BIODIVERSITY

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Active Intentional Management (AIM) for Biodiversity and Other Forest Values

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ABSTRACT

Comparisons of natural and managed forests suggest that neither single-species management nor conventional forestry is likely to successfully meet meeting broad and diverse conservation goals. Biocomplexity is important to ecosystem function and capacity to produce useful goods and services; biocomplexity includes much more than trees of different sizes, species diversity, and individual habitat elements. Managing multiple processes of forest development, not just providing selected structures, is necessary to restore biocomplexity and ecosystem function. Experiments in inducing heterogeneity into second-growth forest canopies not only support the importance of biocomplexity to various biotic communities including soil organisms, vascular plants, fungi, birds, small mammals, and vertebrate predators but also suggest management can promote biocomplexity. At the landscape scale, strategies emphasizing reserves and riparian corridors that do not take into account ecological restoration of second-growth forest ecosystems and degraded streams may be self-fulfilling prophecies of forest fragmentation and landscape dysfunction. Restoring landscape function entails restoring function to second-growth forest. Intentional management can reduce the need for wide riparian buffers, produce landscapes dominated by late-seral stages that are hospitable to wildlife associated with old-growth forests, provide a sustained yield of forest products, and contribute to economic, social, and environmental sustainability.

KEYWORDS: Biodiversity, northern spotted owl, northern flying squirrel, trurrles, keystone complex, mosaics.

INTRODUCTION

Societies demand much of their forests. When people were scarce and primeval forest abundant, forests were impediments to progress. Now people are abundant, forests are reduced in area, and most temperate forests are second growth. As the pace of social evolution quickens, many look to naturalistic forests for wholeness, stability, and permanence. Millenarian forests of the Pacific Northwest, USA, especially provide a “cathedral” that is spiritually evocative. Forested lands, nevertheless, continue to be converted to other uses even as the value of forests as providers of ecological services such as green space, clean water, clear air, and carbon sequestration are increasingly apparent. Furthermore, as citizens in democracies question governments about intragenerational social justice for people, desire for intergenerational equity (leaving a good world to future generations) and ecological justice for other species flourish concomitantly (Ray 1996). Old-growth forests are mostly in reserves and parks, distant from people, and insufficient to provide various values in the amounts desired by people. Young forests with old-forest legacies may sustain biodiversity, and likely will develop high biocomplexity and “naturalness” if left untouched. Biocomplexity does much to promote human experiences of wildness, naturalness, and spirituality. But much second growth, even within reserves, is low in ecological capacity and may not develop into biologically complex forest capable of providing diverse values. Industrial “fiber farms” will not provide most values people seek—orderly plantations suggest human domination over nature devoid of wildness and naturalness with biodiversity intentionally suppressed.

The challenge we face is how to actively and intentionally manage (AIM) extra-reserve forests to provide the diverse values society demands. Comparisons of natural

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and managed forests show single-purpose management is unlikely to provide full societal value because complexity is more important than single habitat elements to both biodiversity and capacity to produce various goods and services (Carey 2003a). Zoning for single purposes accentuates fragmentation and can result in cumulative effects that negatively impact soil, hydrologic processes, aquatic habitats, population viability of rare species, native species through invasion by exotic species, and ability to manage efficaciously and efficiently. Restoring landscape function entails restoring biocomplexity to second-growth forests over broad areas and managing ecosystem and landscape processes and dynamics rather than static ecosystem and landscape structures (Carey 2003b). AIMing for multiple values is guided by empirical, experimental, and modeling data (Carey et al. 1999b). AIM also requires some reference condition by which to judge success. I use the ancient forests of the Pacific Northwest, which by their very persistence over 250-1,000 years demonstrate high biodiversity and robustness in the face of change. In particular, I use the arboreal-rodents keystone complex and forest-floor small-mammal community to set benchmarks and measure success of management.

The arboreal-rodent keystone complex of the northern spotted owl (*Strix occidentalis caurina*), northern flying squirrel (*Glaucomys sabrinus*), ectomycorrhizal fungi, and Douglas- fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*) provides a heuristic framework for analysis. The spotted owl is threatened with extinction due to loss of old (long-lived, resilient, persistent), naturally complex forests. The flying squirrel is the primary prey of the owl. Truffles (sporocarps of ectomycorrhizal fungi) are the primary food of the squirrel. The squirrel disperses truffle spores and associated microorganisms throughout the forest (Li et al. 1986). Mycorrhizal fungi enhance the ability of Douglas-fir (and other trees) to absorb water and nutrients from the soil and receive carbohydrates in return. The fungi move photosynthetic carbohydrates from trees to the mycorrhizosphere, a vast array of microbes, insects, nematodes, bacteria, and other soil organisms. If one expands consideration to the arboreal rodent group, then a broader food web is accounted for. For example, I measured the total and relative biomass (percent due to each species) of three squirrels, the northern flying squirrel, Douglas' squirrel (*Tamiasciurus douglasii*), and Townsend's chipmunk (*Tamias townsendii*), and compared them to the simultaneously high abundances of the three species in old-growth forest. Biomass of this species group is informative because it represents the energy available to medium-sized carnivores and the energy available to consumers in terms of the reproductive output of the forest ecosystem (flowers, seeds, berries, nuts, mushrooms, and truffles). The arboreal rodents are the major prey for most of the medium-sized predators in the forests (including hawks, owls, weasels, felids, canids). Although diets of the three species overlap, the flying squirrel is a truffle and mushroom specialist, the Douglas' squirrel a conifer seed specialist, and Townsend's chipmunk is a fruit generalist, feeding on conifers seeds, seeds and nuts of deciduous trees, berries of shrubs, and truffles (but relegated to areas of high shrub cover). Thus, not only does the combined biomass of these three species reflect fruiting by plants and fungi, it also determines the carrying capacity of the forest ecosystem for a diverse assemblage of the medium-sized predators. Thus, the combined biomass is a measure of ecological productivity and can be measured at the local (stand) scale (Carey et al. 1999b). If one moves up an additional level in the forest ecosystem, for example to vertebrate predators, populations often respond to landscape character or regional variation, like the spotted owl (Carey et al. 1992) and cannot be used to evaluate management at small scales.

Most would agree that the forest floor (litter, humus, and soil) functions as the foundation for sustainability of forest ecosystems. Biological activity in the forest floor depends on a variety of invertebrates that are important in nutrient cycling. Forest-floor invertebrates are abundant in both species and individuals, with species often unknown or poorly described. Still, changes in the structure of biotic communities in response to forest management generally are more informative than changes in the abundances of single species (Carey et al. 1999a, b; Carey and Harrington 2001). Thus, I stepped up one level in the forest ecosystem to measure the most tractable group of organisms dependent on forest-floor diversity and function. I measured the biotic integrity (composition and relative abundance of species compared to old forests) of the forest-floor, small mammal community that includes granivores, herbivores, fungivores, and insectivores, but which is particularly diverse in insectivores in Pacific Northwest forests (Carey and Harrington 2001). In various studies, I incorporated other biotic communities to form expanded complexes that are more comprehensive in evaluating forest ecosystem function than any single species, biotic community, or trophic pathway.

METHODS

Three types of studies sharpen active, intentional management: comparative or cross-sectional studies, experimental studies, and computer simulations. Initially cross-sectional studies focused on comparing natural forests of different ages—young (40 to 80 years), mature (80 to 200 years), and old (>200 years). In the 1980s, an interagency multi-university effort compared biotic communities in old growth, mature, and naturally young studies in multiple physiographic provinces in northern California, and western Oregon and Washington. Communities included vascular plants, belowground fungi, reptiles, amphibians, birds and mammals (summarized in Ruggiero et al. 1991). Then, I expanded these studies to compare managed young, naturally young, and naturally old stands. Similar geographical replication was used to study spotted owl habitat use, owl demography, owl prey bases, and associated food webs in natural and managed forests (summarized in Carey et al. 1992, 1999a; Carey and Harrington 2001; Forsman et al. 1993).

In 1991, drawing from comparative studies, I implemented two experiments in accelerating development of biocomplexity (high plants species diversity, spatial heterogeneity in the understory density and composition, and diversity in the vertical distribution of foliage) by inducing heterogeneity into second-growth forest canopies similar to that in old growth (fig. 1). Maps of natural mosaics had revealed 0.1-0.5-ha patches with a 2:1 ratio of closed to open canopy (Carey et al. 1999a). In the Puget Trough of Washington (summarized in Carey 2003a, Schowalter et al. 2003) and, later, across the Olympic Peninsula (Carey and Harrington 2001), I used variable-density thinning (VDT) to create canopy mosaics in a wide variety of second-growth stands (including previously intensively managed with conventional thinning, passively managed with legacies from old growth, and extensively managed without legacies, with or without thinning). I hypothesized that removing subordinate and codominant trees would make light, water, nutrients, and space available in various amounts to other vegetation though space in the thinned stand. Effects would extend beyond the altered patches of canopy because of low sun angles. Thus, fine-scale canopy heterogeneity would create an even more diverse mosaic of environmental conditions and numerous patch types in the understory (Canham et al.1990). For convenience, I will refer to stands that were subjected to VDT as *mosaics*; stands that originated after clearcutting but that still retained legacies from old growth, such as scattered large live trees, large dead trees, tall stumps of harvested large trees, and large fallen trees and associated biota as *legacy stands*; stands that had been managed intensively for timber with multiple conventional thinnings as *timber stands*; and to the common, extensive second growth as *second growth*.

Finally, in 1994, I led at a team that modeled alternative approaches to landscape management, including AIM to conserve biodiversity (Carey et al. 1999b, Carey 2003b).

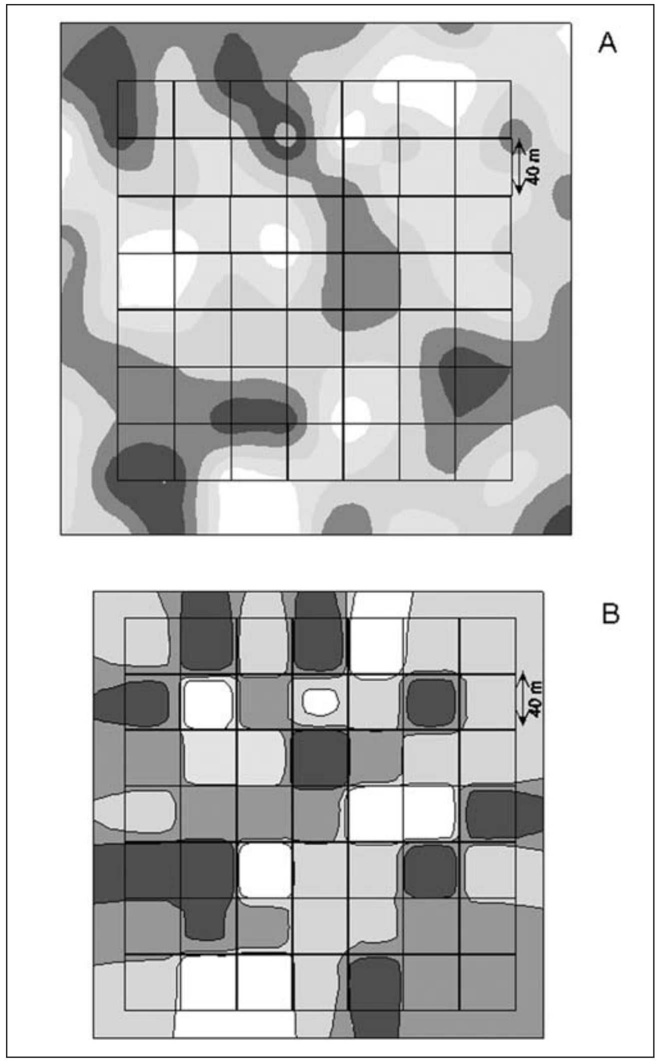

Figure 1—Recreating spatial heterogeneity characteristic of old forests in second-growth forest canopies with variable-density thinning: **(A)** densities of trees >50 cm at 1.5 m above ground in a 280-year-old *Pseudotsuga menziesii* forest, shading (light to dark) represents densities from 3 to 45 trees/ha, based on 225 sampling points (from Carey et al. 1999b); **(B)** relative densities of *P. menziesii* >20 cm at 1.5 m following variable-density thinning of a 56-year-old, second-growth stand; shading (light to dark) represents relative density classes of <3.25, 3.25-4.75, 4.75-6.75, and >6.75 (from Carey et al. 1999c).

Three final scenarios were chosen for full evaluation: protection with no manipulation; management to maximize net present value of wood production; and AIM for a multiplicity of values. Alternatives were evaluated on ability to maintain spotted owls (a threatened species), capacity to support vertebrate diversity (based on 130 species), relative forest-floor function (biotic integrity of the small mammal community), ecological productivity (biomass of arboreal rodents and numbers of deer and elk), timber productivity, and timber revenues.

RESULTS

Comparative Studies

In the Pacific Northwest, natural forests and contemporary managed forests differ in structure, composition, and function (Carey et al. 1999a, Carey and Harrington 2001, Franklin et al. 2002). In natural forests, many ecosystem elements are patchily distributed. These elements include live trees from the preceding stand, large fallen trees, trees with cavities used for denning and nesting, berry-bearing shrubs, deciduous trees, shade-tolerant trees in the midstory, forbs, mosses, and fruiting bodies (truffles) of ectomycorrhizal fungi, among others. Groups of these elements may be correlated in their occurrences due to topographic effects or canopy gap formation and, thus, form distinct patches. Diverse patches, arrayed in fine scale, contribute to emergent properties, such as simultaneously high abundances of potentially competing species that do not occur together in high abundances in large, homogeneous patches. The scale of variation in arrangement that contributes to this synergy is about 0.1 to 0.5 ha, or 80 to 100 m (Canham et al. 1990, Carey et al. 1999a). Biotic legacies from preceding forest, propagules from adjacent stands, forest structuring processes, and development of spatial heterogeneity interact to produce both overall compositional diversity and patch diversity (habitat breadth) (Carey et al. 1999a, Franklin et al. 2000). In contrast, stand tending focused on timber production, particularly even-age management, purposefully reduces spatial heterogeneity and compositional diversity (Carey 2003a, Carey and Harrington 2001). Consequently, the response of species to changes in abundances of particular elements of their habitat varies from place to place in managed forests, in relation to the relative abundances of other ecosystem elements. The diversity and structure (biotic integrity) of vertebrate communities, however, varies consistently in response to complexity of vegetation structure and absence of various compositional elements because spatial heterogeneity and compositional diversity (together, biocomplexity) in plant and fungal communities are prerequisite to preinteractive niche diversification, animal community diversity, and the ability of ecosystems to resist or recover from disturbance (Carey et al. 1999a, Tilman 1999). Forest management can promote biocomplexity (Carey et al. 1999a, b; Lindenmayer and Franklin 2002). Retention of legacies of individual live trees, dead trees, coarse woody debris, or even patches of forest can be used with even-age or uneven-age management systems (Franklin et al. 2000). Variable-retention harvest systems transcend traditional silvicultural conventions (Lindenmayer and Franklin 2002). Thinning influences all structuring processes, including decadence and development of spatial heterogeneity. Thinning with underplanting restores tree species diversity and accelerates canopy stratification and understory development (Thysell and Carey 2001). Retaining decadent trees, wounding trees, and inoculating trees with top-rot fungi promote decadence essential to ecosystem development (Carey et al. 1999a).

Experimental Results

Fungi—Historical management seemingly influenced the structure of soil food webs in experimental second-growth plots, but all retained fungi-dominated soils. Fungal to bacterial biomass ratios were higher in legacy than in timber stands for total biomass and for active biomass, however. Fungal mats covered two-thirds of the forest floor in legacy stands but only one quarter in timber stands. Compared to controls, VDT had no effect on total biomass ratios, but increased active biomass ratios in both timber mosaics and legacy mosaics. Total fungal biomass remained unchanged in timber mosaics but decreased in legacy mosaics. Truffle biomass averaged 0.5 kg/ha (ranging 0-1.8 kg/ha seasonally) in untreated timber and legacy stands. Of 28 species found in timber and legacy stands, 19 were in timber with 7 only in timber stands. Of the 28 species, 21 were in legacy stands with 9 only in legacy stands. *Rhizopogon* was the dominant genus, with a relative frequency of 40 to 47 percent. *Gautieria* and *Leucogaster* were more frequent in legacy than in timber stands and *Melanogaster* and *Hysterangium* were more frequent in timber stands. Truffle production was reduced overall in mosaics (from an overall frequency of 18 percent in control plots to 13 percent in VDT plots) in the short term, with heavily thinned patches most reduced (to 10 percent). Truffle diversity increased to 48 species in mosaics (vs. the 28 species in controls) and productivity quickly recovered. *Gautieria* and *Hysterangium* decreased in abundance in mosaics, but *Melanogaster* increased in species diversity and biomass. A total of 64 mushroom species were found before treatment, 37 (19 mycorrhizal) in legacy stands and 44 (15 mycorrhizal) in timber stands. Richness of ectomycorrhizal mushrooms was highest in legacy stands. After VDT, 108 mushroom species were found in legacy mosaics (vs. 89 species in legacy controls) and 78 species were found in timber mosaics (vs. 65 in timber controls).

Vascular plants—Legacy stands had 27 to 40 species of understory plants compared to 49 to 87 species in the timber stands. Of 91 species in timber stands, 51 were not found in legacy stands and 18 were nonnative species (1 tall shrub, 2 low shrubs, 13 herbs, and 2 grasses). Of 47 species in legacy stands, 4 were not found in timber stands, including the old-growth associate Pacific yew (*Taxus brevifolia* Nutt.), and 1 was nonnative. Community structure differed with management history, with timber stands dominated by aggressive clonal native shrubs and ferns. Timber stands had greater cover than legacy stands for total understory (88 vs. 34 percent), tall shrubs (12 vs. 5 percent), salal (25 vs. 13 percent), swordfern (16 vs. 3 percent), and brackenfern (9 vs. 2 percent). Compared to controls, mosaics initially had reduced understory cover and increased importance of 20 native and 11 exotic species. Two native species decreased in importance. Three years later, understory recovered, species richness increased by 150 percent, only 4 exotic species persisted in importance, and 8 natives increased and 7 natives decreased in importance. Underplanting in mosaics was reestablishing root-rot-resistant trees in root-rot pockets, increasing the resilience of the forest, and in other heavily thinned areas, increasing resistance to spread of root rot. After 10 years, some exotic species persisted at low levels and spatial patterning was beginning to emerge in the understory. Sparse natural regeneration of shade-tolerant conifers, markedly increased abundance of understory hardwoods, and good survival of underplanted trees of high to intermediate shade tolerance portends a rapid increase in understory diversification.

Small mammals—Timber stands had 1.5 times the numbers and 1.7 times the biomass of small mammals in legacy stands. Keen's mouse (*Peromyscus keeni*), a dominant species in natural forests, was rare in both legacy and timber stands. The creeping vole (*Microtus oregoni*) was inordinately abundant in timber stands (3rd ranked) compared to legacy stands (7th ranked) and natural stands. The montane shrew (*Sorex monticolus*) was also inordinately abundant in thinned stands (2nd ranked). Neither management history produced communities typical of natural forests. After VDT, deer mice (*Peromyscus maniculatus*), creeping voles, and vagrant shrews (*Sorex vagrans*) increased in abundance in mosaics. No species decreased in abundance. Northern flying squirrels were twice as abundant in legacy as in timber stands (1.0/ha vs. 0.5/ha). Townsend's chipmunks were opposite (0.2/ha vs. 0.8/ha). Douglas' squirrels were low in abundance in both timber and legacy stands (0.1/ha). Flying squirrels decreased in abundance in legacy mosaics immediately following VDT but recovered within 5 years. Chipmunks increased sharply in legacy mosaics following VDT and remained high. Douglas' squirrels did not respond to VDT in the short term. It remains to be seen whether flying squirrels and Douglas' squirrels will increase over time in mosaics as tree diversity increases and as trees increase photosynthetic activity and allocate additional carbon to seeds and ectomycorrhizal associates. California hazel (*Corylus cornuta* Marsh var. *californica* (A. DC.) Sharp) and bigleaf maple (*Acer macrophyllum* Pursh) are present and may begin producing high quality nuts and seeds for the squirrels in response to the more open canopy and available light.

Wintering birds—Species richness was higher in timber (16.2 ± 1.4 species) than in legacy stands (12.2 ± 1.0 species). Richness remained unchanged in timber mosaics, ranging from 14 to 22 species 3-5 years after VDT. Richness varied annually but was consistently higher in legacy mosaics than in legacy controls in post-VDT years 3 to 5 (annual richness was 12 to 16 species in mosaics vs. 10 to 16 species in controls). The proportion of stand area used increased in mosaics for 2 of 8 abundant species (*Troglodytes troglodytes* and *Melospiza melodia*). No species used legacy more than timber or mosaic stands. Cavity-excavating birds (Picidae) were present, but low in abundance, in all stands.

Modeling Results

Simply protecting second-growth forest caused the landscape to go through waves of forest development. Initially a substantial ecological crunch occurred because of degraded watersheds and oversimplified stands; a long time (200 years) was required for these stands to achieve an old-growth-like condition. Timber management with minimum constraints produced a landscape inhospitable to >20 vertebrate species and allowed no recovery of degraded streams; its sustainability was uncertain, but net present value was maximal. Timber management with riparian reserves drawn from federal guidelines produced relatively narrow, well separated strips of late-seral forest in the long term, unlikely to function fully as late-seral forest because of their continued adjacency to clearcut and young forests; clearcutting was intensified in the available uplands due to removal of streamside and adjacent small patches from forest from management. Biodiversity management, as it was designed to do, produced significant ecological benefits (table 1), including supporting a pair of spotted owls and numbers of deer and elk comparable to the timber management regime. AIM costs were surprisingly low—only a 15-percent loss in net present value compared to maximizing net present value of timber extraction. Assuming (as occurred) increased riparian protection would be mandatory and eliminating costs of improved riparian/mass-wasting management from comparisons, AIM resulted in only a 6-percent decrease in net present value. Other economic values increased: decadal revenues increased by 150 percent, forest-based employment quadrupled, and the wood products manufacturing sector diversified and relied more heavily on high quality wood products and value-added manufacturing (Lippke et al. 1999). Initially, I included a constraint of 30 percent of the landscape in late-seral forest to support one pair of spotted owls; the final shifting steady state mosaic

Table 1—Measures of landscape health in a western Washington landscape in the last 100 years of a 300-year simulation under management to maximize net present value of timber and active intentional management for multiple values

Ecological measure	Timber management	Biodiversity management
Vertebrate diversity (% of maximum possible)	64	100
Forest floor function[a] (% of maximum possible)	12	100
Ecological productivity[b] (% of maximum possible)	19	94
Landscape health (mean of the above 3)	32	98

Source: Adapted from Carey et al. 1999b.

[a] Composition and structure (relative abundances) of the forest-floor small mammal community measured against that in old growth.

[b] Biomass of northern flying squirrels, Townsend's chipmunks, and Douglas' squirrels measured against that in old growth.

maintained >50 percent of the landscape in late seral stages and <15 percent of the landscape was in clearcuts in any decade, resulting in a landscape fully permeable to dispersing late-seral species.

DISCUSSION

The similarity between fungal communities in timber and legacy stands suggests high resiliency in soil food webs in the face of active management (Carey 2003a, Carey et al. 2002, Schowalter et al. 2003). Timber management, however, reduced fungal dominance and macroscopic fungal mats. Mechanical disturbance appears to destroy fungal mats and promote *Melanogaster* over *Hysterangium* and *Gautieria*. Impacts on truffle production, however, were brief. The long-term impacts of forest management on truffle production remain unclear, with inconsistent results from across the region (Carey et al. 2002, Smith et al. 2002). Induced heterogeneity increased sporocarp diversity to a richness that rivals that in old growth (Carey et al. 2002). Retaining unthinned patches in mosaics might help conserve fungal mats and allow their recolonization as increased photosynthetic activity by trees increases the flow of carbohydrates to soil food webs and maintains high fungal diversity.

Conventional thinning produced timber stands with rich understories dominated by clonal natives with numerous exotics present. Achlorophyllous mycotrophs (parasitic plants without chlorophyll) were reduced in abundance by dense understory; retaining unthinned patches in mosaics would help conserve these species. Understories in the timber stands however, lacked both shade-tolerant trees and the spatial heterogeneity associated with diverse vertebrate communities (Carey et al. 1999a). Legacy forests had depauperate understories and low abundances of small mammals and wintering birds. Thus, neither historical management regime had placed stands on a trajectory toward developing the complexity and diversity of old-growth forests. Inducing mosaics increased diversity and abundance of native species but only slightly increased exotics. Underplanting is leading to increased spatial heterogeneity. With time, it now appears likely the course of both timber stands and legacy stands has been altered by inducing heterogeneity, and that development of biocomplexity, although far from ensured, is more likely than before treatment.

Timber and legacy management produced imbalanced mammal communities, with some species common in natural forests low in abundance. Inducing heterogeneity had immediate positive effects on forest-floor mammals, but shade-tolerant midstories and midstory deciduous trees (e.g., *Acer macrophyllum* Pursh) will be required to restore biotic integrity. Chipmunks increased markedly in legacy mosaics with only brief declines in flying squirrels. Flying squirrels remained rare (some of the lowest densities ever recorded in the Pacific Northwest) in the timber stands, perhaps due to open canopies that impede arboreal travel and dense understories that promote excessive chipmunk abundance, make foraging for truffles more difficult, and increase exposure to predation by large gaps between the understory and canopy. Heterogeneity had positive effects on winter birds; cavity-excavating birds, however, remained low in abundance. Promoting deciduous trees early in stand development provides both short-lived trees (e.g., *Alnus rubra* Bong) for cavity excavation and long-lived trees (e.g., *Acer macrophyllum* Pursh and *Arbutus menziesii* Pursh) for mammal dens. Legacy retention and decadence management is essential to maintaining cavity-excavating birds and forest-floor organic matter. It seems that intentional disturbance can produce spatial complexity at the fine scale that is important in restoring and maintaining biodiversity in second-growth temperate, boreal, and tropical forests (Canham et al. 1990; Carey et al., 1999a, b; Franklin et al. 1997, 2000, 2002; Lindenmayer and Franklin 2002).

Natural history, comparative, experimental, and modeling studies have demonstrated the value of diverse ecosystem elements from headwater seeps, to fallen trees, to deciduous components of conifer ecosystems; the importance of spatial heterogeneity in both the vertical and horizontal dimensions and within individual structures; temporal change (seasonal, interannual, decadal, and submillenial dynamics) and adaptation to changing environmental conditions; and small, intermediate, and large-scale disturbances as underpinnings of biodiversity and biocomplexity. Managerially induced homogeneity favors some species over others resulting in a loss of biotic integrity, benefiting globally abundant species and exotic species most. No active management after clearcutting also results in loss of biological diversity and biotic integrity.

Computer simulations suggest that AIMing for multiple values can produce wood, water, clean air, recreational opportunities, revenues to the land manager/owner/trust, biological diversity, viable populations of species associated with old forests, and healthy aquatic systems. Simulations of maximizing wood production and protecting forests without manipulation forecast production of reduced values, even reduced sustainable revenues, higher risks of loss of values, new costs, and lack of opportunity for the ecosystems to adapt to change. AIM, however, requires a suite of techniques, beginning with public involvement in setting goals and strategies to accomplish goals. Next, geotechnical analysis of watersheds can be used to identify areas of unstable slopes and potentially erodable colluvium that, along with headwater seeps and streams, and larger, fish bearing streams, can be buffered appropriately and provide a template for legacy retention. Watershed analysis is followed by design, construction, rehabilitation, and maintenance of efficient and low-impact transportation systems. Intentional ecosystem management then incorporates modern silvicultural methods arrayed into management pathways that provide for directional development of ecosystems in shifting steady-state mosaic landscapes. These methods might include variable-retention harvests systems that emphasize legacy retention as much as wood removal, multispecies planting, precommercial thinning to promote growth and biodiversity, multiple variable-density commercial thinnings to harvest wood while protecting legacies and inducing spatial heterogeneity to stimulate development of biocomplexity, multispecies management that includes deciduous trees and shrubs in conifers forests, and long or alternating long and short rotations. AIM has potential to contribute to economic, social, and environmental sustainability.

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The Effect of Forested Corridors Within Harvested Pine Plantations on Herpetofauna Assemblages

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ABSTRACT

We determined the effect of forested corridors within clearcuts on herpetofauna in the Coastal Plain of South Carolina. We studied four identically managed 20-ha plantations of loblolly pine (*Pinus taeda* L.). Three randomly chosen sites retained a 100-m unharvested forested corridor within a clearcut treatment, and the other site was an unharvested control. All sites were monitored from January 1997 through December 1999 (16 months preharvest and 20 months post-harvest) using two standard drift fence arrays. Forty-two species were captured totaling 2,681 animals, with a recapture rate of 8 percent. The number of anurans and salamanders captured in harvested areas did not differ significantly from the unharvested control. No significant difference was detected pre- and post-treatment evenness, Pielou's J, within treatment and control. Anuran abundance decreased, while Pielou's J increased significantly with increases in the density of post-treatment coarse woody debris. Despite intensive forest management, these sites continued to support diverse and herpetofaunal assemblages.

KEYWORDS: Harvesting, pine plantation, corridors, forest management, herpetofauna, clearcuts.

INTRODUCTION

Interest in using amphibians and reptiles as indicators of forested ecosystem health has increased in the last two decades (Bury and Corn 1988). Amphibians and reptiles are good bioindicators of environmental health because they are exposed to both aquatic and terrestrial environments, have small home ranges, and relatively limited dispersal (Blaustein et al. 1994, Blaustein and Wake 1990, Dunson et al. 1992). Amphibians are integral components of most ecosystems, often constituting the highest fraction of vertebrate biomass (Burton and Likens 1975, Kiester 1971). Amphibians play key roles in forest ecosystems, especially in forest detritus food webs, where they convert small invertebrate biomass into larger sized prey for other vertebrates (Pough 1983). Therefore, declines in amphibian and reptile populations could significantly affect other organisms.

Human activities affect many populations of amphibians, reptiles, and other animals. Forest management is one activity that has drawn particular attention. Although certain forest management practices benefit many wildlife species (Harlow and Van Lear 1981, Harlow and Van Lear 1987, Sweeney et al. 1993, Sweeney and Wigley 1999), their effects on amphibians and reptiles must be better understood before forests can be managed to benefit all wildlife (Raymond and Hardy 1991).

Most research on the effects of forest management on herpetofaunal assemblages was conducted in hilly or rocky terrain. Clearcutting in the Pacific Northwestern United States decreased the number of terrestrial amphibians by reducing the availability of moist microhabitats and by limiting foraging and reproductive opportunities (Dupuis et al. 1995). In North Carolina, clearcuts averaged about 60 percent fewer marked animals than forested controls the first year after harvest, and salamanders were completely absent 2 years post-harvest (Ash 1988). Logging in the southern Appalachians resulted in a decrease in salamander populations of 70 to 80 percent, and clearcutting on USDA Forest Service land in western North Carolina was estimated to eliminate 13.7 million salamanders annually (Petranka et

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al. 1993, Petranka 1994). Several other studies have also reported negative changes in amphibian assemblages following timber harvest (Blaustein and Wake 1990, Bury 1983, Corn and Bury 1989, Harpole and Haas 1999, Pough et al. 1987, Raymond and Hardy 1991).

Although amphibians may be negatively affected by timber management practices, effects may be temporary, as some herpetofaunal populations recover over time (Dodd 1991, Enge and Marion 1986, Pough et al. 1987). In South Carolina, managed pine plantations harbored more amphibian species and individuals than expected (Bennett et al. 1980). Long-term studies found more *Plethodon elongatus* in young stands than older stands, but these sites required increased sampling effort because of understory vegetation (Diller and Wallace 1994). However, interpreting results from most forest management studies is complicated by small sample sizes and lack of pre- and post-treatment data. Here, we report results from the first 3 years of a replicated 20-year experiment designed to examine the effect of Westvaco's Ecosystem-Based Multiple Use Forest Management System (Muckenfuss 1994), which uses maintained forested corridors within clearcuts, on herpetofauna assemblages.

STUDY AREA

We conducted the study in South Carolina on Mead-Westvaco's Jericho and O'Bryan Units located near the towns of Ravenel (Charleston County) and Cottageville (Colleton County), respectively. The study areas had slopes generally <2 percent and elevations of 20 to 30 m above mean sea level. The soil was somewhat poorly drained with a sandy loam or loam surface layer overlaying a heavy clay subsoil, a near-neutral pH and high levels of phosphorus. Sites were very productive for loblolly pine (*Pinus taeda* L.).

Study sites were located in intensively managed, 19-year-old loblolly pine plantations that had been sheared, root raked, and bedded when established. All sites were located within 16.09 km of each other. Mean diameter at breast height and tree density among all sites was 27.9 cm and 672.5 trees/ha, respectively. Vegetation consisted of a canopy of loblolly pine with a midstory of wax myrtle (*Myrica cerifera*), sweetgum (*Liquidambar styraciflua*), red maple (*Acer rubrum*), supplejack (*Berchemia scandens*), and grape (*Vitis rotundifolia*). The understory was composed of grape, poison ivy (*Toxicodendron radicans*), red buckeye (*Aesculus pavia*), heart-leaf (*Hexastylis arifolia*), Easter lily (*Zephyranthes atamasco*), false nettle (*Boehmeria cylindrical*), and Virginia creeper (*Parthenocissus quinquefolia*).

METHODS

To assess responses of herpetofaunal assemblages to timber harvesting with maintained corridors, we chose four identically managed 20-ha sites. Three were assigned at random to retain a 100-m wide unharvested forested corridor running the length of the site within a clearcut (Jericho A, O'Bryan B and C), and the remaining site was assigned as an unharvested control (O'Bryan).

To assess herpetofaunal responses, two standard drift fence arrays (Bury and Corn 1988) were located in the harvested portions (>25 m from the edge) of each treatment site (fig. 1), and within the unharvested control. Arrays were made from 15-m sections of 60-cm high aluminum flashing, placed in the ground 20 cm deep (Corn 1994) and stabilized with stakes anchored to the fence by cable ties (Gibbons and Semlitsch 1981). Pitfall traps were 7.5 L plastic buckets located at the array ends and in the center. We paired two double-ended funnel traps on opposite sides of each leg of the array, 7.5 m from the ends. Funnel traps were constructed of aluminum window screening and were 45.7 cm high x 45.7 cm wide x 76.2 cm long with an interior funnel opening 15.2 cm diameter. We removed all vegetation 30 to 40 cm from the base of the fence and any vegetation within 6 m of the ground that overhung the fence (Dodd and Scott 1994). All drift fence arrays were monitored from January 1997 through December 1999 (16 month preharvest and 20 month post-harvest) with traps opened for the same 8 days in two blocks of four consecutive days each month. When traps were open, they were checked daily for a total of 288 nights.

We recorded the species, sex, life stage, snout-vent length and total length (to the nearest mm), mass (to the nearest 0.01 g using an electronic balance), date and location of all animals captured. Salamanders, anurans, and lizards were cohort-marked by toe-clipping. Snakes and turtles were individually marked with 14 mm PIT tags and shell notching, respectively. After processing, we immediately released all animals on the opposite side of the fence.

We sampled first season (pretreatment) and third season (second year post-treatment) habitat variables to determine their relationship to herpetofaunal responses. We sampled understory vegetation and litter depth on the entire site, using a 60.35 x 60.35 m grid and >50 points/site. Understory vegetation was considered to be within 1.37 m of the ground. Down woody debris and bare soil greater than 5.08 cm in diameter was tallied. Ocular estimates of percentage of understory vegetation, woody debris, and bare soil were made for each point using the Geographic Resource

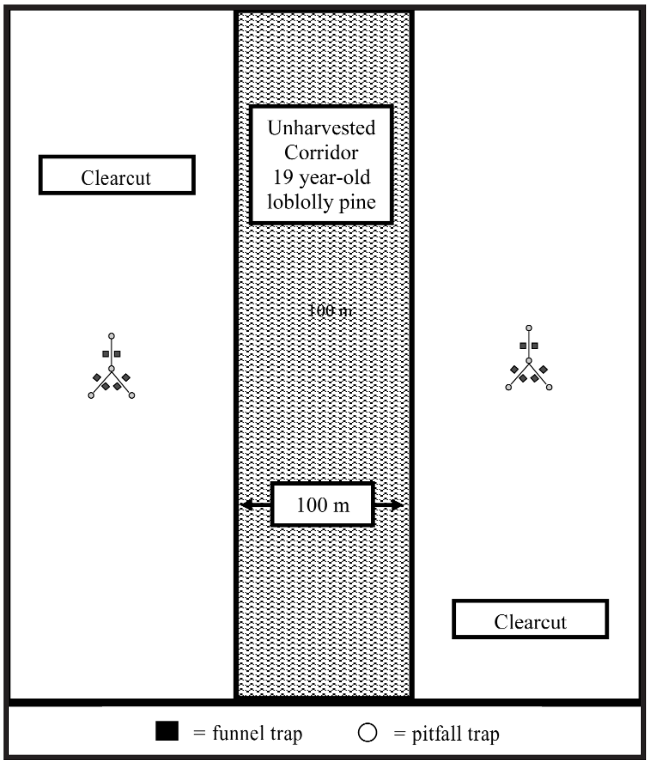

Figure 1—Trapping array for sampling a 20-ha loblolly pine stand for impacts of clearcutting with maintained forested corridors on herpetofauna in the Coastal Plain of South Carolina.

Solutions densitometer. We measured leaf litter depth at each point using a ruler.

We used repeated measures of analysis of variance and Type III sums of squares to test the null hypothesis that monthly capture differences preharvest and post-harvest for groups (anurans, salamanders, lizards, snakes, and turtles) were equal between clearcut/corridor and control. Monthly captures per site were totaled by group to form the sample units.

We examined traits including snout-vent length, total length, and mass for adults of commonly occurring species to determine effects of site and treatment. We used analysis of variance to test for differences in mean values (snout-vent, total length, and mass) by species, pre- and post-harvest. We then used Duncan's Multiple Range test to group means.

We calculated Margalef's species richness and Pielou's evenness index (J) for amphibian and reptile assemblages at each site and used a Students t-test to examine differences among sites and treatments. We used Pearson correlation analysis to examine patterns of herpetofaunal abundance, richness, and evenness in relation to differences in habitat attributes among sites. Significance level was set at $\alpha=0.05$ for all test. All analysis were conducted using the GLM and CORR procedures in SAS (SAS Institute, Inc. 1987).

RESULTS

Forty-two species were captured totaling 2,681 animals with a recapture rate of 8 percent (204). Anurans constituted 84 percent of all animals captured, followed by salamanders (10 percent), lizards (3 percent), snakes (2 percent), and turtles (1 percent) (table 1).

No difference was detected in the number of anurans, salamanders, lizards, snakes, or turtles captured in the harvested areas compared to the unharvested control. Statistical power was ≥46 percent.

We calculated scores for richness and evenness (J) (table 2) and were unable to detect a significant difference among site and between pre- and post-treatment.

Mean snout-vent lengths of marbled salamanders (*Ambystoma opacum*) did not differ significantly by site pretreatment or post-treatment, but total length differed significant by site pretreatment, with site A different from the other sites. However, no significant difference was detected post-treatment. No significant difference was observed pretreatment or post-treatment for mass. Snout-vent length for eastern narrow-mouthed toad (*Gastrophryne carolinensis*) was not significantly different by sites pretreatment or post-treatment. A significant difference was observed by site pretreatment for mass with site A having heavier animals. However, no difference was detected post-treatment. Southern toad (*Bufo terrestris*) did not differ across sites in snout-vent length or mass pretreatment and did not differ post-treatment for snout-vent length. Because of insufficient sample size for southern leopard frogs (*Rana utricularia*) pretreatment, no test of significance was conducted. However, there were no post-treatment differences for snout-vent length or weight for southern leopard frogs. Statistical power for all tests was ≥85 percent.

Anuran abundance was negatively correlated with coarse woody debris ($r = -0.99$, $p = 0.008$) post-treatment. Pielou's J indicia was found to be significantly associated with post-treatment coarse woody debris ($r = 0.98$, $p = 0.01$).

DISCUSSION

Numerous comparative studies have reported declines in herpetofauna population size and species richness within

Table 1—Number of individuals captured in clearcuts by site in intensively managed pine plantations in South Carolina, 1997-1999

	Harvest site A			Harvest site B			Harvest site C			Unharvested control		
	Year											
Species	1997	1998	1999	1997	1998	1999	1997	1998	1999	1997	1998	1999
Acris gryllus (southern cricket frog)	1	--	--	--	--	--	1	--	1	--	--	--
Bufo terrestris (southern toad)	49	36	22	37	31	51	42	23	70	51	11	64
Gastrophryne carolinensis (eastern narrowmouth toad)	27	20	11	76	155	74	267	94	88	66	36	51
Hyla squirella (squirrel treefrog)	2	3	--	2	29	8	8	20	12	2	2	--
Hyla sp. (all other species)	7	4	1	--	10	1	2	9	--	2	--	1
Pseudacris sp. (all species)	1	7	8	--	--	--	3	--	--	1	1	--
Rana clamitans (green frog)	13	--	1	16	--	2	19	--	--	--	--	--
R. utricularia (southern leopard frog)	18	63	23	--	87	46	3	168	163	1	11	19
Scaphiopus holbrookii (eastern spadefoot toad)	--	--	--	1	--	--	--	1	--	--	--	--
All anurans	118	133	66	132	312	182	345	315	334	123	64	135
Clemmys guttata (spotted turtle)	--	--	--	--	--	--	--	2	2	--	1	--
Kinosternon subrubrum (eastern mud turtle)	2	1	--	2	6	1	5	5	5	3	2	1
All turtles	2	1	--	3	7	1	5	7	7	3	4	2
Anolis carolinensis (green anole)	8	1	3	5	3	--	--	1	--	--	5	2
Cnemidophorus sexlineatus (six-lined racerunner)	--	1	--	--	--	--	--	--	--	--	--	--
Eumeces sp. (all species)	3	--	2	--	--	--	1	--	--	3	1	2
Scincella lateralis (ground skink)	1	1	4	1	1	--	2	--	3	3	1	8
All lizards	12	3	9	6	4	--	3	1	3	6	7	12
Coluber constrictor (northern black racer)	2	--	4	--	1	--	3	1	1	--	--	--
Diadophis punctatus (southern ringneck snake)	--	--	--	--	--	--	1	1	1	--	--	1
Lampropeltis getula (eastern king snake	--	--	--	--	--	1	--	2	1	--	--	--
Nerodia sp. (all species)	--	--	--	1	1	--	--	1	2	--	--	--
Thamnophis sirtalis (eastern garter snake)	--	--	2	--	--	--	--	2	1	--	--	--
All snakes	2	1	9	1	2	1	7	7	6	1	3	1
Ambystoma mabeei (Mabee's salamander)	--	--	--	2	1	--	--	--	--	1	--	--
A. maculatum (spotted salamander)	--	--	--	1	1	--	16	3	9	--	1	3
A. opacum (marbled salamander	23	9	4	7	7	5	10	16	26	22	11	38
A. talpoideum (mole salamander)	1	--	--	2	2	2	2	4	13	4	2	3
Eurycea quadridigitata (dwarf salamander)	--	--	--	--	--	--	--	--	--	--	1	--
Plethodon variolatus (SC slimy salamander)	4	1	1	--	--	--	7	1	--	--	--	1
All salamanders	28	10	5	12	11	7	33	24	48	27	25	45

Table 2—Species richness and evenness indices pre- and post-harvest for herpetofauna captured in pine plantations in South Carolina, 1997-1999

Site	Margalef		Pielou J	
	Pretreatment	Post-treatment	Pretreatment	Post-treatment
Harvest site A	2.99	4.51	0.73	0.65
Harvest site B	2.63	3.01	0.52	0.56
Harvest site C	3.03	3.60	0.41	0.56
Unharvested control	3.14	3.35	0.57	0.64

recently clearcut habitats. However, we found no significant differences in the number of amphibian captures among control and treatment sites. This is in contrast with other studies that have documented significantly lower abundance of amphibians in clearcut stands as compared to forested controls (Ash 1988, deMaynadier 1996, Petranka et al. 1993, Sattler and Reichenback 1998). The results of our study, however, are similar to those of Corn and Bury (1991) who documented no significant differences in total abundance of salamanders between clearcuts and forested areas in the Pacific Northwest, but noted that some species appeared to be more sensitive to canopy removal than others. In Kentucky, Adams et al. (1996) reported no observed difference in numerical abundance or species richness among silvicultural prescriptions. In the southern Cascade Range in Washington, all species captured in unmanaged forests also occurred in intensively managed stands (Aubry 2000).

We detected a decline in lizards from the harvested areas which was due to two species, green anole (*Anolis carolinensis*) and ground skink (*Scincella lateralis*). The pooled reptile data showed no declines in abundance, however, which is similar to the findings of Adams et al. (1996), who indicated enhanced species richness and abundance of reptiles following timber harvesting. The presence of slash in clearcuts has been hypothesized to enhance cover for reptiles and their prey in South Carolina (Phelps and Lancia 1995).

We detected a slight decline in the number of amphibians in the second year of the study for the control. The second year of the study partially overlapped with the first year of harvest on the treatment sites. During the second year, there was a moderate drought during the spring and summer months (National Oceanic and Atmospheric Administration 1998). However, there were two 5-cm rains that generated several stagnant pools on the harvested sites, which produced large numbers of anurans. No pools were observed on the control sites. It is hypothesized that the difference in breeding sites was related to a decrease in evapotranspiration in the harvested sites. This hypothesis is also supported by well-water data from the same soil type, which documented an increased water table level following clearcutting (Xu et al. 1999).

We found no significant change in evenness among pre-and post-treatment, while H was significantly different. In Texas, H and J indices were significantly greater in clearcuts than control treatments (Foley 1994). Our work is supported by work in Kentucky that reported no difference among silvicultural prescriptions in abundance or species richness of amphibians, but diversity varied among prescriptions (Adams et al. 1996). Life history traits of various taxa may contribute to discrepancies among studies; therefore, caution should be used when using diversity or evenness indices for comparison between habitats or studies.

We thought that increased exposure of the forest floor to sun and wind would desiccate harvested habitats, thus reducing its quality for amphibians, which would result in decreased growth or health. However, we found no such change in growth or health of commonly occurring species. In another study that examined the effects of a clearcut on individual traits, no significant effects of clearcut were detected on body size or whole body and egg nonpolar lipid percentages for mole salamanders (*Ambystoma talpoideum*) (Chazal and Niewiarowski 1998), although mole salamanders in Chazal and Niewiarowski's study were not exposed to the mechanical disturbance of clearcutting. In our study, we documented no direct effects of timber harvest on individual traits.

We found that the percentage of woody debris was either insignificant or had a negative relationship with abundance. Exposed mineral soil showed no significant relationships. DeMaynadier (1996) also reported similar findings for downed woody debris. DeGraaf and Rudis (1990) found that litter depth was the only variable that was related to herpetofaunal community measures.

Overall, clearcutting had very little effect on the herpetofaunal assemblages in the intensively managed pine plantations we studied. Soil and litter depth may have contributed to the maintenance of herpetofauna populations during and following harvesting activities on these fertile Coastal Plain soils. In addition, the family Ambystomatidae, which comprised a majority of the salamanders in our study, may be more resistant to habitat changes than other genera previously studied. Despite intensive forest management, these sites continue to support diverse and abundant herpetofauna communities.

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Vegetation Response to Alternative Thinning Treatments in Young Douglas-fir Stands

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ABSTRACT

The Young Stand Thinning and Diversity Study was designed to test whether management of young stands, specifically alternative thinning regimes, can accelerate development of late-successional habitat and, therefore, enhance biodiversity while maintaining wood production goals. The study is located in the lower Cascade Range of central Oregon and consists of four replications of four thinning treatments (treatment areas average 30 ha each) in 30- to 50-year-old second-growth Douglas-fir stands (*Pseudotsuga menziesii* (Mirb.) Franco). Treatments include an uncut control, a heavy thin, a light thin, and a light thin with gaps. Unlike traditional thinning, this study sought to maintain overstory diversity by specifically retaining hardwoods and conifer species other than Douglas-fir. A complex set of responses (e.g., vegetation, bird and small mammal populations, mushroom productivity) were measured repeatedly after treatment implementation. Results of vegetation data indicate that only the heavy thinning maintained open canopies for multiple years and accelerated development of large trees, one important component of late-successional habitat. In addition, thinning did not result in higher mortality rates, but decreased density related mortality of Douglas-fir. Also, mortality was low for nondominant overstory species that were intentionally retained during thinning. Finally, vertical canopy structure showed no immediate differentiation among treatments. Tree crowns likely did not have sufficient time to respond, and evidence suggests that retention of nondominant tree species may have prevented simplification of crown structure in low thinnings.

KEYWORDS: Silviculture, Douglas-fir, overstory response, vertical structure, tree growth.

INTRODUCTION

Young, managed Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests are prevalent in the forests of western Oregon, Washington, and British Columbia, often replacing what was once "old-growth" forest (Hunter 2001). Structural features associated with old-growth forests such as large diameter trees and snags, multi-layered canopies, and well-developed understories are often absent from these young stands (Franklin and Spies 1991, Franklin et al. 2002, Halpern and Spies 1995). Such attributes have been identified as key ecosystem components important for late successional wildlife habitat, epiphyte substrate, understory vegetation heterogeneity, and nutrient cycling (Bailey et al. 1998, Forsman et al. 1984, Rambo and Muir 1998, Trofymow et al. 1991). Consequently, young managed forests may not be capable of supporting diverse species assemblages, specifically those dependent on late-successional habitat (Halpern and Spies 1995).

Concern over how to best manage young stands in order to promote late-successional habitat gave rise to the Young Stand Thinning and Diversity Study (YSTDS). This study was designed to test whether management of young stands, specifically alternative thinning regimes, can accelerate development of late-successional habitat and, therefore, enhance biodiversity while maintaining wood production goals. Research questions addressed by this study include effects of thinning on vegetation (Beggs et al., n.d.; Beggs, n.d.; Bohac et al. 1997), small mammals (Garman 2001),

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birds (Hagar et al.1996, Hagar et al. 2004), and soil (Allen 1997). Though the study covers a broad realm of scientific inquiry, this paper focuses on early (5 to 7 years post-treatment) overstory vegetation response to the thinning treatments.

Thinning has traditionally been applied to enhance timber production, not for structural enhancement (Thysell and Carey 2001). Therefore, it remains uncertain if thinning can accelerate development of large trees, a layered canopy, and a well-developed understory. Although tree growth has been shown to increase following thinning (Marshall and Curtis 2002, Staebler 1956), traditional thinning may not sufficiently reduce competition among the largest, most dominant trees to increase their growth and accelerate development of large trees like those important for late-successional habitat. In addition, formation of multi-layered canopies and understory layers may take several years to develop. Tracking the changes in vegetation response over a long period of time is an important feature of the YSTDS. The present study will provide reference information regarding initial vegetation response that will facilitate future interpretation of successional patterns and development.

Vegetation response investigated by this study includes treatment comparisons of (1) changes in overstory canopy cover, (2) growth of the largest 10 to 30 Douglas-fir per hectare, (3) growth of all Douglas-fir ≥ 8 cm in diameter 1.37 m above ground (d.b.h.) (4) vertical crown structure, and (5) mortality of individual tree species.

METHODS

Study Design

The study is located in the lower Cascade Range of central Oregon in 30- to 50-year-old second-growth Douglas-fir stands. It consists of four blocks, with each block containing four treatments (providing a total of 16 treatment units). Blocks are designated as Cougar Reservoir (CR), Mill Creek (MC), Christy Flats (CF), and Sidewalk Creek (SC). Treatment units range in size from 15 to 53 hectares (ha), with a mean of 30 ha. Pretreatment stand exam data indicates that basal area (BA) within blocks was similar among treatment units (Beggs et al., n.d.); therefore, starting conditions were assumed to be similar.

Treatments include an uncut control, a heavy thin, a light thin, and a light thin with gaps (hereafter abbreviated as LtGaps). Treatments were applied between 1995 and 1997. Unlike traditional thinning, these treatments sought to maintain overstory diversity by specifically retaining hardwoods and conifer species other than Douglas-fir. The control maintained stand densities of approximately 650 trees per hectare (t/ha). The light thin reduced stand densities to 250-300 t/ha and was representative of a typical commercial thin. In an attempt to mimic densities believed to be more similar to initial old-growth stand densities (Tappeiner et al. 1997), the heavy thin reduced stand densities to 125 t/ha. Finally, the LtGaps treatment was intended to mimic small-scale disturbance. Therefore, the entire stand was thinned to 250-300 t/ha with 0.2 ha gap cuts spaced evenly throughout the stand (1 gap per 2 ha). This produced the following three subtreatments within the LtGaps treatment: (1) gap: the 0.2 ha gap, excluding the outermost 7.5 m ring; (2) edge: the circular area surrounding the gap cut, extending 7.5 m into the gap and 28.1 m out from the gap/forest interface; and (3) the stand matrix: the remaining portion of the thinned stand (see Sampling and figure 1 for clarification on subtreatment definitions).

Sampling

Vegetation sampling occurred in the summer of 1995, 1996, or 1997, depending on the time of harvest completion. This data will be referred to as "first-year vegetation sampling," inferring it is data describing the earliest post-harvest response. Resampling was completed during the summer of 1999 and again in 2001. These data depict vegetation response 3 to 5 years and 5 to 7 years post-harvest, respectively. Most information presented in this paper will focus on results obtained from the first-year data and 2001 data.

Tree sampling was conducted using 0.1-ha circular permanent plots. For the control, heavy, and light thin treatment units, plots were randomly located along systematically placed transects. Sampling covered approximately 7.5 percent of the area in these treatment units. For the LtGaps treatment units, there was interest in capturing variation occurring among the gap, edge, and matrix. Therefore, in these treatment units, plots were placed in each of 10 gaps, 10 edges, and 10 stand matrix sites that had been randomly selected (30 total plots per treatment unit). Location of subtreatment plots was determined by the radius of the sampling plot (17.8 m; fig. 1). Therefore, gap plots sampled only the gap interior. Edge plots sampled the outermost ring of the gap, the gap/forest interface, and a section of forest extending from the gap into the thinned forest. Stand matrix plots sampled the remainder of the thinned stand that was beyond that which was included in the edge sampling (fig. 1).

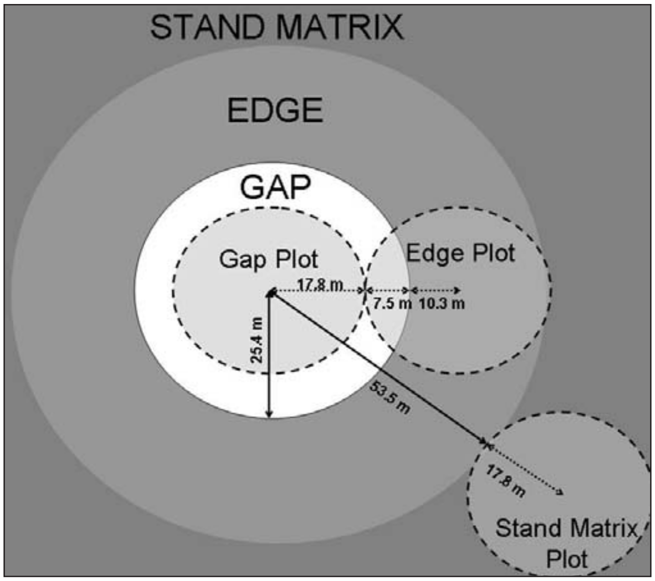

Figure 1—Schematic of sampling plot layout for the treatment light thin with gaps.

In each plot, overstory percent cover was measured centrally and at four cardinal directions. The d.b.h. for all trees ≥ 5cm was also measured in each plot. A random subsample of trees in each plot was measured once for height and crown base height in 1999.

Data Analysis

Canopy cover, absolute growth of the largest 10 to 30 Douglas-fir, relative growth of all Douglas-fir ≥ 8 cm d.b.h., vertical crown structure differentiation, and mortality of individual tree species were compared among treatments. For canopy cover, multiple measurements within plots were averaged to provide plot means. For control, light thin, and heavy thin treatment units, these plot totals were then averaged to provide mean treatment unit values. For the LtGaps treatment units, subtreatments (gap, edge, and matrix) were sampled equally but did not comprise equal proportions of the total treatment unit. To adjust for this and permit comparison with other treatments, weighted averaging was used. Subtreatment means were calculated by averaging plot means within each subtreatment, weighting these averages by the areal proportion of a subtreatment in the total treatment unit, and then averaging these numbers to provide treatment unit means.

Annual growth of each Douglas-fir tree in a treatment unit was calculated by subtracting the first-year d.b.h. from the 2001 d.b.h. and dividing by the number of years between measurements. For all trees ≥ 8 cm d.b.h., relative growth (absolute growth divided by first-year d.b.h.) was used in order to account for differences in average stand d.b.h. among treatments (by removing small trees, thinning inherently increased average stand d.b.h. in thinned treatments relative to controls). To assess growth of the largest 10, 20, and 30 t/ha, the largest 1, 2, and 3 trees from each 0.1 ha plot, respectively, were selected. For these trees, absolute growth was calculated because average d.b.h. of large trees did not differ among treatments. Treatment unit means were calculated by averaging growth of all trees in each treatment unit.

Vertical crown structure was quantified using the Foliage Height Diversity (FHD) index (MacArthur and MacArthur 1961). The FHD is comprised of two components: richness and evenness, making it nearly identical to the Shannon-Weiner diversity index (Margalef 1958). Richness, for this study, was defined as the number of 5-m layers containing canopy vegetation whereas evenness was the relative abundance of vegetation within each layer. Mortality for each species (for list of all species examined, see Beggs et al., n.d.) was calculated as the percentage of all trees (by species) present during first-year vegetation sampling that had died by 2001.

Treatment comparisons were performed by using a randomized complete block model. All multiple comparison tests employed the Tukey-Kramer adjustment with the significance level set at $\alpha = 0.05$ (Ramsey and Schafer 2002). For more complete details on statistical methods and analysis, refer to Beggs et al. (n.d.).

RESULTS AND DISCUSSION

Overstory Cover

The overstory canopy was successfully opened by thinning. Less overstory cover was found in all thinned treatments relative to the control 1 year after thinning (fig. 2). In addition, the heavy thin had less overstory cover than the light thin. By 2001, the heavy thin and LtGaps thin still had less canopy cover than the control, but cover in light thin was no longer significantly different from the control (fig. 2), i.e., a heavy thinning and a light thinning combined with gap formation produced a canopy that remained fairly open for several years. Following a more conventional light thin, however, the open canopy closed relatively quickly. This difference is important because successful establishment of an understory layer is often dependent upon extended periods of canopy openness (Alaback and Herman 1988, Bailey et al. 1998, Klinka et al. 1996). Though some small changes may occur during brief phases of open conditions and persist for extended periods of time (Alaback and Herman 1988, Bailey et al. 1998, Thomas et al. 1999),

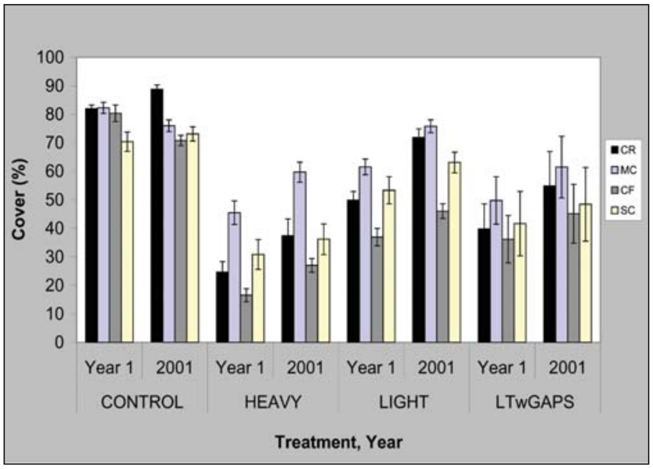

Figure 2—Overstory cover for each treatment unit. CR = Cougar Reservoir, MC = Mill Creek, CF = Christy Flats, SC = Sidewalk Creek Year 1 = 1995, 1996, and 1997 sampling.

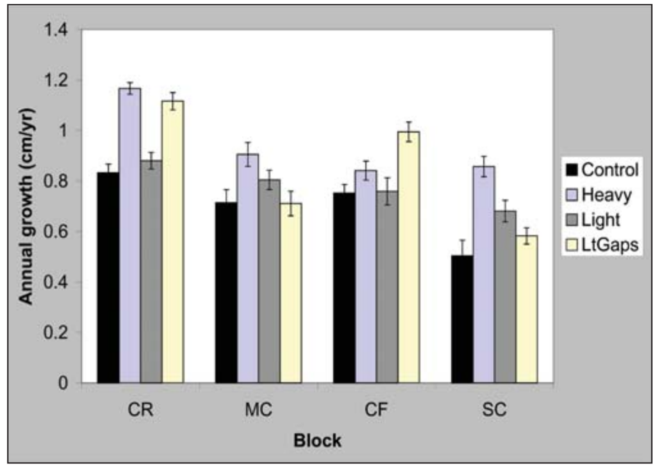

Figure 3—Growth of largest Douglas-fir (20 trees per hectare). CR = Cougar Reservoir, MC = Mill Creek, CF = Christy Flats, SC = Sidewalk Creek.

the rapid closure of the canopy after light thinning may inhibit establishment of a prominent understory layer.

Growth

Growing conditions for the residual trees were most improved following heavy thinning. This is apparent by higher relative diameter growth of all Douglas-fir ≥ 8 cm d.b.h. in the heavy thin than in the control. Also, the LtGaps thin had slightly higher relative growth than the control, likely due to increased resources along gap/edge interface. The light thin did not result in higher diameter growth relative to the control and actually had significantly lower growth than the heavy thin. Heavier thinning also provided favorable growing conditions for the dominant stand components. Only the heavy thin produced higher absolute growth among trees relative to the control for the dominant overstory trees (fig. 3). These results suggest that traditional thinning densities are not low enough to accelerate the development of large trees, an important component of late successional habitat. Apparently the largest trees are already in a dominant position so that the removal of less dominant trees by low thinning has little impact on their growth. This suggests that long-term improvement of growing conditions for the largest trees is only likely to occur by repeated thinning to densities much lower than those of a traditional low thin.

Vertical Structure

Thinning did not result in immediate differentiation of crown layers among treatments. By 2 to 4 years after harvest completion, no difference was found among treatments in the FHD index. When the FHD index was broken into richness and evenness, neither component differed among treatments. Lack of distinction among treatments indicates that crown extension and epicormic branching (Franklin et al. 2002) has not yet begun in these stands. Given the relatively short time of post-treatment response examined, these results are not surprising. However, these results also show that canopy structure was not simplified during the thinning, even though a low thinning was prescribed. Preservation of nondominant overstory species may have permitted retention of lower layers of canopy vegetation, thereby maintaining original partitioning of vegetation in the canopy.

Mortality

High mortality following thinning was not evident for Douglas-fir or other nondominant species. Douglas-fir and all combined hardwoods had significantly higher mortality in the control than in all thinned treatments. Golden chinquapin (*Chrysolepis chrysophylla* (Dougl. ex Hook.)) was the only other species to have a difference in mortality among treatments, with higher mortality in the control than in the heavy thin or the LtGaps thin (mortality in the light thin did not differ from that of the control). These findings suggest retention of non-Douglas-fir species during thinning will not be counteracted by subsequent high post-thinning mortality of these species. Instead, thinning decreased density related mortality in the short term. In addition, the lower mortality of Douglas-fir in the heavy thin relative to the control suggests that a high intensity of thinning, like that applied in this study, did not result in high stand instability or mortality.

CONCLUSION

In summary, thinning dense stands altered overstory conditions in the short term. However, overstory response was affected by the intensity of thinning. Thinning to densities similar to those found in the light treatment of this study produces only temporary opening of the canopy and may not provide sufficient time or resources for understory development or growth of large trees. Higher intensity thinnings may be needed to facilitate future growth of overstory trees and maintain a longer duration of open canopy conditions. Gap openings may also provide some of these benefits. Although initial assessment of vertical canopy structure does not yet show signs of enhanced vertical structure in thinned stands, retention of nondominant overstory species does appear to be beneficial in preventing initial crown simplification following thinning. Though more time is needed to determine true effectiveness of thinning in acceleration of late-successional habitat, early evidence suggests thinning may be an effective tool in management of young Douglas-fir forests.

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Observed Changes in Ground Saturation, Aquatic Invertebrate Densities, and Breeding Bird Populations Following Various Harvesting Intensities

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ABSTRACT

The effects of different harvest treatments on hillslope hydrology, avian populations, and aquatic invertebrates were examined as part of the southeast Alaska Alternatives to Clearcutting project (McClellan et al. 2000). Here, we summarize the response of hillslope soil hydrology, bird, and aquatic communities to the different cutting treatments. Through use of groundwater-monitoring wells, we found that soil saturation on hillslopes increased significantly with harvest intensity at one of two study locations following all cutting treatments. Increased soil saturation may decrease hillslope stability within the harvested area. Aquatic invertebrate drift densities had coefficients of variation between 50 and 85 percent before harvest. Because of such variation, no harvest treatment effect was detected. Headwater streams can be a significant source of aquatic and terrestrial invertebrates to downstream, fish-bearing reaches. However, detecting harvest effects will require greater replication to reduce variability and increase statistical power. Birds associated with large trees were more abundant in stands with higher tree retention at both study locations. Understory species reached maximum abundance at intermediate harvest levels at Hanus Bay but did not differ at Portage Bay. Alterations in sediment delivery or hydrologic response may change the pattern of sediment and woody debris storage and the functioning of aquatic and riparian avian communities further down the slope.

KEYWORDS: Forest management, ground saturation, aquatic ecology, forest passerine ecology.

INTRODUCTION

The Alternatives to Clearcutting (ATC) project was designed to determine ecological effects of different harvest treatments occurring within old-growth forests (McClellan et al. 2000, McClellan 2004). Here, we synthesize results from the hydrology, invertebrate ecology, and avian ecology components of the ATC project. We draw inferences about the potential effects of alternative harvest patterns on physical and biologic interactions in headwater streams. We summarize the individual study results and discuss the potential effects of landslides caused by increased ground saturation to downslope populations that use habitats influenced by sediment and associated accumulations of large, woody debris (LWD).

This paper summarizes effects of forest harvest on (1) ground saturation and hillslope stability, (2) transport (drift) of macroinvertebrates and organic detritus in headwater streams, (3) birds during the breeding season, and (4) potential changes in aquatic communities and riparian bird populations, assuming an increase in landslides.

STUDY SITES, METHODS AND SUMMARY OF RESULTS

Preharvest ATC site evaluation was conducted at Hanus Bay, Portage Bay, and Lancaster Cove; all within southeast Alaska (see McClellan 2004, McClellan 2005 (in these proceedings)). Post-harvest site evaluation was conducted at Hanus and Portage Bay only, as Lancaster Cove has not

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yet been harvested. Forests within the ATC study locations are dominated by western hemlock (*Tsuga heterophylla*) with smaller amounts of Sitka spruce (*Picea sitchensis*), mountain hemlock (*Tsuga mertensiana*), yellow-cedar (*Chamaecyparis nootkatensis*), western redcedar (*Thuja plicata*), and red alder (*Alnus rubra*). Understory plant species include red huckleberry (*Vaccinium parvifloium*), Alaska blueberry (*V. alaskaense*), rusty menziesia (*Menziesia ferruginea*), devil's club (*Oplopanax horridus*), skunk cabbage (*Lysichiton americanus*), and false hellebore (*Veratrum eschscholtzii*). Under natural conditions windthrow is the dominant forest disturbance (Harris et al. 1974, Harris 1989, Nowacki and Kramer 1998), and there is little fire (Noste 1979). Over the last 100 years at the ATC sites, windthrow was not found to be a dominant disturbance mechanism (Hennon and McClellan 2003).

Nine harvest treatments were applied at each ATC study site, ranging from uncut controls (100% retention) to clearcuts (0% retention). Additional treatments included (1) evenly dispersed individual tree selection (ITS) with 75-percent retention (75% ITS), (2) evenly dispersed ITS with 25-percent retention (25% ITS), (3) evenly dispersed ITS with 5-percent retention (5% wildlife trees), (4) clearcut harvest with aggregated reserves at 75-percent retention (75% clumps), (5) aggregated reserves in a thinned matrix at 25-percent retention overall (25% clumps), (6) gap harvest in a thinned matrix at 25-percent retention overall (25% gaps), (7) gaps harvest in an uncut matrix with 75-percent retention overall (75% gaps). Treatments were unreplicated within each of the three sites. Sites were helicopter logged to ensure that disturbances were comparable (McClellan 2000) and to minimize impacts to the ground surface and understory vegetation. A detailed outline of methodology, description of study locations can be found in McClellan et al. (2002, 2004, 2005).

Hydrology

The percentage of soil saturation within hillslope hollows at Hanus Bay and Portage Bay was measured before and after harvest with groundwater monitoring wells. One site within each of the four uniform-harvest treatments, including an uncut control, 75% ITS, 25% ITS, and clearcut was monitored with pressure transducers connected to data loggers. Data loggers, scanning every five seconds, recorded mean water levels at 0.5 to 1.0 hour intervals. Temperature gages were used to identify periods of freezing in the precipitation record. Rainfall was measured either at the site or within 2.0 km of the site if the rain gage at the site was not operating.

Water levels were recorded from 1994 to1998 at Hanus Bay and from 1996 to 2000 at Portage Bay from April through November. The average maximum percentage of saturation measured during a 48-hour storm period was calculated for the seven wells at each site for 20 storms before harvest and 20 storms after harvest. Harvest effects were tested by using analysis of covariance for each of the four treatments. Observations showed that soil saturation levels on hillslopes differed significantly following all cutting treatments at Hanus Bay, but not at Portage Bay (Johnson et al., in review). Use of an infinite slope stability model (Sidle et al. 1985, Terzaghi 1943) indicates that shallow soils at both study locations, with slopes generally < 30° (58%), were not steep enough to be unstable under the observed conditions (Johnson et al., in review). Because many potential harvest sites, including ATC harvest units, occur on steeper slopes than those selected for the hydrology component of this study, we extrapolated the observed soil saturation changes in clearcut sites to steeper slopes using the slope stability model. With this model, we calculated what slope and/or soil depth would result in destabilization in a variety of pre-and post-treatment (saturation) cases. We found that for 100-percent harvest, we estimated a 7-percent reduction in the factor of safety with forests having slope gradients of 35° (70%) with soil depths of 1.25 m (Johnson et al., in review).

Aquatic Ecology

Downstream transport (drift) of macroinvertebrates and organic detritus in headwater streams located within all ATC treatments was compared among streams at all three sites before harvest. Post-harvest data were analyzed only at Hanus Bay. Drifting invertebrates and detritus were captured in drift nets (250 µm mesh size) installed within the stream channels in the spring, summer, and fall. Total stream discharge and discharge through the net were measured and data expressed as invertebrate mass per unit volume.

Mean drift rates both pre- and post-treatment were 220 mg of invertebrate dry mass and 18 g of detritus per stream per day with a range of 130 to 400 mg. There was no relation between harvest treatments and subsequent changes in drift densities (Musslewhite and Wipfli 2004). However, the mean coefficients of variation among streams were 85 percent for preharvest drift densities in the fall and 60 percent for preharvest drift densities in the spring. Based on our understanding of small stream food webs, removing trees to the streambank should impact the aquatic community and terrestrial insect inputs; however, such high natural variation would mask all but the most extreme treatment effects. Although no statistical tests were performed, a trend toward an increase in the proportion of true flies (Diptera), primarily midges (Chironomidae), and a decrease in the proportion of mayflies (Ephermeroptera) was seen in the lower retention treatments. With little research done on insect drift in small southeast Alaska streams, the cause of the large variation in drift is unknown. A better understanding of the fundamental controls on invertebrate growth and production could allow for stratification of research sites and reduced variance. Greater treatment replication would increase the statistical power of the experiments but would greatly increase monitoring costs. Future experiments should have fewer treatments and more replication within each treatment.

Avian Ecology

A census of birds was taken by using the point-count method (Ralph et al. 1995, Verner 1985) three times during the breeding season (mid-May to early July; see McClellan et al. 2000 for details). Because yearly variation within control sites was high for some species, we focused our comparisons within treatments rather than pre- to post-harvest.

Species that use large trees for foraging or nesting (chickadees (*Parus hudsonicus, P. rufescens*), brown creeper (*Certhia americana*), golden-crowned kinglet (*Regulus satrapa*), Pacific-slope flycatcher (*Empidonax difficilis*), rufous hummingbird (*Selasphorus rufus*), Steller's jay (*Cyanocitta stelleri*), Townsend's warbler (*Dendroica townsendi*), varied thrush (*Ixoreus naevius*), woodpeckers (*Picoides pubescens, P. villosus*, *Colaptes auratus*, *Sphyrapicus ruber*)) were more abundant in stands with higher tree retention (75% ITS, 75% gaps, 75% clumps, control) than more heavily harvested treatments (clearcut, 5% wildlife trees, 5% gaps, and 25% clumps) at both study locations; 25% ITS at Portage Bay; fig. 1).

Species that nest or forage primarily in the understory (dark-eyed junco (*Juncus hyemalis*), fox sparrow (*Passerella iliaca*) hermit thrush (*Catharus guttatus*), orange-crowned warbler (*Vermivora celata*), olive-sided flycatcher (*Contopus borealis*), pine grosbeak (*Pinicola enucleator*) ruby-crowned kinglet (*Regulus calendula*), Wilson's warbler (*Wilsonia pusilla*) winter wren (*troglodytes troglodytes*)) were more abundant in several treatments receiving intermediate levels of harvesting (25% ITS, 25% gaps) than in either more heavily harvested (clearcut, 5% wildlife trees) or lighter harvested treatments (75% ITS, 75% gaps, 75% clumps), but only at Hanus Bay (fig. 1).

CONSERVATION OF AQUATIC AND AVIAN SPECIES

Lessons Learned From Alternatives to Clearcutting

The effects of harvest pattern varied widely among the different ecosystem components and between the two locations. There was no evidence that transport of aquatic invertebrates and detritus varied among treatments. Groundwater varied with treatment, but only at one study location. Birds that use large trees differed at both study locations, but understory birds were only affected at one study location. Because treatments were unreplicated at each location, our ability to separate treatment response from natural spatial variation was limited, and there may have been effects below the detection threshold. Conditions such as valley location, total number of trees cut within catchments draining to wells, and the proximity of wells to the trees that were cut varied widely among sites and may have influenced soil saturation more than the cutting treatments (Johnson et al., in review). Bird and invertebrate densities differ widely over space, and this variation is not well quantified with single treatments. Variation in time also limited the power of our analyses. Interannual variation in bird and invertebrate populations is large, and the pre- and post-harvest observation periods were too short to capture the full range of natural annual variation. Additional replication at the treatment level would increase statistical power and enable us to detect more subtle responses. Longer pre- and post-harvest sampling periods should also improve the sensitivity of the analyses.

Physical and biological characteristics of watersheds differ greatly within the Tongass National Forest, and have been used to create a classification system of ecological subsections (Nowacki et al. 2001). The ATC studies presented here confirm that this variation affects the response of key ecosystem components to different harvest regimes. We cannot generalize our findings Tongass-wide based on only two locations. We cannot describe the total magnitude of variation or its geographic pattern. Because we lack a fundamental understanding of the natural variation in ecosystem responses across the Tongass, our ability to detect management effects is compromised. More harvest locations encompassing a greater representation of watershed types are necessary to determine whether the ecological subsections defined in Nowacki et al. (2001) provide a useful context to predict ecosystem response to varying harvest regimes. The potential for a landscape perspective to increase our ability to predict management effects by stratifying

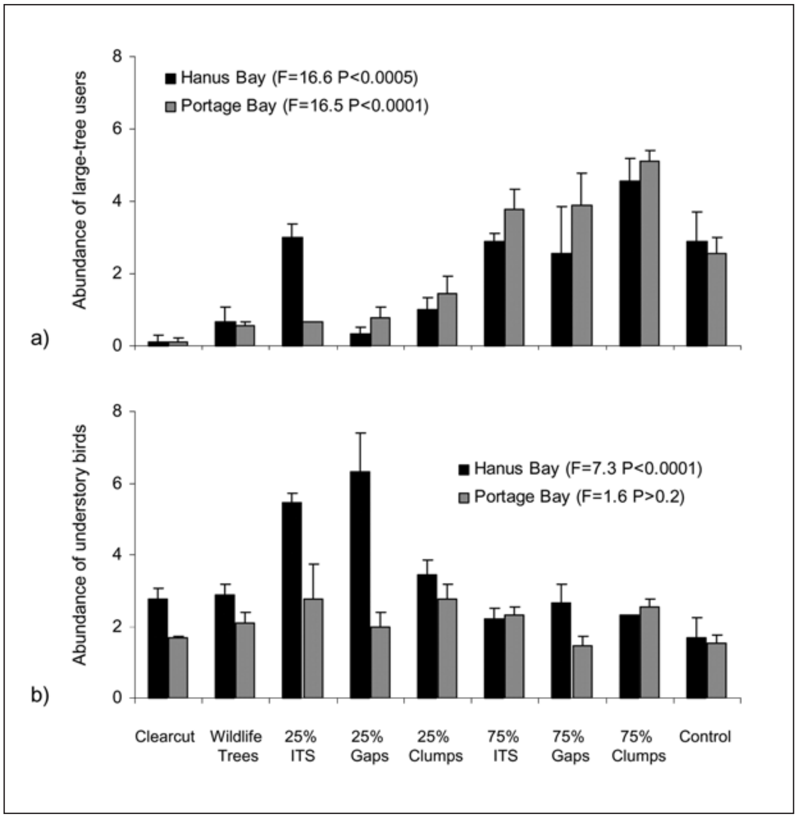

Figure 1a—Mean (± SE) abundance (number of detections per 7850 m^2) of avian species that use large trees for foraging or nesting in sites harvested in various silvicultural patterns at Hanus Bay and Portage Bay. Species include chickadees (*Parus hudsonicus, P. rufescens*), brown creeper (*Certhia americana*), golden-crowned kinglet (*Regulus satrapa*), Pacific-slope flycatcher, (*Empidonax difficilis*), rufous hummingbird (*Selasphorus rufus*), Steller's jay (*Cyanocitta stelleri*), Townsend's warbler (*Dendroica townsendi*), varied thrush (*Ixoreus naevius*), and woodpeckers (*Picoides pubescens , P. villosus, Colaptes auratus, Sphyrapicus ruber*).

Figure 1b—Mean (± SE) abundance (number of detections per 7850 m^2) of avian species that nest or forage in understory vegetation in sites harvested in various silvicultural patterns at Hanus Bay and Portage Bay. Species include dark-eyed junco (*Juncus hyemalis*), fox sparrow (*Passerella iliaca*), hermit thrush (*Catharus guttatus*), orange-crowned warbler (*Vermivora celata*), olive-sided flycatcher (*Contopus borealis*), pine grosbeak (*Pinicola enucleator*), ruby-crowned kinglet (*Regulus calendula*), Wilson's warbler (*Wilsonia pusilla*), winter wren (*Troglodytes troglodytes*).

watersheds into ecologically similar units is promising, and future research should be incorporated into that context.

Synthesis of Hydrology and Avian and Invertebrate Ecology

Although the ATC project did address the potential impacts of some natural physical disturbances such as windthrow (McClellan 2004), it did not successfully assess disturbances from landslides. The groundwater study was originally designed to determine the effects of harvest regime on landslide potential, but the chosen study sites were not steep enough to fail under any groundwater conditions. Increased slope failure is one of the best documented effects of past clearcutting (Bishop and Stevens 1964, O'Loughlin and Zeimer 1982, Sidle 1985, Sidle et al. 1985, Swanston 1970, Swanston and Marion 1991, Wu et al. 1979), and the downstream impacts can be profound. Although the hillslope hollows monitored in this study were not steep enough to generate landslides, it is inevitable that forest harvest will continue within areas where soil depths and slopes do make them vulnerable to failure if soil saturation increases. Because landslides can greatly affect stream and riparian habitats, and because some changes in soil saturation did occur for all cutting alternatives at one of the study locations, we will briefly discuss the potential consequences of slope failures on stream and riparian habitats and the organisms depending on them.

Most landslides generated on steep deglaciated slopes of southeast Alaska enter streams and deposit at the base of wide U-shaped valleys on slopes between 5 and 20 degrees (Gomi et al. 2001, Johnson et al. 2000, Swanston and Marion 1991); slopes similar to those selected for ATC study sites. Thus, although landslide initiation may occur in steep, nonfish bearing reaches, the ultimate effects often occur in lower gradient, fish-bearing reaches. The changes in sediment and woody debris associated with landslide deposition areas may change riparian vegetation type, LWD distributions, and channel morphology; factors that influence food and habitats of mammals, birds, fish, and invertebrates in steep headwater streams. In second-growth forest, more alder was found in stream channels impacted by landslides (Johnson and Edwards 2001). Forests with a higher percentage of alder export more aquatic invertebrates to downstream reaches (Wipfli and Musselwhite 2004) and have a higher nesting density of birds (De Santo et al., n.d.). Mobility of the large wood was found to be associated more with mass movements than with channel characteristics (Gomi et al., in review), therefore the number and size of LWD jams may be influenced by landslides resulting

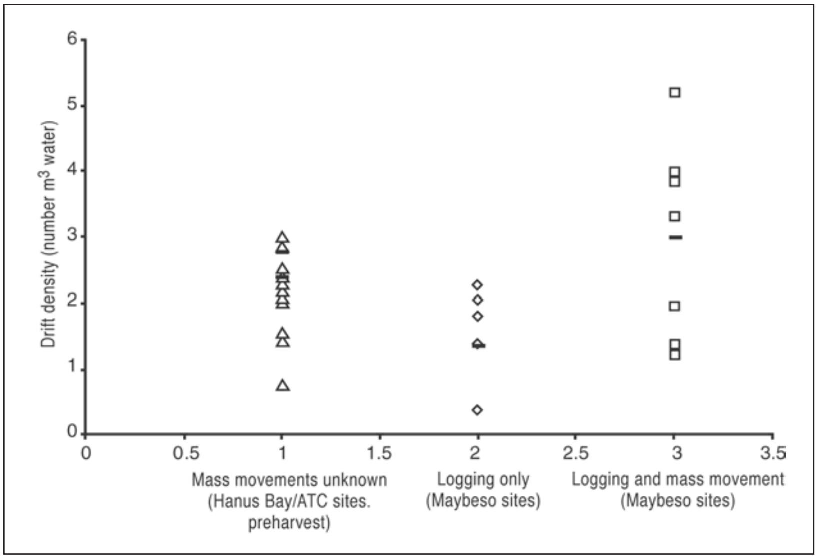

Figure 2—Mean macroinvertebrate drift densities of headwater streams with three different disturbance histories. Streams are located in Hanus Bay, Alaska ("some known disturbance") and the Maybeso Experimental Forest, Alaska ("logging only" and "logging and landslides"). Hanus Bay drift density data are from Musslewhite and Wipfli (2004); Maybeso drift density data are from Wipfli and Musslewhite (2004).

from increased soil saturation on steeper slopes. Large woody debris is important in retaining leaf litter and other food inputs and in creating pools, which increase the quality of habitat for fishes and invertebrates (Bryant et al., in press; Woodsmith et al., in press).

Landslides can influence stream invertebrate community structure and drift rates. Although these links were not made in the ATC study, studies within the Maybeso Experimental Watershed, located in southeast Alaska, indicate how landslides and stream invertebrates are linked. Streams in the Maybeso valley with at least one landslide had significantly higher (t-test, p=0.035) mean invertebrate drift densities than streams subjected to riparian logging only (fig. 2). Drift densities in landslide-impacted Maybeso streams were similar to those seen before harvest in ATC streams in Hanus Bay, which contained streams with some known landslide history (t-test, p=0.35). In addition, baetid mayflies and chironomid larva, the most abundant taxa found in ATC streams, are known to be early and abundant colonizers following disturbance (Mackay 1992). Their abundance in ATC drift samples before harvest may indicate the adaptation of headwater stream communities to frequent disturbance.

Landslides may also impact riparian bird species. Although streams were avoided during ATC avian censuses, some bird species have positive associations with streams. For example, the diet of the winter wren (Van Horne and Bader, 1990) consists of many of the terrestrial and aquatic invertebrates found at ATC headwater streams (Musslewhite and Wipfli, 2004). Wrens tend to cluster their territories around streams in southeast Alaska and British Columbia (De Santo et al. 2003, Waterhouse et al. 2002, respectively) with territory size inversely related to amount of stream bank (De Santo et al. 2003). Wrens in southeast Alaska often nest in downed wood or stream banks (30% of 175 nests in downed wood; 14% of nests in stream banks; De Santo et al. 2003 and unpublished data[5]) as they do in British Columbia (Waterhouse 1998, Waterhouse et al. 2002). In southeast Alaska, live trees and moss growing on live trees were the foraging substrate most commonly used by wrens, but most individuals also foraged on downed wood, stumps, logs and slash (De Santo et al., n.d.). That wren abundance in clearcuts was equal to or greater than that in controls at both ATC sites may be because structural forest attributes, including understory vegetation, moss cover, and presence of small streams, are more influential in determining the spatial distribution of wrens than forest

[5] On file with M. McClellan, Pacific Northwest Research Station, 2270 Sherwood Lane, Suite 2A, Juneau, AK 99801, USA.

age (Waterhouse et al. 2002). Where harvest occurs in landslide prone hillslopes, landsliding could alter the availability and quality of bird habitat.

In summary, impacts on slope stability or aquatic invertebrates occurred, but the power of the research to detect impacts was limited by the lack of replication, high natural variation, and short sampling interval. Impacts on some avian species were evident, indicating a need for further sampling to elucidate probable habitat associations and effects of timber harvest. It is clear from this research that data on the fundamental ecological properties of basins across the large span of the Tongass National Forest are needed to serve as a reference to quantify the impact of various management practices. The ecological subsections defined in Nowacki et al. (2001) may serve as a useful guide to stratify rivers into similar classes and thereby partition some of the variance encompassed by the studies described here; however, a longer period of measurement will also be needed to be able to detect treatment effects in the face of high interannual natural variability.

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Results From Green-Tree Retention Experiments: Ectomycorrhizal Fungi

Daniel L. Luoma[1] and Joyce Eberhart[2]

ABSTRACT

The ecosystem effects of natural disturbances differ dramatically from those engendered by even-aged management practices that emphasize commodity production. Because forest management activities can reduce ectomycorrhizal (EM) fungus diversity and forest regeneration success, management approaches are needed to sustain these essential forests organisms. We present selected results from experiments that test biodiversity assumptions behind current guidelines for ecosystem management. We examine contrasts in structural retention as they affect biodiversity and sporocarp production of EM fungi—a functional guild of organisms well suited as indicators of disturbance effects on below-ground ecosystems.

Overstory removal significantly reduced EMF sporocarp production but, in contrast to the initial hypothesis, the effects were not always proportional to basal area retained. The effect of spatial pattern of retention varied between retention levels and mushroom and truffle sporocarp groups. Management implications include the need to address the conservation of rare truffle and mushroom species in a manner that recognizes their different responses to forest disturbance. We also raise the hypothesis that fire suppression may favor mushroom production over truffle production. Because fire seems to be important in the reproductive evolution of EMF, our results also add further impetus to the development of management plans that seek to restore forest health from the effects of decades of fire suppression.

Experimental results suggest using dispersed green-tree retention in combination with aggregated retention to maintain sporocarp production. Such a mix ameliorates disturbance effects and may maintain higher levels of sporocarp production in the aggregates by reducing edge effects. It remains unclear how short-term reductions in sporocarp abundance will affect EM fungus populations for future forests. After disturbance, spores are a form of legacy and key to enabling adaptations by other species in the face of environmental change. Long-term silvicultural experiments are essential for monitoring trends in the EM fungus community.

KEYWORDS: Fungi, hypogeous, epigeous, mycorrhizae, biomass, diversity, disturbance.

INTRODUCTION

Fungi profoundly affect nearly all terrestrial ecological processes and events; accurate information on the fungal component is required to adequately understand how ecosystems function (Trappe and Luoma 1992). When fungal mycelia form particular associations with a host plant's fine roots, a symbiotic organ forms, called a mycorrhiza. Through this structure the plant provides carbohydrates to the fungus, which in turn facilitates uptake of nitrogen, phosphorus, other minerals and water to the plant (Allen 1991, Marks and Kozlowski 1973, Smith and Read 1997). The fungus also protects plant roots from attack by pathogens and the effects of heavy metal toxins, promotes fine root development, and may produce antibiotics, hormones and vitamins useful to the plant (Smith and Read 1997). Mycorrhizal associations are vital to the existence of most vascular plants (Smith and Read 1997, Trappe 1987).

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Availability of mycorrhizal fungi determines patterns of primary plant succession on new soils such as moraines, fresh volcanic deposits, and mine spoils (Allen 1991, Cázares 1992, Helm et al. 1999, Trappe and Luoma 1992). Mycorrhizal fungi are associated not only with increased plant productivity but also with developing community diversity following disturbance (Allen et al. 1995, Cázares 1992). Mycorrhizal fungal species differ in their ability to provide particular benefits to their hosts, and their presence and diversity change during plant succession (Cázares 1992, Helm et al. 1996, Mason et al. 1983, Trappe 1977). A diversity of mycorrhizal fungi is likely essential for successful shifts of geographic range by plants due to climate change (Perry et al. 1990).

The ectomycorrhiza type is characterized by a mantle of fungal hyphae encasing the root tips of the associated plant. Ectomycorrhizae are characteristic of the Pinaceae and Fagaceae which dominate most forests in the Pacific Northwestern United States, and are required for survival of these hosts in field soil (Trappe and Luoma 1992). We are able to reasonably infer the mycorrhizal status of diverse forest fungi by their placement in certain fungal genera (Molina et al. 1992, Trappe 1962) despite the reservations expressed by Arnolds (1991). Ectomycorrhizal fungi (EMF) form a functional guild linking primary producers to soil systems, are important in ecosystem response to disturbance (Janos 1980, Perry et al. 1989), and may be sensitive indicators of environmental changes (Arnebrant and Söderström 1992, Arnolds 1991, Termorshuizen and Schaffers 1987, Termorshuizen et al. 1990). Ectomycorrhizal fungi mostly produce macroscopic sporocarps in the form of mushrooms and truffles (epigeous or above-ground fruiting bodies and hypogeous or below-ground fruiting bodies, respectively). Sporocarps produce the spores that disseminate the species and provide for genetic recombination within and among populations.

Forest Management

Studies from the Pacific Northwest indicate that forest management activities can reduce ectomycorrhizal fungi, forest regeneration success, and influence patterns of plant succession (Amaranthus et al. 1994; Harvey et al. 1980a, 1980b; Waters et al. 1994; Wright and Tarrant 1958). Development of management approaches to sustain these essential organisms in forests has been hampered by a lack of knowledge of EMF community structure, diversity, and spatial and temporal variability across stands and landscapes.

Many EMF species, especially those that produce truffles, are also important dietary items for vertebrates and invertebrates: some small mammal species rely on them for over 90 percent of their diet (Carey et al. 1999, Claridge et al. 1996, Hayes et al. 1986, Jacobs 2002, Maser et al. 1978, Maser et al. 1985). Truffle species diversity provides necessary nutritional diversity to the diet of mammal mycophagists (see review by Luoma et al. 2003). Small mammals, in turn, form important links in the trophic structure of forest ecosystems as prey for raptors (e.g., owls and goshawks) and mammalian carnivores (e.g., martens and fishers) (Carey 1991, Fogel and Trappe 1978, Hayes et al. 1986, McIntire 1984).

Few studies have examined silvicultural effects on EMF sporocarp production (Colgan et al. 1999, Waters et al. 1994). Although EMF sporocarps do not reveal as complete a picture of the below-ground EMF community as root tip studies (Dahlberg et al. 1997, Gardes and Bruns 1996, Horton and Bruns 2001, Yamada and Katsuya 2001), silvicultural effects on sporocarp production mirror the effects found in root-tip studies: species diversity and community composition can change dramatically. Thinning affects the composition and diversity of EMF in the stand as well as the frequency of sporocarps (Carey et al. 2002, Colgan et al. 1999, Waters et al. 1994). For example, stands that were heavily thinned showed increased dominance by one fungal species. Thinning also reduced truffle biomass, frequency of truffles, and shifted overall species composition (Carey et al. 2002, Colgan et al. 1999). However, total truffle biomass and frequency of sporocarps may recover 10 to 17 years after thinning, whereas shifts in species dominance persist longer (Waters et al. 1994).

Green-tree retention, the practice of leaving live, structurally-sound, large trees in a stand after extracting timber, is an alternative forest management method designed to accelerate the development of late-successional forest characteristics in young, managed stands (Aubry et al. 1999). The Demonstration of Ecosystem Management Options (DEMO) experiment is a long-term study designed to examine the effects of different levels and patterns of green-tree retention on multiple forest attributes (see Aubry et al. 1999). Studies of disturbance effects on the below-ground ecosystem are relatively rare. These studies are critical to forest managers seeking to incorporate basic ecological knowledge into forest management policies and practices. Here we focus on initial results from an experiment that tested some of the assumptions behind the current guidelines for ecosystem management as they affect a functional guild of organisms (EMF) that are well suited as indicators of disturbance effects on the below-ground ecosystem. Detailed reporting of the DEMO fungi study can be found in Luoma et al. (2004).

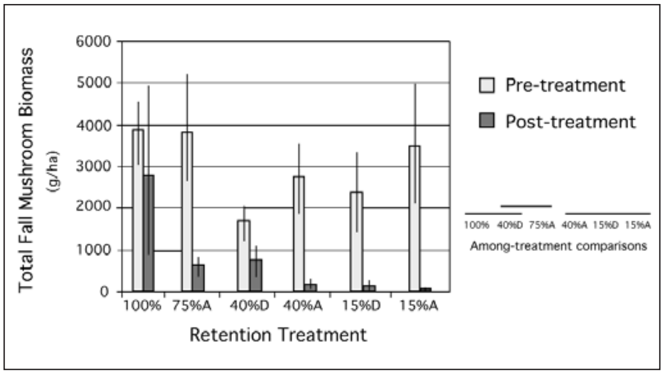

Figure 1—Mean total ectomycorrhizal mushroom standing crop biomass for the fall samples from three DEMO blocks. Standard errors are indicated by vertical bars. Among-treatment comparisons were derived from multiple analysis of variance, repeated measures contrasts of the time * treatment interaction using transformed data. Treatments (see methods) without a shared horizontal bar above them are significantly different at $p \leq 0.1$ (adapted from Luoma et al. 2004, used with permission). D = dispersed retention; A = aggregated retention.

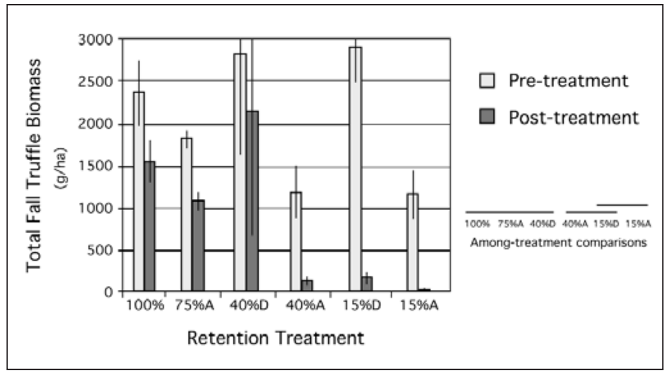

Figure 2—Mean total truffle standing crop biomass for the fall samples from three DEMO blocks. Standard errors are indicated by vertical bars. Among-treatment comparisons were derived from multiple analysis of variance, repeated measures contrasts of the time * treatment interaction using transformed data. Treatments (see methods) without a shared horizontal bar above them are significantly different at $p \leq 0.1$ (adapted from Luoma et al. 2004, used with permission). D = dispersed retention; A = aggregated retention.

The DEMO Experiment

The objective of the DEMO fungi study was to compare pre- and post-treatment standing crop biomass of EMF sporocarps within no harvest, 75-percent, 40-percent (dispersed and aggregated), and 15-percent (dispersed and aggregated) retention treatments. The DEMO experiment replicated six green-tree retention treatments in six geographic locations (Aubry et al. 1999). The treatments consisted of four levels of live tree retention (15, 40, 75, and 100 percent of existing live-tree basal area), with two patterns of retention, aggregated (A) and dispersed (D), applied to the 15- and 40-percent retention treatments. The aggregated pattern consisted of residual trees retained in clumps of about 1 ha and the dispersed pattern has residual trees homogeneously dispersed throughout the unit. For the 75-percent retention treatment, all of the harvest occurred in approximately 1-ha patches dispersed throughout the unit. Fungal sporocarp sampling was limited to 3 blocks.

Study Area

General environmental characteristics of the sites are described by Halpren et al. (1999). The Butte block is located on the Gifford Pinchot National Forest in southwestern Washington. The Dog Prairie and Watson Falls blocks are located on the Umpqua National Forest in southwestern Oregon. Prior to harvest, all blocks were dominated by *Pseudotsuga menziesii* (Mirb.) Franco. The importance of other tree species varied by block (Halpren et al. 1999).

RESULTS

Luoma et al. (2004) found that total fall biomass exceeded total spring biomass for both epigeous and hypogeous sporocarps except in the Watson Falls block where spring biomass of *Gautieria* was a major contribution to greater spring hypogeous sporocarp biomass. In particular, total fall mushroom biomass decreased significantly in the 40-percent aggregated, 15-percent dispersed, and 15-percent aggregated treatments as compared to the other treatments (fig. 1). No treatment effect was detected on the fall mushroom standing crop in the 40-percent dispersed treatment (fig. 1). Total fall truffle biomass was significantly reduced in the 40-percent aggregated, 15-percent dispersed, and 15-percent aggregated treatments as compared to the control, 75-percent aggregated, and 40-percent dispersed treatments (fig. 2). No treatment effect was detected on the fall truffle standing crop in the 40-percent dispersed treatment (fig. 2).

DISCUSSION

Relevance to Ecosystem Management

Standing crop data are useful to interpret the role of fungal species as a food source for animals or the energy expanded in an ecosystem for species reproduction. Standing crop of hypogeous sporocarps may underestimate actual sporocarp biomass productivity because animals utilize a proportion of the fruiting bodies (Luoma et al. 2003). The degree of underestimation is most pronounced at periods of low productivity, when consumptive pressure on the available food resource is proportionally high (North et al. 1997).

When both epigeous and hypogeous species are simultaneously assessed, new understanding of overall diversity phenomena emerges. For example, the more equable biomass distribution of hypogeous sporocarps compared to epigeous between spring and fall (Luoma et al. 2004, Smith et al.

2002) has important implications to mycophagous mammals. Fungal diversity in the diet of such animals appears to be nutritionally important (Claridge et al. 1999, Johnson 1994, Maser et al. 1978). Clearly, animals that depend on fungi as major food items (Fogel and Trappe 1978, Luoma et al. 2003) could not rely on epigeous fungi for diet diversity over the spring. Quite possibly, the decline in populations of some mycophagous animals could relate to decline in diversity of the fungal populations due to habitat disturbance (Claridge et al. 1996, Pyare 2001). Results from the DEMO study show that truffle genera important in the diets of small mammals were significantly affected by the treatments (Jacobs 2002).

Maintenance of EM fungal diversity is important for ecosystem health and resilience (Amaranthus 1997; Amaranthus and Perry 1987, 1989; Perry et al. 1990). Disturbance, whether natural or human caused, can drastically alter populations of EM fungi (Amaranthus et al. 1990, 1994, 1996; Colgan 1997; Pilz and Perry 1984; Schoenberger and Perry 1982).

The Secotioid Syndrome

Some sporocarps have morphology that is intermediate between truffles and mushrooms. Such sporocarps have been referred to as "secotioid" (Singer 1958). In addition to epigeous secotioid taxa, Thiers (1984) included all truffle-like taxa in his analysis of the "secoitioid syndrome." He proposed that in the Mediterranean and semi-arid climates of the western United States, high summer temperatures combined with extended drought stress were primary drivers in the evolution of hypogeous sporocarp formation (i.e., the truffle form). Bruns et al. (1989) documented that such morphological divergence (from mushroom to truffle) can proceed relatively rapidly, possibly as a result of selective pressures on a small number of developmental genes. Hibbett et al. (1994) present a case in which a simple secotioid phenotype, arising from a mutation at one locus, has persisted over a wide geographic range in wet environments that presumably do not exert the selective pressures that drive the secotioid syndrome toward evolution of more strongly sequestrate (Trappe et al. 1992) sporocarps. Baura et al. (1992) speculate that such mutations will not persist long in a population. Kretzer and Bruns (1997) found that secotioid forms of the important EM mushroom genus *Suillus* evolved at least twice and have persisted for evolutionarily significant periods of time over a wide range of summer-dry habitats in the western United States. They noted that the selective forces that favor a secotioid lineage were unclear.

We propose that results from Luoma et al. (2004) represent the first experimental evidence to support Thiers' hypothesis (1984). Even the relative small (1 ha) gaps created in the 75-percent aggregated retention treatment significantly reduced fall production of EM mushrooms in the surrounding uncut forest. Those same gaps, however, did not significantly reduce truffle production. The formation of gaps likely influenced the thermal properties, humidity, and evapotranspiration of the remaining intact forest (e.g., Zheng and Chen 2000). Based on these results, we extend Thiers' hypothesis to encompass the influence of fire in the broader context of forest disturbance in the summer-dry climates of the western United States. Fire is an important agent for producing the patterns of forest fragmentation (e.g., Heyerdahl et al. 2001) that would select for hypogeous sporocarp production via the "secotioid syndrome."

CONCLUSIONS

Even though green-tree retention can preserve ectomycorrhiza diversity (Stockdale 2000), sporocarp production and EM species richness was significantly reduced at all levels of retention except the control. These effects, however, differed by season and treatment (Luoma et al. 2004).

The DEMO study demonstrated the importance of pretreatment sampling. Experimental units within blocks were intended to be as similar as possible in overstory vegetation and site characteristics (Aubry et al. 1999), yet pretreatment results showed that uniformity of fungal populations in forests based on stand structure alone can not be assumed (see also Cázares et al. 1999).

Management implications include the need to address the conservation of rare truffle and mushroom species in a manner that recognizes their different responses to forest disturbance. We also raise the hypothesis that fire suppression may have favored mushroom production over truffle production. Because fire seems to be important in the reproductive evolution of EMF, our results also add further impetus to the development of management plans that seek to restore forest health from the effects of decades of fire suppression (Agee 1997).

Luoma et al. (2004) also found that overstory removal significantly reduced EMF sporocarp production but, in contrast to their initial hypothesis, the effects were not always proportional to basal area retained. The effect of spatial pattern of retention varied between retention levels and mushroom and truffle sporocarp groups. Though not directly studied in the DEMO experiment, Luoma et al. (2004) concluded their results supported the use of dispersed green-tree retention in combination with aggregated retention when maintenance of sporocarp production is a goal. Continuing study of retention level and spatial pattern relationships is important for development of scientifically sound silvicultural techniques for use in the pursuit of science-based forest management.

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Green-Tree Retention in Managed Forests: Post-Harvest Responses of Salamanders

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ABSTRACT

Salamanders often are proposed as indicators to monitor ecological ramifications of forest management practices because of their high site tenacity and environmental sensitivity. In this study, we examined post-treatment responses of salamanders to variation in the level and pattern of live trees retained during harvest within the Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) region of the Pacific Northwest of the United States. Forested units within six blocks were subjected to six different treatments ranging from 100- to 15-percent basal area retention in dispersed or aggregated patterns. When analyzed with randomized block ANOVA, salamander capture trends associated with the level and pattern of basal area retained were not evident up to 2 years after harvest for any of the four most abundantly captured species: ensatina (*Ensatina eschscholtzii*), western red-backed salamander (*Plethodon vehiculum*), northwestern salamander (*Ambystoma gracile*), and rough-skinned newt (*Taricha granulosa*). However, habitat analyses employing multiple regression indicated that salamanders were encountered most frequently when coarse woody debris (CWD) volume and/or herb cover were high; tree basal area was of secondary or limited importance in predicting salamander numbers. Results suggest that salamanders respond to altered environmental conditions induced by forest management that are not directly tied to basal area. Additionally, because we have the silvicultural tools to manipulate forest ground conditions, it should be possible to manage structural features important to salamanders while also extracting timber.

KEYWORDS: Basal area retention, forest structure, Pacific Northwest, salamanders, tree harvest.

INTRODUCTION

Consideration of biodiversity goals along with timber production recently intensified in managed forests in the Pacific Northwest (PNW) of the United States as concerns grew over the steady conversion of once abundant, structurally diverse, old-growth forests into single-species, even-aged, short-rotation plantations. Many land managers hoping to conserve and enhance biological diversity in production forests, are using a range of uneven-aged management options to promote complex forest structure. However, the effectiveness of these options in conserving forest-associated species are poorly studied. Thus, the data necessary to help direct selection of appropriate management scenarios conducive to both timber production and biological diversity are largely lacking.

In 1992, the Demonstration of Ecosystem Management Options (DEMO) study was initiated by the U.S. Congress in the Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) region of the PNW to examine ecological consequences of harvests where live trees are retained in different levels and patterns (Halpern and Raphael 1999). Animal groups studied in this large-scale, long-term experiment are terrestrial and arboreal small mammals, bats, birds, and salamanders. This paper focuses on initial DEMO results obtained for salamanders.

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Because salamanders are abundant in many forest ecosystems (Hairston 1987, Merchant 1972, Pough et al. 1987) and may have biomass comparable to birds and mammals (Burton and Likens 1975), they likely contribute significantly to natural forest processes. Additionally, their narrow habitat tolerances (Blaustein 1994) may allow them to respond rapidly to changing environmental conditions. Many species also have limited mobility and high site tenacity (Welsh 1990). Collectively, these features may make salamanders a suitable indicator species group for monitoring ecosystem consequences of forest management (McLaren et al. 1998, Welsh and Droege 2001).

In this study, we hypothesized that salamanders would respond in three ways to partial forest harvests that alter the level and pattern of green-tree retention. First, because salamanders are forest-dwellers, we projected that their numbers would correlate with tree basal area, i.e., abundance would decrease with increasing harvest intensity. Second, if salamander abundance does correlate with tree basal area, then harvest level rather than harvest pattern should have a greater impact on salamander numbers within a standardized area. Finally, because terrestrial salamanders largely reside in and on the ground, we predicted that their numbers would relate more closely to forest floor characteristics than to tree basal area, particularly immediately post-harvest when disrupted ground conditions resulting from logging are most pronounced.

METHODS

Study Area and Design

Six harvest treatments within low- to mid-elevation (range: 244 to 1614 m) Douglas-fir dominated forests provided the framework of the study: 100-percent basal area retention (control); 75-, 40-, and 15-percent aggregated retention (75%A, 40%A, 15 %A); and 40 and 15 percent dispersed retention (40%D, 15%D). In 75%A units, harvest areas were restricted to three 1-ha patches; in 40%A units, trees were retained in five 1-ha patches; in 15%A units, trees were retained in two 1-ha patches. Treatments were replicated six times across four blocks in western Washington (Capitol Forest, Butte, Paradise Hills, Little White Salmon) and two blocks in western Oregon (Watson Falls, Dog Prairie) for a total of 36, 13-ha treatment units. Units were harvested in 1997 and 1998. See Aubry et al. (1999) and Lehmkuhl et al. (1999) for additional study site and treatment details.

Forest Structure and Salamander Sampling

Sampling in each unit was based on a permanent 9 x 7 or 8 x 8-point grid with 40-m spacing between adjacent points. Vegetation characteristics were measured at approximately half the grid points in each of the treatment units both before and after harvest. Variables examined in this study included: tree basal area (≥5.0-cm d.b.h.), percentage of herb cover (100 percent maximum; <1-m height potential), percentage of tall shrub cover (100 percent maximum; >1-m height potential), and coarse woody debris (CWD) volume (≥10-cm diameter). Additional information on vegetation sampling is published in Halpern et al. (1999) and McKenzie et al. (2000).

We placed one 72- x 15-cm pitfall trap at each grid point on each unit to sample salamanders for 2 years both before and after harvest. Traps were open approximately 4 weeks during the months of October and November (sampling period range: 26 to 32 days) when ground surface activity of salamanders is high due to fall rains (Olson 1999). All captured salamanders were identified to species, and live individuals were temporarily removed from the units to prevent multiple sampling of individuals within a trapping season.

Statistical Analyses

For each four-week trapping session, salamander samples were standardized to number of individuals captured per 1000 trap nights per unit. To examine effects of retention treatments on salamander abundance, we used randomized block one-way ANCOVAs on differences between means of the two before and after harvest samples. Tree basal area and elevation were used as covariates. Differences among means were tested with Tukey's Studentized Range (HSD) Test. One-way ANOVAs were used to test for differences in forest characteristics among treatments both before and after harvest, and pre- and post-harvest correlations among forest characteristics also were determined. Additionally, we used multiple-regression analyses with blocks as indicator variables to identify which of the four forest features sampled correlated with mean post-harvest salamander captures. We also included elevation as a possible predictor variable. Data were transformed for analysis when necessary and analyses were performed using SAS/STAT software (SAS Institute Inc. 1989). Significant differences were tested at $\alpha = 0.1$ except for the HSD Test ($\alpha = 0.05$).

RESULTS AND DISCUSSION

Preharvest treatment means (N = 6) for tree basal area, percentage of herb cover, percentage of tall shrub cover, and CWD volume all were similar among treatment units (p-values ≥ 0.38; table 1). Herb cover decreased with increasing basal area before harvest ($r = -0.53$, $p < 0.001$), most likely as a consequence of reduced light availability

Table 1—Pre- and post-harvest means for structural characteristics of 13-ha forest units treated to different levels and patterns of tree basal area retention

	Forest characteristics							
	BA (m^2/ha)		Herbs (%)		Tall shrubs (%)		CWD (m^3/ha)	
Treatments	Pre	Post	Pre	Post	Pre	Post	Pre	Post
100%	66.5	68.1	36.2	33.9	21.3	21.0	150.6	145.8
75%A	63.7	51.4	37.7	33.0	26.5	17.7	184.3	157.8
40%D	61.8	29.5	36.4	22.5	24.4	11.4	159.4	114.6
40%A	69.0	27.1	32.8	19.7	30.0	20.4	191.6	151.6
15%D	67.5	11.8	30.1	14.6	26.6	9.8	157.0	141.3
15%A	72.0	11.5	31.4	13.4	23.3	12.3	293.2	265.0

Note: BA = tree basal area; Herbs = cover of vegetation with <1-m height potential; Tall shrubs = cover of vegetation with >1-m height potential; CWD = volume of coarse woody debris with ≥10-cm diameter. Treatments include: 100% basal area retention; 75, 40, and 15% aggregated retention (75%A, 40%A, 15%A); and 40 and 15% dispersed retention (40%D, 15%D). Each treatment was replicated in six blocks in Douglas-fir dominated forests in western Washington and western Oregon.

on the forest floor as tree crown cover increased with increasing basal area (e.g., Bailey et al. 1998). Conversely, herb cover decreased with decreasing basal area after harvest ($r = 0.38$, $p = 0.02$) when elevated timber removal resulted in more extensive ground disturbance (Halpern and McKenzie 2001). Although tall shrub cover had positive correlations with herb cover both before ($r = 0.48$, $p = 0.003$) and after harvest ($r = 0.49$, $p = 0.002$), it did not mimic basal area trends (p-values > 0.60) as did herb cover. CWD volume had a marginally significant negative relationship with herb cover after harvest ($r = -0.3$, $p = 0.07$), but no apparent trends with basal area (preharvest: $r = -0.13$, $p = 0.46$; post-harvest: $r = -0.21$, $p = 0.22$).

Eight salamander species were encountered during the study, but no species was captured in all units before harvest, and few were captured in all six replicates of a treatment after harvest (table 2). Three species were found exclusively in one unit each following harvest: Cascade torrent salamander (*Rhyacotriton cascadae*), Pacific giant salamander (*Dicamptodon tenebrosus*), and clouded salamander (*Aneides ferreus*). The western red-backed salamander (*Plethodon vehiculum*) was found only on the low elevation units of Capitol Forest. Additionally, control units had from 42 percent fewer captures to 99 percent more captures in the post-harvest period compared with preharvest (table 3). These data highlight the large temporal, spatial, and geographic variability associated with the distribution of salamanders and their aboveground activities even in the absence of forest management. This variability significantly complicates interpretation of salamander abundance estimates and forest management impacts (Pechmann et al. 1991).

Trapping resulted in the capture of 3657 individual salamanders at the rate of 13.8 captures per 1000 trap nights before harvest and a similar rate of 14.3 captures per 1000 trap nights after harvest (range: 0 – 82.2 captures per 1000 trap nights per treatment unit). Randomized block ANCOVA did not detect a significant difference in capture rate change across treatments before or after harvest ($p = 0.48$), and captures did not differ between aggregated and dispersed units at the same harvest level. Only three of the eight salamander species encountered were found in sufficient numbers and in sufficiently wide distributions to analyze species captures with ANCOVA. These species were ensatina (*Ensatina eschscholtzii*), northwestern salamander (*Ambystoma gracile*), and rough-skinned newt (*Taricha granulosa*) (table 3). There were no significant treatment effects for any of these species on the change in capture rates before and after harvest (p-values > 0.21), including no capture differences between dispersed and aggregated treatments. Collectively, these results caused us to reject the hypotheses that salamanders would decrease in abundance with increasing harvest intensity, and that harvest level would have a greater impact on salamander numbers than harvest pattern.

Salamanders may not have exhibited immediate post-harvest responses to basal area reduction for several reasons. Because of high site tenacity, small home ranges, fossorial tendencies, and high life potential (up to several decades; Bowler 1977), salamander populations may have delayed aboveground responses to forest management activities (Bunnell et al. 1997). High temporal and spatial capture variability of salamanders also may reduce our ability to detect significant differences due to treatments.

Table 2—The percentage of 13-ha forest treatment units (N = 6 units per treatment) in which individual salamander species were trapped before and after harvest

Species	All units preharvest	Post-harvest Units					
		100%	75%A	40%D	40%A	15%D	15%A
Ensatina (T) (*Ensatina eschscholtzii*)	86.1	100.0	100.0	83.3	83.3	100.0	100.0
Western red-backed salamander (T) (*Plethodon vehiculum*)	22.2	16.7	16.7	16.7	16.7	16.7	16.7
Northwestern salamander (A) (*Ambystoma gracile*)	83.3	83.3	66.7	100.0	66.7	83.3	100.0
Rough-skinned newt (A) (*Taricha granulosa*)	63.9	66.7	66.7	33.3	66.7	83.3	66.7
Cascade torrent salamander (A) (*Rhyacotriton cascadae*)	8.3	0.0	16.7	0.0	0.0	0.0	0.0
Long-toed salamander (A) (*Ambystoma macrodactylum*)	13.9	27.8	16.7	16.7	16.7	0.0	16.7
Pacific giant salamander (A) (*Dicamptodon tenebrosus*)	13.9	0.0	0.0	0.0	16.7	0.0	0.0
Clouded salamander (T) (*Aneides ferreus*)	27.8	0.0	0.0	0.0	0.0	0.0	16.7

Note: Basal area retention treatments in Douglas-fir dominated forests are identified in table 1. Salamander species followed by (A) are those that breed in water and have both aquatic and terrestrial phases; species followed by (T) breed and live their entire lives on land. Species are ordered from most to least abundant before harvest.

Additionally, treatments in the DEMO study were based on percentage of basal area reductions rather than absolute basal areas. Thus, variability in unit basal areas prior to harvest was maintained among units within treatments following harvest. The range of absolute basal area conditions within treatment units likely contributed to the variability in salamander captures among replicates within treatments.

Results from the regression analyses that focused on captures and absolute measures of forest structure suggest that salamander captures reflect variation in forest features beyond basal area. Regression analysis of post-harvest salamander captures (per 1000 trap nights) versus forest structural features resulted in significant correlations (p-values ≤ 0.0001) for the four most abundant species. Captures for both the western red-backed salamander and the rough-skinned newt were positively correlated with herb cover ($r^2 = 0.95$ and 0.62, respectively). The northwestern salamander also was captured more frequently as herb cover increased, but this species had a negative relationship with tree basal area ($r^2 = 0.66$). Ensatina had positive correlations with both basal area and CWD volume ($r^2 = 0.85$). When all species were examined together, CWD volume was the forest characteristic of those measured that best predicted total salamander captures ($r^2 = 0.78$, $p = 0.0001$). Tall shrub cover was not a significant correlate for any of the four most abundant salamander species analyzed individually or for salamanders as a group.

The regression analyses also indicated that combinations of tree basal area, herb cover, and CWD volume were better able to predict captures of the fully-terrestrial salamanders, ensatina and western red-backed salamander, than captures of the aquatic breeders, northwestern salamander and rough-skinned newt ($r^2 \geq 0.85$ and 0.66, respectively). Whereas the presence of aquatic species on land is tied foremost to the presence of nearby aquatic breeding areas (Gibbons and Semlitsch 1981), terrestrial salamanders are impacted by events on land throughout their entire lives. This tighter link with conditions on land suggests that terrestrial salamanders may better reflect changing forest conditions than aquatic species. Yet, for all species in this study, ground cover in the form of CWD or herbaceous vegetation was a significant habitat component. Coarse wood debris and herbs appear to provide salamanders with cover, ameliorate microclimatic conditions, and serve as a source of and/or foraging surface for arthropod prey (Butts and McComb 2000, Jaeger 1978). Basal area, however, was a poor correlate of the forest structures examined in the study in the first few years following harvest; this probably resulted

Table 3—Mean salamander captures and the percent distribution of captures across basal area retention treatments before and after harvest

Species	Captures		Basal area retention treatments													
			100%		75%A		40%D		40%A		15%D		15%A	
	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
Ensatina (T)	7.20	7.03	20.1	21.3	16.2	14.1	14.8	11.6	13.2	13.5	15.1	18.7	20.6	20.8
Western red-backed salamander (T)	3.25	4.50	11.3	24.8	14.9	14.8	11.3	11.5	17.4	17.0	15.4	15.9	29.7	15.9
Northwestern salamander (A)	2.33	2.00	7.9	10.8	17.9	25.8	17.9	13.3	15.7	15.8	8.6	13.3	32.1	20.8
Rough-skinned newt (A)	0.55	0.58	15.1	25.7	18.2	25.7	18.2	8.6	9.1	14.3	24.2	14.3	15.1	11.4
Cascade torrent salamander (A)	0.20	0.08	0.0	0.0	41.7	100.0	0.0	0.0	8.3	0.0	50.0	0.0	0.0	0.0
Long-toed salamander (A)	0.17	0.15	40.0	11.1	10.0	22.2	0.0	11.1	10.0	11.1	0.0	0.0	40.0	44.4
Pacific giant salamander (A)	0.08	0.02	0.0	0.0	20.0	0.0	0.0	0.0	40.0	100.0	40.0	0.0	0.0	0.0
Clouded salamander (T)	0.02	0.02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	100.0	100.0

Note: Basal area retention treatments are described in table 1. Each post-harvest treatment consists of six replicated units in Douglas-fir dominated forests, for a total of 36 preharvest units. Salamander scientific names are provided in table 2. Aquatic salamanders (A) have both aquatic and terrestrial life stages; terrestrial salamanders (T) have no aquatic phase. Salamander captures are standardized to number of captures per 1000 trap nights.

from harvest activity-induced forest floor disturbance (Halpern and McKenzie 2001). Consequently, captures of individual salamander species also correlated poorly with tree basal area after harvest because salamanders are more impacted by forest structural features near the ground. These results support our third hypothesis concerning close salamander relationships with forest floor structure.

Results of this study suggest that salamanders are able to tolerate major habitat disruptions resulting from harvest, at least during the immediate post-treatment years. They also respond to altered environmental conditions induced by forest management that are not directly tied to basal area. Because salamander captures, and presumably salamander populations, are significantly correlated with CWD and herbaceous vegetation, it should be possible to harvest timber while simultaneously managing forest floor habitat features important to salamanders during forest regeneration.

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Fate of Taxa After Variable-Retention Harvesting in Douglas-fir Forests of the Northwestern United States

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ABSTRACT

Regeneration harvests that retain differing proportions of initial stand basal area have been proposed as an alternative to clearcutting as a way to conserve biological diversity in forests. Variable-retention harvesting has been mandated on federal lands in the Pacific Northwest region of the United States, but its efficacy for maintaining species diversity is largely untested. The DEMO (Demonstration of Ecosystem Management Options) study was initiated to evaluate the effects of differing levels and patterns of retained trees on various groups of forest-dwelling organisms. Six treatments were implemented in Douglas-fir forests at each of six locations (blocks) in western Washington and Oregon: 100-percent retention (control), 75-percent aggregated retention (three 1-ha gaps cut within the treatment unit), 40-percent dispersed retention (regular distribution of residual trees), 40-percent aggregated retention (five uncut 1-ha forest aggregates), 15-percent dispersed retention (regular distribution of residual trees), and 15-percent aggregated retention (two uncut 1-ha forest aggregates). Herbs, ectomycorrhizal sporocarps (mushrooms and truffles), canopy arthropods, amphibians, forest-floor small mammals, and breeding birds were sampled before and soon after treatments were implemented. In this paper, we analyze the probability that taxa detected before treatment were detected after treatment. ANCOVA suggested that the probability of confirming the persistence of mushroom, truffle, and bird taxa in each unit after treatment was directly related to the proportion of live trees retained. Orthogonal contrasts indicated that most treatment effects were attributable to differing levels of retention, although fewer bird species were detected after harvest in dispersed-retention treatments compared to aggregated-retention treatments. Regression analysis with continuous stand structural variables also indicated that lower residual stand densities resulted in a significantly greater probability of detection failure for all groups, and that additional variation in the response of canopy arthropods and birds was accounted for by covariates representing differences in vertical structure. Further assessment of both the loss of initial taxa and the appearance of new taxa will be conditioned on possible changes in detection probabilities and known and hypothesized habitat relationships or trophic interactions.

KEYWORDS: Variable retention, biodiversity, timber harvest, crown area profile, stand structure.

INTRODUCTION

Variable-retention harvesting may enable economically viable timber extraction while maintaining elements of biological diversity that might be lost if stands were clearcut (Franklin et al. 1997). Retained trees are intended to serve at least three functions: (1) maintain forest-dwelling taxa until a new stand is established, (2) provide forest connectivity in harvested areas, and (3) enhance the structural diversity of the future stand. Federal forest-management strategies in the northwestern United States (the Northwest Forest Plan) specify the retention of live trees in at least 15 percent of each harvest unit, with at least 70 percent of retention in forest aggregates 0.2-1.0 ha in size (Tuchman et al. 1996). To date, few studies have been conducted to evaluate the extent to which different levels or patterns of retention maintain late-seral forest species in regenerating stands. The DEMO (Demonstration of Ecosystem Management Objectives) study was established in the mid-1990s to address this knowledge gap with a controlled harvest

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experiment implemented by a multidisciplinary team of researchers and forest managers (Aubry et al. 1999).

Variable-retention treatments may fail to maintain forest biodiversity for at least two reasons: local extirpation of species or reduction of populations below viable levels. In this paper, we focus on the former by determining which taxa were detected prior to variable-retention harvest treatments, then assessing their apparent fate several years after harvest. There are potential limitations to this kind of analysis. Simple sampling error may confound our assessments of the presence or absence of taxa both before and after harvest. In addition, variable-retention harvests create post-treatment environments that may differ substantially in composition and structure from pretreatment environments. Consequently, apparent treatment effects on the persistence of taxa may reflect local extirpations, or may be due to sampling error or changes in detection probabilities, even if populations remain stable (Gu and Swihart 2004). In this paper we limit our analysis to testing for (1) the effect of retention level and spatial pattern on the probability of detecting the persistence of taxa several years after harvest, and (2) the effects of residual stand structural parameters on the probability of detection. Future work will address the causal mechanisms of treatment effects and the likelihood that our results reflect local extirpations vs. alternative explanations.

METHODS

Six study locations (blocks) were selected to represent a diversity of mature forests (65-170 years) dominated by Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco); Aubry et al. 1999). Two blocks were located in the Cascade Range in central Oregon, three in the Cascade Range in southern Washington, and one in the Coast Range in southwestern Washington. Elevations ranged from ca. 200-1700 m and slopes varied from gentle to steep, with a broad range of aspects represented (table 1). Three blocks (Butte, Capitol Forest, and Watson Falls) were in the western hemlock (*Tsuga heterophylla*) forest zone, one (Little White Salmon) was in the grand fir (*Abies grandis*) zone, one (Dog Prairie) was in the white fir (*A. concolor*) zone, and one (Paradise Hills), was in the Pacific silver fir (*A. amabilis*) zone (Franklin and Dyrness 1973). At each block, five harvest treatments and a control were randomly assigned to 13-ha experimental (treatment) units. Treatments differed by the level (percentage of initial basal area) and spatial pattern (dispersed vs. aggregated) of retained trees as follows: (1) 100 percent: 100-percent retention (control); (2) 75%A: 75-percent aggregated retention (three circular, 1-ha patch cuts in an uncut matrix); (3) 40%D: 40-percent dispersed retention (uniform spatial distribution of residual trees); (4) 40%A: 40-percent aggregated retention (five circular 1-ha forest aggregates in a cut matrix); (5) 15%D: 15-percent dispersed retention (uniform distribution of residual trees); and (6) 15%A: 15-percent aggregated retention (two circular 1-ha forest aggregates in a cut matrix). Residual trees in the dispersed treatments were selected from larger and more stable dominants and co-dominants.

Before and after treatment in each unit, DEMO researchers sampled overstory and understory vegetation, sporocarps (mushrooms and truffles) of ectomycorrhizal fungi, canopy arthropods, amphibians, forest-floor small mammals, and diurnal breeding birds (Aubry et al. 1999). For most taxa, sampling was conducted within a grid of 63 or 64 sampling points with 40-m spacing. Herbaceous and woody plants were measured on permanent fixed-area plots (Halpern et al. 1999, 2005). Mushrooms and truffles were tallied by genus or species on temporary and permanent fixed-area plots (Cazares et al. 1999, Luoma et al. 2004). Canopy arthropods were sampled by collecting one branch in each third of the canopy of one Douglas-fir tree. In 15- and 40-percent aggregated-retention treatments, the sample tree was located near the middle of a forest aggregate (Progar et al. 1999); after harvest, an additional tree was sampled at the edge of the aggregate (Schowalter et al., in press). Resulting samples were sorted to the lowest taxonomic level possible. Amphibians and small mammals were sampled with pitfall traps located at each grid point (Lehmkuhl et al. 1999). Traps were opened for approximately 4 weeks during October and early November between the onset of fall rains and the beginning of snow accumulation. Diurnal breeding birds were sampled by tallying all species detected within 50 m of four point-count stations in each treatment unit, with stations ≥160 m apart and ≥80 m from the stand perimeter. Point-counts were completed within 3 hours of dawn, and repeated six times during the spring breeding period (Lehmkuhl et al. 1999).

Basal area, tree density, canopy depth, and canopy cover were computed from the plot data before and after harvest, along with various sets of indices of vertical diversity in crown area profile (Dubrasich et al. 1997). Individual crown profiles and implied canopy cover were estimated from existing crown-width and crown-profile equations applied to individual trees (Dubrasich et al. 1997, Hann 1998). The first set of indices was based on the crown profile, irrespective of species. Vertical uniformity in this profile was expressed in three ways: the standard deviation of crown area among 0.5-m height intervals (SD_{CA}); Shannon index of diversity among 0.5-m height intervals (H'_{CA}), with crown area as the measure of abundance; and Pileou index of evenness among the same 0.5-m height intervals (J'_{CA}).

Table 1—General topographic and forest attributes for each of the six experimental blocks in the DEMO study. Minimum and maximum values represent treatment unit means.

Location/ Block	Elevation (m)	Slope (%)	Aspect	Stand age (yr)	Site index (m at 50 yr)
Oregon Cascade Range: Umpqua National Forest					
Watson Falls	945-1310	4-7	flat	110-130	40-43
Dog Prairie	1460-1710	34-62	SW	165	30
Washington Cascade Range: Gifford Pinchot National Forest					
Butte	975-1280	40-53	E-SE	70-80	27-32
Little White Salmon	825-975	40-60	NW-NE	140-170	30
Paradise Hills	850-1035	9-33	variable	110-140	26-33
Washington Coast Range: Washington State Department of Natural Resources					
Capitol Forest	210-275	28-52	variable	65	37-41

Because canopy diversity may result either from stratification of species into separate layers or from presence of most species in all layers, a second set of indices was computed for the crown area profile of each species separately. Weighted average indices (SD_{sp}, H'_{sp}, and J'_{sp}) were then determined by applying the total crown volume of each species as the weight. The third set of indices was based on species diversity (H'), species evenness (J'), and species richness (S) at each 0.5-m height interval in the stand. The weighted mean of each measure was then computed across height intervals by using total crown area at a given height as the weight (H'_{ht}, J'_{ht}, S_{ht}).

To reduce the potentially confounding influences of sampling error and variation in detection probabilities among treatments and among taxa, we eliminated wildlife species from consideration that were not effectively sampled with the methods used, had spatial requirements that exceeded the size of the treatment units, or were not associated with forested habitats. We eliminated amphibians altogether because only a few species were appropriate to include in our analyses. For most taxa, only individuals that could be identified to the species level were included. However, due to the challenges involved in identifying many arthropods and fungi to species, we also included higher level taxonomic designations for those groups. The primary response variable for assessing potential losses in biological diversity was the probability of detection failure; i.e., that a taxon present before harvest was not detected after harvest. We analyzed data for each taxonomic group as a whole, and also conducted separate analyses on the subset of herbs, small mammals, and birds that we considered to be closely associated with late-seral forest conditions.

The response variable in all analyses was binary with respect to detection after harvest, with individual taxa serving as observational units. We therefore performed analyses of covariance (ANCOVA) in the context of generalized linear models, specifying a binomial regression with logit link function (McCullagh and Nelder 1989). The preharvest measure of abundance for each taxon was introduced as a covariate to help account for differing probabilities of detection between common and rare taxa. Overall treatment effects were tested by reduction-in-deviance tests between a reduced model with only block effects and the covariate vs. a full model with an added set of treatment indicator variables. A set of *a priori* orthogonal contrasts was also computed to test for differences among specific treatments (table 2; Zar 1984). The ANCOVA was then reformulated by retaining block effects but replacing categorical treatments with continuous variables representing residual stand structure, initial stand structure, and change in stand structure. If a significant set of stand structural variables was found for a taxonomic group, we reintroduced a set of indicator variables representing the treatments to test for residual treatment effects. The effects of stand structural variables indicated by the binomial regressions were assessed by first plotting the predicted probability of detection failure on stand-density reduction, while holding other predictor variables constant. The probabilities were then predicted across the same range in stand density reduction for one to three additional levels of a second predictor to produce several lines on the same graph.

Table 2—Orthogonal contrasts in the DEMO study for treatment effects analyzed with ANCOVA

Name	Treatment contrast		
Treatment	100%	vs.	75%A, 40%D, 40%A, 15%D, 15%A
Level	40%D, 40%A	vs.	15%D, 15%A
Pattern	40%D, 15%D	vs.	40%A, 15%A
Interaction	40%D, 15%A	vs.	40%A, 15%D
75%	75%A	vs.	40%D, 40%A, 15%D, 15%A

RESULTS

The harvest treatments produced residual basal areas ranging from 8.4 to 99.5 m^2 ha^{-1}; however, residual basal area for a given treatment varied two-fold among blocks due to differences in initial basal area (fig. 1). Canopy cover was closely correlated with basal area, ranging from approximately 15 to 96 percent. However, cover was greater for a given level of retention in the dispersed treatments due to the uniform spacing of residual trees and lack of crown overlap.

Analysis of covariance indicated significant treatment effects for mushrooms, truffles, and birds ($P \leq 0.026$; table 3). When bird and herb species with an affinity for late-seral forest conditions were analyzed separately, there was a significant treatment effect for both groups ($P \geq 0.01$; table 4). Orthogonal contrasts showed that significant treatment effects for mushrooms, truffles, and late-seral herbs were driven by greater detection failure in harvested units relative to controls (tables 3 and 4). For all birds, a greater proportion of species was "lost" from the 15-percent treatments than from the 40-percent treatments ($P = 0.002$), and a greater proportion was "lost" from the dispersed treatments than from the aggregated treatments ($P = 0.030$). Pretreatment abundance was a very significant covariate in all ANCOVAs ($P < 0.001$).

Residual basal area was not a significant covariate in any of the ANCOVAs ($P > 0.05$), although correcting for differences in residual basal area by adding this covariate strengthened the significance of many of the orthogonal contrasts (data not shown).

The probability of "losing" an herb species increased with both harvest intensity (greater reduction in tree density) and residual basal area of Douglas-fir (fig. 2). As with the other taxonomic groups, the probability of detection failure for an herb species increased significantly as its preharvest abundance declined (fig. 3). Stand-density reduction had a similar effect on mushrooms and truffles, but for this group the aspect of stand density that best captured the effect was change in canopy cover. Canopy arthropods were most responsive to descriptors of the crown area profile. Detection failure rates increased with greater reduction in species-specific canopy diversity (decline in H'_{sp}). The strongest predictor for canopy arthropods, however, was height-specific canopy species richness (S_{ht}); treatments imposing a greater decline in average tree species richness for a fixed canopy height resulted in an increase in the probability of detection failure. Similar to herbs, birds responded most strongly to reductions in stand density, particularly as measured by change in total basal area. For birds, however, detection failure rate was significantly higher in dispersed than in aggregated treatments (fig. 4). Residual canopies that were characterized by greater diversity (lower SD_{CA}) had a higher probability of bird species "loss."

When categorical treatment variables were tested in models containing stand structural attributes, no marginal treatment effects could be detected. One or two stand structural variables performed about as well as categorical treatment variables alone.

DISCUSSION

Correct interpretation of the "effects" of canopy structural variables requires an understanding of how variable-retention treatments change crown area profiles. For example, the dispersed-retention treatments tended to produce profiles with a more uniform vertical distribution of crown area, so although crown volume was substantially reduced, SD_{CA} indicated greater vertical diversity. If the treatment units were relatively uniform in structure, aggregated-retention

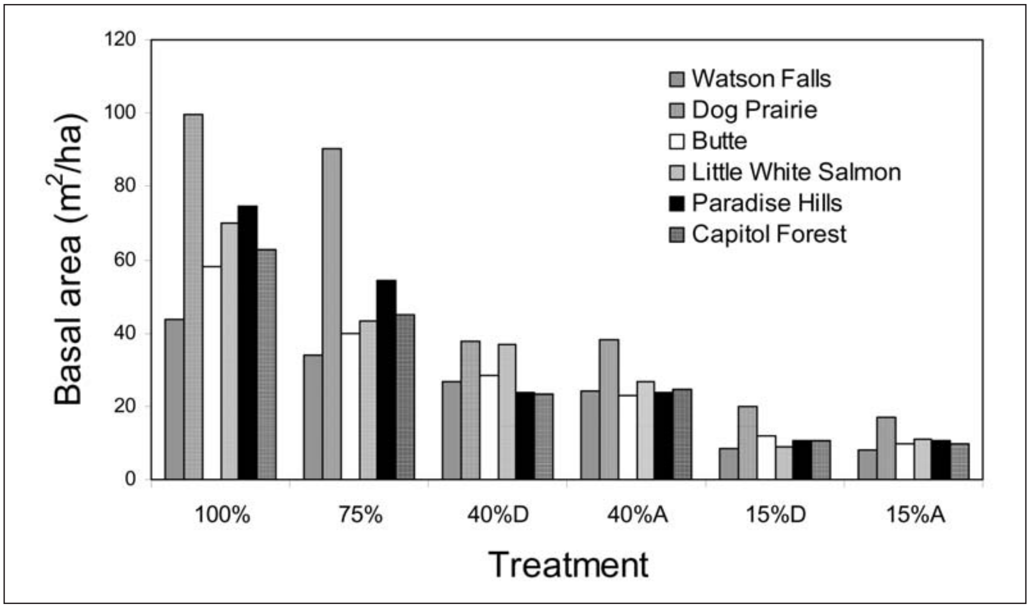

Figure 1—Residual basal area for each treatment unit in each block of the DEMO study. D = dispersed; A = aggregate.

Table 3—Test of treatment effects on the probability that a taxon present before harvest was not detected after harvest. *P*-values are shown for the null hypothesis of no treatment effect. Only significant ($P < 0.05$) and marginal ($0.05 < P \leq 0.15$) test results are shown.

Test	Herbs	Shrubs	Mushrooms	Truffles	Canopy arthropods	Forest-floor small mammals	Diurnal breeding birds
Treatment effect	—	—	0.002	0.026	—	—	0.006
Orthogonal contrasts							
Treatment	—	—	0.001	0.005	—	—	—
Level	—	0.069	0.017	0.109	—	—	0.002
Pattern	—	—	—	—	—	—	0.030
Interaction	—	—	—	—	—	—	—
75 percent	—	—	0.011	0.067	—	—	—

Table 4—Test of treatment effects on the probability that a taxon associated with late-seral forest conditions that was present before harvest was not detected after harvest. *P*-values are shown for the null hypothesis of no treatment effect. Only significant ($P < 0.05$) and marginal ($0.05 < P \leq 0.15$) test results are shown.

Test	Late-seral herbs	Late-seral shrubs	Late-seral, forest-floor small mammals	Late-seral diurnal breeding birds
Treatment effect	0.005	—	—	0.010
Orthogonal contrasts				
Treatment	0.002	—	—	—
Level	—	—	—	0.038
Pattern	0.110	—	—	—
Interaction	—	—	—	—
75 percent	0.028	—	—	—

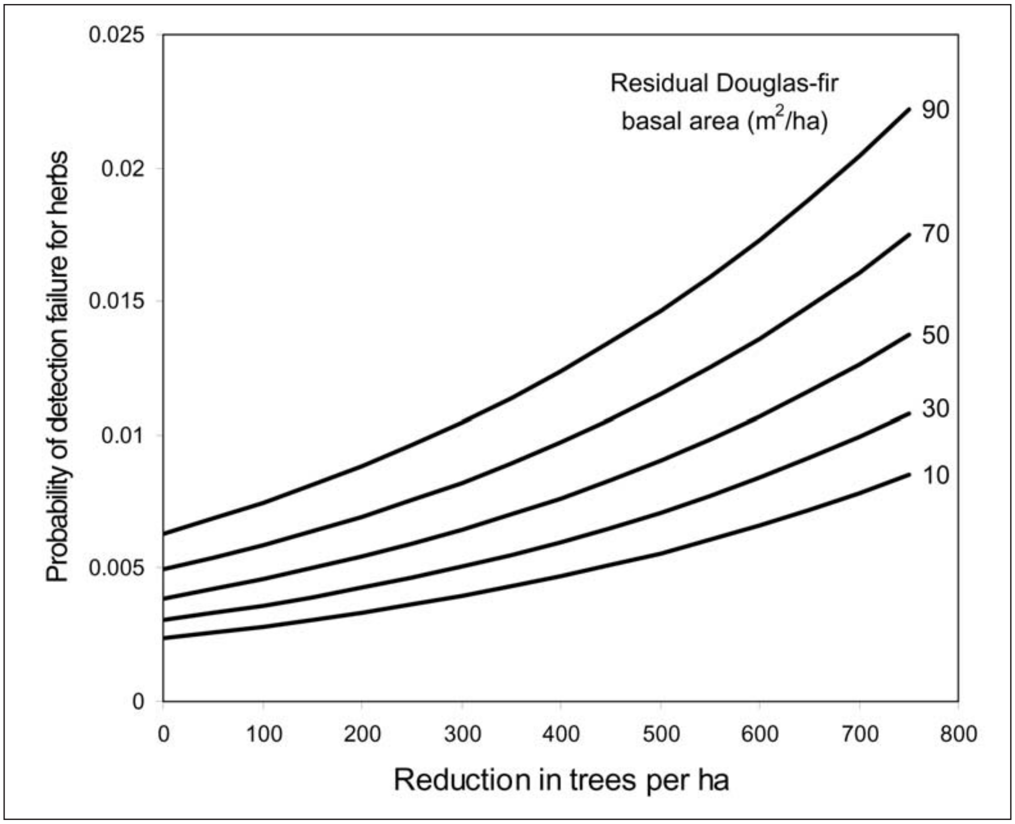

Figure 2—Trend in probability that an herb species present before harvest was not detected after harvest, as a function of reduction in stand density and residual Douglas-fir basal area. Preharvest frequency was also a predictor in the binomial regression and was held constant at 30 microplots per treatment unit.

treatments should have had little effect on the diversity of the average crown-area profile. However, these treatments obviously increased horizontal variation in stand density and other aspects of stand structure. Also, because dominant and codominant trees were retained in the dispersed treatments, structure was homogenized. The strong correlation between variables representing stand density and those representing vertical canopy structure made assessing their relative importance difficult, however. For example, the probability of detection failure for birds increased with residual canopy diversity, a result attributable to a combination of greater tree density and more even distribution of crown volume among canopy layers. As stand density is reduced, intensity of disturbance and impacts to the structure of forest-floor habitat, food sources, and microclimate increase. Greater residual canopy diversity may either indicate lower effective canopy cover (because canopy diversity increases with decreasing retention) or perhaps a reduced probability of detecting a species that is present (Gu and Swihart 2004).

The short-term function of residual live trees as refugia for different forest species was substantiated in a general way by early results of the DEMO variable-retention experiment. As noted earlier, apparent "loss" of taxa can be attributed to several alternative explanations besides local extirpation. For plants and fungi these include sampling error, burial by slash, destruction of only above-ground plant parts by harvesting, reduced sporocarp production, and temporary dormancy induced by microclimatic stress (e.g., Lessica and Steele 1994). For small mammals and birds, failure to detect a species after harvest may have been due to a real change in abundance induced by the harvest treatments, but also to sampling error or a change in probability of detection (Gu and Swihart 2004, Hopkins and Kennedy 2004). For example, if the treatments affected home range sizes or travel patterns of small mammals or the behaviors of birds, treatment effects could be partly attributable to changes in detection probabilities.

For each taxonomic group, the proportion of taxa that were not detected after timber harvest depended directly or indirectly on the degree of disturbance represented by each harvest treatment. In some cases (mushrooms, truffles, and birds), this effect was evident in the orthogonal contrasts

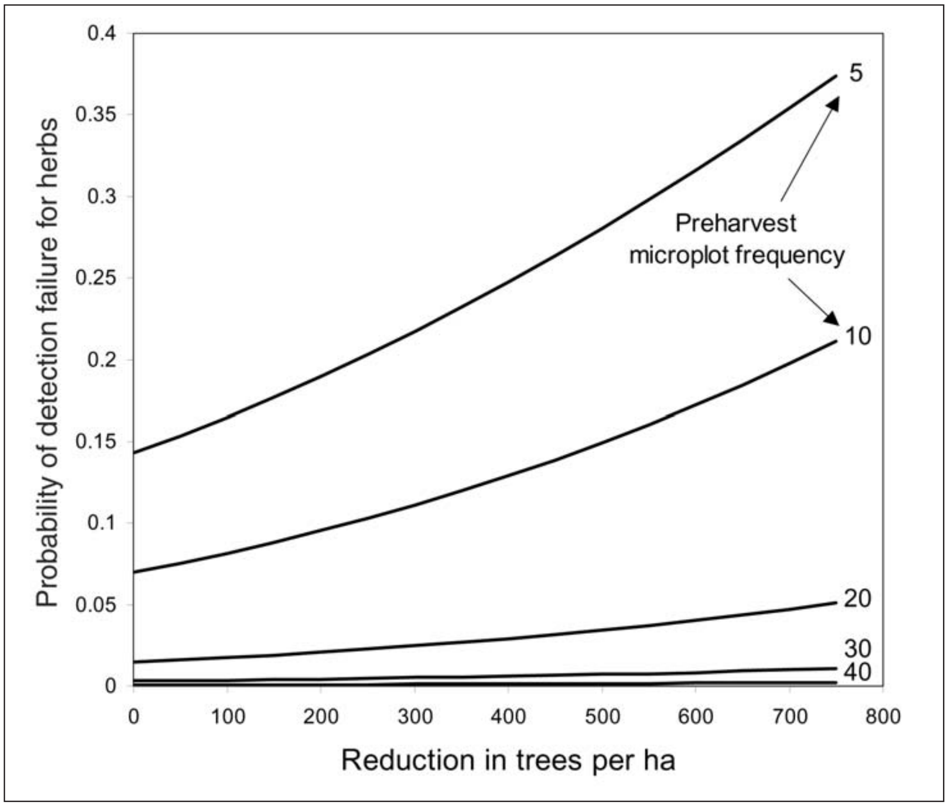

Figure 3—Trend in probability that an herb species present before harvest is not detected after harvest, as a function of reduction in stand density and preharvest microplot frequency. Residual Douglas-fir basal area was also a predictor in the binomial regression and was held constant at 30 m^2/ha.

embedded in each ANCOVA. However, continuous structural variables introduced in the regression analyses revealed direct or indirect effects of variable-retention harvesting that could not be detected with ANCOVA. To some degree, a single covariate representing residual stand density was redundant with the categorical treatment variables, as indicated by the lack of significance for residual basal area in the ANCOVAs. Sets of continuous covariates, however, depicted more detailed aspects of stand or habitat structure (i.e., structural variability within the same nominal treatment), to the extent that categorical treatment variables could add no further explanatory power to the regression models.

For at least one taxonomic group (birds), the more widespread disturbance in dispersed treatments resulted in greater species "loss" than in aggregated treatments at corresponding levels of retention. Although a continuous variable representing variation in horizontal stand structure may have been capable of accounting for this effect, categorical representation in the ANCOVA was adequate for DEMO given the standardized shape and size of forest aggregates.

Subtle variation in residual stand structure at a given level of retention may have important implications that vary with the habitat requirements of individual species. Taxa that are restricted to the forest floor, such as herbs and ectomycorrhizal sporocarps, were influenced primarily by changes in stand density. Conversely, taxa utilizing more of the vegetation profile, such as canopy arthropods and birds, responded to both changes in stand density and more subtle variation in crown-area profile. Small mammals responded in a manner that was intermediate between these groups. Factors such as risk of wind damage constrained the potential range in residual tree size and stand structure that could be achieved with variable-retention treatments, but some variation in design could have had a significant effect on residual species diversity, at least for a limited time after harvest. For example, leaving a wider range of tree sizes in dispersed-retention treatments would have enhanced vertical diversity, but this objective may not be achievable if trees from lower crown classes in the original stand are more susceptible to windthrow. However, variable-retention treatments that promote variation in the size and species of retained trees seem capable of retaining a greater variety of wildlife species immediately after harvest. Such modifications of

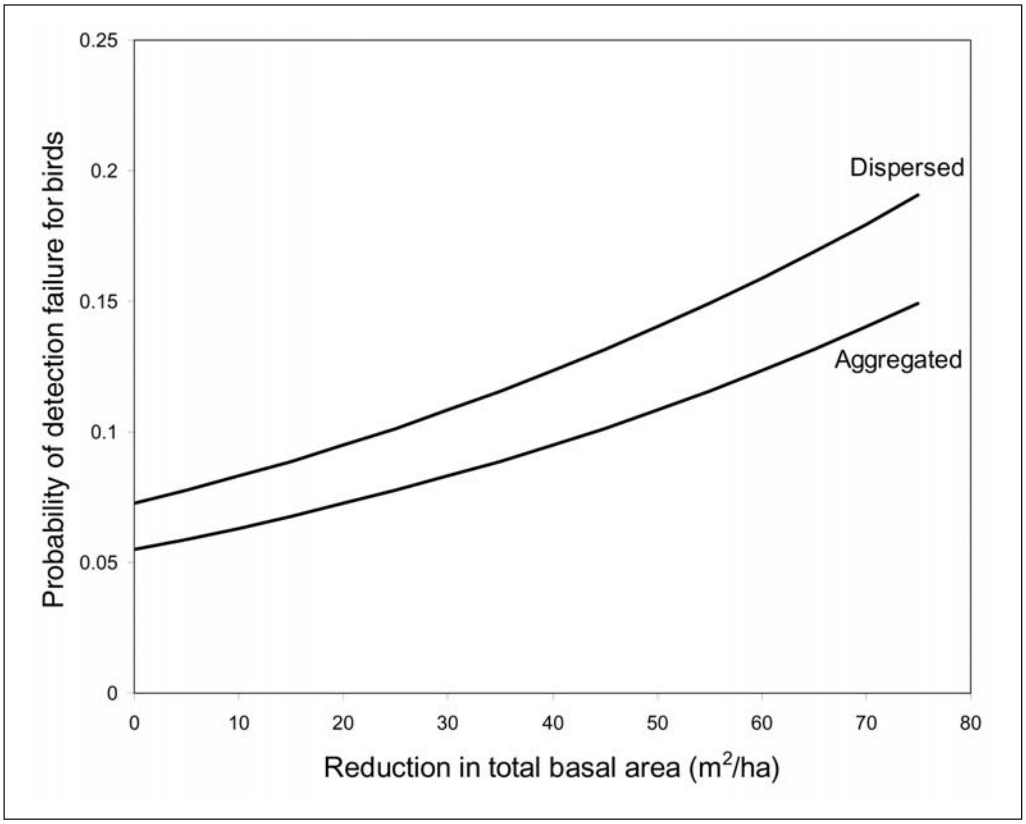

Figure 4—Trend in probability that a bird species present before harvest was not detected after harvest, as a function of reduction in total basal area and pattern of retention (aggregated vs. dispersed). Preharvest bird count was also a predictor in the binomial regression and was held constant at two observations per treatment unit.

the variable-retention treatments implemented in this study may promote greater structural diversity and, hence, greater retention of late-seral species in future stands.

Further analyses will focus on hypothesized interactions and trophic relationships among various taxa, applying such approaches as path analysis and structural equation modeling. However, further inferences are limited by the relatively low degrees of freedom resulting from the small number of treatment units in the DEMO study. This statistical issue is one that typifies many other forest ecological experiments conducted on an operational scale. On-going field work to monitor responses of various taxa will facilitate interpretation of short-term responses in the context of longer term changes in presence and abundance, and should clarify some of the mechanisms by which populations are responding to the variable-retention treatments.

ACKNOWLEDGMENTS

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Effects of Forest Structural Retention Harvest on Resource Availability and Habitat Utilization of Bark-Foraging Birds

Maria Mayrhofer,[1] David Manuwal,[1] and Juraj Halaj[2]

INTRODUCTION

Forest harvest with structural retention is gaining importance as a forest management tool in the Pacific Northwest. An important element of this silvicultural treatment is the preservation of late-successional forest characteristics to conserve biodiversity. Effective implementation requires a thorough understanding of how changes in forest structure affect ecological processes and community development. The Demonstration of Ecosystem Management Options (DEMO) study is an attempt to provide empirical evidence to evaluate the effects of green-tree retention in the Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) dominated forests of the western United States. From the standpoint of ecological research, the DEMO study design provides a unique opportunity to study species and habitat interactions at scales relevant to forest management.

Previous studies on experimental silvicultural techniques in Oregon and Canada have reported positive effects on bird community composition and abundance with increasing retention of forest legacy components. However, recent literature reviews of several studies on forest fragmentation and thinning point out that most studies recorded changes in bird abundance but failed to investigate underlying ecological mechanisms. Understanding the biological processes that determine avian viability would provide insight on the effects of various silvicultural treatments and help make inferences on forest management strategies.

The negative effects of timber harvest on the abundance of cavity-nesting birds such as the brown creeper (*Certhia americana*), red-breasted nuthatch (*Sitta canadensis*), and hairy woodpecker (*Picoides villosus*) has been demonstrated by previous studies. The post-harvest assessment of the DEMO bird community study reported a significant decline in the brown creeper, a bark-foraging species, and a decreasing trend in the red-breasted nuthatch in the 40-percent and 15-percent aggregated and dispersed retention treatments. The brown creeper prefers old-growth forest conditions with a continuous canopy cover and high stand density. It is categorized as a forest-interior species that is sensitive to edges and forest thinning activities. Thus the brown creeper is a good indicator species for the effects of the various levels and patterns of forest harvest in the DEMO units. However, the ecological mechanisms that underlie the brown creeper's habitat needs have not been well studied.

The habitat factors that influence bird abundance include suitable nest sites, presence of predators, food abundance, and the quality of foraging sites. The tree retention treatments of the DEMO units affect these factors to different degrees. The availability of adequate nest sites in the form of snags is a limiting factor to the abundance of cavity nesting birds such as hairy woodpeckers and red-breasted nuthatches. Preferred nesting snags and trees usually have large diameters (>50cm d.b.h.) and advanced decay or heart rot. In addition, brown creepers often prefer sloughing bark so they can create their nest-cavity under the bark slab. Forest management typically reduces tree density, removes snags, creates more edge, and fragments habitat. Nest sites close to edges, in particular, can be subject to increased predation. Bird nests in forests are under predation pressure from a large and diverse community of predators. The abundance and community composition of nest predators depends on the type and quality of the habitat and the influence of the surrounding landscape.

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Arthropods are the main food source for bark-foraging bird species. Changes in forest structure following harvest result in alterations of microclimatic regimes and, consequently, the quality of arthropod habitat. These changes can cause differences in arthropod abundance and guild composition and thus food availability for bark-foraging birds among DEMO retention treatments. Specific habitat resources can differ in importance to the birds, and can constitute an ultimate threshold that determines the presence or absence of a bird species. Analyzing the variety of resources that a bird uses in its home range can provide insights into the importance of specific habitat attributes. The main objective of this study was to determine if the various degrees and patterns of timber harvests intended to retain certain aspects of forest structure affected the resource availability and habitat utilization of the bark-foraging birds brown creeper, red-breasted nuthatch and hairy woodpecker.

STUDY DESIGN

The design of this study is based on the methods of Marzluff et al. (2004). A resource utilization distribution (UD) can assess the relative importance of specific habitat resources for a bird such as the brown creeper and demonstrate its sensitivity to other habitat characteristics. A UD of the DEMO units will provide insights on the usefulness of varying levels and spatial distributions of retained green trees and snags for brown creepers. The UD can be expressed as a probabilistic density function of bird resource use. Kernel density estimation is the optimal technique to define resource UD with adequate spatial point sampling. Each mapped brown creeper territory was defined as a 100-percent fixed Kernel home range boundary that estimates a 100-percent probability that the bird will be located in this territory. During the 2003 breeding season, we collected 30 to 230 relocation points per brown creeper territory, which is an adequate sample size for Kernel home range estimates. Using the fixed Kernel estimates with least squares cross validation, we generated the UD with the ArcView extension "Animal Movements." The ArcView extension "Focal Patch" calculates the density of a bird's UD at each sampling point, which estimates the intensity of the bird's habitat use. Using ArcView, we can spatially reference the following resource attributes of the DEMO units: percent forest cover, percent edge/patch size at aggregated retention treatments, food abundance in form of arthropod abundance and guild composition, and nest predator abundance and community composition. The ArcView extension "Focal Patch" intersects with "Patch Analyst" and measures resource attributes at each grid cell and associates them with the density of the brown creeper UD at every sampling point.

The relationship between DEMO resource attributes and the brown creeper UD can be further analyzed by using multiple linear regression techniques. The statistical software package, R, uses the ArcView data to generate a resource utilization function (RUF). The coefficients of the RUF regression provide a measure of the intensity of DEMO resource use by the brown creeper. The Kernel point estimates introduce spatial autocorrelation into the data, but are accounted for by the error term in the regression model. The coefficients of habitat attributes, such as forest cover or food abundance, can be tested against each other for their significance in the bird's resource use. To compare the probabilities of resource use of all sampled brown creepers for possible population inferences, the individual birds (RUFs) need to be considered as independent samples with similar choice selection among DEMO forest resources. Coefficients that are not available in a bird's territory range can be set equal to zero. It is possible to pool only individual birds that have the same coefficients of habitat resources available in their territory. Using the pooled bird data, we can calculate an average RUF to make inferences on the use of various DEMO forest resources at a population level. This information could be used in forest management decisions.

METHODS

During the breeding season of 2003 (May through mid-August), we attempted to capture all brown creepers that established a territory at the 5 treatment units of the two Oregon DEMO blocks, Watson Falls and Dog Prairie. The 5 experimental harvest retention treatments consisted of a 100-percent control unit, 40-percent and 15-percent dispersed, and 40-percent and 15-percent aggregated forest units. The Douglas-fir dominated forests were all beyond the stem exclusion phase, but among the different treatment units, the stands included a variety of disturbance histories and ages. Male brown creepers were caught with mist-nets by attracting them with territorial call-backs and bird decoys. The birds were banded with U.S. Fish and Wildlife aluminum leg bands and with two additional colored leg bands. To improve individual bird identification in the field, the creepers' two central retrices were also color-marked with duct tape the same color as the birds' leg band combination. Throughout the breeding season, all brown creepers were spot-mapped by using the existing grid system to determine the location and size of the birds' territories at the DEMO treatment units. In addition to spot-maps, we used global positioning system (GPS) units to map habitat use of brown creepers whose territories extended into the adjacent forest beyond the DEMO experimental units. This method ensured that the entire brown creeper territory was mapped and thus all habitat use investigated.

Due to logistic constraints, we were only able to spot-map hairy woodpecker and red-breasted nuthatch activities on the DEMO treatment sites and the adjacent forest. Territory use in the forest beyond the edge of the experimental units was not mapped, and the woodpeckers and nuthatches were not color banded. We conducted nest searches for all 3 target species, and the nesting progress was checked every 4 to 7 days. To determine the breeding status and reproductive success, we recorded the Vickery Index for every territory. Nest site characteristics were assessed by using circular vegetation plots (0.05 ha radius) around each nest tree. Encounters with the following potential nest predators were spot-mapped at all DEMO sites: Douglas squirrels (*Tamiascurus douglasii*), chipmunks (*Tamias* sp.), house wrens (*Troglodytes aedon*) and corvids (*Cyanocitta stelleri*, *Corvus corax*, *Perisoreus canadensis*). Food availability in the form of bark arthropods was sampled by crawl traps installed on the trunks of 20 live trees and 20 snags in every treatment unit. To accurately determine food abundance, we refined our arthropod samples to match the diet preferences of our focal bird species using fecal sample analysis obtained from the captured and banded brown creepers and a priori knowledge.

RESULTS

At the time of this abstract submission, the second field season of this study is in progress. Preliminary results on brown creeper utilization distribution will be presented during this workshop.

ACKNOWLEDGMENTS

This research is a component of the Demonstra-tion of Ecosystem Management Options (DEMO) study. Funds were provided by the USDA Forest Service, PNW Research Station to Oregon State University and to the University of Washington.

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Using Spatially Variable Overstory Retention to Restore Structural and Compositional Complexity in Pine Ecosystems

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ABSTRACT

Increasingly, forest managers incorporate overstory retention into silvicultural prescriptions for forests traditionally managed for single-cohort structure. The ecological benefits of retention may come at the cost of reduced growth of tree regeneration because of competition with residual trees. An important question in retention research, and its application, is how spatial pattern of retention (e.g., dispersed, aggregate) influences resource availability and heterogeneity, competitive environments, and regeneration dynamics. Recently, we initiated two operational-scale experiments in pine ecosystems (longleaf pine (*Pinus palustris* Miller) in southern Georgia, USA and red pine (*Pinus resinosa* Aiton) in northern Minnesota, USA) to address questions about the influence of retention pattern on resource availability and tree regeneration. These experiments address the hypothesis that resource availability at the stand scale will be highest with aggregate retention rather than dispersed retention because of nonlinear relationships between competitor abundance and target plant response. In both studies, our goal is to test approaches for restoring age diversity in single-cohort stands, while minimizing competitive inhibition of the new cohort. Our initial results show clearly that spatial pattern of retention has a significant effect on stand-scale resource availability and regeneration growth.

KEYWORDS: Structural complexity, biological legacies, overstory retention, longleaf pine, red pine, regeneration, plant competition, productivity.

INTRODUCTION

There is a fundamental truth that is apparent when contrasting forests that develop naturally (i.e., in response to natural disturbances and development processes) to those that develop in response to management: *nature generates complex forest stands, whereas management (for wood production) simplifies them*. The complexity inherent in natural stands is embodied in diverse (in a relative sense) age and size structures, diverse tree composition (again, in a relative sense), abundant large coarse woody debris, diverse understory, shrub, and ground layer plant communities, and spatially variable (horizontal and vertical) patterns in these attributes.

Complexity in natural stands is characteristic of all forests, including those that initiate following stand-replacement disturbances. Such forests rarely display the simplified structure that is characteristic of clearcuts, but rather include a rich legacy of biological (or biologically derived) structures that survive the disturbance and provide critical habitat, sustain important functions, and influence recovery processes in the post-disturbance ecosystem.

Incorporating biological legacies into regeneration harvest prescriptions has emerged as an important principle of ecological forestry and is being implemented in many regions (Franklin et al. 1997, Lindenmayer and Franklin 2002, Palik and Zasada 2002, Vanha-Majamaa and Jalonen

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2001). More specifically, retention of live, large overstory trees during regeneration harvest is one of the more obvious ways that foresters incorporate the legacy concept into management, particularly in forests that traditionally are managed for single-cohort structure.

Retention prescriptions for large trees (indeed, for all types of legacies) must address three questions: what species and sizes to retain, how many trees to retain, and what spatial pattern of retention to use—e.g., dispersed or spatially aggregated. It is the third question regarding spatial pattern of retention and influence on ecosystem responses that we are exploring. Spatial pattern of retention is receiving scrutiny (Franklin et al. 1997) because some ecological objectives may be sustained by dispersing retained trees whereas other objectives may be sustained by aggregating retained trees. Additionally, spatial pattern of retention may lead to profound differences in growth and productivity of trees (Palik et al. 1997, 2003). Our premise is that the ecological benefits of retention may come at the cost of reduced growth rates of tree regeneration because of competition with residual overstory trees. On the other hand, retention may provide a tool to better control the diversity of resource environments and the mixture of species regenerating in the stand.

Recently, we initiated two operational-scale experiments in pine ecosystems (longleaf pine (*Pinus palustris* Miller) and red pine (*Pinus resinosa* Aiton)) to address the influence of retention pattern on resource availability and tree regeneration. Similar experimental approaches were used in both study forests. In both studies, our goal was to test approaches for introducing structural complexity into simplified stands, while minimizing competitive inhibition of the new cohort of pines.

STUDY AREAS

The longleaf pine experiment was conducted in southern Georgia, USA, on the property of the Jones Ecological Research Center. The red pine study is being conducted in northern Minnesota, USA, on the Chippewa National Forest. Both study systems occur on deep, loamy sand soils. Historical age structures for the two systems are characterized by two to many cohorts of the dominant pine species. Longleaf pine forests are almost exclusively mono-dominant, whereas red pine forests historically contained admixtures of jack pine (*Pinus banksiana* Lamb.), eastern white pine (*Pinus strobus* L.), and several hardwood species.

EXPERIMENTS

The general experimental design is similar for both pine systems (fig. 1). Four retention treatments are assigned randomly within replicate blocks and include uncut control, dispersed retention, small-gap cutting, and large-gap cutting (the latter two treatments result in aggregate residual tree retention). The retention treatments are cut in similar residual basal areas (12 m^2/hectare (ha) for longleaf pine and 18 m^2/ha for red pine). Stand sizes are 2 to 3 ha for longleaf pine and 15 to 25 ha for red pine. The different residual basal areas in the two studies reflect natural differences in stocking for the two forest types; i.e., longleaf pine stands typically have low basal area, relative to red pine stands. Moreover, the differences in stand sizes correspond to the average patch sizes of management units on the two properties. Gap sizes are approximately 0.1 (small) and 0.3 (large) ha. Approximately 5 small and 2 large gaps were cut in each treatment stand for the longleaf pine study, while in the red pine experiment, gap numbers averaged around 28 (small) and 18 (large), respectively in each unit. Additionally, for the red pine experiment, the residual matrix between gaps was lightly thinned.

After harvest, nursery-grown seedlings were planted across the range of residual overstory basal areas in each treatment stand. For the longleaf pine experiment, only longleaf pine seedlings were planted, since these systems are typically mono-dominant. For the red pine experiment, equal mixtures of red pine, eastern white pine and jack pine were planted because all three pines can occur in these systems.

We are also examining the competitive interaction of understory vegetation and a new tree cohort. In both experiments, half of planted seedlings receive understory competition control (herbicides in longleaf pine; manual cutting in red pine), whereas the other half receive no understory control. We were unable to use herbicides to control the understory on the national forest study area. Here we report only on results from the understory control treatment.

Response variables measured include survival and growth (ground level diameter, biomass) of the planted pine seedlings and light (as a percentage of photosynthetically active radiation (PAR) in the open measured by using a LICOR LAI 2000 sensor), nitrogen (using resin exchange beads), and water availability (using time domain reflectometry (TDR)). Additional variables being measured in the red pine experiment include natural regeneration establishment and growth, biomass productivity of the residual

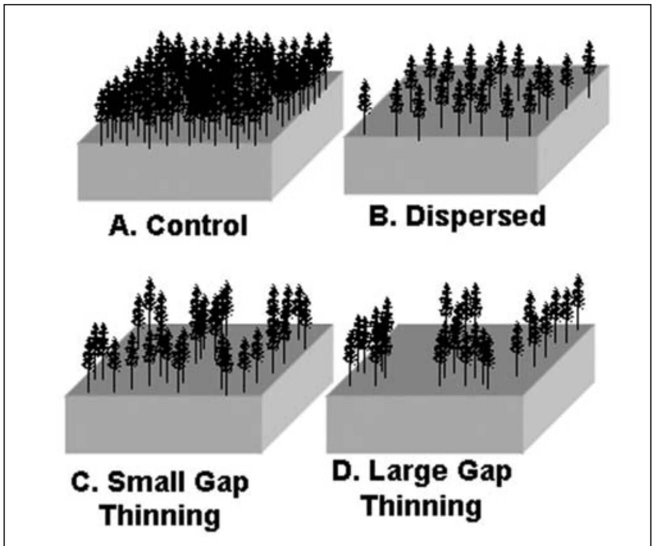

Figure 1—Conceptual representation of overstory retention treatments that differ in spatial pattern of residual trees. (A) undisturbed forest, (B) dispersed retention, (C) small gap thinning, (D) large gap thinning. Treatments B to D have the same residual overstory basal area. Note: in the red pine experiment only, some thinning also occurred in the residual matrix between the gaps.

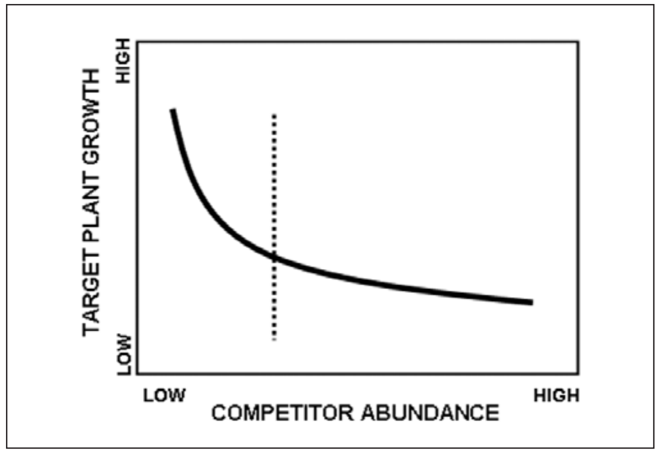

Figure 2—Conceptual relationship between competitor abundance and target plant growth.

cohort, shrub and herbaceous layer productivity, plant richness and community composition, tree pathogen responses, coarse woody debris dynamics, and songbird communities.

HYPOTHESES AND CONCEPTUAL BACKGROUND

Our experiments address the hypothesis that the availability of resources at the stand scale and heterogeneity will be higher with aggregate retention than dispersed retention because of a nonlinear (negative exponential) relationship between competitor abundance and target plant response (Palik et al. 1997). Research on plant competition has demonstrated the shape of this plant interaction relationship (fig. 2), in which target plant growth is low across a wide range of competitor abundance and only increases (exponentially) below threshold levels of low competitor abundance. It is hypothesized that the growth relationship reflects the pattern of availability of two or more limiting resources; e.g., it is only at low competitor abundance that light and nitrogen are both abundant (Palik et al. 2003).

We have used this neighborhood-scale relationship (i.e., the area immediately around the target plant) to generate stand-scale hypotheses about resource availability and growth of regeneration under different spatial patterns of retention (Palik et al. 1997, 2003). Specifically, given a fixed (and low) level of residual basal area in harvest stands, we predict that resource availability to a new cohort of trees will increase slowly from an uncut (control) stand, through dispersed retention, to small-gap aggregate retention, and will only increase exponentially where aggregate retention provides for many large openings in the stand (fig. 1). Similarly, we predict that new cohort (target plant) growth will follow a similar pattern, increasing exponentially only with aggregate retention that incorporates many large openings in the stand. The theoretical basis for these stand-level hypotheses is this: only when retention is aggregated, and large openings exist, will sufficient competitive neighborhoods be far enough away from overstory competitors to be free of extreme resource competition, i.e., they fall far to the left on the interaction curve (fig. 2).

RESULTS AND SUMMARY

Our research in the longleaf pine system is complete, and we have reported on this work elsewhere (Battaglia et al. 2002, 2003; Jones et al. 2003; Palik et al. 2003). In contrast, our experiment in red pine is ongoing; harvest treatments were installed in winter 2002-2003, and seedlings were planted in spring 2003. In the following sections, we summarize major findings of the completed longleaf pine experiment and highlight some of the features of the red pine experiment currently in progress.

Longleaf Pine Ecosystems

At the neighborhood-scale, resource availability and seedling growth of longleaf pine varied across the range of overstory abundance, expressed as overstory abundance index (a distance weighted measure of total basal area with 15 m of the seedling), according to the hypothesized negative exponential relationship (fig. 2). Nitrogen availability

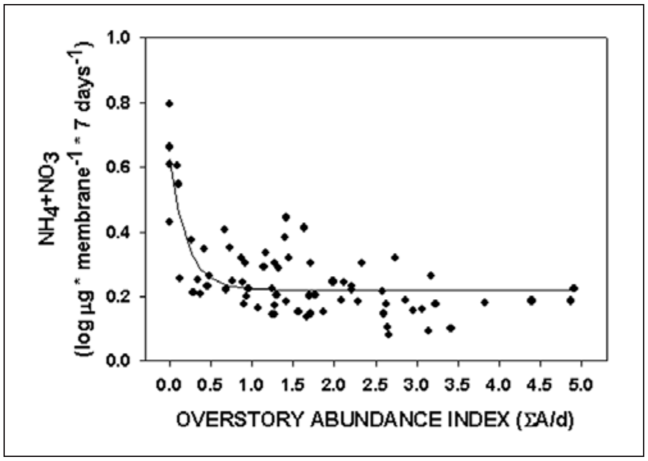

Figure 3—The response of nitrogen availability (expressed as NH_4+$N0_3$) to overstory abundance index (a distance weighted measure of total basal area with 15 m of the sample point) in a longleaf pine woodland (from Palik et al. 2003).

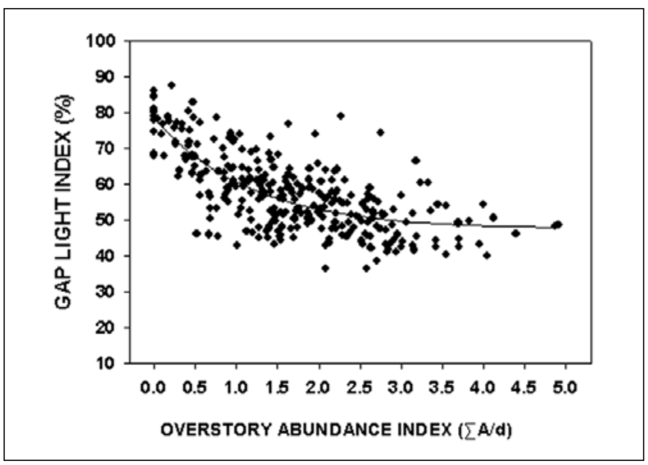

Figure 4—The response of gap light index (expressed as percentage of an open condition) to overstory abundance index (a distance weighted measure of total basal area with 15 m of the sample point) in a longleaf pine woodland (from Palik et al. 2003).

(fig. 3) increased only marginally across a wide range of overstory abundance, but increased more rapidly at low overstory abundance. Gap light index responded similarly, although the increase in light began at a higher overstory competitor abundance than with nitrogen (fig. 4). Above-ground seedling biomass growth responded similarly to nitrogen. Biomass increment was low and nearly constant across a wide range of overstory competitor abundance and increased exponentially at low overstory abundance (fig. 5). The response of below-ground biomass increment was similar to above-ground biomass increment (data not shown).

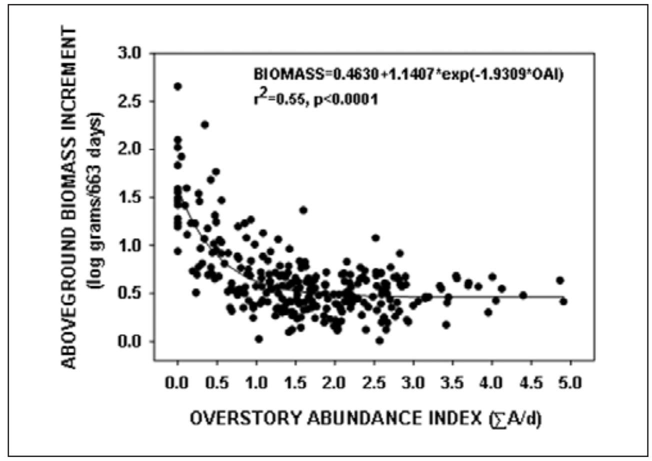

Figure 5—The response of above-ground longleaf pine seedling biomass increment to overstory abundance index (a distance weighted measure of total basal area with 15 m of the seedling) in a longleaf pine woodland (from Palik et al. 2003).

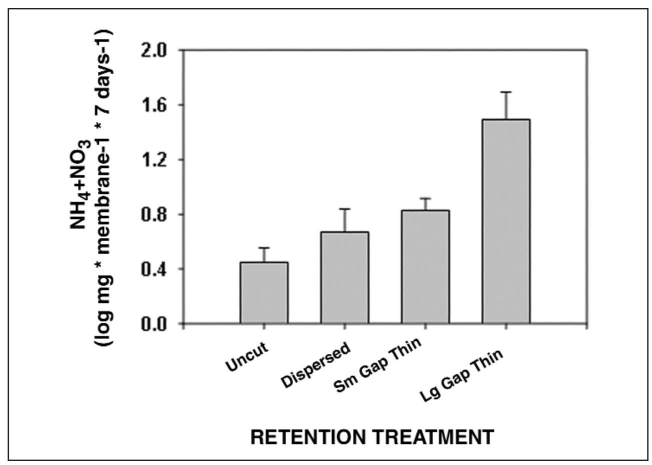

Figure 6—Soil nitrogen availability (expressed as NH_4+$N0_3$) in four retention treatments in a longleaf pine woodland. Values are means (±se) of three replicates (from Palik et al. 2003).

At the stand scale, resource availability and seedlings growth responses were as hypothesized. Nitrogen availability was lowest in the control stands, increased with dispersed retention and small-gap retention, and was greatest with large-gap retention (fig. 6). Similarly, gap light index increased across the same treatment array, although with more muted differences among treatments (fig. 7). Finally, the above-ground seedling biomass increment was similar among control, dispersed retention, and small gap thinning treatments, but increased substantially in the large gap thinning treatment (fig. 8). Our major findings, based on these results, are listed below:

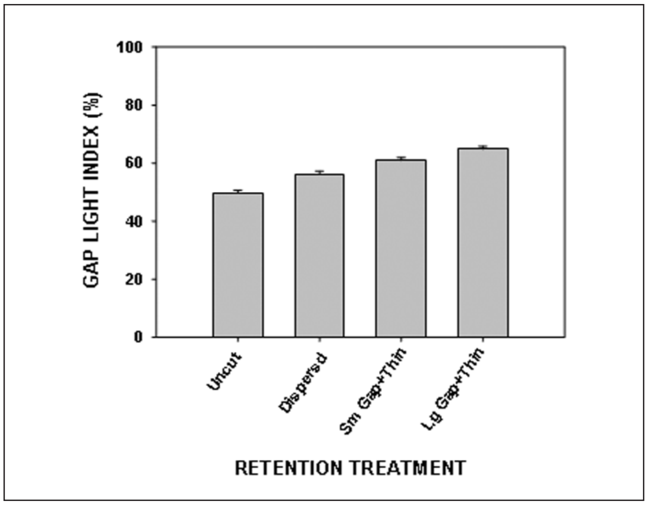

Figure 7—Gap light index (expressed as percentage of an open condition) in four retention treatments in a longleaf pine woodland. Values are means (±se) of three replicates (from Palik et al. 2003).

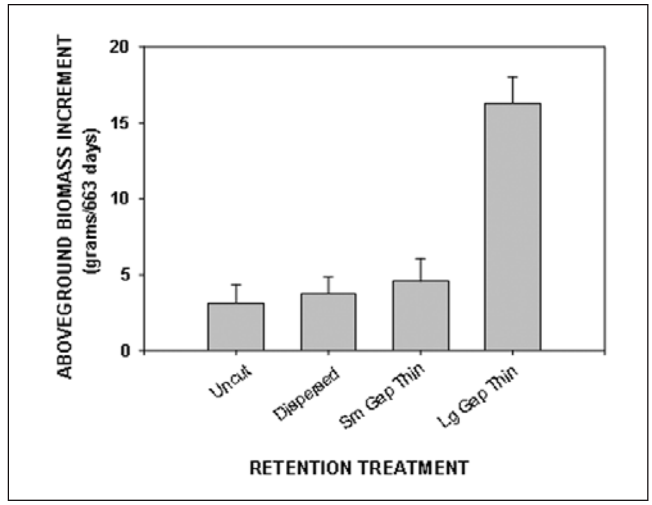

Figure 8—Above-ground longleaf pine seedling biomass increment in four retention treatments in a longleaf pine woodland. Values are means (±se) of three replicates (from Palik et al. 2003).

- All treatments retained significant structural complexity similar to mature overstory longleaf pine systems.

- Resource availability and competitive environments in the understory differed dramatically, depending on spatial arrangement of retained trees.

- Retaining trees in aggregates, through large-gap cutting, provided the most favorable resource environment for regenerating longleaf pine seedlings (in the absence of competing herbaceous and shrub competition).

- We expect similar (although muted responses) in resources and regeneration growth in the presence of competing herbaceous and shrub vegetation.

Red Pine Ecosystems

As with the longleaf pine experiment, we are examining resource availability in the red pine experiment and new cohort growth. Additional objectives, not included in the longleaf pine study, include an examination of

- Regeneration survival and growth of pine species differing in shade tolerance

- Development of natural regeneration

- Recruitment of large dead wood

- Productivity patterns and trade-offs between cohorts and among structural layers

- Responses of shrub and herbaceous plant communities to treatments

- Responses of red pine shoot pathogens to treatments

- Migratory songbird responses

We predict similar neighborhood and stand-scale responses in resource availability and seedling growth to retention treatments, and consequently, the same set of major summary points are likely to be true for the red pine system. Additionally, we have posed a set of stand-scale hypotheses for some of the other response variables listed below:

1) **Residual cohort production.** Among the three harvest treatments, productivity of the residual cohort will increase from large-gap retention to small-gap retention to dispersed retention because of decreasing inter-tree shading and greater ability to preempt soil resources.

2) **Shrub-herb preemption.** Among the three harvest treatments, resource capture and productivity of shrub and herb layers will be maximized with large-gap retention and minimized with dispersed retention because of decreased ability of the residual tree cohort to capture resources in the former treatment.

3) **Residual tree blowdown.** Among the three harvest treatments, blowdown of residual trees will be maximized with dispersed retention and minimized with gap-retention because of decreased mutual protection and support among neighboring trees with the former treatment.

4) **Ground layer plant communities.** Composition of ground layer plant communities will be less altered with aggregate (gap-based) retention relative to dispersed

retention because a greater percentage of the stand will be left in an unharvested (or minimally harvested) condition.

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Wildlife Response to Fire and Thinning Treatments in Ponderosa Pine

Steve Zack,[1] Kerry Farris,[2] and Luke George[3]

There is growing consensus that ponderosa pine (*Pinus ponderosa* P.&C. Lawson) forests need widespread management with prescribed fire and thinning. The structure and natural processes of these forests have been dramatically altered in the past 100-plus years primarily due to fire suppression and large-tree logging. The park-like appearance of these pine forests, which were the result of frequent, low-intensity fires, have been replaced by dense stands of pines and encroaching fir (*Abies* spp.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) throughout its range. Likewise, the avifauna of these pine forests has been affected, with several species considered in decline (e.g., white-headed woodpecker (*Picoides albolarvatus*), olive-sided flycatcher (*Contopus cooperi*), and white-breasted nuthatch (*Sitta carolinensis*)). We are collaboratively engaged (with scientists from the USDA Forest Service, Pacific Southwest Research Station in northern California and as part of the National Fire and Fire Surrogate research) in experimental forest research at several locations in California and Oregon to assess the wildlife response to alternative forest treatments of thinning and prescribed fire in comparison to controls (continued fire suppression).

In general, we are finding that a foraging guild of bark gleaners (woodpeckers, nuthatches, creepers, and chickadees) tend to respond positively to thinning and prescribed fire, whereas a foraging guild of leaf gleaners (vireos, tanagers, and warblers) tend to respond negatively to such treatments. Further, we are beginning to understand how *Picoides* woodpeckers respond to fire treatments, and how interactions between fire, woodpeckers, bark beetles, and fungi can result in snag decay that facilitates the possibility of cavity excavation. Snags with cavities are a critical wildlife resource. It seems clear that working with, and not against, fire helps create structures and revitalizes processes important to wildlife.

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David Marshall and Robert Curtis (retired), both from the Pacific Northwest Research Station, examine stand structure on research plots in Washington State Capitol Forest. *Photo by Tom Iraci*

INTERNATIONAL WORKSHOP

Balancing Ecosystem Values

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Proceedings

FIRE AND FIRE SURROGATES

Photo by Tom Iraci

Restorative Management of Boreal Forest Stands With and Without Fire: An Experimental Approach

Timo Kuuluvainen,[1] Saara Lilja,[1] and Ilkka Vanha-Majamaa[2]

There is considerable interest in using fire, or imitating the effects of fire, to restore natural forest features and biodiversity to managed forest. We are carrying out experimental research in southern Finland to determine the effects of different restorative cutting treatments, with and without fire treatment, on forest structure and regeneration, biodiversity, and timber production. Our motivation is to test various approaches that will meet economic and biodiversity goals of forest management. The experiment is set up in mature, managed spruce-dominated stands on mesic sites; for sampling the stands are further divided in two parts, a moist and a dry biotope. The experimental treatments consist of shelterwood-type cutting (60 m^3/ha standing retention trees) with four levels of down retention trees (5, 30, and 60 m^3/ha) and a burning/no-burning treatment. The treatments were replicated three times, resulting in 24 stands and 48 sample quadrates 20 x 40 m.

The pretreatment inventories were done during the field season 2001. The cutting treatments were conducted in winter 2002 and the burnings in summer 2002. Post-treatment inventories of the described variables were carried out between 2002 and 2004. The follow-up variables include stand structure, tree mortality and regeneration, forest floor microhabitat diversity, ground and field layer vegetation, humus and soil, epixylic lichens and bryophytes, invertebrates, coarse woody debris and polypores, and soil microbes. The first post-treatment results of the experiment are presented.

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Effects of Thinning and Prescribed Fire on Understory Vegetation in Interior Pine Forests of California

Martin W. Ritchie[1]

ABSTRACT

An array of treatments were applied to ponderosa pine (*Pinus ponderosa*) stands in northeastern California in a large-scale study designed to evaluate the effects of management on accelerating late-successional attributes. Treatments include both mechanical manipulation (thinning) and prescribed fire. The three factors associated with late-successional stands considered in this study are the diameter distribution, species composition, and fire frequency. A potential treatment response is the change in understory composition following treatment. Understory vegetation was sampled in the second growing season after the burn treatment, and 3 to 5 growing seasons after mechanical treatments. A total of 79 distinct species were identified within the treatment units. Species richness appeared to be unaffected by treatments. The distribution of species within treatment units was affected, but response was limited to only a few species. In general, understory species appeared to be slightly more evenly distributed in treated units. Shrub species, primarily snowbrush (*Ceanothus velutinus*), were more common in all treated areas, and white fir (*Abies concolor*) was less common, particularly in burned treatments. Bitterbrush (*Purshia tridentata*) was not found through much of the study area, so conclusions for this key browse species are difficult to make. There was no short-term effect of treatments on pine regeneration. The infrequency of pine regeneration, however, suggests the need for long-term evaluation.

KEYWORDS: Ponderosa pine, white fir, thinning, prescribed fire.

INTRODUCTION

The distribution of stand structures found throughout the range of ponderosa pine has been altered over the last 150 years. Past harvesting practices, fire exclusion, and grazing have all played a role in the transition to a landscape often dominated by young, dense stands and attendant increases in fire-intolerant species (Agee 1994, Arno 2000).

Changes in stand structure and species composition affect resilience to disturbance events induced by insects or fire. Dense stands have a greater risk of bark-beetle-induced mortality (Oliver and Uzoh 1997) and reduced rates of individual tree growth (Cochran and Barrett 1995, Oliver and Edminster 1988). Heavier concentrations of surface fuels and ladder fuels make stands more susceptible to stand-replacing fires.

Past treatment history has a significant effect on current stand structure (Parsons and DeBenedetti 1979). Treatments such as thinning from below and prescribed fire may affect more than the growth and mortality of the dominant canopy. Busse et al. (2000) have found that prescribed burning in interior pine stands of central Oregon had no significant effect on herbaceous vegetation cover, and that shrub cover only declined slightly. Exclusion of fire is generally thought to enhance conditions for survival of white fir (Burns and Honkala 1990, Hopkins 1981). Thinning and prescribed burning may create more favorable conditions for regeneration of ponderosa pine in these stands (Roy 1983). Reestablishment of fire may reduce levels of white fir survival in the understory. By the same token, disturbance from fire or thinning may increase levels of some common shrubs found in the understory (Conard et al. 1985, Gratkowski 1962).

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Studies are needed to evaluate these effects over time, to obtain a better understanding of both short-term and long-term effects (Halpern and Spies 1995). With changing focus of management in the Pacific Northwest, we need a better understanding of the effects of treatments on individual species and plant communities. The Northwest Forest Plan (USDA and USDI 1994 a,b) established a network of adaptive management areas throughout the Pacific Northwest.

The Goosenest Adaptive Management Area in northeastern California was designated for the evaluation of treatments designed to accelerate development of late-successional features. The stands of this adaptive management area have been influenced by the past management activities of the Long-Bell Lumber Company and the Klamath National Forest.

During the 1930s, the Long-Bell Lumber Company removed the largest, older ponderosa pine trees from much of what is now the Goosenest Adaptive Management Area. Subsequently, the Klamath National Forest obtained these harvested areas, and over the last 50 years conducted varying degrees of sanitation and salvage harvesting. Stands exhibiting late-successional characteristics are no longer a prominent feature of this area.

An interdisciplinary team of scientists from the Pacific Southwest Research Station worked with managers from the Klamath National Forest to implement an experiment with prescriptions geared toward restoring late-successional forest conditions (Ritchie, in press). The prescriptions involve application of combinations of mechanical (thinning) treatments and prescribed fire. The experiment is a large-scale effort, with 20 treatment units comprising over 1200 ha in the study.

Thinning and prescribed fire will have direct effects on stand structure as well as fuel levels. However, the composition of the understory vegetation may also be significantly affected as the canopy is opened (Peek and others 2001), and this can be an important element of treatment response. This study evaluates effects of these thinning and burning treatments on understory vegetation abundance and species richness.

METHODS

This study was installed in the Goosenest Adaptive Management Area (GAMA) in northeastern California (fig. 1) at lat 41˚34' N, long 121˚41' W. The GAMA is located on the Goosenest Ranger District of the Klamath National Forest. Much of the region was harvested by Long-Bell Lumber Company during the 1930s. The removal of most valuable pine at that time, coupled with the exclusion of fire resulted in a dense understory of white fir, a species that appears to have previously been sparsely distributed throughout the area.

Elevations in the treated areas range from 1480 to 1780 m. Slopes are gentle, generally with a northwest aspect. Soils in the study area are sandy loams or loams derived from volcanic ash. Pumice overburden is common in the area. Annual precipitation ranges from 13 to 51 cm and averages 28 cm.

Measured in 1996, pretreatment stand quadratic mean diameters ranged from 24.6 to 41.1 cm with a median of 29.0. Basal area ranged from 26.6 to 58.3 m2/ha, with a heavy concentration of fir in smaller diameter classes. Red fir (*Abies magnifica*) is present in the higher elevation stands, but in this study we will make no effort to distinguish between red and white fir in the overstory. The mean density for trees between 10 and 30 cm diameter was 250 stems/ha for true fir and 130 stems/ha for ponderosa pine. The disparity between the two is even higher when considering stands with no recent sanitation/salvage work.

Experimental Design

Treatments were developed by an interdisciplinary team, with a goal of accelerating the development of late-successional conditions. Three goals served as a focus for the prescription: (1) accelerating the growth of the largest trees in the stand while reducing density-related mortality, (2) increasing the proportion of pine in these increasingly fir-dominated systems, and (3) introducing low-intensity fires through prescribed burning.

Twenty experimental units were identified, each 40 ha in size. Pretreament conditions varied within the experiment (table 1), but there were no statistically significant differences between any of the recorded stand variables (p-values > 0.65). The experiment features four treatments, with each treatment replicated 5 times in a completely randomized design. The four treatments are (1) a control, (2) a large-tree emphasis thinning, (3) a pine emphasis thinning, and (4) a pine emphasis thinning followed by prescribed fire.

The treatment units were given a treated buffer strip approximately 100 m in width, where possible. Roads within the units were avoided, although in some instances secondary roads do pass through treatment units. Treatments

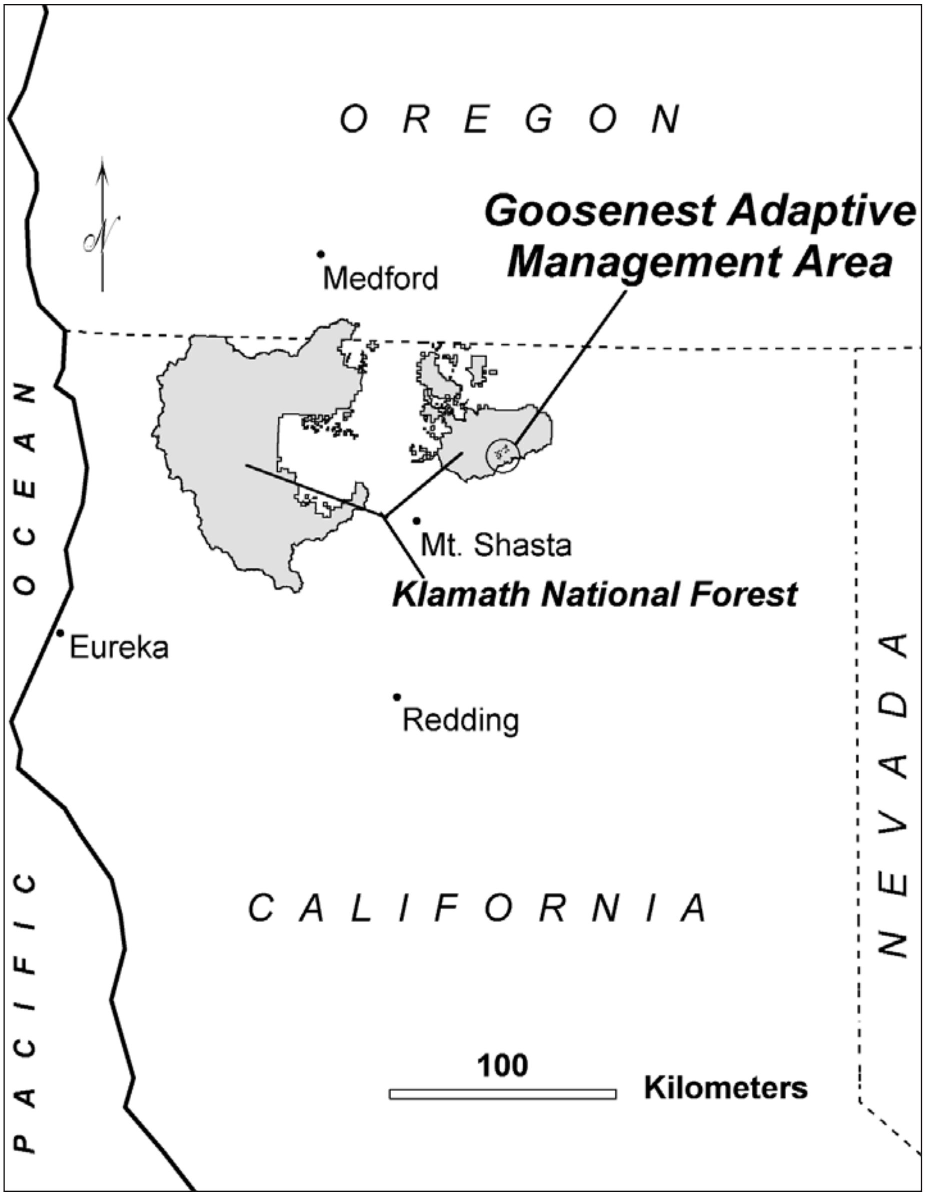

Figure 1—Location of the Goosenest Adaptive Management Area.

were applied randomly to the units; the randomization resulted in a higher concentration of large-tree units among the lower elevations than the concentration of other treatments.

The silvicultural prescription (Ritchie 2005) for pine emphasis was designed to reduce stand density while retaining most ponderosa pine trees. The large-tree emphasis prescription was intended to reduce stand density by thinning smaller diameter trees without regard for species. Both prescriptions were test-marked prior to implementation to evaluate the effectiveness of the marking guidelines.

The pine-emphasis prescription was designed to increase the proportion of pine, with a residual density target of 80-percent basal area, by retaining existing pine and regenerating pine in planned openings. The prescription called for thinning with a "D+5" rule (spacing in feet is equal to diameter in inches plus 5), with the constraint that all ponderosa pine dominants and codominants greater than 30 cm were retained regardless of spacing. Also retained were all sugar pine (*Pinus lambertiana*) trees and snags greater than 38 cm.

In pine-emphasis treatments, approximately 15 percent of the area was regenerated to pine by harvesting openings

Table 1—Pretreament quadratic mean diameter, basal area, stand density index (Reineke 1933), and density summaries for 20 stands in the Goosenest Adaptive Management Area study

	Minimum	1st Quartile	Median	3rd Quartile	Maximum
QMD, cm	24.6	28.2	29.0	39.4	41.1
BA, m^2/ha	26.6	30.3	37.0	42.9	58.3
SDI, trees/ha	474	523	689	822	1084
N, trees/ha	217	385	568	778	902

Table 2—Distribution of burn severity (percentage of area) following prescribed fire within each of the burned units at the Goosenest Adaptive Management Area

	Burned units				
Condition	**3**	**6**	**13**	**15**	**17**
Unburned openings	13	13	14	14	14
Light ground char	51	51	48	57	54
Moderate ground char	30	31	22	19	17
Deep ground char	6	5	16	10	15

and planting ponderosa pine seedlings. All trees in these designated openings were removed with the exception of the occasional sugar pine. Openings were generally located in dense thickets of fir, or other areas with a very sparse distribution of pine, and ranged in size from 0.15 to 1.35 ha.

Fire has been reintroduced to 5 of the 10 pine-emphasis units subsequent to thinning. Although mortality was low, the fire burned with light to moderate intensity through most of the units (table 2). Application of prescribed fire is to be maintained at a rate consistent with known fire history of the study area, commensurate with the ability to carry surface fire through the stands. A study is underway to determine historical periodicity of fires in the GAMA. Fuel levels are to be reevaluated periodically to determine the optimum time to reburn.

The large-tree treatment was a thin from below (larger diameter trees were retained), regardless of species, to a residual spacing of 5.5 to 7.6 m. Because changing species composition was not a specific goal of the large-tree treatment, no regeneration openings were created. The large-tree emphasis treatment creates greater vertical and horizontal uniformity than the pine emphasis treatment, but will likely promote the development of stands with a greater proportion of true fir than observed in the pine emphasis treatment. Although there was no specific guide to favoring pine in this treatment, there was still some effect on species distribution since there is a higher proportion of fir in the smaller diameter classes.

In all units, whole trees greater than 10 cm in diameter were transported to the landing with low ground pressure equipment minimize the buildup of surface fuels. Nonmerchantable material was chipped and processed off site. Cut trees less than 10 cm in diameter were left on site. The control units have received no treatment and are to be kept free of any subsequent stand treatments. Fire suppression activities are to continue in the area.

Total volume removed was 17.6 million board feet plus an additional 62 thousand green metric-tons of biomass. Due to the scale of the project, three field seasons were required to complete mechanical treatments (1998, 1999, and 2000). Prescribed fire was applied in October 2001.

Treated units ranged from 20- to 30-percent crown cover post-treatment (excluding consideration of planned openings, which had a cover of zero). Thinning treatments reduced crown cover approximately 25 percent. Cover in the controls ranged from 45 to 60 percent. For pine-emphasis treatments, the proportion of fir basal area was reduced roughly 20 percent (fig. 2). In the large-tree treatments, proportion of fir basal area was reduced about 7 percent, on average.

Sampling Design

Sampling of understory vegetation consisted of an array of six, 0.5 m^2 plots located systematically (at 6-, 21-, 36-, 64-, 79- and 93-meter mark) along a 100 m transect. There were 17 to 19 such transects in each treatment unit, with each transect spaced on a systematic grid. Grid points are surveyed at a 100 m square spacing and every other grid point is sampled for understory vegetation. Thus the center point of each transect was on a 141 m square spacing throughout the treatment unit. Transects were oriented diagonally, with a random choice determining whether each transect was NE/SW or NW/SE. At each of the sampling

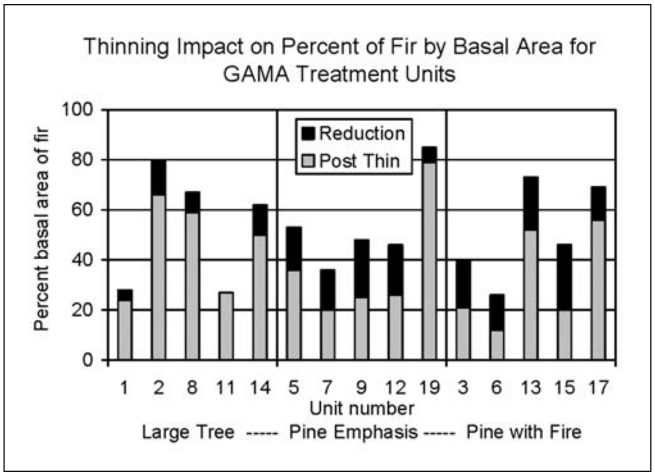

Figure 2—Reduction in proportion of white fir basal area and post-treatment proportion of white fir.

plots, the crew recorded all species present on each plot and estimated cover for the most common plant species. Trees above 1.3 m in height were not included in the sample. Understory vegetation was sampled between mid-May and mid-July, 2003. A total of 79 species were observed on 2,142 fixed area sample plots (table 3). Reference for species identification is Hickman (1993).

ANALYSIS

Species Richness

Stability of species richness was tested by using a bootstrap approach. Data were sampled from each of the treatment units, starting with an intensity of sample size of 5, up to 100, in increments of 5. For each level of sampling intensity, 500 samples were taken. The average species richness was calculated for each unit at each level of sampling intensity. Treatment effects on species richness were also analyzed using a Kruskal-Wallis (nonparametric) test.

Species Frequency

Species distribution was tested by evaluating the frequency observed in the sampling frame. Average number of species recorded within transects varies by treatment. This is a standard one-way analysis of variance with contrasts defined a priori for comparing

Treated vs. untreated
Thinned and burned vs. unburned
Thinned and burned vs. thinned only

Individual species trends in plot frequency were evaluated for several common understory species and species groups (forbs, grasses, shrubs and trees). The grasses-group included grass-like sedges (genus *Carex*). The index for groups was calculated as the mean, over each transect, of the number of species in each group in each 0.5 m^2 plot. This value was then averaged for the entire treatment unit.

Frequency for individual species was calculated as the sum over each transect, then averaged for the treatment unit across grid points. Individual species considered were white fir, bitterbrush, goldenbush and snowbrush. Other species were present but in insufficient numbers to reveal any trends.

RESULTS

The bootstrap evaluation of richness as a function of sample size shows that richness is still increasing at sample size of 100 (fig. 3). Although the number of species differs between units, all show the same pattern, with increasing richness through the entire range of sample sizes. This suggests that our samples (ranging in size from 102 to 114) of half square meter plots are inadequate to fully capture species richness at this scale. That is, with a larger sample size we would have observed a greater value for species richness. To fully capture species richness, we need to either increase sample size or area of the sample frame.

Furthermore, variability in richness is greater among treated units. The range in richness values is less for the controls than for each of the other treatment units. The maximum for thinned and thinned-and-burned units was between 30 and 40, while the control maximum is less than 30.

With sample size nearly constant across our treatment units, we considered richness conditional on the observed sample size. Graphic analysis shows a trend toward slightly lower values for control plots (fig. 4). However, a nonparametric Kruskal-Wallis test indicated no statistically significant effect of treatments on species richness (p-value = 0.648).

Frequency analysis of variance for species groups (trees, shrubs, grasses, and forbs) yielded few well-defined trends (table 4). However, the mean index for tree species was higher for control than treated (0.18 vs. 0.088) and higher for unburned than burned (0.131 vs. 0.047). Also, the frequency of shrub species was higher for treated than control units (0.40 vs. 0.15). When analyzed as a group, forbs and shrubs had no significant differences in treatment means.

Only four of the individual species showed any significant (p-value < 0.10) difference in frequency response (table 5). The analysis of variance for most species yielded p-values greater than 0.50.

Table 3—Most common plant species, by plant form, and frequency observed in 2,142 sample plots on the Goosenest Adaptive Management Area study

Common Name	Scientific Name	Group	Count
Miner's lettuce	*Claytonia perfoliata*	Forb	114
Maiden blue-eyed Mary	*Collinsia parviflora*	Forb	438
Quill cryptantha	*Cryptantha affinis*	Forb	260
Squirreltail	*Elymus elymoides*	Forb	178
Groundsmoke	*Gayophytum diffusum*	Forb	588
Mountain violet	*Viola purpurea*	Forb	94
California needlegrass	*Achnatherum occidentale*	Grass	442
California brome	*Bromus carinatus*	Grass	71
Cheat grass	*Bromus tectorum*	Grass	4
Western fescue	*Festuca occidentalis*	Grass	33
Kentucky bluegrass	*Poa pratensis*	Grass	3
Long-stolon sedge	*Carex inops ssp. inops*	Grasslike	256
Ross' sedge	*Carex rossii*	Grasslike	448
Bitter dogbane	*Apocynum androsaemifolium*	Shrub	72
Greenleaf manzanita	*Arctostaphyllos patula*	Shrub	79
Squawcarpet	*Ceanothus prostratus*	Shrub	67
Snowbrush	*Ceanothus velutinus*	Shrub	140
Goldenbush	*Ericameria bloomeri*	Shrub	144
Bitterbrush	*Purshia tridentata*	Shrub	40
Wax currant	*Ribes cereum*	Shrub	55
Creeping snowberry	*Symphoricarpos mollis*	Shrub	79
White fir	*Abies concolor*	Tree	205
Incense-cedar	*Calocedrus decurrens*	Tree	5
Ponderosa pine	*Pinus ponderosa*	Tree	26

The response of tree frequency is demonstrated primarily in white fir. Ponderosa pine regeneration was found to be a relatively rare event at this early stage of the study. White fir frequency was substantially reduced by both thinning and burning treatments (fig. 5). When expressed as stocking percentage, white fir in the control was approximately eight times that found in the thinned and burned units (table 6).

Snowbrush treatment effect was highly significant (table 5); control units had the lowest frequency of snowbrush (fig. 5). Density in thinned and burned units averaged roughly 40 times the density of controls (table 6). There were no significant effects on goldenbush (p-value = 0.46). Statistically significant treatment effects were found for bitterbrush (p-value = 0.015). However, this finding is tempered by the pretreatment distribution of this species. An analysis of pretreatment frequency for this species found significant differences. Bitterbrush was more prevalent on the large-tree emphasis treatment units both before and after treatment.

Among the grasses, sedges and forbs, only groundsmoke showed any significant treatment effect (p-value=0.02), with an increase in frequency for all treated areas.

CONCLUSION

In general, observed changes in understory vegetation were subtle. Effects of both thinning and prescribed fire may take several years to fully develop. Understory species were more evenly distributed within treated plots, and there is some suggestion that species richness may also increase in treated areas. With time, these responses may become more striking, particularly with reapplication of treatments.

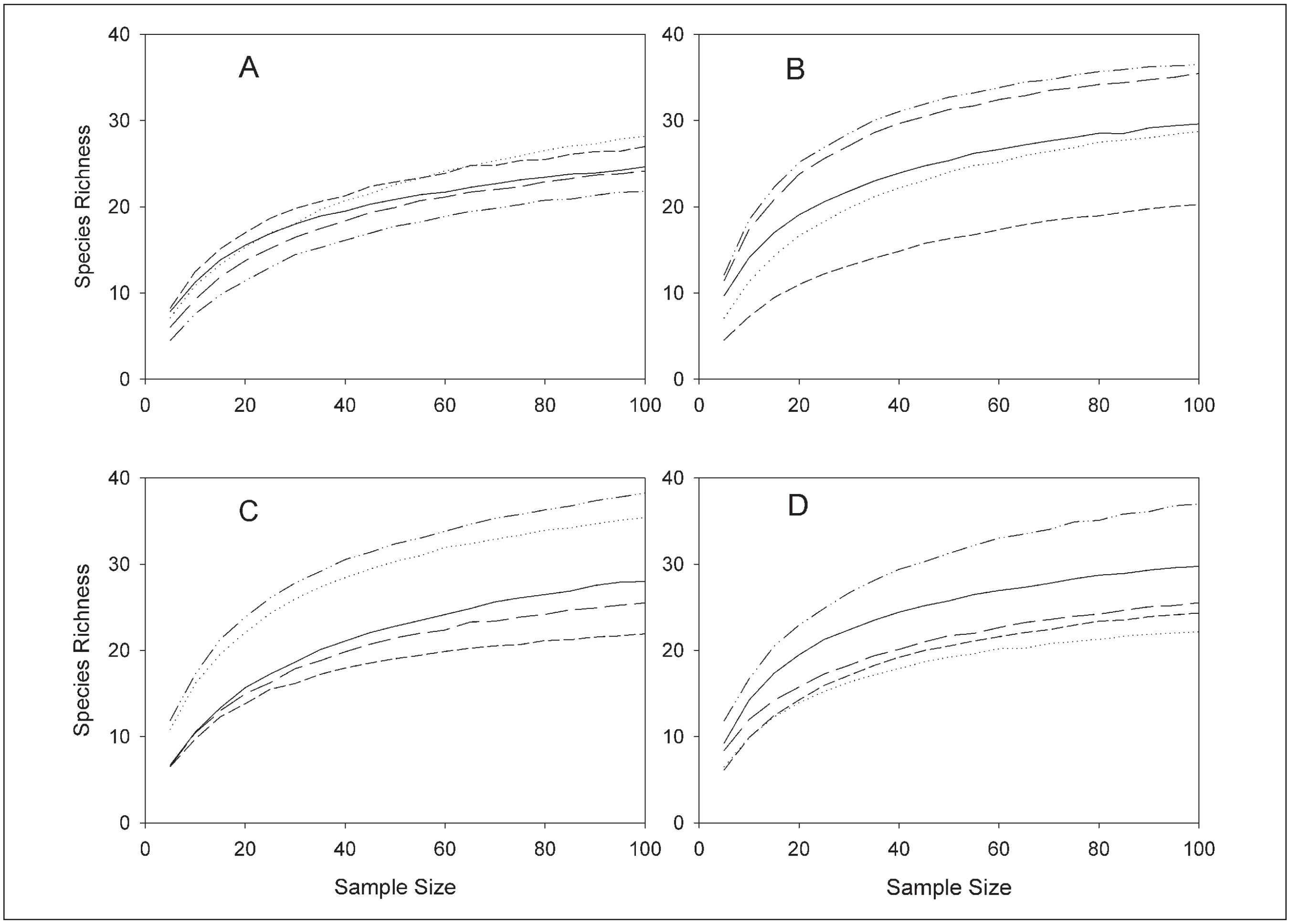

Figure 3—Relation between species richness and number of sample plots for each of the treatment units (each unit is a separate line) in (A) control, (B) large-tree emphasis, (C) pine emphasis and, (D) pine emphasis with prescribed fire treatments.

Table 4—P-values associated with analysis of frequencies of species groups and specific, single degree-of-freedom contrasts. Significant (p-value < 0.10) comparisons are in italics

Contrasts	Trees	Shrubs	Grasses	Forbs
Pine vs. pine+fire	0.115	0.900	0.148	0.298
Treated vs. control	*0.004*	*0.007*	0.524	0.131
Burn vs. unburned	*0.004*	0.232	0.252	0.169
Overall treatment p-value	*0.007*	*0.050*	0.410	0.333

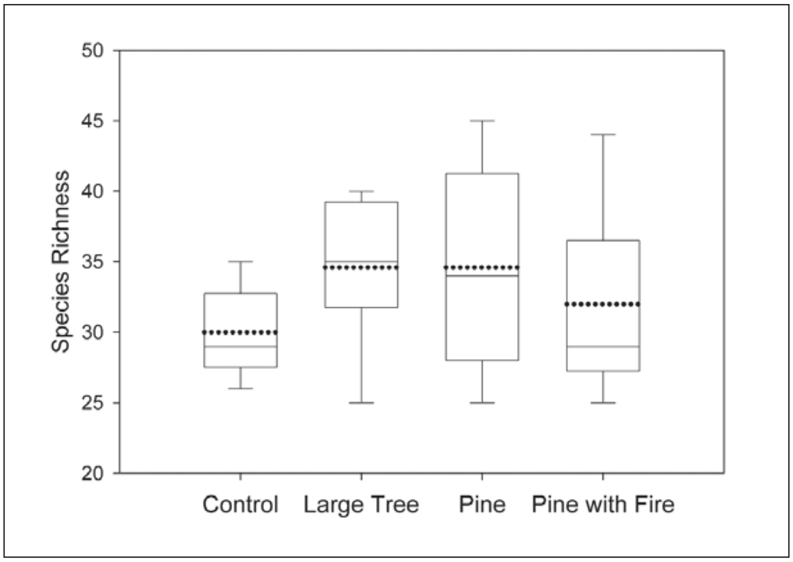

Figure 4—Box plot of species richness over treatment; solid lines are quartiles and median, treatment mean is dotted line.

Table 5—Results of ANOVA for individual species frequency, sorted by p-value; species not listed had p-values > 0.50

Species	Treatment M.S.[a]	Error M.S.	F	P-value
Snowbrush	0.00197	0.000148	13.32	0.0001
White fir	0.00207	0.000230	9.01	0.0010
Bitterbrush	0.000699	0.000146	4.77	0.0147
Groundsmoke	0.00116	0.00267	4.36	0.0200
Squirreltail	0.000695	0.000345	2.01	0.1532
California brome	0.000126	0.000111	1.31	0.3653
California needlegrass	0.000511	0.000555	0.92	0.4538
Goldenbush	0.000269	0.000301	0.90	0.4641

[a] M.S. = mean square

Richness in this forest type, and at these scales, requires a more intense sampling method. A thorough sample of a 40-ha unit may not be feasible without a significant increase in time and effort. Our sampling of this study area took a two-person crew more than 10 weeks, effectively spanning the growing season for many species. Because many observed plants are ephemeral, results may be influenced by time of measurement. At this scale, it is probably not possible, from a practical perspective, to quantify richness without estimating the true value. Because richness is a value influenced by rare events, larger plots and a larger sampling crew are needed to accurately quantify this measure of diversity.

Frequency values are not independent of plot size, and these plots are small for some classes of vegetation (Bonham 1988: 91). Comparisons, therefore, with other studies employing a different sampling strategy are not possible.

White fir is a common understory species in our untreated (control) units, however our treatments, particularly prescribed fire, have significantly reduced the occurrence of this species. This suggests favorable prospects for keeping this species at reduced levels, particularly with planned repeated applications of prescribed fire.

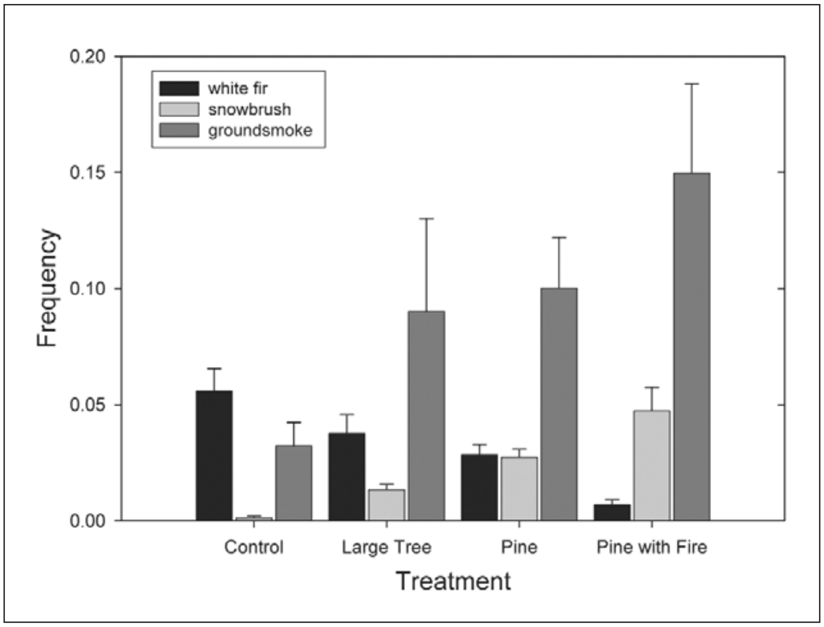

Figure 5—Frequency of white fir, snowbrush and groundsmoke related to treatment (control, large-tree emphasis, pine emphasis and pine emphasis with prescribed fire).

Table 6—Plant stocking, expressed as a percentage of the area with plants on a 0.5 m^2 plot, ± standard error, by treatment

Treatment	White fir	Bitterbrush	Snowbrush
Control	0.93 ± 0.16	0	0.02 ± 0.01
Large tree	0.63 ± 0.14	0.40 ± 0.18	0.22 ± 0.04
Pine emphasis	0.47 ± 0.07	0.01 ± 0.01	0.46 ± 0.06
Pine emphasis + fire	0.11 ± 0.04	0.01 ± 0.01	0.79 ± 0.17

Natural regeneration of ponderosa pine is largely absent throughout the study area at this time, although survival of planted openings is generally high. Since the openings were ripped to a depth of approximately 20 cm prior to planting, these openings are generally devoid of other vegetation at the time our samples were taken.

As a group, shrubs responded positively to the treatments, particularly snowbrush. Results for bitterbrush, a key browse species, were inconclusive. The study area is a transition zone for the species, with observations only made on 6 of the 20 treatment units, and all of these were at the lowest elevations of the study. Since three of the large-tree treatment units are in this lower elevational range, there is a somewhat spurious treatment effect found in bitterbrush.

The combination of burning and thinning in this study produced a strong response in snowbrush. The combination of fire and more open stand conditions created a favorable environment for germination and growth. Snowbrush is known to respond to burning (Halpern 1988). Existing plants sprout vigorously after fire; seeds can persist in the soil for long periods and germinate when favorable conditions are presented (USDA 1988). Similar response to thinning and burning has been shown elsewhere in pine forests of northeastern California (Borsting 2002).

With the exception of groundsmoke, which increased in frequency in treated areas, grass and forb species did not appear to respond significantly to thinning or prescribed fire treatments at this early stage of the experiment.

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A Collaborative Approach to Forest Management: Using a Landscape-Level Dynamic Simulation Model as a Tool to Enhance Communication Among Diverse Landowners

Christine M. Stalling[1]

ABSTRACT

Forest management tools have evolved from stand-level, silvicultural applications to increasingly diverse tools addressing multiple scales of analyses and disciplines. Given the complexity of these multiple scales, successful communication across all ownerships is necessary for effective land management applications. In this study, the utility of using the landscape dynamic simulation model SIMPPLLE (SIMulating Patterns and Processes at Landscape scaLEs) is tested as a science communication tool emphasizing social values in a collaborative approach to fuels management. Landowners and stakeholders in the Swan Valley, northwestern Montana, will participate in workshops focusing on the use of modeling to help visualize the effects of management activities on the Swan Valley landscape, both spatially and temporally. Methods of integrating and communicating ecological, social, and economic principles associated with the landscape will be applied using the SIMPPLLE model in conjunction with support from the Swan Ecosystem Center (SEC). The SEC is a nonprofit citizen's organization whose mission is to maintain their community by focusing on partnerships that encourage sustainable use and care of public and private lands in the Swan Valley. The "Upper Swan Valley Landscape Assessment," coordinated by SEC, serves as a guide to sound stewardship plans in the Swan Valley and will be used to provide information for model exercises.

KEYWORDS: Landscape, simulation model, collaboration, communication, management.

INTRODUCTION

Collaborative partnerships, strategic planning, and public-private cooperation have become prevalent in natural resource management during the last decade. Wildland fuels have been accumulating during at least the past 50 years due in part to wildland fire management policies. As demonstrated by recent wildland fires, large fuel loads contribute to intense fire behavior and increase resistance to control. Consequently, property and natural resources have been destroyed, costs of fire management have increased, fire dependent ecosystems have deteriorated, and the risks to human life continue to escalate. The Swan Valley in northwestern Montana is no exception. The threat of wildfire is one of the many challenges facing the Swan Valley today. The impacts of human population growth, private timber agency land sales, threats to wildlife corridors due to forestry practices and population growth, and fire in the wildland-urban interface continue to present complex issues for land managers. In addition, conflicting views regarding desired future conditions on the landscape provide barriers to land management. The Swan Valley community is well represented by public and private interests and is closely involved in managing their valley for "the ecological integrity of the land and the resources it supports" (Swan Ecosystem Center 2004). Successful, science-based management in such a community can only be accomplished when all landowners and stakeholders understand the reason for any proposed plan of action and agree with the planned actions. Communication must occur between managers, researchers, landowners, and stakeholders before active management can take place.

Communication is more complex than simply sending a message to a receiver who then assimilates the information; it is a complex process in which the receiver interprets the

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message based on personal experiences (Weber and Word 2001). Because groups and individuals learn and understand in many different ways, it is necessary to adjust methods of communication accordingly. The level of pre-existing knowledge within a group can influence whether a certain approach used to communicate science and the associated impacts of management activities will succeed or fail. Studies in adult learning show that a hands-on approach is an effective method for reaching adult audiences (Parkinson et al. 2003). Shindler and Toman (2003) indicate "…effective, inclusive communication strategies must not only provide information but also focus on how people come to understand forest conditions…" Despite the substantial knowledge base of ecological and socioeconomic values and the use of models and modeling approaches to represent these phenomena, the ability of resource managers to successfully communicate this knowledge is lacking.

Land managers recognize the need to schedule and evaluate management activities across multiple ownerships when planning management applications on landscapes (Bettinger and Sessions 2003). Yet, environmental decision-making is a complex endeavor that requires an integrated and multidisciplinary approach, which includes public participants and stakeholder groups that can affect or be affected by management decisions (Videira et al. 2003). In order for all stakeholders to play an active and informed role in any collaborative planning effort, they must first understand the basic ecological science related to vegetation change and the spatial relationships of vegetation and processes such as fire across landscapes. Thus, landowners and stakeholders would benefit from tools to help them better understand the links between the patterns and processes of the larger landscape and their influences on the ecology of land units at smaller scales, such as the stand level. The landscape dynamic simulation model, SIMPPLLE (Chew et al. 2004), has potential as a tool for communicating the science of landscape change and the influences of natural processes and management treatments both temporally and spatially. Landowners and stakeholders in the Swan Valley are engaged in the ecology and management of their landscape and are interested in exploring the use of landscape level modeling to help develop their management plans. The research proposed in this paper is designed to explore the applicability of the SIMPPLLE model as a tool to aid communication and collaboration of landscape level management alternatives among residents and stakeholders in the Swan Valley.

Some funding for this project was provided to the Swan Ecosystem Center (SEC) through a challenge cost share agreement with the Flathead National Forest (funded by the U.S. Forest Service Washington Office). More funding is being sought through the Joint Fire Sciences Program for an 18-month study period in which the model will be introduced to members of the Swan Valley community; workshops and/or interviews will be conducted to gain a baseline measure of landscape-level knowledge among participants; management plans will be developed by using the model and data representative of the Swan Valley; and interviews/workshops will then be used as a follow-up to measure whether the model was a useful tool in this process.

OBJECTIVES

The utility of a landscape model will be tested as a tool to communicate natural resource management concepts and to enhance stakeholder participation in collaborative planning efforts, specifically for the identification of wildland fuel treatments in the Swan Valley, western Montana. The approach is first to test the current version of the SIMPPLLE model as a science communication tool among stakeholders including private landowners, Montana Department of Natural Resources and Conservation (DNRC), the Flathead National Forest, and Plum Creek Timberlands; second to test the utility of the model as a tool to facilitate a collaborative approach to wildland fuels management; and third to develop a land management plan for a community forest in the Swan Valley by using the model to consider influences of natural processes and management treatments on the landscape beyond the community forest. Workshops and interviews will be used to gauge the level of participant understanding of the basic science related to vegetation change, fire, and population influences on management issues before and after running the model. Participants will then use the model to develop wildland fuel treatment alternatives for the Swan Valley; interviews will follow to analyze whether the model helped participants reach agreement on the management plans.

The upper Swan Valley includes approximately 100 000 hectares of federal, state, and private land. The checkerboard ownership is distributed between the Flathead National Forest (67 percent), Plum Creek Timber Company (24 percent), nonindustrial private (9 percent), and the DNRC (less than 1 percent); private and corporate lands occur primarily in the valley bottom while public lands are in the upper elevation and wilderness areas (Swan Ecosystem Center 2004). This heavily forested landscape, composed of a narrow valley through which the Swan River winds, is surrounded by the Mission Mountain Wilderness directly west and the Swan Range to the east. The valley is characterized by ecologically significant wetlands, threatened and endangered species habitat, wildlife connectivity, and high natural biodiversity. The town of Condon is centrally located in the upper Swan Valley; approximately 550 permanent valley residents live in Condon and the surrounding area. A 1993 community profile (Cestero and Belsky 2003) describes the permanent residents of the upper Swan Valley as a group with diverse interests: approximately 25 percent of the population is struggling to make a living by holding more than one job, while the fastest growing part of the population is retired. With the majority of ownership held by nonresidents (Flathead National Forest and Plum Creek Timber Co.), declining timber harvests influencing local economics, and increasing interest in the ecology and management of the Swan Valley and associated wildlands, residents of the Swan Valley have been experiencing substantial "growing pains" over the past decade. In response to the contentious economic and environmental issues associated with these growing pains, an ad hoc citizen's group was formed in 1990. In 1996, the Swan Citizen's Ad Hoc Committee developed the Swan Ecosystem Center, a nonprofit organization located in a former Forest Service building. The purpose of the center is to "represent the community in partnership with the Forest Service" (Cestero and Belsky 2004).

The Swan Ecosystem Center is important to the upper Swan Valley community because the organization provides a forum for residents and stakeholders to discuss ecological, social, and economic issues. Forest stewardship and timber management practices were tested by members of the SEC in conjunction with the Flathead National Forest in a ponderosa pine restoration project on SEC land; long-term monitoring of the site by SEC committee members continues and lessons learned from the experiment continue to provide educational opportunities. This project and a similar one on private land were used to illustrate the links between community well-being and forest health. During the restoration process and subsequent monitoring, resident volunteers have gained immeasurable knowledge of fire ecology, forest stewardship logging, and Forest Service regulations in the design and implementation of the project.

Through these community projects, civil dialogue and a growing trust has been built among former adversaries. Residents feel that interactions with Plum Creek and the Forest Service, with SEC often facilitating, indicate a willingness to address residents' concerns; residents now feel they are gaining greater influence in valley land management decisions (Cestero and Belsky 2003). The SEC's role is important to this study because of the trust they have already established in the Swan Valley community; they will provide guidance in the participant selection process as well as data collection and project support.

Knowledge of ecology and management of forests at the stand level is evident among members of the Swan Valley community. The community's commitment to managing the upper Swan Valley as an ecosystem is outlined in the landscape assessment (Swan Ecosystem Center 2004). This study will provide a possible approach to moving from stand-level ecology and management practices to the broader scale interactions and connectivity of individual vegetation units with the landscape-level patterns and processes that are shaped by fire, insects and disease events as well as prescribed treatments.

METHODS

Workshops and interviews will be used to gauge the level of participant understanding of the basic science related to vegetation change, fire, and population influences on management issues before and after running the model. Participants will then use the model to develop fuel treatment alternatives for the Swan Valley; interviews will follow to analyze whether the model enhanced participation and thereby helped participants reach agreement on the management plans.

MODEL APPROACH

An integrative approach to landscape level planning and management should be accomplished by bringing together knowledge from many disciplines. SIMPPLLE is a management tool developed to help land managers integrate the best available knowledge of vegetation change resulting from disturbance processes such as fire, insects, and diseases as well as fire suppression and management treatment activities (Chew et al. 2004). The model uses landscape-level vegetation data to represent vegetation change as a result of the interacting disturbance processes and management actions. The model's spatial link to geographic information systems (GIS) and the dynamic approach to modeling landscapes help users visualize how a landscape can change over time and space with and without the influence of management applications. The model has been used as a tool by natural resource managers and researchers but its utility with nonexperts has not been tested.

COMMUNICATION AND COLLABORATION

Public participation is critical to sound environmental policy and decisionmaking (Videira et al. 2003). An informed public is more capable of meaningful participation than a public that bases decisions on other criteria such as trust, shared community knowledge, or rumors. The communication process between expert and nonexpert groups may or may not result in greater understanding of the environmental issues that a community must address. Experts can raise public awareness of environmental issues by acknowledging and addressing the often-implicit assumptions that block understanding of science by less informed members of the public (Weber and Word 2001).

To communicate the science of landscape-level management, it is necessary to understand current perceptions and existing knowledge in a community of landowners and stakeholders. One-on-one interviews will be conducted with participants selected by the Swan Ecosystem Center for this study to establish a baseline level of understanding of forest management and ecology held by this group. Interview questions will be developed with help from Dr. Kari Gunderson, Resource Management Scientist with the Aldo Leopold Wilderness Research Institute, and Dr. Jill Belsky, Professor of Rural and Environmental Sociology at the University of Montana, Missoula. Following these interviews, I will lead a 2 to 3 hour workshop at the SEC in which modeling exercises will be used to display how SIMPPLLE represents landscape change spatially and temporally. During the workshop, discussions and questions regarding model approach, ecology, and management issues will be addressed and a follow-up interview will be conducted to establish whether the model was helpful as a tool to aid in communicating issues and concerns among participants.

I will lead a second 2 to 3 hour workshop in which the model will be used to help the same participants develop management alternatives for fire risk reduction, maintenance of wildlife corridors, influences of private timber company land sales, and desired future conditions. Follow-up interviews will be conducted to establish whether the model was helpful in bringing the participants to consensus in developing the proposed management alternatives. If participants are able to come to consensus on a management plan for their community forest, those alternatives will be applied and monitored on the community forest.

PRODUCTS

Two publications will be developed; one will detail an approach to testing a landscape dynamic simulation model as a tool to communicate science-based concepts associated with dynamic landscapes and fuel treatments to community members and stakeholders. A second publication will detail the use of a landscape dynamic simulation model as a tool for collaborative planning of fuels treatments within a landscape assessment framework.

The SIMPPLLE model is available for use and can be downloaded from a publicly accessible Internet website or an ftp site. Installation instructions exist for a wide variety of platforms. As a result of this study, it will be possible to improve system application to a broader client base.

ACKNOWLEDGMENTS

Anne Dahl, executive director of the Swan Ecosystem Center, continues to provide invaluable cooperation and support for this project. Kari Gunderson, resource management specialist for the Aldo Leopold Wilderness Research Institute, continues to donate her time and expertise to this project as well as the myriad needs of the Swan Valley. Jimmie Chew, forester with the Rocky Mountain Research Station and developer of the SIMPPLLE model, continues to provide expertise toward this research project.

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Implementation of the Fire and Fire Surrogate Study—A National Research Effort to Evaluate the Consequences of Fuel Reduction Treatments

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ABSTRACT

Many fire-dependent forests today are denser, contain fewer large trees, have higher fuel loads, and greater fuel continuity than occurred under historical fire regimes. These conditions increase the probability of unnaturally severe wildfires. Silviculturists are increasingly being asked to design fuel reduction treatments to help protect existing and future forest structures from severe, damaging, and expensive wildfires. The consequences of replacing the historical role of fire with fuel reduction treatments, such as underburning with prescribed fire, cutting with mechanized equipment like a feller-buncher, or a combination of both, remain largely unknown and require innovative operational-scale experiments for improved understanding. The Fire and Fire Surrogate (FFS) study is a large manipulative experiment designed by an interdisciplinary team of federal agency and academic researchers to address ecological processes, economic viability, and operational consequences of different fuel reduction treatments. Replicated at 13 installations on federal and state lands extending from the eastern Cascade Range in Washington to the southern coastal plain in Florida, this study is likely the largest operational-scale experiment ever funded to test silvicultural treatments designed to balance ecological and economic objectives for sustaining healthy forests. This paper describes the study objectives and research approach, provides a status of work at the different sites, and presents initial results of changes in stand structure and related understory vegetation as an example of the broad comparisons that this study allows. These initial among-site comparisons highlight the potential value of network-wide meta-analyses for determining the scale at which common themes emerge.

KEYWORDS: Fire and Fire Surrogate study, fuel reduction, overstory-understory interactions, meta-analysis.

INTRODUCTION

Many fire-dependent forests—especially those with historically short-interval, low- to moderate-severity fire regimes—contain more small trees and fewer large trees, have higher fuel loads, and greater fuel continuity compared to conditions under historical fire regimes (Agee 1993, Arno et al. 1997, Barden 1997, Caprio and Swetnam 1995, Cowell 1998, Kilgore and Taylor 1979, Swetnam 1990, Taylor and Skinner 1998, Van Lear and Waldrop 1989, Waldrop et al. 1987, Yaussy and Sutherland 1994). These conditions are the result of fire exclusion and suppression, past livestock grazing and timber harvests, tree recruitment after farm abandonment (especially in the southern United States), and changes in climate (Arno et al. 1997, Skinner and Chang 1996). Collectively, these conditions contribute to a general deterioration in forest ecosystem integrity and an increase in the probability of unnaturally severe wildfires

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(Stephens 1998). Silviculturists are increasingly being asked to design fuel reduction treatments that reduce the stand basal area and the density of small trees, remove fire-sensitive trees, reduce the accumulation of woody debris, and increase the height to live crowns to help protect these forests from severe wildfire and at the same time meet a host of other resource objectives. Strategies for managing forest fuels to reduce the incidence of these expensive and damaging wildfires include underburning with prescribed fire, cutting live and dead trees and removing logs with mechanized equipment like feller-bunchers, or a combination of both. The consequences of implementing these strategies remain largely unknown. Innovative operational-scale experiments that evaluate the effects of alternative management practices involving fire and mechanical or manual surrogates for natural disturbance events are essential for improved understanding of management decisions.

A team of federal, state, university, and private scientists and land managers designed the Fire and Fire Surrogate (FFS) study, an integrated national network of long-term studies, with support from the USDA/USDI Joint Fire Science Program and the national Fire Plan. The national network currently includes 13 sites on federal and state lands extending from the Cascade Range in Washington to Florida (table 1). These 13 sites represent ecosystems with frequent, low-severity natural fire regimes. At each site, a common experimental design was used to facilitate broad comparison of treatment effects. This FFS network likely represents the largest operational-scale experiment ever funded to test silvicultural treatments designed to balance ecological and economic objectives for sustaining healthy forests. Details of the network and links to individual sites are available at the web site http://www.fs.fed.us/ffs/. In this paper, we report on the study objectives and research approach, provide a status of work at the different FFS sites, and present initial results of changes in stand structure and related understory vegetation for a subset of the sites.

STUDY OBJECTIVES OF THE FIRE AND FIRE SURROGATE STUDY

The FFS study was designed to quantify the ecological and economic consequences of fire and fire surrogate treatments across a number of forest types and conditions in the United States. Specific objectives are listed below:

1. Quantify the initial effects (first 5 years) of fire and fire surrogate treatments on specific core response variables within the disciplines of vegetation, fuel and potential fire behavior, soils and forest floor, wildlife, entomology, pathology, and treatment costs and utilization economics.

2. Establish and maintain an integrated national network of long-term interdisciplinary studies using a common “core” design that facilitates broad applicability of results yet allows each site within the national network to be independent for statistical analyses and modeling, and allows flexibility for addressing locally-important issues.

3. Designate FFS research sites as demonstration areas for technology transfer to professionals and for the education of students and the public.

4. Develop an integrated and spatially-referenced database and archive data from all network sites; facilitate developing interdisciplinary and multiscale models, and integrate results across the network.

5. Over the long term, continue to monitor the results of treatments, repeat treatments where appropriate, develop and validate models of ecosystem structure and function, and refine recommendations for ecosystem management.

RESEARCH APPROACH

The FFS study is implemented on land administered by the USDA Forest Service, the USDI National Park Service, various university experimental forests and education centers, state parks, and state forests. The core experimental design for the FFS study includes common treatments, similar treatment replication and plot sizes, and common response variables for all research sites in the network. The four treatments used at 12 of the 13 sites include (1) untreated control, (2) prescribed fire only, with periodic repeated burns, (3) mechanical thinning, with periodic repeated thinning, and (4) mechanical thinning followed by prescribed fire, with this combination repeated as necessary. Treatments at the Sequoia National Park site consisted of an untreated control, an early-season burn, and a late-season burn, which are the principal landscape-scale treatment options on lands managed by the National Park Service. The FFS treatments span a range of realistic management options, and they likely will provide a range of ecological effects. Implementation of the active (noncontrol) treatments at each site was guided by a desired future condition or target stand condition uniquely defined for each site such that, if impacted by a head fire under 80th percentile weather conditions, at least 80 percent of the basal area of overstory trees would survive. Treatments were replicated at each of the sites at least three times in either a completely randomized or randomized block design. Each treatment unit was at least 10 ha and surrounded by a

Table 1—Location and current status of sites in the Fire and Fire Surrogate study network

FFS site	Forest type	Location	Treatment year
Blodgett	Ponderosa pine and white fir	Central Sierra Nevada, California	2002
Hungry Bob	Ponderosa pine and Douglas-fir	Blue Mountains, northeastern Oregon	1998-2000
Jemez Mountains	Ponderosa pine	Northern New Mexico	In progress
Lubrecht Forest	Ponderosa pine and Douglas-fir	Northern Rockies, western Montana	2002
Mission Creek	Ponderosa pine and Douglas-fir	Central Cascades, Washington	In progress
Ohio Hills	Mixed oaks	Southern Ohio	2001
Sequoia	Ponderosa and sugar pine, white fir	Southern Sierra Nevada, California	2002
Solon Dixon	Longleaf pine	South central Alabama	2003
South Carolina Piedmont	Loblolly and shortleaf pine	Northwestern South Carolina	2001-2002
Southern Appalachian	Hickory, oaks, and shortleaf pine	Southwest North Carolina	2002-2003
Southern Cascades	Ponderosa pine	Southern Cascades, northern California	2000-2002
Southern Coastal Plain	Longleaf and slash pine	Central Gulf Coast, Florida	2001
Southwest Plateau	Ponderosa pine	Northern Arizona	2003

similarly treated 50-m buffer. Assignment of treatment to each of the units was completely random. This requirement for randomization is central to the conduct of science but has not often been a part of large operational-scale studies involving land-management agencies.

Core variables encompassed several broad disciplines, including vegetation, fuel and potential fire behavior, soils and forest floor, wildlife, entomology, pathology, and treatment costs and utilization economics (FFS Study Plan 2001). Some 400 response variables were selected for monitoring, with the majority spatially referenced to a 50-m square grid of permanent sample points established and maintained in each treatment unit.

Funding for the FFS has come from home institutions and agencies, the USDA through a National Research Initiative competitive grant, the National Fire Plan, and primarily the Joint Fire Science Program.

The FFS study has three organizational tiers. The first tier is site leaders or managers who ensure uniformity of layout and implementation across all disciplines at a single site. The site managers, along with group leaders for the study disciplines (entomology, economics, fuels, pathology, soils, vegetation, and wildlife) belong to the Science and Management Integration Committee (SMIC), the second tier in the organization. The third tier is a five-member Executive Committee (a national network manager, two disciplinary group leaders, and two site managers) selected by the SMIC. Initially, the SMIC developed comprehensive study plans guiding study implementation at each site, noting any justifications for and deviations from the agreed-upon national FFS standard in implementation or monitoring. In addition, the SMIC is responsible for ensuring that (1) site-level studies are progressing according to project guidelines, (2) data collection protocols and analysis remain consistent and state-of-the-art, (3) data are properly archived and managed, and (4) integration is occurring at all levels. Site managers have responsibility for ensuring data are collected appropriately and are maintained in local databases, while the SMIC oversees the creation of a central national FFS database. The Executive Committee is responsible for project oversight, distribution of funds, and reporting to the Joint Fire Science Program Governing Board.

CURRENT STATUS OF WORK AT FFS RESEARCH SITES

Most of our field effort began in early 2000. The initial set of treatments has been completed at 11 of the 13 sites (table 1), and measurement of responses is ongoing. Prescribed fire for fuel reduction has been used at all sites; however, the season of application, intensity of burn, and frequency of burn varied across sites. For example, even though the Hungry Bob site in northeastern Oregon and the Lubrecht Forest site in western Montana have similar stand histories, stand structure, and vegetation composition, burns at Hungry Bob were conducted in October, whereas burns at Lubrecht Forest were conducted in May and June. Burns at the Southern Coastal Plain site in Florida were scheduled as an early-season treatment on a 3-year return interval; the second iteration of burns was completed early in the spring 2004. At most sites, mechanical fuel treatments generally consisted of removing small-diameter stems in a low thinning by using a combination of single-grip harvester and forwarder. All trees to be harvested were marked prior to harvest activities. At the Southern Coastal Plain site, the accumulation of fuels was in understory layers and consisted primarily of herbaceous matter rather than overstory layers; therefore, the mechanical treatment employed a roller drum chopper. At Blodgett in California, large stem diameters required an initial commercial thinning from below by hand-falling, with logs yarded to landings by rubber-tired skidders. Next, live and dead understory stems were masticated by using an excavator with a disk-type cutter head, with masticated material left on site. In all but the Ohio, Alabama, North and South Carolina, and Florida sites where litter decomposition occurs rapidly, the combination treatment of mechanical thinning followed by prescribed fire required waiting a full season for activity fuels to cure before burning.

Pretreatment data has been collected on all sites, as has most of the first year post-treatment data. Our first challenge within the FFS network was to portray immediate or short-term changes resulting from treatments. These short-term changes are likely of general interest to managers concerned with how conditions changed as a result of treatment. To answer this question, we used both pre- and post-treatment data and focused on the difference between pre- and post-treatment values. We also used pretreatment data as a covariate. At the site level, univariate analysis of variance for the change in each response variable was a first means of evaluating treatment differences. The second challenge is to predict longer term differences among treatments. These long-term differences are likely of general interest to managers concerned with how response variables change over time in response to the treatments; some Southern sites could also consider how the chosen variables change in response to multiple entries. To answer these longer term questions, pretreatment data is likely of little benefit. At this time, our effort in the FFS network has been confined to addressing the first challenge; over time, we will transition to considering the second challenge.

Because typical univariate analyses test a single potential causal pathway among variables through the direct effect of each predictor, our understanding of the overall system complexity is limited to the number of predictors we examine. In addition to univariate analyses, we intend to use multivariate ordination techniques such as nonmetric multidimensional scaling (NMS) and indicator species analysis (McCune and Grace 2002) for comparing species composition across treatments. A path analysis technique being considered for both site-level and network-level analysis uses indirect effects in structural equation modeling, and may help elucidate previously unrealized relationships (Quinn and Keough 2002). Finally, the SMIC recognized at the onset that the strength of the FFS network could best be realized through the calculation of effect sizes in a meta-analysis.

PRELIMINARY EVALUATION OF CORE RESPONSE VARIABLES

We began assessing the results of treatments the first growing season after full implementation of all treatments at each site. One initial question we addressed was the degree to which the active treatments (prescribed fire, mechanical thinning, and the combination of prescribed fire and mechanical thinning) resulted in similar stand structure. Based on analysis of variance, our active treatments resulted in a reduction in basal area at the Blodgett, Hungry Bob, Lubrecht Forest, Ohio Hills, and South Carolina Piedmont sites ($p < 0.05$) (table 2). The reduction was generally one-third to one-half of pretreatment basal area.

Another important indication of treatment success is the difference in height to live crown, or the height of the lower live branches, because this metric influences the transition of surface fire into tree crowns. Prescribed fire often kills small trees and prunes lower branches or scorches the lower crown of larger trees, whereas mechanical treatments can be more selective in removing only the small trees. Burning significantly increased the height to live crown ($p < 0.05$) at two sites in the western United States (Hungry Bob and Sequoia), but not in two sites in the eastern United States (Ohio Hills and South Carolina Piedmont) (table 2). Fuel types and pretreatment crown closure likely are responsible for the lack of treatment effects on lower crown heights at our southern and eastern sites. Surface fuels decay readily in the high moisture regimes of the Ohio, Alabama, North and South Carolina, and Florida sites, and a greater proportion of the material that burns is live understory compared to more western sites. In addition, the greater pretreatment height to base of live crown (table 3) suggests that self-pruning of lower branches occurs more frequently at these sites, restricting lethal heat to the lower live tree canopies

Our initial results of changes in understory (non-tree) species richness were highly variable (table 2). The lack of universally significant differences between treatments lends support to the hypothesis that sites dominated by natural high-frequency, low-severity fire regimes typically contain plant communities that undergo little floristic change after treatments modeled on historical fire disturbance (Metlen et al. 2004).

Table 2—Significance of pairwise orthogonal contrasts (planned a priori) after univariate analysis of variance for the change in basal area, post-treatment lower crown height, post-treatment understory vascular species richness, and log density for selected Fire and Fire Surrogate study sites

	Blodgett	Hungry Bob	Lubrecht Forest	Ohio Hills	South Carolina Piedmont	Southern Coastal Plain
Change in basal area (m^2-ha^{-1})						
Control vs. three active treatments	0.001	0.001	0.001	0.001	0.011	NS
Burn or thin vs. combined thin and burn	.115	.370	.001	.004	.053	NS
Burn vs. thin	.072	.015	.004	.001	.974	NS
Lower crown height (m)						
Control vs. three active treatments	NA	.004	.011	.899	.277	NS
Burn or thin vs. combined thin and burn	NA	.057	.015	.359	.077	NS
Burn vs. thin	NA	.017	.102	.788	.575	NS
Understory species richness (species·m^{-2})						
Control vs. three active treatments	NA	.107	.683	.001	.859	NS
Burn or thin vs. combined thin and burn	NA	.666	.044	.024	.786	NS
Burn vs. thin	NA	.748	.014	.026	.502	NS
Log density (number·ha^{-1})						
Control vs. three active treatments	NA	.003	NS	NA	.761	NS
Burn or thin vs. combined thin and burn	NA	.066	NS	NA	.383	NS
Burn vs. thin	NA	.001	NS	NA	.478	NS

• NA = Data not currently available for analysis.

• NS = Analysis of variance indicated no treatment effect, therefore contrasts unwarranted.

Finally, log density was selected as one metric for assessing the reduction in ground fuels, along with more traditional measures of volume and mass of litter and duff components, because log density has direct implications for changes in wildlife habitat values. Our first-year assessment indicated a significant reduction of log density with active treatment at Hungry Bob ($p = 0.003$) (table 2) with burn units containing few logs compared to thinned units. At other sites, log number did not change significantly with treatments. This likely is due to the propensity of fire to reduce the volume and mass, but not totally consume many downed logs, especially if the burns are conducted when moisture content within logs is relatively high. The number of downed logs likely will change over time at all sites as recently killed trees gradually fall to the forest floor.

Our efforts to date have focused on analyzing the individual site and building the network database to facilitate cross-site comparisons by using meta-analysis techniques. For example, we conducted a meta-analysis comparing the live crown height in first-year postburn units with live crown height in first-year control units from six sites. We used MetaWin version 2 (Rosenberg et al. 2000) with means and standard deviation data to calculate Hedges' *d* effect size and nonparametric estimates of the sampling variances for each study based on a fixed effects model. Nonparametric variances were calculated because they may be less constrained by the assumptions based on large sample sizes (Rosenbery et al. 2000). The confidence interval bounding the overall effect size was calculated based on 999 iterations of resampling. Our preliminary meta-analysis failed to indicate a significant network-wide treatment effect (an increase in height to live crown); the overall effect size was 0.4067 with a confidence interval spanning 0.0 (table 3). Yet significant and meaningful differences are anticipated by the third year post-treatment as lower crowns, once scorched by burning, continue to die back. These conflicting results from individual sites draw attention to the value of the FFS study as a large-scale experiment and suggest the value of meta-analysis across the network. Meta-analysis may be useful for determining which variables show similar response across sites, which variables require local interpretation, and the scale at which common themes emerge.

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Table 3—Results of a meta-analysis comparing the first-year post-treatment height to live crown for control and burn only treatments at selected Fire and Fire Surrogate study sites

Site	Mean height, control	Mean height, burn	Effect size (Hedges' *d*)	Nonparametric variance
Blodgett	7.5	7.4	-0.1358	0.6667
Hungry Bob	3.3	7.3	1.6667	.5000
Lubrecht Forest	8.1	7.4	-.2423	.6667
Sequoia	4.7	9.8	1.6543	.5000
South Carolina Piedmont	11.2	11.0	-.1753	.6667
Southern Coastal Plain	12.4	10.3	-1.3596	.7500

Mean effect size	95% confidence interval
0.4067	-0.413 to 1.226

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An overstory retention harvest on the Washington State Capitol Forest. *Photo by Tom Iraci*

CLOSING PERSPECTIVES

Photo by Tom Iraci

Balancing Ecosystem Values: A Manager's Perspective

Lynn Burditt[1]

Thank you for the opportunity to be with you this week. My remarks will be in three segments: national, regional, and local priorities; workshop observations; and some take-home messages.

PRIORITIES

As an agency, the Forest Service has a focus on four major threats facing the Nation's forests[2]— fire and fuels, invasive species, loss of open space, and unmanaged outdoor recreation. Within the Pacific Northwest Region, we have focused priorities that improve our responsiveness to these four threats. Chief Dale Bosworth, at the 2004 Ecological Society of America meeting, referred to the "era of ecological restoration and recreation." The Forest Service, and we as scientists and managers, must take a long-term view. We must ask ourselves to balance our studies such that we not only improve insights for current managers but also design studies that will answer the needs of the next decades, questions not fully understood or articulated today.

WORKSHOP OBSERVATIONS

Innovative experiments for sustainable forestry generate key insights for the future. This workshop compiles a diverse potential from which to draw knowledge. The range of posters, presentations, and field experiences include an impressive diversity of geography, sponsorship, study type (retrospective, model, field based), as well as spatial and temporal complexity. Participants demonstrate a shared understanding of the essential need to communicate effectively with a variety of public, the importance of rapid translation and transfer of information, and the power of field experiences and visual tools to convey understanding.

Critical elements for continued success include strong partnerships, descriptive language, and effective integration.

Science-Manager Partnerships

Landscape-scale studies and field-based experiments require a strong relationship among managers and scientists. It is imperative that participants value the expertise each brings to any partnership. Landscape studies entail a long-term commitment of a land base for the purpose of generating information. These studies generally require a landscape size that exceeds an "experimental forest" or a "research natural area." Management designations that reflect the purposes of the study can ensure succeeding managers are aware of the informational values being generated. This facilitates ongoing management commitment to long-term, landscape scale studies (e.g., Miller Creek in Montana and the long-term ecosystem productivity sites in the Pacific Northwest[3]).

Scientists and managers who work at the interface have a shared interest in the information that is developed; however, there may be different expectations, timing needs, and reward systems. Study design and implementation require a willingness to work collaboratively on study questions, meet administrative requirements, complete the appropriate scale of environmental analysis, incorporate appropriate pretreatment data collection, and facilitate grant proposals and funding needs.

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[2] Collins, Sally. 2004. Environmental services: making conservation work. Speech at Outdoor Writers Association of America. June 22. Spokane, WA.

[3] Some long-term study areas are incorporated into Land and Resource Management Plans; for example, the Miller Creek Demonstration Forest on the Flathead National Forest in Montana and the Long-Term Ecosystem Productivity Sites on the Willamette and Siskiyou National Forests in Oregon.

Large-scale studies and long-term research face challenges similar to those many counties face: voters may be willing to fund the capital construction of a detention or treatment facility; however, they expect the governmental entity to fund ongoing operations without new taxes. Over time, we have implemented several landscape-scale studies. We must be vigilant about the investments needed. Dedicated follow-through is essential for scientists and managers to maintain credibility with public. Vegetative manipulation implemented for large-scale studies generally includes a need for the activity to generate information. Whether focused on applied research or adaptive management, completion of data collection, analysis, and information synthesis is imperative.

Labguage: How We Convey Meaning?

Silvicultural system language has a strong foundation and conveys particular images. For example the terms "pre-commercial" and "commercial thinning" communicate an economic objective. Although this is a useful element, the terms may not adequately describe the management objectives. In an area where late-successional habitat conditions are the focus, the primary emphasis is to create specific habitat conditions. Terminology that effectively communicates this objective may not reflect the economic aspect. The workshop dialog regarding the necessity for new terminology and whether the descriptors we use need to reflect the mix of ecosystem values and objectives should continue.

Models

The use of simulation models and computer capability has expanded our capacity to analyze large amounts of information. The resultant plethora of models generates several concerns: assessing which model a manager should use for analysis, unwillingness to give up a model, a perception that each scientist has to generate his or her own model (associated with the reward system), and management (in)ability to collect and maintain data needed to facilitate model use. These models generate graphic and pictorial displays that are a wonderful communication tool. We need to highlight their value for understanding and comprehension while ensuring people do not lose touch with the real world (and why we all care about the land) as a tradeoff.

Science-Based Information or Opinion?

We live in a world of change. During the 1980s, the Forest Service was beginning to develop concepts of "legacy," whereas today it is common to hear, "We have learned that what we leave on the land is more important than what we take away." We have transitioned our management from a focus on outputs to a clear emphasis on outcomes.

Whether a scientist intends it, the potential exists for the individual to be seen as or used as a guru (spiritual guide or teacher). When this occurs, some parties may view the information as *prescriptive* rather than providing a basis to inform *management decisions*. One example relates to insights Andy Carey shared regarding active intentional management. A group of proponents has formed who feel the only viable thinning prescription is variable density – no matter what the land management objective, the geographic setting, the landscape context, the mix of vegetation conditions, or the economic implications. These proponents see the concept as a prescriptive approach that should be applied in all settings. While we continue to develop an understanding of public values, we must ensure we use science-based information appropriately.

INTRGRATION AND WHERE WE NEED TO FOCUS

It is imperative that we consider the kinds of information we will need 10 or 20 or 30 years from now. How can we design and implement the integrated research questions that will inform those needs? We must improve our capacity to integrate efforts. The concepts of adaptive management and research-management partnerships are articulated on many fronts. To a large extent, current success and energy arises from the individuals at the local level who have a shared sense of dedication to the outcome. Within our federal agencies, we must continue to search for an enhanced institutional model if we truly want to increase our capacity for adaptive management.

Landscape-scale studies require large investments of energy and resources. We must assess the lessons we are generating and examine our commitment to follow through for long-term information generation. This workshop focused on innovative experiments for sustainable forestry helps identify the proliferation of field studies that exist. The studies discussed here represent major investments. To best facilitate outcomes from these investments, perhaps we need to create a vehicle to place these studies into context, synthesize key findings, provide integrative lessons, enhance collaboration, and provide advice on which studies have long-term merit.

The dedication and commitment you have each made to generate new information for society is impressive. I appreciate the opportunity to visit with you today and look forward to an ongoing conversation.

Where Can We Go From Here?

E. David Ford[1]

INTRODUCTION

First I wish to thank the meeting organizers for developing such an excellent and timely program of papers, which combined with the field tours, has enabled us to analyze many important issues facing the development of silvicultural and management approaches to multipurpose use forestry. I also thank the organizers for inviting me to provide remarks to summarize the meeting and make a synthesis. Of course this is too large a task for one person—and I am still wondering why I agreed to attempt it! So what I offer are some comments about specific issues that arose during the meeting and a description of why an effort to synthesize our current knowledge is necessary, and how I think it might proceed.

SPECIFIC ISSUES

The value of experiments established over the last decade has been that treatments have been replicated, and that contrasting manipulations have been applied. For example, DEMO contrasts quantity and pattern of removal in a factorial. Nevertheless, having been involved in the planning process for DEMO, I am relieved that some response variables clearly show treatment effects when analyzed using ANOVA (see Maguire et al. 2005). I am relieved because it was apparent at the start of the work, given the distribution of the replicates across considerable differences in landscape, that the assumption of homogeneity of variance required for ANOVA was not likely to hold. For detecting that there *are* differences between treatments, textbooks reassure that ANOVA is robust—and so it seems to be in this case.

However, we want to know not just that there are differences between treatments but what it is about the treatments that produce differences in response. This more detailed analysis is proving complex (see Maguire et al. 2005), and it seems likely we will have to take a many-sided approach in examining for relationships between structure of the communities created and the response of organisms and physical and chemical properties. In particular, responses are likely to vary within treatment replicates and to change over time—in practice the treatments have created a set of different systems. In this case we can use various biometric techniques, e.g., longitudinal analysis (Diggle et al. 1994), and use what we have so far considered to be replicates as repetitions of stochastic processes, and be very glad that we have these repetitions! So, although the experiment was originally established as synchronic, i.e., treatments compared at the same time because measurements were made before and after treatments were applied, we also have possibilities for analysis as a diachronic experiment, i.e., before and after.

Given the time and expense concerned in collecting the data, and the importance of the results for the development of silvicultural techniques, it would be a terrible waste if modern biometric techniques were not used. This is likely to involve a detailed collaboration between scientists and biometricians, particularly those interested in stochastic processes. I strongly urge that analysis be followed through to the formation of statistical models wherever possible, and that this should be done before planning new experiments or extending measurements in the present series. For variables where there are responses to the treatments, we need to move on to analyze the causal processes involved. For variables where there are no responses, or responses that are unclear, it is import to reconsider whether the correct measurements are being made rather than simply continuing an established protocol.

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During the meeting there were a number of interesting papers from scientists working in other parts of the world, particularly the tropics and in the hardwood zones of central and eastern North American. A common thread in this work, either explicitly stated or implicit, was to consider silvicultural treatments as creating gaps and to use "gap theory" as a theoretical foundation for the work (e.g., Canham and Marks 1985, Runkle 1985). A note of caution for scientists working in the tall conifer forests in western North America—it is not clear what role canopy gaps play in the regeneration of these forests (Canham 1989), nor how and when they come to form (Franklin et al. 2002). The initial regeneration of shade tolerant species does not require gaps and may actually be inhibited by them if ground vegetation develops.

SYNTHESIS

At the meeting, both in the talks and on the field trips, we have seen two types of experiments that ask different questions. Experiments into *silvicultural design* ask, "Can we use a silvicultural system other than clear felling to obtain a timber yield and yet satisfy a requirement for non-timber objectives, particularly public perception?" Experiments of *scientific analysis* ask, "What are the ecological consequences of removing timber in different amounts and different spatial arrangements?" The difference in the types of experiment is frequently apparent in the types of treatment applied. It is also apparent in the response variables measured. I have a plea to make: Can the type and rates of growth of natural regeneration and planted stock be measured in both types of experiment? Not only could this be an important way to relate these different types of experiment, but these regeneration parameters are essential for reaching a goal of sustainability, and they will determine the ecological pathway followed after manipulations. Applying silvicultural treatments without measuring the effects and fate of regeneration means that understanding these treatments is postponed too long into the future. Applying treatments to gain an ecological understanding, but restricting measurement to just the supposed response variables of interest reduces the possibility of developing a causal theory and reduces the utility of these results for silvicultural purposes.

I stress the need for developing a causal theory. By this I mean a theory for the effect of timber operations on biological and ecological processes and the growth of timber itself. The alternative to a causal theory is one based on repeated empirical investigations. We do not have time to develop an empirically based theory where we apply treatments and sit back and wait for results. Furthermore, experience in forestry suggests that variation in forest type and structure can affect the results of experiments even when we think we are experimenting within the same forest type.

A causal theory for this purpose would define accepted information and the concepts used; it would also define variation in treatment—response according to conditions. Causal theories also contain questions and we can expect them to change as knowledge improves (Ford 2000, Chapter 11). The definition of a causal theory would be more than typically is found in an academic review, in particular, it would define uncertainty. I suggest that the development of such a theory for the conifer forest of the Pacific Coastal region should be an international task, very suitable for IUFRO. After all IUFRO was founded in response to similar requirements in European forestry. One reason for an explicit definition of a causal theory is to ensure the use of appropriate models. Unfortunately we all too frequently see the requirement for a theory relapse to some model that is available and/or in current use.

IN CONCLUSION

In response to the question,"Where can we go from here?" I offer three suggestions:

1. Specify the theory of regeneration in conifer forests in relation to potential manipulations and their effects on timber and nontimber objectives. This is a task of synthesis, and I suggest it would best be conducted as an international exercise. And, although scientists are clearly essential for specifying known and likely interactions between processes, the theory should have a clearly specified management purpose.
2. Postpone additional large-scale initiatives until a theory is specified and focus on analysis of results from current experiments. Analysis of results from recent experiments is essential for developing a theory, this analysis is not complete and has not been integrated with existing information and theories on forest development, habitat creation, and management.
3. Anticipate investment in Living Forest Laboratories. The various experiments established in this forest type are likely to be important in the future because treatments have been repeated. At least some of these experiments should be carefully managed and deciding which ones should depend upon the results of my first suggestion.

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Summary of "Realities in Timber Management" Discussion at 2004 IUFRO Workshop

Russ McKinley[1]

The Oregon Department of Forestry claims Oregon forests grow 10 billion board feet (bf) annually; 4 billion bf are cut annually. How long can we continue to store this extra fiber? How long should this situation continue, especially in our dry forests where extra carbon is recycled mostly by fire rather than by decomposition?

The biggest forcasting problem, as I see it, on federal land is that the federal government (USDA Forest Service and USDI Bureau of Land Management) is not accurately predicting the consequences of forest succession without restoration management and the consequences of succession with different types of management intervention. If current (deficient) simulations are not viewed as credible among federal, state, and industry biologists, they will have little use. I applaud your efforts to create the data and models for improved simulations.

While I was reading your abstracts, I remembered a discussion Dr. Arnie Skaugset and I had about the stream surveys we did as an industry. He showed the Oregon Forest Industry Council that industry riparian areas were not statistically different from state or federal areas. Then he told me that natural variation is so large that it will be very expensive to quantify the footprint of management on the landscape. I see each of you struggling with this same sampling problem as well.

In my opinion, experience over the past 10 to 15 years in dry (mostly federal) forests shows the following:

Forest succession without restoration produces fire prone landscapes (Condition Class III). In many cases, this places watersheds as well as wildlife habitat and species at risk.

Managed disturbance (forest restoration) can be more effective than natural disturbance (uncharacteristic wildfire) in maintaining watershed and wildlife values.

There are many locations where the long-term benefits to watersheds and wildlife values of forest restoration outweigh the short-term risks of such restoration. It's apparent to me that in most cases, in dry, fire-prone forests "no action (no restoration management) is not good for wildlife." I was not surprised to see that the review team for the status of the northern spotted owl (*Strix occidentalis caurina*) identified uncharacteristic wildfire as the major cause of habitat loss for the northern spotted owl from 1994 to 2004. I'll feel a lot better about this when there is broader understanding and acceptance of this reality, especially in the biological community. Until this happens, there will likely be very limited economic, social and biological benefits from our "at risk" fire-prone forests with little restoration being accomplished at a landscape scale.

So why would biologists want to participate in this discussion? What follows are some things I have observed in southern Oregon.

Losses on the Biscuit fire:

- Marbled murrelet (*Brachyramphus marmoratus*) critical habitat, 96 million acres, 65 percent (heavy/high mortality)
- Northern spotted owl critical habitat, 69 million acres, 63 percent (heavy/high mortality)
- Coho salmon (*Oncorhynchus kisutch*) watersheds, 338 million acres, 65 percent (heavy/high mortality)
- Late-successional reserve, 160 million acres, 68 percent (heavy/high mortality)

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Seventy-seven percent of Oregon's forests (24.5 million acres) are in Condition Class II or III.

Recent review of the northern spotted owl on dry site Douglas-fir showed "uncharacteristic wildfire" to be the single largest threat.

Without restoration, Class II becomes Class III. If private land cannot be used for forestry, it will be developed for home sites. If the land was appropriate for agriculture or industry, it is probably already in use. This, in my experience, is not good for wildlife.

Forest certification forces landowners to consider wildlife and the effect of the management plan on landscape biodiversity. Our biologists helped us develop our plan but really struggle to quantify the good or negative aspects of the plan. We pass our audits and feel good about our plans but struggle to quantify value. I keep asking, and the biologists want to know also.

Given the mix of government, large owners, and small owners, we may be providing the landscape we need. We can't quantify what we do; we as a society can't agree on biodiversity objectives and for some groups; this confusion helps them generate revenue. I truly believe there is no better private use of forestland that has to pay taxes than forestry. The research you are doing will over time help us improve through the Sustainable Forest Certification process.

I spent parts of 2 days reviewing your conference abstracts. As scientists, I understand that you need to produce peer-reviewed replicable research. As a result of my review, here are some observations for your consideration.

Much of the results were stand-level results. This makes sense to achieve proper statistical precision. The work you are doing is absolutely critical to preserving a broad range of habitat. I suggest it is time to combine many of these models so they can be used for landscape assessment. We recently completed a landscape assessment for northern spotted owls in southwest Oregon. It was a very large effort, but it did produce results that interested managers and biologists. We have a new appreciation for the importance of forest restoration management in protecting the habitat of northern spotted owl over many decades. Natural variation made this work much more difficult than anticipated. Getting clean data sets on 800,000 acres took much more time than anticipated. We found a "light touch on the land" was worse than no action, and bolder approaches had much better results at controlling uncharacteristic wildfire. This was better in the long run for northern spotted owls. The strategic problem of where you *can* apply treatments and then the strategic problem of where you *should* put the treatments are challenges we will struggle with for at least the rest of my career. Quantifying the "footprint of any kind of management" on the landscape is a very stiff challenge.

When I reviewed my thoughts with Steve Mealey, Boise's manager of ecology, he pointed out that at the November 2003 Risk Conference in Portland, Mark Finney reported that his modeling work with FARSITE indicates that less than or equal to 25 percent of a landscape can be treated to effectively reduce fire risks across the entire landscape, as long as the treatments are strategically placed. If such treatments are not strategically placed, as much as 50 percent of the landscape must be treated to create resiliency. Strategic placement of treatments for effectiveness in reducing resiliency requires landscape-level assessments and planning. This strongly supports concerns about the need to plan and assess ecosystem conditions at the landscape scale.

Increasingly, professional cultures control how our work is used in decisionmaking. Learning to work within those cultures in a collaborative way makes the difference in whether the good work you produce will ever get used. In our case, if the biologists do not loudly acknowledge that wildlife habitat is at risk from lack of restoration management, I doubt your work will have much value.

Until biologists and interdisciplinary teams loudly ask for designed and carefully executed restoration management for "at risk" forests, I predict there will be little economic and related social value generated from the harvest of timber on federal lands.

Large fires originating on Class II and III forests will continue to damage our company timberlands. These forest, vulnerable to lightening, are growing in size and number. Working collaboratively with biologists could introduce controlled restoration management and limit the size and damage of these wildfires.

We need you and the biologists to be successful. I worry that it may take a long time. The mill capacity to support federal land management programs is mostly gone. The economic pressure to produce an annual cut is almost gone. Predictable succession patterns without forest restoration management will create forests that will burn with uncharacteristic intensity and waste useable wood and damage habitat.

The Healthy Forest Restoration Act says to me that accurate simulation analysis of forest succession with different types of management intervention is both *desired and required* for successful forest ecosystem management and timber sales on federal lands. The door is open. Make your work count!

When your collaborative work with biologists and interdisciplinary teams is successful, management strategies will develop that could create economic opportunity. These opportunities can be pursued and social benefits will result. If this challenge is not met, the rest of the world will gladly supply our wood products. No shortages will occur for long at the home improvement centers, just lost opportunities to provide for our own needs sustainably.

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A Public Land Manager's Perspective: The Need for Innovative Experiments for Sustainable Forestry

Gretchen Nicholas[1]

Good morning. The two areas I will address are my general impressions of the conference and the relevance of the work I have seen with regard to Washington Department of Natural Resources' (DNR) needs.

DNR has two major roles: natural resources protector and land manager. As resource protector, DNR provides regulation of forest practices and fire protection for state and private forested lands in Washington, regulates mining reclamation, and assists rural and urban private landowners accomplish healthy ecosystems on their small forests or urban properties.

As land manager, DNR is steward of 2.4 million acres of the state's aquatic lands – bedlands, tidelands, many beaches, and navigable lakes and rivers. In addition, DNR is a manager of 2.1 million acres of forestland and 800,000 acres of farm and grazing lands, and additional commercial properties. These are state trust lands, managed in trust to provide revenue for multiple public benefits including funding construction of public schools statewide and universities and funding services in many counties. The income earned from these lands is paid directly to the specific beneficiaries and amount to about $150 million a year. My comments today will be made from the perspective of a land manager of these proprietary holdings.

First, my general impressions:
I heard three major themes at the conference and will illustrate those themes with quotes from the various speakers or participants.

The first theme relates to the relative dearth of information on anything other than regeneration harvest, particularly in the Pacific Northwest region of the United States. This is an important issue because many forestry organizations are beginning to experiment with new harvest methods such as variable retention harvest and biodiversity pathways, yet there is little experimental evidence to guide the silviculture. Jerry Franklin described the current status of this issue in his introductory remarks. He said, "We have framed two end points…Clear-cut management and no management, but nothing in between. But now we are starting to work there." However, "…ecological science [with regard to this area in between] is a series of theoretical constructs that are untested… We need some data on these systems…"

I agree with Franklin's comments and think he would add that data are only one part of the problem. In many cases we lack unifying theories, or constructs, to guide our actions. One example is the arena of biodiversity. As Steve Zack said, "Biodiversity is bean counting"… but not always… "It is inappropriate to confuse the conservation interest in maximizing biodiversity with efforts like those in the restoration of ponderosa pine where the emphasis is in restructuring the habitat and reinstalling fire. The effect might be local reduction in biodiversity ("species richness") as the habitat is simplified with treatments, but, if we are truly concerned with the habitat and processes, then prior higher richness was an artifact of fire suppression, and our restored ponderosa pine forests have the 'relevant' richness." This comment by Zack illustrates the complexity of such issues as ecosystem restoration and preservation of biodiversity.

With all this uncertainty, it is nice to know that there is some certainty in life. Dan Luoma summed up his entire presentation in the simple, certain, and bumper-sticker-ready comment – "No fungi, no forests."

The second theme relates to the need for long-term research. It often seems that just when a research project

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starts gathering the information we need over a long enough time frame to be valuable, the funding is gone. Many times at this conference researchers have mentioned that funding sources often seem fickle and driven by political interests. I overheard Klaus Von Gadow making the remark, "All important decisions are made in 10 minutes." Although that humorous comment is not entirely true, it is true that many decisions on funding are made by managers like me, who have a great number of priorities to balance and goals to achieve. We are usually looking for projects that meet our long- or short-term goals. This search for relevance brings me to the third theme.

The third theme I heard is that partnership between managers and scientists is needed to achieve relevance. That type of partnership is demonstrated by research such as the Blue Ridge study (silvicultural options for young-growth Douglas-fir forests) in DNR's Capitol Forest near Olympia This study examines biotic responses to alternative silvicultural regimes such as group selection, patch cuts, thinning and regeneration harvest with leave trees. The presenter for that study, Dave Marshall, emphasized that this study originated at the request of field personnel.

I offer some advice for the scientist embarking on a partnership with managers in research. As Lisa Ganio emphasized, it is important to develop a focused understanding of the question that must be answered before a study is designed. Once the data are being analyzed, it is too late. Lisa also pointed out that the broader the scope of inference of a study, the more valuable the research results will be.

Producing results when the results are needed also is important to maintaining funding. As Mike McClellan accurately observed, "We need to produce something in the short term as well as the long term. Use retrospective studies to provide interim guidance, if necessary." I would add, as suggested by several speakers, modeling with adequately validated models can also provide near-term information.

A good example of the combined use of modeling with retrospective studies as a starting point, followed by the installation of long term studies, can be found with the Biodiversity pathways approach developed by Andrew Carey and his colleagues at the U.S. Forest Service. The concept of "biodiversity pathways" is based on the four possible types of major studies: (1) literature review and synthesis to produce theory; (2) retrospective (or comparative/cross-sectional) studies to provide quantitative guidance for treatments; (3) prospective (or experimental studies like the Forest Ecosystem study in its 13th year and the Olympic Habitat Development Study, with plots up to their 7th year post-treatment) to test hypotheses under the new theory; and (4) simulation studies (the Washington Forest Landscape Management Project) to integrate various models and compare alternative pathways with respect to social, economic, and ecological results); DNR just completed some of its own in-house simulations studies in its Sustainable Harvest analysis.

The second major topic I will discuss is the relevance of the research presented at this workshop to the DNR. The DNR recently completed a final Environmental Impact Statement (EIS) for management of 1.4 million acres of forested state trust lands west of the Cascade Range. The Board of Natural Resources, which represents the beneficiaries and makes major policy decisions regarding DNR-managed lands, decided in September 2004 to implement the preferred alternative. The preferred alternative specifies that in areas targeted for habitat development, a large proportion of the annual acreage planned for harvest will be devoted to a biodiversity pathways approach. This will amount to approximately 20 to 30 percent of the 15,000 to 20,000-acre annual harvest. Results will be monitored to assure that the dual objectives of income to the trusts and habitat creation are being met. But long-term success depends on long-term information, and this new direction creates a need for increased information in the following areas:

- Progression of forest structural stages and modeling of stand growth in response to biodiversity management
- Growth of regeneration in patch and group selection scenarios
- Sampling methods for multiple age stands

This information will help determine whether DNR is actually creating the stand features it wishes to, and ultimately through monitoring, will determine whether the result is improved habitat.

Forest inventory is expensive, and the DNR has invested extensively in good inventory samples and GIS information. Yet future necessary information will be more extensive than our current techniques can address. We must find efficient ways to meet future need for new types of data without increasing information collection costs. One arena that shows promise is the laser imaging technology called LIDAR (light detection and ranging).[2] There are many information needs that can be filled by LIDAR. For example, measurements of vegetation structure can be used to infer stand development stages, Steve Reutebuch discussed on-going research that he and colleagues are pursuing that would provide tree-, stand-, and landscape-level structure measurements that are geospatially referenced and compatible with existing GIS stand information. If successful it could result in an effective means to monitor forest structural changes over time in response to planned treatments.

Research can be made more relevant to operations if researchers develop thresholds whenever possible. For example Gordon Bradley's work on visual impacts of various types of silvicultural activities has been extremely useful to the DNR in determining types of harvest patterns to use in visually sensitive areas. We know from his studies that scattered leave trees are visually pleasing to most of the individuals surveyed. The minimum number of leave trees per acre required to create the positive viewer reaction would be useful "threshold" information to help in guideline development. With better data on the incremental impacts of increased levels of overstory on various understory species, we could frame appropriate leave tree targets in visually sensitive areas. The collaborative (Washington Department of Natural Resources/Pacific Northwest Research Station) Overstory Density Study at Capitol Forest will provide such needed information.

Another area in which DNR needs more information about thresholds relates to public concerns about cumulative impacts within a landscape and how that affects biodiversity. The DNR has substantially addressed this issue through its multi-species habitat conservation plan. In addition the DNR has adopted a modified variable retention strategy and has recently completed a final EIS that calls for a substantial implementation of the Biodiversity pathways approach developed by Andrew Carey and his colleagues. As I previously mentioned, these strategies represent the best efforts of scientists to use various research methods to develop answers about how to conduct harvest in forests and still preserve biodiversity. More information is needed about the impacts of various levels of deviation from historic levels of disturbance, which would allow for a better understanding of landscape and stand level targets. Bob Seymour took a step in that direction with his work on indexing activity to historic frequency and extent of disturbance. Equally promising is some of Fred Bunnell's work that tackles the biodiversity issue from a strategic level by setting criteria for landscape level activities. I look forward to reviewing his work more closely and to seeing what information he uses to quantify his targets.

In conclusion, there are tremendous operational research opportunities on DNR-managed state trust lands. As the Capitol Forest tour demonstrated, we are able to complete the harvest of experimental designs expeditiously, we have excellent resource information and tracking systems, and we are experimenting with innovative methods of forest management. We have had very successful collaboration with the Pacific Northwest Research Station thus far, and look forward to strengthening that relationship and others that can move us all forward in the management of healthy, diverse and productive forestlands.

[2] LIDAR (LIght Detection And Ranging) uses the same principle as RADAR. The LIDAR instrument transmits light out to a target. The transmitted light interacts with and is changed by the target. Some of this light is reflected or scattered back to the instrument where it is analyzed. The change in the properties of the light enables some property of the target to be determined. The time for the light to travel out to the target and back to the LIDAR is used to determine the range to the target. (http://www.ghcc.msfc.nasa.gov/sparcle/sparcle_tutorial.html)

Public perceptions of silvicultural options for harvesting young-growth forests are presented by Gordon Bradley (University of Washington) during field tours of the Capitol Forest Blue Ridge study. *Photo by Robert Harrison*

Photo by Charley Peterson

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