non-breeding). A density estimate of 0.22 $cats/km^2$ calculated based on mean home range size and a mean home range overlap of 27.24%. Home range and density results from this study were consistent with those from a 1978 study conducted on the same study area and indicate similar population dynamics. Availability of 6 habitat types defined by structural differences resulting from timber management was determined from Landsat digital imagery. Habitat analysis based on availability within the study area showed young stands (12-25 years old) to be the most and harvest-age stands (50-70 years old) to be the least preferred. Differential use of habitat ($P < 0.05$) was detected for 3 cats based on availability within individual home ranges. Young stands were the most consistently preferred. Clearcuts (2-5 years old) and harvest-age stands were mostly avoided. Preferred habitat appeared to be related to abundant understory vegetative cover and prey availability. The final report for this study is included as Appendix C in the West-side Study results volume.

EAST-SIDE STUDIES

Preliminary field work indicated that a study of pond-breeding salamanders was logistically more difficult in northeastern Washington, and an alternative project to examine the consequences of forest fragmentation on the dispersal of 2 frog species (Columbian spotted frog [*Rana luteiventris*] and Pacific tree frog [*Hyla regilla*]) was developed. Because of budget reductions during the course of the project, we were unable to fully support both studies with TFW funding. Because of their potential importance, we sought supplemental funding from other sources and continued both projects as graduate research studies.

We conducted additional studies to extend our understanding of the processes affecting population dynamics and community structure at the landscape level. These studies included an examination of the use of forest strips as corridors by small mammals (thesis completed by Lisa A. Nordstrom, WSU), analysis of dispersal and genetic subdivision of small mammals that are forest specialists (dissertation research in progress by Stephen G. Mech, WSU), analysis of the community structure and trophic relationships of shrews (thesis by Robert E. Griffith, EWU), and evaluation of the effects of patch size and forest type on predation of artificial nests.

POPULATION STRUCTURE FOR 2 ANURANS (*RANA LUTEIVENTRIS* AND *HYLA REGILLA)*

The question of how the movement patterns of amphibians are affected by forest harvesting is important to understanding the population dynamics and probabilities of local extirpation for these species. Direct methods of examining this question (e.g., mark-recapture) were not cost-effective and we believed would provide insufficient data. Consequently we began examining molecular methods for analyzing the genetic "tags" of individuals. That way, we could sample populations across the landscape and infer the movement of individuals by the presence or absence of these tags within the sample populations. We examined multilocus DNA fingerprinting, DNA sequencing, and microsatellite variation. As work progressed, we realized that our initial question had to be reframed by asking whether local populations had become genetically differentiated.

We developed 7 microsatellite markers and studied their frequency distributions in 8 *Rana luteiventris* (3 loci) and 3 *Hyla regilla* (4 loci) populations. Allele sizes appeared to conform to a stepwise mutation model except 11% of allele differences probably resulted from larger mutations. Most loci were in Hardy-Weinberg equilibrium, although we found evidence of null alleles for 3 *R. luteiventris* loci. Significant regressions of $\hat{M}$ on geographic distance ($P < 0.001$) suggested a neighborhood size of 19 to 274 frogs for *R. luteiventris* populations. Using these estimates with models for effective population size, we estimated mutation rate to be near 10^{-4}. New statistics derived specifically for microsatellite data appear to have higher variances than estimators based on an infinite allele model.

We investigated a multilocus detection method for measuring microsatellite variation simultaneously at many loci. This technique reveals a series of alleles within a restriction fragment profile, but potential measurement errors require grouping alleles of similar size into bins. Mean band-sharing and heterozygosity were highly correlated ($r = -0.99$) and the former increased continuously with increasing bin width. We used replicate gels to calibrate a binning algorithm and found significant differentiation between *R. luteiventris* (N = 5) and *H. regilla* (N = 2) populations ($P \leq 0.008$), which was consistent with results from single locus markers. Contrary to published findings, band-sharing statistics do not exhibit excessive levels of covariance. This work is being prepared for submission to *Conservation Biology* and *Molecular Ecology* and is included as Appendix A of the East-side Study results found in Volume 3 of this report.

DETECTING EARLY DIFFERENTIATION BETWEEN SMALL, NONEQUILIBRIUM POPULATIONS

Genetic markers are important for describing population structure and making inferences about migration, but there may be limitations when using these tools to analyze smaller, nonequilibrium populations. We use a simulation model to evaluate how quickly and consistently we might detect genetic isolation between small populations resulting, for example, from recent habitat fragmentation. Genetic drift is the only evolutionary force in the model and reproduction replaces mortality at each iteration. All populations are initialized with similar allele frequencies to mimic isolation of previously panmictic populations. In nearly all cases, initial mean differentiation is rapid and approximately linear whereas the variance for mean differentiation is a function of population size, number of loci, and number of alleles. Detecting early differentiation (≤30 iterations) between groups of populations is feasible when populations are small (e.g., N = 50) and several markers are employed (e.g., ≥6 loci, 2 alleles each). With more markers (≥8 loci) we can detect statistically significant differences within a single iteration. A more biologically meaningful level of significance (i.e., equivalent to <1 migrant/generation) can be detected within 31 iterations using 10 hypervariable markers (10 alleles each). For larger populations (N > 300), we may be unable to detect biologically significant differentiation until after 150 iterations. Although genetic isolation can be detected between groups of small populations of approximately equal size, pairwise comparisons between individual populations may produce conflicting interpretations despite identical model parameters. The final report for this study is included as Appendix B in the East-side Study results volume.

Small Mammal Use of Corridors in a Fragmented Landscape

Corridors have been advocated as a means of ameliorating the effects of habitat fragmentation on wildlife populations and communities. Their usefulness, however, depends on several factors, including degree of fragmentation, corridor size, habitat quality, and especially the habitat requirements and mobility of the particular species. Distributions of small mammal species were analyzed across clearcut, regenerating, and closed-canopy forest habitats in northeastern Washington. Red-backed voles were identified as a forest-restricted species, able to perceive corridors in the landscape. Use of corridors by red-backed voles was found to be highly variable due to interyear variation in population density. When populations were high, corridors provided additional habitat for colonization. Corridor stands, however, contained lower quality habitat than forest stands, lacking habitat characteristics preferred by red-backed voles. At lower population levels, the corridors only provided a dispersal route between the larger, more preferred forest stands. Variation in habitat quality and corridor size appeared to influence vole abundance within corridors. Size of forest patches did not affect vole presence or abundance due to differences in habitat surrounding forest patches. Thus, the relative importance of corridors to red-backed vole distribution and persistence remains uncertain. However, these connective strips do serve a function as additional habitat in periods of high population density and are capable of facilitating movement between forest patches for forest-interior small mammal species. (The final report is a modification of a thesis written by Lisa A. Nordstrom, WSU, and is included as Appendix C of the East-side Study results volume. The report is being prepared for submission to *Conservation Biology*).

STRUCTURE OF SHREW COMMUNITIES

The role of interspecific competition in determining the distribution and abundance of species has been the subject of considerable debate. Some authors have argued that species communities follow "assembly rules" determined by competitive and niche relationships. Recent work on shrew (Family Soricidae) communities in eastern North America supports the view that assembly rules may be operating for functional groups of species based on body size. In northeastern Washington, 5 species of shrews occur in managed forests. Because the ecology of these species is not well understood, we made them a focal point of our small mammal surveys. We developed a computer randomization for determining the distribution of possible shrew communities consisting of up to 5 species. We then tested for differences between the observed and expected distributions for each of 3 years. This allowed us to consider changes due to temporal and spatial variation in the distributions of species. This analysis provides little evidence of competition as a mechanism structuring the shrew community. (The final report presents a preliminary analysis examining the evidence for assembly rules for shrew communities. This work is the thesis research of Robert E. Griffith, EWU, and is included as Appendix D of the East-side Study results volume.)

NEST PREDATION IN MANAGED FORESTS

Nest predation is considered a major influence on the population dynamics (Ricklefs 1969) and community relationships (Martin 1988) of many bird species. Habitat fragmentation can result in increased nest predation because of the creation of edge habitats that allow predators to invade the interior of forested areas that were previously inaccessible (Wilcove 1985). In 1994, we conducted an experiment on 30 stands (5 stands of each combination of forest type and stand size) to examine how rates of predation on artificial nests are influenced by habitat and patch size. Predation rates did not differ among patch sizes, but did increase over time. Predation increased with increasing forest structure (clearcut < regeneration < closed canopy). This result was surprising and suggested that the type of predator is an important element in understanding the differences in predation across stands. Mammalian predators like the red squirrel (*Tamiasciurus hudsonicus*) may be responsible for much of the predation that we observed. Funding for this work was provided by Eastern Washington University. This work will be submitted to *Ecological Applications* by Hallett and O'Connell. A draft manuscript is included as Appendix E in the East-side Study results volume.

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APPENDIX

Development and analysis of a landscape-level GIS database for assessing wildlife use of managed forests

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GIS'93
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Vancouver
British Columbia
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Abstract

Analysis of wildlife response to managed forest landscapes is being conducted at the Pacific Northwest Research Station, USDA Forest Service in Olympia, Washington through a research project funded by the Washington State Timber, Fish, and Wildlife (TFW) Cooperative. Remote sensing and geographic information systems (GIS) were used extensively in the first phase of this project to evaluate landscapes and to select study sites for sampling wildlife. To select study sites that met specific habitat conditions, we assessed landscapes in southwestern Washington by integrating classified Landsat imagery with other GIS base layers and landscape pattern indices. This analysis demonstrates the integration of remote sensing, GIS, and statistical analysis for evaluating landscapes. The ability to characterize and screen large areas using this methodology may prove to be an invaluable tool for large-area analysis in forestry and wildlife research.

Introduction

Landscape pattern and forest fragmentation in the Pacific Northwest are growing concerns for those charged with management of wildlife species, especially species known to be dependent on one seral stage of forest (typically old growth). Studies have recently been conducted that examined the relationship between forest fragmentation and wildlife abundance in *unmanaged* forest habitats of the Pacific Northwest (Lehmkuhl, *et al.* 1991), but little attention has been given to studying this relationship in intensively managed landscapes.

A research program has been initiated by the Washington State Timber, Fish, and Wildlife (TFW) Cooperative to examine the abundance and diversity of wildlife in intensively managed forests. A primary goal of this research is to assess the influence that stand age, size, and position within the landscape have on wildlife populations. A necessary pre-requisite for this assessment is the ability to discriminate between landscapes according to the age, size, and pattern of forest stands, and the intensity and manner in which these landscapes have been logged. To discriminate between landscapes in an experimental framework, objectively derived measures must be used to quantify the differences in structural pattern among landscapes such that the differential effects of management intensity and history on wildlife diversity and abundance can be assessed.

Tools have been developed by researchers in the emerging field of landscape ecology that provide an objective means to quantify the structure of forested landscapes and their component stands through the use of digital maps and imagery in a GIS environment. Several measures of landscape pattern have been developed that are easily calculated using GIS data (Forman and Godron 1986; O'Neill, *et al.* 1988; Milne 1988; Turner 1990). Examples of these indices are the relative dominance of cover types in the landscape (dominance), the adjacency pattern of land cover types, whether clumped or dispersed (contagion), and the complexity of shapes on the landscape (fractal dimension) (O'Neill, *et al.* 1988; Milne 1988). These so called 'information theoretic indices' provide a useful means for quantifying the structural pattern of landscapes, and can be used to describe the extent to which landscapes have been altered by human activities (O'Neill, *et al.* 1988;Turner 1988).

O'Neill, *et al.* (1988) described the pattern of land cover types in the eastern U.S. using information theoretic indices and were able to assess the degree of human influence on large landscapes. Lehmkuhl, *et al.* (1991) used pattern indices to describe levels of forest fragmentation around old-growth stands and related the effect of fragmentation to species diversity and abundance. Lehmkuhl and Raphael (in press) used pattern indices to describe habitat around northern spotted owl (*Strix occidentalis caurina*) nests and were able to infer selection for lower levels of fragmentation than in randomly selected areas. Ripple (1991) used pattern indices to describe the changing nature and degree of fragmentation of a forested landscape over time.

These analyses demonstrate the application of pattern indices to describe the structural conditions of landscapes and how these structural conditions may influence ecological processes. These indices are also useful, however, for multivariate discrimination of structural pattern among landscapes. No previous effort has attempted to develop an *a priori* gradient of landscape pattern with which to screen large areas. The use of this methodology may allow a more thorough assessment of the type and extent of human disturbance at the landscape scale and the consequent effects on the functioning of ecological systems.

Development of a database to describe intensively managed landscapes

The first objective was to locate intensively managed forest stands that fell into a specific range of sizes, seral stages, and landscape configurations in which to sample wildlife communities. Target landscapes were those consisting predominantly of second-growth forest in shrub/sapling (clear-cut) pre-canopy closure (young), and mature (harvestable) seral stages. Seven combinations of age and size of forest stands were initially formulated, and each would be replicated eight times for a total of 56 sites. Large patches of mature, young, and clear-cut stages, smaller patches of mature, young, and clear-cut stages, and very small patches in the mature stage ('Upland Management Areas' or UMAs) would be sampled from landscapes over a gradient of landscape pattern.

Our initial geographical area of interest consisted of all of the managed Douglas-fir (*Pseudotsuga menziesii*) dominated forests west of the Cascade crest in Washington State. In order to search large areas for the target landscape configurations, we developed a GIS methodology by which we were able to classify landscapes by age class, subdivide the landscape by watershed, calculate the pattern or character of the landscape, and examine areas for their management intensity and potential as study areas. Through the use of this methodology we were able to screen a large area of southwestern Washington for landscapes that met our research needs, and then array them according to their landscape pattern.

Base data were derived or collected from various sources to provide a framework for evaluating landscape parameters and to conduct a suitability analysis for selecting study sites. One of the most important base layers needed for this analysis was a map of forest seral stages. Due to the large extent and multiple ownerships in our target area, no existing data source could provide the geographic coverage, standardized forest stage classification, or up-to-date representation that we required. We therefore relied on Landsat Thematic Mapper (TM) imagery to create a surrogate seral-stage classification based on species, canopy structure, and age.

A combination of supervised and unsupervised techniques was used to classify TM imagery. We first conducted an unsupervised clustering with ERDAS (ERDAS Inc., Atlanta, GA) image processing software to identify the range of spectral reflectances. We identified clusters by integrating existing Washington State Department of Natural Resources (DNR) land-use polygon data with the TM imagery. Vector polygon data was overlaid on the imagery using ARC/INFOs (ESRI, Redlands, CA) 'Live Link'. We first examined the spatial extent of stands from vector polygons, and then identified the type of stand from polygon attributes of stand age and dominant species. Unsupervised clustering enabled us to identify spectral variation within polygons (stands) that were coded with a single age or dominant species.

We selected training areas for supervised classification from the identified clusters and by on-screen digitizing. This method of selecting signatures captured the spectral variation within forest age classes (used as surrogates for forest seral stage), and allowed for classification of hardwoods, water, urban areas, grasslands, and snow and ice.

Table 1: *Age classes used to classify imagery and corresponding forest seral stage.*

AGE	SERAL STAGE
3-8 years	clear-cut, shrub/sapling, open canopy
10-20 years	young forests, pre-canopy closure
50-80 years	mature forests, closed canopy
> 80 years	old forests, multistoried canopy
other ages, other non-forest	outside ages of interest, non-forest

The spectral signatures developed from unsupervised clustering and supervised training were combined, merged, and selected for use in a supervised classifier (maximum likelihood). The resulting classification was recoded to four age classes of interest and one 'Other' class that included all non-forest classes as well as the 20-50 year-old forests which were not of interest. The four age classes correspond to forest seral stages found in managed landscapes (table 1), and were interpretable from satellite imagery.

The final classification was filtered using a 3x3 roving window. Isolated pixels were blended into the class that represented the majority of class values in the 9-pixel scanning window. This produced a somewhat cleaner image that was less influenced by the noise produced by edge pixels between two different stand types. This filtered classification was also resampled to 100x100 m (1 ha) from the original 25-m pixels (0.125 ha) using a nearest-neighbor resampling algorithm. To reduce processing constraints for this spatially extensive dataset. Although some loss of resolution was inevitable, the proportions of land cover types mapped as 0.125-ha cells remained the same after resampling and mapping to 1 ha cells.

Subdivision of the study area by watershed

To quantify the range of landscape pattern, we subdivided the age-class map of the study area into watersheds from polygon data. Initially, we used the boundaries of major river basins provided by the State Water Resource Inventory Area (WRIA) classification. These basins were approximately 60,000 to 81,000 ha in area. This divided our study area into 68 whole and partial watersheds for analysis. This number was reduced to 19 watersheds by selecting only those whole watersheds that were west of the Cascade crest and east of the Puget Trough physiographic province (Franklin and Dyrness 1973). This ensured that all of the watersheds under consideration were in similar environmental zones, and that the major differences between landscapes would be management history (figure 1).

As a preliminary assessment of landscape condition, we calculated indices of pattern on these watersheds. Percentage area in each age class and three measures of pattern were calculated on each of the 19 watersheds in our study area using indices available in IDRISI (Clark

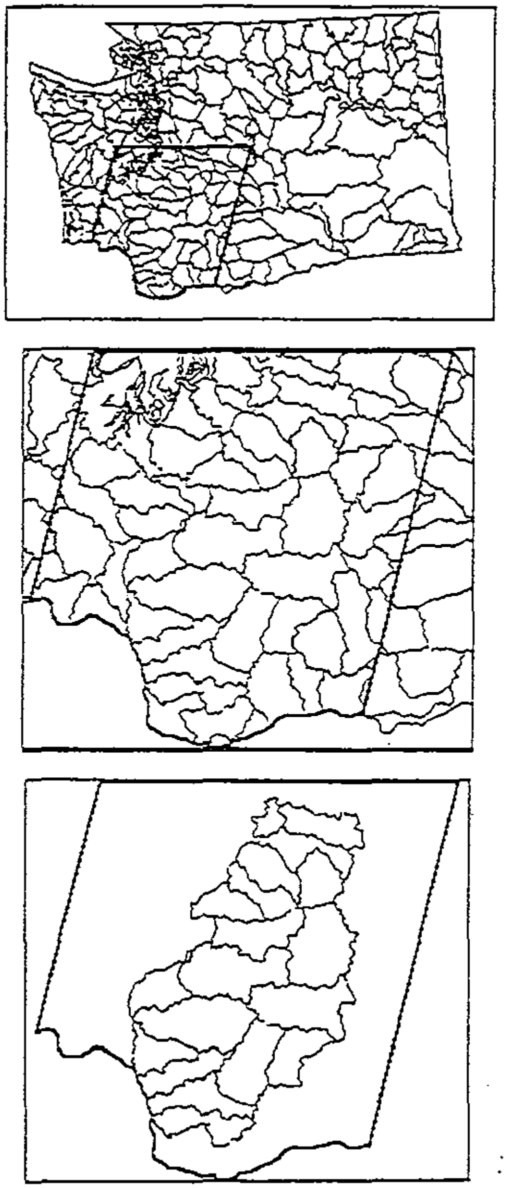

Figure 1: *Top: Initial area of interest (southwestern Washington) and boundaries of major river basins. Middle: Close-up of the 68 whole and partial river basins in the area of interest. Bottom: Boundaries of 19 major river basins remaining after initial selection for managed landscapes west of Cascade crest.*

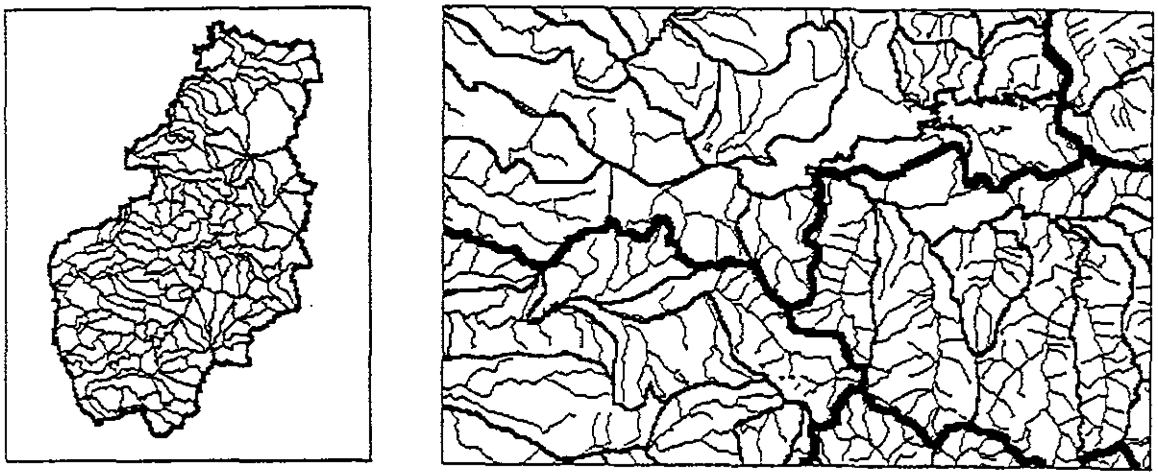

Figure 2: *Left: Subdivision of major river basins into sub-basins. Right: Close-up of major basins, sub-basins, and stream networks.*

University), a pc-based GIS. Diversity, dominance, and fragmentation indices were calculated using IDRISIs moving window procedure which determined an index value for each individual pixel within the watershed. A single index for each watershed was calculated as the average of values for all pixels. The range of index values among watersheds was narrow, with most of the variation in landscape pattern occurring within watersheds at this scale. It was therefore necessary to map smaller watersheds that would provide a more useful unit of analysis.

To divide the 19 major river basins into smaller watersheds, we delineated sub-basins of 4,000 to 12,000 ha for pattern analysis. We followed DNR guidelines for delineating Watershed Administrative Units (WAUs) within each of the major watersheds. United States Geological Survey (USGS) 1:100,000-scale topographic maps were used to manually interpret the boundaries between 3rd-and 4th-order stream basins. These interpretations were digitized and edited using ARC/INFO. This resulted in 119 sub-basins mapped by age class.(figure 2).

Preliminary screening

Since our main goal in this analysis was to establish study sites only intensively managed forest stands, we first eliminated areas not meeting this criterion. Land within national parks, national monuments, wilderness areas, and similarly reserved areas were excluded first. These were masked out of the study area by overlaying a map of major public land ownerships (from DNR). In addition to the preliminary screening for land use, another mask was created that removed areas from consideration that were at elevations above the Douglas-fir zone; a binary mask was created of areas above and below 1,500 m using USGS 1:250,000 Digital Elevation Model (DEM) data. The incorporation of these two masks eliminated four sub-basins and portions of other basins from consideration and effectively served as a suit-ability model for these components of our search criteria.

A third component of the selection criteria used for our initial screening was clear-cut to old growth ratio. We eliminated those sub-basins from consideration that had greater than 20% old-growth forests and less than 20% clear-cut as these areas would not be representative of intensively managed landscapes. This resulted in the elimination of 30 sub-basins from consideration, leaving 85 sub-basins consisting primarily of intensively managed forests.

One additional screening was necessary to remove sub-basins (landscapes) that were heavily influenced by ash or mud flows from the Mount St. Helen's eruption of 1980. These landscapes are anomalous in vegetative regrowth and the presence of wildlife because vegetation pattern has not been structured primarily by forest management practices. This resulted in 79 sub-basins in which to search for study sites.

Generation of a landscape pattern gradient

Area in each forest age class, the distribution of stand sizes, and three indices of landscape pattern (dominance, contagion, and fractal dimension) were calculated and evaluated to determine the range of available stand conditions in the remaining 79 sub-basins. Dominance and contagion were computed using the algorithms provided in the SPAN program (Turner 1990). The fractal dimension was calculated by the perimeter-area method described by Milne (1988). These three pattern indices were tabulated with percent area in each age class to provide variables on composition and configuration of each landscape that could be used as a basis for discrimination. The position of sample sub-basins on a landscape pattern gradient was examined by reducing information in the original variables to principal components (PCA), clustering landscapes based on PCA component

Table 2: *Principal components of landscape structure in 79 sub-basins in western Washington, and rotated loadings of variables on components.*

VARIABLE	COMPONENT 1	COMPONENT 2	COMPONENT 3	COMPONENT 4
Eigenvalue	2.8308	1.6400	1.0813	.9739
% Variation	35.38	20.50	13.52	12.17
% Clear-cut	-0.0722	0.9699	-0.0760	0.0343
% Young	-0.1865	-0.0592	-0.0417	-0.9632
% Mature	-0.3982	-0.7535	0.3139	0.1570
% Old Growth	-0.7325	-0.1124	-0.4419	0.1915
% Other	0.8639	-0.0704	-0.0790	0.3811
Dominance	0.2538	0.6987	0.3957	0.3213
Contagion	0.0187	-0.0921	0.8577	0.0392
Fractal Dimension	0.6149	0.3138	-0.0405	0.1874

loadings, and using discriminate analysis to test the strength of the classification.

We conducted principal components analysis using the NCSS statistical package (Hintze 1990) to reduce the description of landscape composition and pattern from the eight original variables to four independent components. Four components were retained in the analysis, even though the eigenvalue for the fourth component was less than 1.0. This fourth component explained an additional 12 percent of the variance resulting in approximately 82 percent of the total variance in the data set being explained (table 2).

Component loadings showed that sub-basins differed primarily along an urban/patch complexity gradient characterized by the amount of "Other" (urban, middle age 20-50 yr forest) land-type, a lack of old growth, and patch shape complexity (fractal). A second independent component further differentiated sub-basins along a clear-cut gradient based on the dominance of recent clear-cut area. The third component of landscape pattern was a patch clumpiness gradient based on the contagion of forest types, or the spatial complexity of juxtaposition of types. The fourth component of pattern differentiated sub-basins based on the absence of young forest.

Classification of sub-basins using cluster analysis

Cluster analysis was used to classify the 79 sub-basins into one of five groups. The k-means algorithm implemented by the NCSS statistical package was used to assign landscapes to groups based on the loadings on the four components described above. The cluster means for each group are given in table 3. Discriminant analysis conducted on the derived groups determined that the groupings were significantly separable and all observations except one were properly classified.

The five groups represent different landscape configurations and establish a gradient of landscape pattern resulting from differing intensities of fragmentation and varying natural and cultural influences. The first group or landscape-structure type was dominated by landscape

Table 3: *Mean principal component scores for groups of watershed sub-basins in western Washington.*

GROUP	COMPONENT 1 urban/complex	COMPONENT 2 clear-cut	COMPONENT 3 clumped	COMPONENT 4 not-young	N
1	-0.7432	2.1044	0.0278	-0.0371	9
2	-0.7451	-0.6421	0.3735	0.8400	22
3	0.7904	0.0642	-0.3042	0.2422	31
4	-0.2083	0.0272	0.2750	-1.604	16
5	1.913	-1.009	-3.438	-0.2948	1

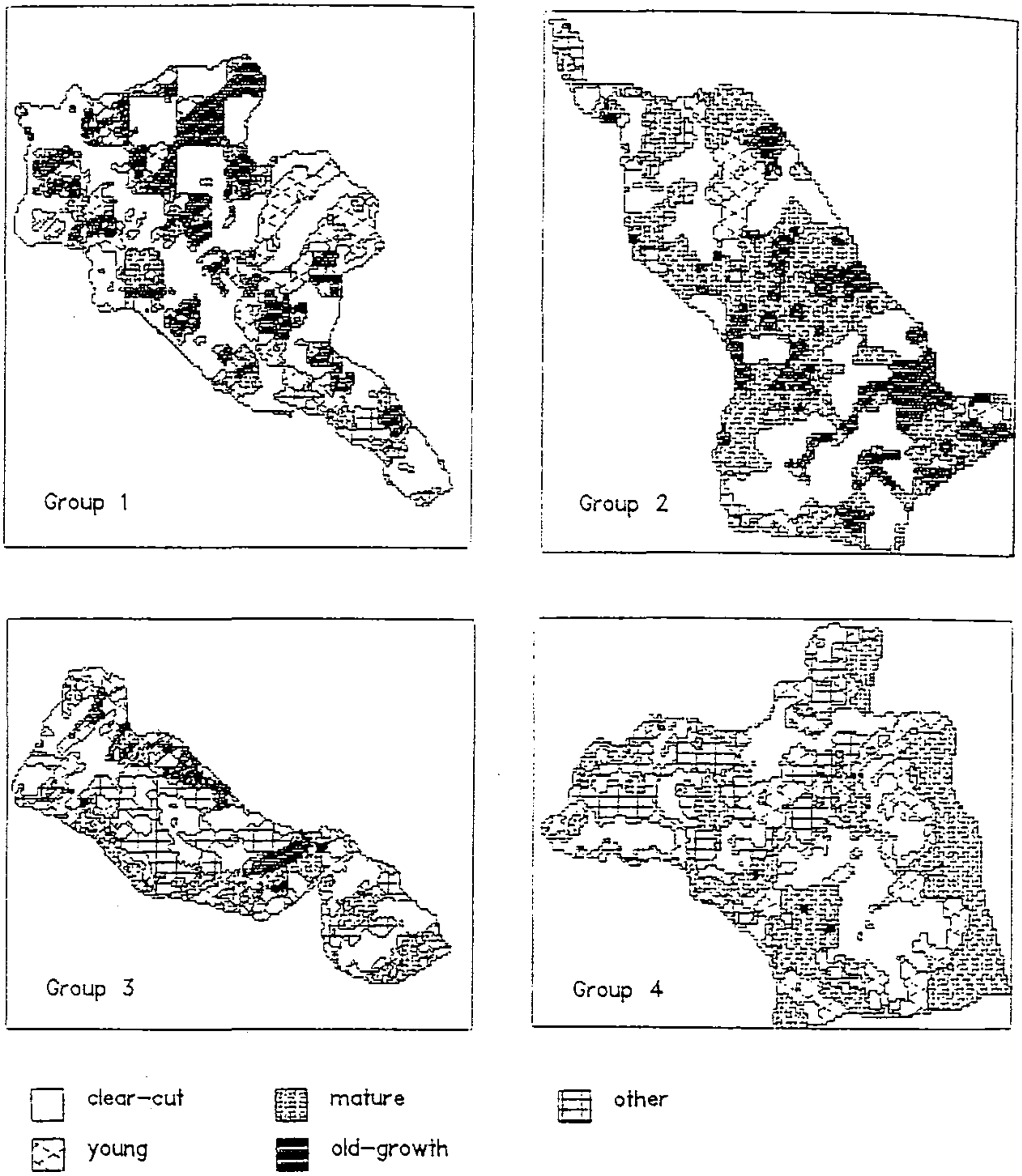

Figure 3: *Example landscape pattern types in 4 of 5 landscape pattern groupings.*

component 2, or the amount of recent clear-cut area in the landscape. These sub-basins had a large proportion of the landscape in clear-cuts, and low contagion or clustering of types. Group 1 was characterized by landscapes with dispersed patches of large clear-cuts in a matrix of older forests (figure 3), such as occurs in the checkerboard pattern of public/private ownership commonly found in parts of the western U.S.

The second group was composed of sub-basins that were high in components 4 and 3, which included areas that had large patches of older managed forests, with some non-forest components, and few young forest components. These landscapes had high dominance and contagion of types suggesting large patches of interconnected older forest, with some clear-cutting, but very few patches of young forest. These landscapes may have been logged early in the century and left idle until recently, or have been recently logged (figure 3).

The third structure group included landscapes that were dominated by non-forest classes, had complex edge characteristics, and low contagion or a dispersed pattern of patches. These landscapes may be heavily influenced by urban or agricultural land-use patterns, or the pattern of patches may reflect natural influences, such as rugged topography, the presence of lakes or rivers, or natural disturbances (figure 3).

The fourth group was composed of landscapes dominated by younger seral stages with high contagion or clumpiness. This landscape type was typical of managed forests in varying stages of re-growth. These landscapes had approximately equal proportions of patches in the clear-cut, young, and mature seral stages with little if any older growth. This suggests a managed landscape harvested in rotations (figure 3). The fifth group consisted of only one landscape that was anomalous because it consisted primarily of contiguous patches of the "Other" class, but also has approximately equal proportions of the clear-cut and mature classes.

Implementation of the landscape structure gradient

The landscape gradient described above was used to target specific areas for field visits to determine their potential as study sites. Two to three landscapes in each group were visited to field check the structure types as determined from satellite imagery and multivariate analysis; these visits confirmed the existence of the described structure types. We then focused our efforts on the fourth structure type (the intensively managed forests) as the landscapes of interest for wildlife sampling. These landscapes provided similar seral stages, an intensive management history, and a variety of patch sizes and configurations from which to choose. This enabled us to eliminate landscapr-level variance from the sampling scheme and to concentrate instead on stand-level influences.

After completing the initial search using this methodology, access rights to private managed forests were granted by the Champion and Weyerhaeuser timber companies. Site selection was accomplished by using forest stand maps developed from aerial photography and by extensive field surveys.

Conclusions

The development of this methodology enabled us to efficiently and effectively search a large area for specific landscape patterns or habitat conditions of interest. Locating suitable landscapes using conventional methods would otherwise have been very time consuming, if not impossible. The use of landscape pattern indices to develop gradients of landscape pattern proved invaluable in targeting our search to specific areas. This analysis demonstrated the applicability of integrating remote sensing, GIS, and pattern indices to characterize landscapes over broad regions. This methodology may also be applicable to forest managers, wildlife conservation efforts, and regional studies to characterize and search for specific landscape patterns or habitat conditions.

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