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An Experiment for Assessing Vertebrate Response to Varying Levels and Patterns of Green-tree Retention

Abstract

The emphasis of forest management in the Pacific Northwest has shifted recently from the production of timber resources to the maintenance or restoration of biological diversity and ecosystem functioning. New standards and guidelines for management emphasize the retention of forest structures (live trees, logs, and snags) to reduce logging impacts, to enrich reestablished stands with important structural features, and to enhance connectivity across forest landscapes. However, little is known about the effects on wildlife of varying the level and spatial distribution of retained structures in forests of western Oregon and Washington. Replicated and controlled experiments within the Demonstration of Ecosystem Management Options (DEMO) study are beginning to quantify the effects of varying the level and spatial aggregation of green-tree retention during forest harvest on a variety of ecosystem components (e.g., vertebrates, invertebrates, vegetation, fungi), as well as snow hydrology and social perceptions of these types of regeneration harvests. Eight replicate blocks of six experimental treatments have been established on the Umpqua National Forest in Oregon, and on the Gifford Pinchot National Forest and Capitol State Forest in Washington. The objectives of the wildlife studies are to quantify patterns of species richness, evenness, and relative abundance of birds, small mammals, bats, and amphibians before and after harvest to examine short-term treatment responses. Pre-treatment sampling has been completed on all sites, harvest treatments are in progress, and post-treatment sampling has begun. In this paper, we present an overview of our hypotheses and methods, and document the occurrence and relative abundance of species prior to harvest of the study blocks. Long-term studies of vertebrate response, habitat associations, and trophic interactions are planned. Results will inform managers on the consequences of alternative forest management strategies.

Introduction

Forest management in the Pacific Northwest during most of the last 50 years has emphasized the production of timber resources. Traditional practices of clearcut logging and artificial regeneration typically produce simplified stand structures that differ from the structural, compositional, and functional complexity of naturally regenerated forests, primarily in the absence of a significant legacy of large live and dead trees and down woody debris (Franklin et al. 1997). The loss of the old-growth legacy and attendant forest complexity has negatively affected the viability of associated species (Harris 1984, Mannan and Meslow 1984, Lehmkuhl and Ruggiero 1991, Ruggiero et al. 1991, FEMAT 1993) and has spurred the development of new forest management strategies for species conservation (e.g., USDA and USDI 1994).

New approaches for managing public forest lands seek to remedy problems associated with past management practices by integrating the maintenance of ecological values with commodity production (DeBell et al. 1997, Kohm and Franklin 1997, Tappeiner et al. 1997). Among these techniques are green-tree or variable retention systems. Retention of green trees in harvested stands has often been translated into the retention of a few

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scattered "wildlife" trees, with little apparent benefit to wildlife (McComb et al. 1993b). More sophisticated retention techniques that vary the level and spatial pattern of retained trees have been proposed to mimic the effects of natural forest disturbance, in particular the retention of large structural elements including live trees, snags, and down wood (FEMAT 1993, Franklin et al. 1997). The objectives of these techniques are to maintain ecosystem structure and function, enrich reestablished stands with structural features that would otherwise be absent, and enhance connectivity across the landscape (Franklin et al. 1997).

Presumably, vertebrate species that persisted within disturbance regimes of the pre-European settlement period may be expected to persist in managed forests if those regimes are mimicked. However, it is unclear how species will respond to our attempts to mimic natural disturbance regimes, as there are few data on the relationships between the levels and spatial patterns of green-tree retention and the short-term persistence or the long-term recolonization of species in harvested stands (McComb et al. 1993a, Chambers 1996, Hansen and Hounihan 1996, Franklin et al. 1997). Retrospective research has focused on species found in unmanaged late-seral and younger forests (e.g., Raphael 1988, Ruggiero et al. 1991, Rosenberg and Anthony 1993, McGarigal and McComb 1995, Gomez and Anthony 1996, Hagar et al. 1996), recent clearcuts (Medin 1986), or stands with evenly-distributed partial retention of green trees (Vega 1993, Hansen et al. 1995, Hansen and Hounihan 1996). Retaining more green trees might maintain components of stand structure or function that will allow some species to persist in harvested areas, but may result in poorer persistence of other species (Hansen et al. 1995, Hansen and Hounihan 1996, Chambers et al. *in press*). Retaining trees in undisturbed refugial patches likewise might result in smaller initial impacts or faster long-term recovery, depending on the life history and population structure of the species. Although the results of relevant experimental studies with pre- and post-treatment data are beginning to emerge (e.g., Chambers et al. *in press*), no studies have been published that examine the effects on vertebrate communities of simultaneously varying both the level and spatial pattern of green-tree retention (Franklin et al. 1997).

As a component of the Demonstration of Ecosystem Management Options (DEMO) study (Aubry et al. 1999), we have been studying responses of forest vertebrate species to experimentally manipulated levels (percentage of basal area) and patterns (dispersed vs. aggregated) of green-tree retention. Our primary goals are to: (1) quantify short-term changes in community composition and abundance of birds, small mammals, bats, and amphibians; (2) identify the vegetation and other habitat attributes associated with observed changes in species occurrence and abundance; and (3) quantify functional trophic relationships (e.g., fungivory) of select species groups.

In this paper we provide an overview of our study area, describe our experimental and sampling designs, briefly review the ecology and habitat relationships of vertebrate groups targeted for study, describe hypothetical short-term (5-10 yr) responses of vertebrates to treatments and the analytical techniques to test these hypotheses, briefly summarize pre-treatment results, and describe the challenges and limitations of implementing an experimental study of this scope on vertebrates. This paper does not contain a detailed analysis of pre-treatment data. Hypothetical responses of vertebrates to the retention treatments are based on current knowledge of species' ecologies and the hypothesized responses of habitat elements to these treatments as described by Halpern et al. (1999). Aubry et al. (1999) provide detailed descriptions of overall project goals, experimental design and harvest treatments, and the full scope of DEMO research (vegetation, fungi, invertebrates, social perceptions, snow hydrology).

Methods

Study Area

Study sites occur in Douglas-fir (*Pseudotsuga menziesii*) dominated forests on the Umpqua National Forest in western Oregon, and on the Gifford Pinchot National Forest and Capitol State Forest in western Washington (see Aubry et al. 1999). The eight study blocks (four in each state) provide a broad geographical and ecological scope of inference (Aubry et al. 1999). Within each block, six study units were selected from the same or adjacent watersheds to reduce differences in physiographic and stand characteristics, and to reduce differences resulting from past management. In contrast, blocks vary widely in biophysical setting and past management. Variation within and among blocks is accounted for in the sampling design and analytical techniques (see below).

Among the blocks, study sites range from 200-1700 m in elevation, slopes vary from steep to gentle, and nearly all aspects are represented. Stands encompass a variety of disturbance histories, ages, and successional stages, but all have developed beyond the stem-exclusion stage of forest development (*sensu* Oliver and Larson 1990). All stands on the Umpqua National Forest have been thinned or salvaged to some degree, whereas those on the Gifford Pinchot National Forest are all natural unmanaged forests. Two blocks (Layng Creek, Oregon and Capitol State Forest, Washington) are second growth as a result of previous harvest. Forest understories are compositionally diverse, but many blocks share the same dominant herbaceous and woody species. Halpern et al. (1999) provide detailed site descriptions and an overview of the variation among blocks.

Our results should be applicable to portions of the western Cascade and Coast Ranges with similar biophysical conditions and species distributions. Many, but not all, of the vertebrate species of interest occur throughout both regions. Where biogeographic variation in species distributions and responses to treatments may occur (e.g., Carey and Johnson 1995), the block design of the experiment will help to control for some regional differences in treatment effects.

Experimental Design

A randomized block design allows us to control for biophysical differences across the regional study area. At each of eight study blocks (replicates) five harvest treatments and a control were randomly assigned to 13-ha treatment units. After extensive survey of candidate stands across the potential study area, 13 ha proved to be the maximum stand size that could accommodate the minimum sampling grid (described below) and provide for homogenous within-stand biophysical conditions. Treatments vary in the level of retention of live trees (15-100% basal area) and in the pattern of retention (trees uniformly dispersed vs. aggregated). The six treatments are: 100% retention (control); 75% aggregated retention (trees harvested in three circular, 1-ha gaps); 40% dispersed retention throughout the stand; 40% aggregated retention (as five undisturbed, circular, 1-ha patches); 15% dispersed retention; and 15% aggregated retention (as two undisturbed, circular, 1-ha patches). All snags will be retained in forest aggregates. In harvested portions of treatment units, existing snags will be retained where possible, and 6.5 snags/ha will be created from live trees (see Aubry et al. [1999] for details).

Sampling Design and Analytical Methods

Sampling for vertebrates and vegetation within each treatment unit occurred on a permanently marked 8 x 8 or 7 x 9 grid, depending on the shape of the stand, with 40-m spacing between grid points and a 40-m buffer to the edge of the treatment unit. This grid configuration was chosen to conform to sampling methods for arboreal rodents, which require the largest sampling area among all species studied, and to allow for systematic sampling of treatments. Carey et al. (1991) recommend 10 x 10 grids with 40-m spacing (16 ha) to estimate the density of northern flying squirrels (*Glaucomys sabrinus*), but indicate that smaller grids (e.g., 8 x 8 or 7 x 9, [13 ha]) are adequate if the goal is to calculate relative abundance indices rather than density (see also Carey et al. [1996]). Because larger grids could not be accommodated given the sizes of intact stands, we accepted the smaller grid (see Abbott et al. [1999], Aubry et al. [1999] for additional details).

Field methods were designed to index habitat use by numbers of detections or individuals captured. We assume that use will be strongly correlated with changes in abundance for most species of interest. For some species with large ranges of movement relative to the area of treatment units (e.g., bats and some birds), use may not be closely tied to changes in abundance. Thus, the most ecologically meaningful information will be gained from species that occur at moderate to high densities and that exhibit local patterns of movement relative to the size of treatment units.

Data were collected for 2 yr prior to treatment to quantify species occurrences and relative abundances. Comparison of pre- with post-treatment data in testing treatment responses will allow us to control for spatial variation in vertebrate occurrence and abundance among treatment units within blocks. Comparison of data between treatment and control units will control for temporal variation. Although comparisons of pre- and post-treatment data from manipulated and control stands enable us to separate annual variation in species abundances from the effects of harvest treatments, an understanding of long-term (> 5 yr) responses will require repeated measurements over longer time frames.

We analyzed the power of the experimental design to detect treatment effects on a subset of bird and small mammal species. We chose a focal species (northern flying squirrel), four common and one wide-ranging bird species, and one moderately abundant terrestrial small mammal species that we knew to be well sampled and could be anticipated to be either greatly or minimally affected by treatments. We used pre-treatment data to estimate mean and variance of captures. Treatment response was hypothesized to be at least a 40% (birds) or 50% (small mammals) reduction in captures between the control and one treatment—the 40% dispersed retention treatment—based on the interactions of individual species life histories, habitat relationships, and the level of canopy removal. Nearly all treatment effects are expected to result in greater declines; thus, the test understates the expected magnitude and frequency of treatment effects and gives conservative power estimates.

Treatment effects on individual species abundances (for species meeting minimum criteria for occurrence or abundance in treatment units) and on community attributes of species richness and evenness will be tested as the difference between pre- and post-treatment values among treatment and control units using randomized block analysis of variance (ANOVA) (Skalski and Robson 1992) or nonparametric analogs. Repeated-measures ANOVA will be used to test for long-term treatment effects. Similarity in faunal associations among and within treatments will be examined by clustering or ordination techniques. Relationships between vegetation or other habitat attributes (e.g., volume of woody debris) and species presence or abundance will be examined across all replicates primarily by regression analysis.

Hypotheses, Methods, and Preliminary Results

Birds

Review

Some bird populations are declining in the Pacific Northwest (Sharp 1990, 1992), in part as a result of timber harvesting that shifts tree age-class distributions from old to young and alters natural disturbance regimes (Franklin and Forman 1987, Spies and Franklin 1989, FEMAT 1993). There are concerns for cavity-nesting birds (Morrison et al. 1986, Chambers et al. 1997), neotropical migrants (Hagan and Johnston 1992, Martin and Finch 1995), the northern spotted owl (*Strix occidentalis caurina*) (Forsman et al. 1996), and marbled murrelet (*Brachyramphus marmoratus*) (Carter and Morrison 1992, Nelson and Sealy 1995, Ralph et al. 1995) that have fueled public debate over forest management in the Pacific Northwest.

Significant research has been devoted to bird populations in natural, unmanaged Douglas-fir forests of western Washington and Oregon (Carey et al. 1991b, Gilbert and Allwine 1991a, Huff et al. 1991, Manuwal 1991), and there have been some retrospective studies of birds in managed forests of the region (Artman 1990, McGarigal and McComb 1992, Hansen et al. 1995, Hagar et al. 1996, Hansen and Hounihan 1996). However, replicated manipulations that include pre- and post-harvest sampling (e.g., Bosakowski 1997, Chambers and McComb 1997, Chambers et al. *in press*) have been limited in size and scope.

Bird species associated with mid- to late-successional forests that use large trees for foraging or as nest sites (Brown 1985) typically decline in response to harvest (Franzreb 1977, Keller and Anderson 1992, Chambers 1996, Hansen and Hounihan 1996), although intensity of harvest affects responses differently. Compared with clearcutting, 25% retention of green-tree basal area offers benefits (foraging and nesting habitat) to some of the species associated with old-growth forests (Chambers 1996, Chambers et al. *in press*).

Neither Chambers (1996) nor Vega (1993) detected a significant change in bird species richness among retention-treatment, late-successional forest, and clearcut stands ≥8 ha in area, although the composition of the bird community changed after harvest. However, retention stands provided breeding habitat both for birds associated with early seral stages and for some species associated with late-successional forest. The retention of large trees apparently provided foraging substrates and in some cases breeding sites for some species associated with old-growth forest. However, less intensive harvesting (approximately 30% reduction in wood volume) resulted in decline or loss of fewer bird species associated with late-successional forests (Medin and Booth 1989, Chambers 1996, Chambers et al. *in press*). Thus, a variety of stand treatments may be necessary to maintain habitat for different species.

Hypotheses

Hypothesis 1—*Abundance of canopy-dwelling birds will decline with decreasing levels of tree retention; aggregation of retention will reduce this effect (Table 1).* Species whose primary habitat is closed-canopy forest, such as hermit/Townsend's warbler (*Dendroica occidentalis/D. townsendi*) and the golden-crowned kinglet (*Regulus satrapa*), likely will decline in abundance with the loss of canopy habitat. Aggregation of patches will allow for some persistence in treatment units.

Hypothesis 2—*Abundance of birds associated with understory vegetation will decline with decreasing levels of tree retention in the short term (<5 yr); abundances will increase in the longer term. Aggregation of retention will ameliorate the effects of decreasing retention (Table 1).* Some vegetation layers, understory shrubs in particular, are predicted to decline in the short-term after harvest (Halpern et al. 1999), but will recover or be replaced by other understory species over long periods of time (>5 yr).

Hypothesis 3—*Primary and secondary cavity-nesting birds will decline with decreasing levels of tree retention and by dispersing the pattern of retention (Table 1).* Increased levels of harvest will result in marked reductions in snags in treatment units with ≤40% retention (Halpern et al. 1999). Old decayed snags, which are valuable for some cavity-nesting birds (northern flicker [*Colaptes auratus*], red-breasted nuthatch [*Sitta canadensis*]), will be lost in treated areas, and creation of replacement snags from sound trees (see Aubry et al. 1999) will have little effect in the short-term. Aggregation of retained trees in undisturbed refugia will likely ameliorate the effect of declining retention on cavity-nesters that use open canopy habitat (e.g., chestnut-backed chickadees [*Parus rufescens*]).

Methods

Bird community composition, species richness, and relative abundance were estimated in each stand from four point-count stations using the modified variable circular plot method described by Reynolds et al. (1980). Spot mapping was conducted during point counts, following procedures by Ralph et al. (1993), to examine locations of territories of selected species (Table 2) relative to canopy and forest floor characteristics before and after harvest.

Bird surveys began in late April and ended by early- to mid-July. Each treatment unit was visited six times during the breeding season. Surveys were evenly spaced throughout the breeding season to capture variation in breeding phenology among bird species. Abundance was expressed as a detection rate (mean number of birds detected within 75 m per visit) to facilitate comparison of species abundance among stands. Count stations were at least 160 m apart, thus 75 m was chosen as a conservative maximum detection distance to avoid recounting individual birds. We combined detections of hermit warblers and Townsend's warblers because of extensive hybridization between the two species (Rohwer and Wood 1998). We later will assess habitat quality and verify breeding status of all birds by using a reproductive index (Vickery et al. 1992) to determine whether species are nesting in the study stands.

Power analyses for four common species showed adequate power in the point-count sampling design to detect changes in detection rates of hermit/Townsend's warblers (0.99), golden-crowned kinglets (0.99), dark-eyed juncos (*Junco hyemalis*) (0.97), and brown creepers (*Certhia americana*) (0.96). Power was inadequate (0.52) for the less common and wider-ranging gray jay (*Perisoreus canadensis*).

Summary of Pre-treatment Results

Using the point-count data, we selected those species for which some analyses might be possible in one or more sites. Criteria for selection were species that were observed within 75 m of the observer and species that had 30 or more detections in a block in either of the two pre-treatment sampling sessions. A threshold of 30 detections was evident in the frequency distribution of pre-treatment detections and was thus used to separate common from uncommon species. In Washington, 15 of 51 species detected fit the criteria for analyses and in Oregon, 12 of 70 species (Table 2). There should be adequate power to

TABLE 1. Predicted short-term (5-10 yr) responses of vertebrate species to green-tree retention treatments in the Demonstration of Ecosystem Management Options study. Response is the predicted change in abundance from measured pre-treatment values: 0 = no detectable effect, - = decline in abundance, + = increase in abundance. Single, double, or triple entries indicate hypothesized responses that are weak, moderate, or strong, respectively. Double entries separated by a comma indicate the range of uncertainty in predicted response.

Species	Level and Pattern of Retention				
	75%	40% Dispersed	40% Aggregated	15% Dispersed	15% Aggregated
Birds					
Hammond's flycatcher	+	+	0	-	–
Pacific-slope flycatcher	-	–	–	—	—
Chestnut-backed chickadee	-	–	-	–	-
Red-breasted nuthatch	-	–	-	—	–
Brown creeper	-	-	-	—	–
Winter wren	-	–	–	–, —	–, —
Hermit thrush	0	–	-	—	-
Swainson's thrush	0, -	0, -	0, -	-	-
Varied thrush	-	–	-	—	–
Golden-crowned kinglet	-	–	-	—	–
Black-throated gray warbler	0	0	0	-	0, -
Hermit/Townsend's warbler	-	–	-	—	–
MacGillivray's warbler	+	+	+	++	++
Wilson's warbler	0	-	-	-, –	-, –
Dark-eyed junco	0	0	0	0	0
Mammals					
Montane shrew[1]	0	-, 0	-, 0	-, 0	-, 0
Fog shrew[1]	0	0	0	0	0
Trowbridge's shrew[2]	0, -	-, –	-, –	–, —	-, —
Vagrant shrew[1]	0, +	+	+	++	++
Shrew-mole[2]	0, -	–	-, –	—	–, —
Coast mole[2]	0	-	-	-	-
Siskiyou chipmunk (OR)[3]	0	0	0	-	-
Townsend's chipmunk[3]	0	0	0	-	-
Douglas' squirrel[3]	0	–	-, –	—	–, —
Northern flying squirrel[3]	-	–	-, –	—	–, —
Deer mouse[1]	0, +	+, ++	+, ++	+, +++	+, +++
Forest deer mouse (WA)[2]	0	-	-	-, –	-, –
Bushy-tailed woodrat[3]	0	-	-	-	-
Southern red-backed vole (WA)[2]	-	-	-	–	–
Western red-backed vole (OR)[2]	-	-	-	–	–
Creeping vole[1]	0, +	+	+	++	++
Amphibians					
Northwestern salamander	0	-	-	–	–
Rough-skinned newt	0	-	-	–	–
Ensatina	0	0, -	0, -	-	-
Tailed frog (WA)	-	–	–	—	—

[1] Generalist species or species associated with early-successional stages of forest development
[2] Species associated with closed-canopy forests
[3] Arboreal or semi-arboreal species

TABLE 2. Within-block pre-treatment rank abundance of 17 songbird species with >30 detections/block/year in the DEMO experimental green-tree retention study in Oregon and Washington during 1995 and 1996. Abundance is inversely correlated with rank (i.e., 1 indicates highest abundance).

		Oregon Blocks				Washington Blocks			
Species	Common Name	Watson Falls[1]	Little River	Layng Creek	Dog Prairie[2]	Butte	Little White Salmon	Paradise Hills	Capitol Forest
Dendroica occidentalis/D. townsendi	Hermit/Townsend's warbler[3]	1	1	2	1	1	1	1	7
Regulus satrapa	Golden-crowned kinglet[3]	4	2	7	2	2	3	2	5
Troglodytes troglodytes	Winter wren[3]	is[4]	5	1	is	4	5	5	1
Sitta canadensis	Red-breasted nuthatch[3]	2	3	5	5	8	2	3	10
Junco hyemalis	Dark-eyed junco[3]	5	6	is	4	3	4	6	13
Certhia americana	Brown creeper[3]	6	4	4	3	5	6	4	6
Empidonax difficilis	Pacific-slope flycatcher[3]	is	9	8	is	7	12	12	2
Parus rufescens	Chestnut-backed chickadee[3]	7	7	3	is	6	8	7	4
Wilsonia pusilla	Wilson's warbler	is	is	9	nd[5]	nd	11	is	3
Catharus guttatas	Hermit thrush[3]	8	8	is	is	9	7	8	is
Empidonax hammondii	Hammond's flycatcher[3]	3	10	6	is	is	is	is	is
Oporornis tolmiei	MacGillivray's warbler[3]	9	is	10	nd	is	is	nd	is
Perisoreus canadensis	Gray jay	is	is	is	is	10	9	10	11
Ixoreus naevius	Varied thrush[3]	is	is	is	nd	is	is	9	9
Carduelis pinus	Pine siskin	is	is	is	is	11	10	11	is
Catharus ustulata	Swainson's thrush[3]	is	is	is	is	nd	is	is	8
Dendroica nigrescens	Black-throated gray warbler	nd	is	is	nd	nd	is	nd	12
Mean annual total detections		1768	1742	1550	1353	1335	1830	1641	1882

[1] 1994 and 1995 data
[2] 1996 data only
[3] Species additionally sampled with spot mapping
[4] is = insufficient sample size (i.e., < 30 detections per block)
[5] nd = not detected

show treatment effects for at least the four species found in significant numbers in all blocks.

Hermit/Townsend's warblers, golden-crowned kinglets, red-breasted nuthatches, and brown creepers were fairly abundant on all blocks in both states (Table 2). As expected, species that occupy large territories (e.g., grouse, woodpeckers, and corvids) generally were detected in small numbers and are not included in the list. Of the 15 species in Washington blocks, four were either not detected or detected only in small numbers on some blocks. This was true for seven species in Oregon.

Some elevational trends in species abundance also were evident. For example, winter wrens (*Troglodytes troglodytes*), Wilson's warblers (*Wilsonia pusilla*), and Pacific-slope flycatchers (*Empidonax difficilis*) generally were most abundant in low-elevation blocks (Capitol Forest, Layng Creek). Resulting regional and elevational differences in bird species composition and abundance between blocks in Oregon and Washington will make future analysis over the total array of treatments potentially difficult for some species.

Arboreal and Terrestrial Small Mammals

Review

The small mammal community of the Pacific Northwest is one of the richest in North America (Harris 1984, Raphael 1990). Several species, such as the Trowbridge's shrew (*Sorex trowbridgii*), shrew-mole (*Neurotrichus gibbsii*), forest deer mouse (*Peromyscus keeni*), western red-backed vole (*Clethrionomys californicus*), and red tree vole (*Phenacomys longicaudus*), occur only in western British Columbia, Washington, Oregon, or northern California (Hagmeier and Stults 1964, Simpson 1964). These species are well adapted to older forests (Harris 1984, Aubry et al. 1991) and typically occur in greatest abundance in naturally regenerated mature forests (Carey and Johnson 1995).

Arboreal and terrestrial small mammals play important ecological roles in forest ecosystems. They form key links in many food webs as consumers and prey, and are important dispersers of plant propagules (Harris 1984). For example, northern flying squirrels, red tree voles, and woodrats (*Neotoma* spp.) are primary prey for the northern spotted owl and other forest owls (Forsman et al. 1984). Some arboreal and terrestrial rodents consume the sporocarps of ectomycorrhizal fungi and disperse fungal spores and nitrogen-fixing bacteria (Fogel and Trappe 1978, Li et al. 1986) and thus represent an important link in the nutrient cycling and productivity of forests.

Arboreal and semi-arboreal species in the study areas are consistently found in closed-canopy forest that has developed beyond the stem-exclusion stage. This pattern is generally attributable to the greater compositional and structural diversity found in old forest age classes (Carey 1991, 1995). These species can, however, be relatively abundant in younger stands with diverse understory composition and woody debris legacies (Doyle 1990, Rosenberg and Anthony 1993, Carey 1995). However, they are heavily influenced by traditional forest management practices that remove part or all of the forest canopy (Carey 1991, 1995).

The diversity of terrestrial small mammals in the study areas precludes a simple generalization about abundance in relation to forest age and habitat features. Many investigators have studied small mammals in old-growth forests, in naturally regenerated younger stands, and in recent clearcuts (<20 yr old) (Hooven and Black 1976, Ramirez and Hornocker 1981, Martell 1983, Van Horne 1983, Raphael 1988, Morrison and Anthony 1989, Ruggiero et al. 1991, Carey 1995, Carey and Johnson 1995). Several species (red-backed voles, Trowbridge's shrew, shrew-moles) are most abundant in closed-canopy forests that have developed beyond the stem-exclusion stage, where sparse herb and shrub layers provide limited resources for these species. Several other species are habitat generalists (deer mouse [*Peromyscus maniculatus*], montane shrew [*Sorex monticolus*]), or are most abundant in dense, ground-level vegetation characteristic of early-successional communities and streamside areas (creeping vole [*Microtus oregoni*], Pacific jumping mouse [*Zapus trinotatus*]). Clearcut logging produces unfavorable habitat for closed-canopy species until canopy closure. Long-term adverse impacts on persistence (over several rotations) may occur if harvest techniques eliminate inputs of coarse woody debris and other resources. Conversely, high-intensity disturbances will lead to an increase of early-successional small mammal species. Generalist species also may increase in abundance with disturbance as a direct effect of habitat changes and indirectly if the disturbance adversely affects competitors.

Although the general changes in the composition of small mammal communities can be predicted following clearcutting, (Hooven and Black 1976, Morrison and Anthony 1989) responses to less intensive harvest are not clear. Recent studies have examined small mammal response to green-tree retention harvests (Waters and Zabel 1995, Chambers 1996, Von Treba et al. 1998), but these studies were not designed to provide broad inference for Pacific Northwest species. In particular, little information is available to compare effects of dispersed vs. aggregated retention, or to predict patterns of response across a gradient of increasing disturbance intensity. Similarly, the ability of retention patches to serve as refugia for closed-canopy species is unknown.

Hypotheses

Hypothesis 1—*Abundance of arboreal rodents will decline with decreasing levels of retention of green trees, although aggregation of retention will reduce this effect (Table 1).* Decreasing retention will result in marked reduction in arboreal canopy habitat and den snags in treatment units with <40% retention (Halpern et al. 1999). Old decayed snags, which are most valuable for denning sites, will be lost in treated areas, and creation of replacement snags from sound trees will have little benefit in the short-term. Similar declining trends in the number of vegetation layers and the cover of the dominant understory shrubs (e.g., *Acer circinatum*, *Berberis nervosa*, *Gaultheria shallon* and *Vaccinium* spp.)—especially ericaceous shrubs—will contribute to the hypothesized response (Carey 1995).

Northern flying squirrels likely will show greater responses to treatments than chipmunks (*Tamias* spp.), which are semi-arboreal (Carey 1991). However, aggregation of retained trees in undisturbed refugia likely will ameliorate these effects to some extent. Declining abundances of arboreal rodents also will be associated with declines in the diversity and abundance of ectomycorrhizal fungi as a consequence of canopy removal (Cazares et al. 1999). Northern flying squirrels, which are obligate fungivores, are expected to experience greater treatment effects than chipmunks, which have more diverse diets.

Hypothesis 2—*Terrestrial small mammals associated with closed canopy (post stem-exclusion) forests will decline with increasing levels of disturbance (Table 1); treatment-level abundances will not differ between dispersed and aggregated harvests of the same intensity.* We expect clumped populations of these species will be found in retention patches of aggregate treatment units, with few individuals occupying the harvested matrix. In dispersed retention patches, these species will be found in low abundance throughout the unit.

Hypothesis 3—*Small mammal species associated with early-successional habitats and habitat generalists will increase or have no change in abundance with disturbance intensity.* The relative abundance of these species will increase with the proportion of basal area removed. Patterns of dispersion for these species will be the inverse of the patterns hypothesized for closed-canopy species.

Methods

Methods for live-trapping arboreal rodents were modified from Carey et al. (1991a). Tomahawk 201[1] traps (Tomahawk Live Trap Company, Tomahawk, Wisconsin) were used to sample an 8 x 8 or 7 x 9 sampling grid within each treatment unit. Sampling occurred in the fall to estimate reproductive status, relative abundances, and consumption of hypogeous fungi by flying squirrels and chipmunks. Captures were reported as individuals captured per 100 trap nights, adjusted by 0.5 trap nights for sprung traps; however, mark-recapture population estimates will be made if further analysis shows that capture data can meet the proper assumptions.

Sampling for 2 wk on all units during 1994 and 1995 indicated that flying squirrel abundances were too low (averaging 0-1 squirrel per grid) in three of eight blocks for future analysis of treatment effects, and that an additional 2 wk of sampling resulted in about 25% more captures of individual squirrels. As a result, we discontinued sampling arboreal rodents in the four blocks with low to marginal abundance, but doubled the sampling period to 4 wk at the remaining four blocks. Power analysis showed that doubling the trapping sessions to 4 wk on four blocks would result in power $\geq$0.90 for detecting a 50% reduction in abundance.

To estimate the species and quantities of fungi consumed, fecal pellets were collected from flying squirrels, chipmunks, and red-backed voles on the Watson Falls block in Oregon and the Butte block in Washington. Frequency of occurrence and relative abundance of fungal spores in feces will be compared to the concurrent frequency and abundance of fungi sampled as part of the DEMO mycological studies (Cazares et al. 1999).

Pitfall traps were used to sample terrestrial small mammals at each point in the grid. Only pitfall traps were used because they sample those species whose habitat associations are least understood, i.e., the insectivores. Most other small mammal species, although less efficiently captured by pitfall traps, are generally caught in sufficient numbers for statistical analyses (Aubry et al. 1991); power analysis of preliminary pitfall captures for deer mice (*Peromyscus* spp.) showed power to be ≥0.99 for detecting 50% reductions in abundance. Power to estimate effects on Trowbridge's shrew and vagrant shrew (*Sorex vagrans*), which were more abundant than deer mice, is expected to be similar.

Pitfall traps were operated as removal traps to provide data comparable to previous studies in unmanaged forests (Ruggiero et al. 1991) and to ongoing studies in managed forests in the Pacific Northwest (K. B. Aubry, pers. comm.). Traps were partially filled with water as recommended by the American Society of Mammalogists (1987), opened for 28 days in each treatment unit, and checked once a week between September and November. Numbers caught were reported as an uncorrected catch per unit effort index (captures/100 trap nights).

Summary of Pre-treatment Results

Ten rodent species were captured using arboreal rodent trapping methods during 1995-96 (Table 3). Chipmunks were the most frequently captured group. Townsend's chipmunk (*Tamias townsendii*) was abundant on all Washington blocks and on the Layng Creek block in Oregon. The Siskiyou chipmunk (*T. siskiyou*) was the most abundant species on the Watson Falls block in Oregon. Altogether, 568 northern flying squirrels were captured. Small numbers of bushy-tailed woodrats (*Neotoma cinerea*) were captured on several blocks. Douglas' squirrels (*Tamiasciurus douglasii*) were captured infrequently on all blocks, but this species is probably underrepresented in the data due to trap avoidance (Carey et al. 1991a).

Captures of terrestrial small mammals were similar in Oregon and Washington, with shrews (*Sorex* spp.), deer mice (*Peromyscus* spp.), and

TABLE 3. Within-block pre-treatment rank abundance of rodent species captured in Tomahawk live traps in the DEMO experimental green-tree retention study in Oregon and Washington during 1995 and 1996. Capture ranks (1 = highest abundance) were determined from the total number of individuals captured for each species within a block.

Species	Common Name	Oregon Blocks		Washington Blocks	
		Watson Falls	Layng Creek	Butte	Capitol Forest
Tamias townsendii	Townsend's chipmunk[1]	na[2]	1	1	1
Tamias siskiyou	Siskiyou chipmunk[1]	1	na	na	na
Glaucomys sabrinus	Northern flying squirrel	2	2	2	2
Tamiasciurus douglasii	Douglas' squirrel	3	3	5	3
Spermophilus lateralis	Golden-mantled ground squirrel	4	—[3]	na	na
Spermophilus beecheyi	California ground squirrel	5	—	—	—
Neotoma cinerea	Bushy-tailed woodrat	6	4	3	—
Spermophilus saturatus	Cascade golden-mantled ground squirrel	na	na	4	—
Tamias amoenus	Yellow-pine chipmunk	—	—	6	—
Sciurus carolinensis	Eastern gray squirrel	—	—	—	4
Mean annual total individuals		251	286	321	153

[1] Distributions of Townsend's and Siskiyou chipmunks in Oregon according to Verts and Carraway (1998).
[2] na = species' range outside of sample blocks
[3] — = not captured

red-backed voles (*Clethrionomys* spp.) comprising approximately 80% of the capture totals (Table 4). Rank order of mammalian captures was fairly consistent among blocks, although the southern red-backed vole in Washington had high abundance in dense-canopy, high-elevation blocks (Paradise Hills and Butte) and intermediate abundance in open-canopy (Little White Salmon) or low-elevation (Capitol Forest) blocks.

Given the pre-treatment capture totals it appears that we will have sufficient samples of Townsend's chipmunk, Siskiyou chipmunk, and the northern flying squirrel to test for treatment effects (Table 3). For the terrestrial small mammals we should have sufficient samples for Trowbridge's shrew, vagrant shrew, fog shrew (*Sorex sonomae*, after Carraway 1990), montane shrew, southern red-backed vole, western red-backed vole, deer mouse, and forest deer mouse (Table 4). Three other species (shrew-mole, creeping vole, Pacific jumping mouse) may be included in the statistical analysis depending upon their post-treatment responses.

Bats

Review

There is growing concern for the future of microchiropteran bats in intensively managed forest landscapes throughout the world (e.g., Lunney et al. 1985, Thomas 1988, Limpens and Kapteyn 1991, Thomas and West 1991). Two issues are of special importance: (1) reduction or loss of suitable day-roost sites in young forests and (2) changes in foraging ecology through alterations in the composition and abundance of prey and in the configuration of foraging space. Most work to date has addressed the first of these two issues.

The relationships of forest-dwelling bat species to forest structures and ultimately the silvicultural systems that produce them may be the least known for all vertebrate groups in the Pacific Northwest. Radiotelemetry indicates that bats use structures associated with old forests by roosting within the fissures and under the flakes of bark on large living trees and within the cracks and openings of large snags (Barclay et al. 1988,

TABLE 4. Within-block pre-treatment rank abundance of terrestrial small mammal species captured in pitfall traps for the DEMO experimental green-tree retention study in Oregon and Washington during 1995 and 1996. Abundance is inversely correlated with rank (i.e., 1 indicates highest abundance).

		Oregon Blocks				Washington Blocks			
Species	Common Name	Watson Falls	Little River	Layng Creek	Dog Prairie	Butte	Little White Salmon	Paradise Hills	Capitol Forest
Sorex trowbridgii	Trowbridge's shrew	1	1	1	1	1	1	3	1
Clethrionomys gapperi	Southern red-backed vole	na[1]	na	na	na	2	5	1	4
Clethrionomys californicus	Western red-backed vole	2	2	2	2	na	na	na	na
Sorex vagrans	Vagrant shrew	3	5	7	4	12	7	11	10
Peromyscus maniculatus	Deer mouse	4	3	4	3	5	4	6	6
Sorex sonomae	Fog shrew	5	4	3	5	na	na	na	na
Neurotrichus gibbsii	Shrew-mole	6	7	5	7	8	10	11	8
Microtus oregoni	Creeping vole	7	6	8	6	7	9	9	7
Mustela erminea	Ermine	8	10	10	10	9	11	10	11
Zapus trinotatus	Pacific jumping mouse	9	8	6	9	—[3]	—	—	—
Scapanus orarius	Coast mole	9	9	9	8	—	—	—	—
Peromyscus keeni	Forest deer mouse	na	na	na	na	3	2	2	2
Peromyscus species[2]	Juvenile deer mouse	—	—	—	—	6	3	5	3
Sorex monticolus	Montane shrew	na	na	na	na	4	6	4	5
Glaucomys sabrinus	Northern flying squirrel	—	—	—	—	10	12	8	9
Tamias townsendii	Townsend's chipmunk	—	—	—	—	11	8	7	12
Mean annual total individuals		633	1144	687	744	662	581	374	1038

[1] na = species' range outside of sample blocks
[2] Taxon not identified to species level
[3] — = not captured in all blocks within a state

Christy 1993, Frazier 1997). When studied in forests with continuous canopy ranging from 55 to 700 yr-of-age, echolocation calls were 2.7 to 5.7 times more frequent in older (>210 yr) than in younger forest (Thomas and West 1991). When studied over the much smaller range of stand ages found on private, intensively managed forest lands (from clearcut areas to about 65 yr-of-age), echolocation calls were about two times more frequent over clearcut areas than in young forest (Erickson and West 1996).

Concomitant with greater echolocation call frequency over clearcut areas is a shift in composition of calls to a higher proportion of larger bodied, non-*Myotis* species, a pattern also found by Hayes and Adam (1996). While there is little doubt that intensive forest management can lead to loss of forest structures that are important to bats and could result in bat population decline, the effects of partial harvests and green-tree retention on bat use of forested areas are not clear (Fenton et al. 1998). Given the provision of suitable day roosts, bat populations may do quite well. Critical work on the characteristics of roosts is very recent for this region (Campbell 1993, Erickson 1993, Ormsbee 1996, Frazier 1997). No studies have addressed bat habitat use on a large-scale experimental basis.

Hypotheses

Hypothesis 1—*Use of green-tree retention units will increase with harvest intensity for non-Myotis species.* As flight space for larger-bodied and faster flying bats is created, echolocation calls of silver-haired (*Lasionycteris noctivagans*), hoary (*Lasiurus cinereus*), and Townsend's big-eared bats (*Corynorhinus townsendii*) will be detected at higher numbers. The number of echolocation calls from *Myotis* species will not change. Thus, the number of echolocation calls summed across bat species will increase with harvest intensity.

Hypothesis 2—*Feeding rates will increase with harvest intensity.* Clearcut areas function as commuting space and as marginal areas for feeding. Closed-canopy stands function primarily for roosting. As harvest intensity increases, the frequency of "feeding buzzes" (a characteristic call indicating prey capture) should increase.

Methods

Indices of bat activity were estimated with automated divide-by-N detectors (Anabat II detectors and delay switches, Titley Electronics, Ballina, N.S.W., Australia). The systems were set to record at dusk and shut down 8-9 hr later. Several detectors were operated simultaneously at blocks within each state. Each block was surveyed for a 2-day period each month. Sampling for echolocation calls began in late June or early July depending upon the timing of warm weather. We found that a minimum of 6 nights of sampling per site (excluding windy or rainy nights) was necessary to account for the high variation in recorded calls among nights. Although it is generally not possible to distinguish repeated calls made by an individual bat from individual calls of many bats, the technique offers a very good method for determining presence, relative use, and behavior (feeding vs. passover) of bats in an area.

To provide information on age, sex ratios, reproductive condition, and species identification of *Myotis* species, we captured bats near habitual flyways, streams, and ponds with mist nets and collapsible harp traps (Kunz and Kurta 1988). Upon release, echolocation calls of captured bats were recorded to augment our library of reference calls.

Summary of Pre-treatment Results

At present we can identify the following five species or species groups (Erickson and West 1996): (1) big brown bat, (2) hoary bat, (3) silver-haired bat, (4) Townsend's big-eared bat, and (5) *Myotis* species (*M. evotis, M. volans, M. keenii, M. ciliolabrum, M. californicus, and M. lucifugus*).

A total of 3,924 calls were detected during the pre-treatment sampling. In Washington, the *Myotis* group accounted for 98% of the 2,102 recorded detections (Table 5). Other species were rarely detected. Detections at the Little White Salmon block accounted for over half (62%) of all detections in Washington and 33% of calls for both states. In Oregon, the *Myotis* species group accounted for 97% of the 1,822 recorded calls.

Nightly activity was highest on the Little White Salmon block, which has relatively open-canopy stands, with an average of ≥17 calls per night. This range of activity is as high as that found in old-growth (Thomas and West 1991). The Little River block was the second most active site with just over eight calls per night. The least active blocks were the relatively closed-canopy Butte and Paradise Hills blocks with <3 calls per night.

TABLE 5. Pre-treatment mean detection rate (echolocation calls/night), and standard error of mean, for all bats, all *Myotis* species, and all non-*Myotis* species in DEMO experimental green-tree retention blocks in Oregon (1995 and 1996) and Washington (1995).

Block	Total $\bar{x}$ (se)	*Myotis* $\bar{x}$ (se)	Non-*Myotis* $\bar{x}$ (se)
Oregon			
Watson Falls	7.89 (11.22)	7.71 (11.20)	0.11 (0.72)
Little River	8.21 (10.39)	8.14 (10.32)	0.07 (0.30)
Layng Creek	4.92 (6.43)	4.43 (5.66)	0.45 (2.55)
Dog Prairie	4.06 (9.97)	4.06 (9.97)	0
Washington			
Butte	2.53 (11.04)	2.45 (10.98)	0.06 (0.23)
Little White Salmon	17.81 (29.58)	17.26 (29.16)	0.51 (2.99)
Paradise Hills	1.56 (2.86)	1.55 (2.96)	0.04 (0.19)
Capitol Forest	6.90 (14.87)	6.90 (14.87)	0

As expected, feeding activity was low within all blocks. Of the 3,924 detections, only 35 were identified as feeding activity. Little White Salmon had the highest number of feeding buzzes, accounting for 66% of the feeding activity.

Amphibians

Review

In regions west of the Cascade crest, amphibians dominate the Pacific Northwest herpetofauna. Not only are there more species of amphibians than reptiles, but they are more distinctive in their endemism (Nussbaum et al. 1983). In headwater streams and riparian areas of the Pacific Northwest, amphibian adults and larvae are the top predators, easily exceeding the numbers and biomass of other forms (Bury 1988). Densities of amphibians can be very high, up to 12 per m^2 in moist areas (Leonard et al. 1993).

Several amphibian taxa are being revised and a number of cryptic species have been described (Good 1989, Good and Wake 1992, Green et al. 1997). Recent identification of new species underscores the disjunct nature of amphibian distributions and the potential for extinction. Although there have been studies of habitat relationships in unmanaged forests for the systematically stable species of the region (Raphael 1988, Aubry and Hall 1991, Corn and Bury 1991, Gilbert and Allwine 1991b), we know little about habitat associations for many of the newly identified species in unmanaged forests. Information in managed forests is particularly sparse for most species. Well-replicated studies of amphibian occurrence in managed forest are needed to investigate the effects of several commonly employed silvicultural practices, including short-rotation harvest, thinning, and green-tree retention. Several species have special status in Oregon and Washington: Cascade frog (*Rana cascadae*), tailed frog (*Ascaphus truei*), spotted frog (*Rana pretiosa*), clouded salamander (*Aneides ferreus*), Oregon slender salamander (*Batrachoseps wrighti*), torrent salamanders (*Rhyacotriton cascadae* and *R. variegatus*), Larch Mountain salamander (*Plethodon larselli*), Dunn's salamander (*P. dunni*), and Van Dyke's salamander (*P. vandykei*). Information from our studies will be important for conservation of these sensitive species.

Hypotheses

Hypothesis I—*Abundance of amphibians will decline with harvest intensity.* We expect a general decline for all species with increasing harvest intensity, with the exception of the western red-backed salamander (*Plethodon vehiculum*) in Washington where the species has shown an ability to persist in clearcut areas. We expect a sharp decline in species richness, but this may occur over a longer time frame than the initial 2-yr postharvest sampling period. Several amphibian species are long lived and are capable of withstanding adverse conditions for prolonged periods. Thus, we may see a lag response for this pattern, underscoring the need to resample these sites in the future (perhaps at 5 and 10 yr after the initial postharvest sampling).

Hypothesis 2—*Amphibians will persist in forest patches in aggregated retention units, but most species will decline in harvested areas within aggregated retention units and in dispersed retention units.* We expect that the retained forest patches may be sufficient to permit survival of a portion of the original amphibian community, perhaps until the forest canopy closes again. We expect the numbers of amphibians persisting in the forest patches to be positively correlated with the total area remaining in patches.

Hypothesis 3—*Individuals captured in the harvested portions of the aggregated retention units and in the 15% dispersed retention units will show poorer body condition compared to individuals found within the forest patches or in the 40% dispersed retention units.* Adverse conditions in harvest areas should lead to poor body condition for individuals residing there. We would expect reduced reproduction and lower than expected values for body-condition indices (such as a regression of weight vs. snout-vent length).

Methods

Amphibians were sampled concurrently with terrestrial small mammals during fall using pitfall traps. Traps were opened after the onset of fall rains when amphibian surface activity increases from the relative inactivity of summer. After identifying species and recording information on body dimensions, mass, sex and reproductive condition, amphibians were toe-clipped and released or held under cool, moist conditions for the remainder of the sampling periods then returned to their point of capture. As with small mammals, numbers caught were expressed as an uncorrected catch of individuals per unit effort index (captures/100 trap nights).

Pitfall trapping is a good technique for capturing surface-active amphibians, but is not very effective at sampling species that limit their movements to surface cover or the interior of large woody debris. Fortunately, only two species limit their surface activity, the clouded salamander and the Oregon slender salamander, both of which occur only in Oregon. Time-constrained searches and cover-board sampling techniques to target these species initially were planned, but were not conducted because of budget constraints and the localized distributions of the species.

Summary of Pre-treatment Results

The rank ordering of amphibian captures showed considerable variation among blocks and between states (Table 6). Amphibian abundance and diversity were about two times higher in Washington than in Oregon. Five species were common to all blocks in Washington, but only two species were present on all blocks in Oregon. In Washington, species composition varied considerably among blocks (Table 6). This high variability will make statistical detection of post-treatment effects difficult. However, the ensatina (*Ensatina eschscholtzii*), which comprised 54% of all amphibian captures across the eight blocks should be a suitable species for detecting treatment effects across both states. In Washington, the northwestern salamander (*Ambystoma gracile*) and the tailed frog were also common enough to suggest that treatment effects will be detectable.

Limitations and Challenges

In an undertaking of this kind and magnitude one expects challenges. We have experienced several difficulties thus far; many were expected and others were not. For example, we anticipated that the patchiness of species' distributions would make uniform replicates impossible. There are species in each vertebrate group that are found only in subsets of the study blocks. This will result in lower power for treatment comparisons. In addition, the scale of silvicultural operations does not always mesh well with the scales of species' biology. Given the small size of harvest units relative to the home ranges of larger-bodied species, some species will not respond exclusively to the effects of the experimental treatments. They may show partial responses to the treatments, but may be influenced by conditions adjacent to the treatment areas. Our choices of species for the study were made partly with this in mind. We chose those species for which we had the best chance of measuring a response attributable to the treatments, i.e., those species with limited movement or those tied to small areas by seasonal territoriality. This approach cannot be entirely successful, and while the scaling will be acceptable for small mammals, amphibians, and several avian species, it may be unsatisfactory for others such as the larger cavity-nesting birds.

TABLE 6. Within-block pre-treatment rank abundance of amphibian species captured in pitfall traps for the DEMO experimental green-tree retention study in Oregon and Washington during 1995 and 1996. Abundance is inversely correlated with rank (i.e., 1 indicates highest abundance).

		Oregon Blocks				Washington Blocks			
Species	Common Name	Watson Falls	Little River	Layng Creek	Dog Prairie	Butte	Little White Salmon	Paradise Hills	Capitol Forest
Ensatina eschscholtzii	Ensatina	1	3	1	1	2	1	5	1
Plethodon vehiculum	Western red-backed salamander	—[1]	—	2	3	—	10	—	2
Ascaphus truei	Tailed frog	—	—	—	—	1	3	1	6
Ambystoma gracile	Northwestern salamander	2	1	5	2	3	2	2	3
Taricha granulosa	Rough-skinned newt	3	2	3	—	6	5	4	4
Ambystoma macrodactylum	Long-toed salamander	—	—	—	—	—	4	7	—
Rhyacotriton cascadae	Cascade torrent salamander	—	—	—	—	—	8	3	—
Rana aurora	Red-legged frog	nd[2]	nd	nd	nd	5	10	11	5
Rana cascadae	Cascades frog	nd	nd	nd	nd	—	6	6	—
Hyla regilla	Pacific tree frog	4	—	—	3	4	10	9	—
Rana spp.	Unidentified frog	5	6	—	3	nd	nd	nd	nd
Dicamptodon tenebrosus	Pacific giant salamander	—	5	6	—	—	8	8	—
Bufo boreas	Western toad	—	—	—	—	7	6	9	—
Aneides ferreus	Clouded salamander	—	4	4	3	na[3]	na	na	na
Plethodon dunni	Dunn's salamander	—	—	6	—	na	na	na	na
Mean annual total individuals		55	114	90	140	56	204	132	467

[1] — = not captured
[2] nd = *Rana* species not differentiated in Oregon blocks
[3] na = species' range outside of sample blocks

We primarily will use indices of abundance as treatment response variables rather than indicators of realized fitness (e.g., juvenile survival). However, because density or abundance estimates can be misleading indicators of habitat quality (Van Horne 1983), we will attempt to estimate indices of fitness for vertebrates when feasible. Nest searches and territory mapping in conjunction with counts will help to determine treatment effects on the number of breeding bird territories. Determination of the age class and reproductive condition of small mammals and amphibians captured in pitfall traps will provide estimates of age-class distributions and reproductive condition in treatment units. Age-class distributions and estimates of reproductive condition in live-caught rodents (chipmunks, flying squirrels) will help to indicate realized fitness relative to treatments. The presence or absence of bat roosts will indicate sensitivity of bats to treatments.

The costs associated with these studies are very high. It has been necessary to rework the sampling design in response to budgetary shortfalls. These considerations have led to reduced sampling efforts for arboreal rodents (reductions in trapping intensity per period and loss of seasonal sampling), small mammals (elimination of simultaneous live trapping), and amphibians (elimination of time-constrained searches).

Our final concern recognizes that not all responses to treatments will occur within the initial 2-yr post-harvest sampling period. We expect lag responses for several species groups within the amphibians, arboreal rodents, and birds. We expect species packing in the aggregated retention patches for species in all of these groups, a phenomenon that may not begin to dissipate until the second or even third post-harvest year. These concerns point to the need for intensive, long-term sampling to understand how key forest species or species groups will respond to stand-level manipulations, the effects of which may be manifested over decades. Funding agencies must likewise be willing to provide continuing support if they wish to successfully manage for species whose responses to harvest activities may be dynamic over long periods of time.

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Endnote

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