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Dry Forests of the Northeastern Cascades Fire and Fire Surrogate Project Site, Mission Creek, Okanogan-Wenatchee National Forest

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Abstract

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The Fire and Fire Surrogate (FFS) project is a large long-term metastudy established to assess the effectiveness and ecological impacts of burning and fire "surrogates" such as cuttings and mechanical fuel treatments that are used instead of fire, or in combination with fire, to restore dry forests. One of the 13 national FFS sites is the Northeastern Cascades site at Mission Creek on the Okanogan-Wenatchee National Forest. The study area includes 12 forested stands that encompass a representative range of dry forest conditions in the northeastern Cascade Range. We describe site histories and environmental settings, experimental design, field methods, and quantify the pretreatment composition and structure of vegetation, fuels, soils and soil biota, entomology and pathology, birds, and small mammals that occurred during the 2000 and 2001 field seasons. We also describe the implementation of thinning treatments completed during 2003 and spring burning treatments done during 2004 and 2006.

Keywords: Dry forest, Washington, Cascade Range, vegetation, fuels, soils, soil biota, entomology, pathology, wildlife.

Summary

The Mission Creek Fire and Fire Surrogate (FFS) project was initiated in 2000. Since that time, pretreatment measurements have been completed and analyzed and are summarized in this report. Treatments have also been applied, although they were completed in two phases: thinning and four of the prescribed burns were completed in 2002–2004, and the final two prescribed burns were completed in 2006. This research paper summarizes the implications of the pretreatment results and the implementation of treatments and posttreatment progress.

The forests of the Mission Creek study site are representative of larger dry forest landscapes of the West. Mission Creek is within Ecological Subregion 11 (ESR 11) of the interior Columbia River basin, and the primary scope of inference for the study results. The subregion extends along the lower elevation forested lands of the eastern Cascades of Washington and Oregon, and on an easterly arc to the Kettle Range in northeastern Washington. The Mission Creek results combined with those from the Lubrecht FFS site in the northern Rocky Mountains of northeastern Montana and the Hungry Bob FFS site in the Blue Mountains of northeastern Oregon expand the FFS scope of inference to much of the dry forest in the interior Columbia River basin.

The forests of the Mission Creek study site are representative of larger dry forest landscapes of the West.

This dry forest is codominated by Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and ponderosa pine (*Pinus ponderosa* Dougl. ex Laws). About 10 percent of the forest composition consists of grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.) and western larch (*Larix occidentalis* Nutt.). Among FFS sites, it is less ponderosa pine-dominated than dry forests of the Southwest, has less lodgepole pine (*Pinus contorta* Dougl. ex Loud.) than dry forests of the northern Rocky Mountains, and has lower tree species richness than sites in the Sierra Nevada. Among FFS sites, Mission Creek is closest in tree composition to the Hungry Bob site in northeastern Oregon, but our experimental units appear to have had fewer logging entries and higher basal area. Compared to historical stands in ESR 11, current units have higher tree density and basal area, more small trees, and more tree layers. All of these features are likely to be affected by treatments of thinning or prescribed fire.

There are few significant vegetation differences among Mission Creek units. However, there is substantial within-unit variation. Overstory vegetation is generally similar across all pretreatment units in terms of tree density, snag density, basal area, stand density index, sapling density, species composition, canopy cover, bulk density, and base height. The site is fairly species rich with 124 understory species, most with low cover and frequency, but graminoids and shrubs dominate the understory. Serviceberry (*Amelanchier alnifolia (Nutt.)* Nutt. ex M. Roem.),

snowberry (*Symphoricarpos albus* (L.) S.F. Blake), and rose (*Rosa* spp.) are the dominant shrubs, and pinegrass (*Calamagrostis rubescens* Buckley) and elk sedge (*Carex geyeri* Boott) are the dominant graminoids. Understory community structure differs among treatment units, and understory vegetation is correlated with physical and biological elements, such as elevation and overstory tree density. Dry forests of Montana have shrub-forb understory dominance, and the Oregon and Southwest sites are forb-graminoid dominated.

Current fuel conditions at Mission Creek show moderate variability among units, but few significant differences. The ranges of biomass for 1-hr, 10-hr, and 100-hr fuels are within the ranges of other dry forests, but the larger fuels (1,000-hr+ [>7.62 cm diameter]) are 25 to 65 percent of levels in other dry forests. Fire behavior is affected primarily by the smaller dead fuels plus herbs and shrubs, so the low levels of coarse woody debris have few fire behavior implications. Projections of fire behavior show a majority of units would be affected by severe fire behavior (torching and crowning) under worst-case weather using either the Northern Forest Fire Laboratory models or custom models where shrub depth was used to define fuelbed depth. Less severe fire behavior is predicted if fuelbed depth is defined by dead fuel particle height.

Projections of fire behavior show a majority of units would be affected by severe fire behavior under worst-case weather.

Soils of Mission Creek are largely mollisols, with some alfisols, inceptisols, and entisols, derived primarily from sandstone. The A horizons, on average, are deep for a forest soil (up to 88 cm), but quite variable across units; Bw and Bt horizons are also quite variable. Soils are well-structured with moderate bulk densities, usually loamy to sandy loamy at the surface, grading to increasingly sandy or clayey texture at depth. Hydrophobicity is common at the surface owing to exudates from leaf litter coating surface soil particles. Carbon and nitrogen are high in these soils, with a high base saturation as well. Nutrients appear less limiting for plant growth than moisture, and it appears that by June, soil moisture levels are significantly drawn down by evapotranspiration.

Soil enzymes and microarthropods are variable across and within units. Both can affect site productivity by influencing organic matter input and decomposition. Sampling on a landscape gradient of ridgetop, sideslope, open bare area, and valley bottom showed some differences by landscape position, probably associated with moisture availability. Soil enzyme activity was relatively low in the surface soil samples, generally higher with increasing Douglas-fir dominance, and not significantly different by landscape position. The soil microarthropod fauna density was dominated by oribatid mites (84 percent) and Collembola (springtails, 7 percent). The total populations of these two microarthropods did not differ by landscape position, but some of the less abundant microarthropods were more often found in valleys than on ridges. There were some positive correlations with available moisture and negative relationships with increasing bulk density for both soil enzymes and soil microarthropods.

Root pathogens were almost absent at Mission Creek with only one sample (on an asymptomatic tree) showing the presence of a root rot. This reflects the dry conditions at the study site, and would not generally apply to moist forests, even in the same ecoregion. Analysis of fungal endophytes showed that they exist in large woody roots of live conifers. Almost half of asymptomatic Douglas-fir and ponderosa pine sampled contained fungal endophytes, generating an interesting question of how they might influence the dynamics of pathenogenic fungi in the rhizosphere of forest tree roots. Bark beetle activity was largely due to western pine beetle (*Dendroctonus brevicomis*) and mountain pine beetle (*D. ponderosae*) attacking ponderosa pine. Although some large centers of activity were found, the percentage of trees affected by bark beetle attack was relatively low in the pretreatment sampling. Dwarf mistletoe was found on all but two units, and was present on both Douglas-fir and ponderosa pine. Every overstory class was affected, but the density of trees affected was relatively low. Infection rates ranged from 0 to 8.8 percent of unit basal area. Woodpecker foraging was observed on less than 1 percent to more than 8 percent of snags or trees larger than 20 cm diameter.

The most common birds detected at Mission Creek were the chipping sparrow (*Spizella passerina*), dark-eyed junco (*Junco hyemalis*), mountain chickadee (*Poecile gambeli*), Nashville warbler (*Vermivora ruficapilla*), pine siskin (*Carduelis pinus*), red-breasted nuthatch (*Sitta canadensis*), Townsend's warbler (*Dendroica townsendi*), western tanager (*Piranga ludoviciana*), and yellow-rumped warbler (*D. coronata*). Species assemblages did not differ across units. Nesting success was highest for cavity nesters, intermediate for ground nesters, and lowest for foliage nesters, representing decreasing ability to provide safe sites for young birds. Bark gleaners (predominantly red-breasted nuthatch), brown creeper (*Certhia americana*), and hairy woodpecker (*Picoides villosus*) exhibited preference for larger trees, with the first two showing no preference for whether the tree was live or dead, and the woodpecker preferentially selecting larger snags. Responses of focal species to potential treatment were projected as being positive for the chipping sparrow, white-headed woodpecker (*Picoides albolarvatus*), western bluebird (*Sialia mexicana*), and Cassin's finch (*Carpodacus cassinii*), and negative for the hermit thrush (*Catharus guttatus*), western tanager, and brown creeper.

About two-thirds of the 21 expected species of small mammals were captured at the Mission Creek study site. The deer mouse (*Peromyscus maniculatus*) was the dominant small rodent (51 percent of captures), the yellow-pine chipmunk

(*Neotamias amoenus*) was the dominant arboreal or semiarboreal rodent (21 percent of captures), and the Trowbridge's shrew (*Sorex trowbridgii*) was the most common insectivore. The deer mouse and yellow-pine chipmunk were codominant if the larger size of the chipmunk was factored in. Other species individually accounted for <5 percent of the catch and occurred in <50 percent of the stands. Other than chipmunks, few arboreal or semiarboreal species were captured, probably because of a lack of cover (low numbers of snags, logs, and mistletoe) and food (particularly truffles and lichens for northern flying squirrels [*Glaucomys sabrinus*]). Mammal assemblages were similar among the assigned treatment groups of stands, but assemblages did differ among units. Half the units were relatively mesic habitats and supported a richer assemblage of small mammals that included all the captured species, compared to the other relatively species-poor dry units. Management practices that open and dry stands, reduce large down wood, and shift understory dominance to grass likely will shift mammal species assemblages to favor species associated with the dry end of the gradient.

Planning for the fire and surrogate (thinning) treatments began in 2000. The thin treatment was designed as a low thinning to reduce basal area so the residual stand would have a clumped distribution of roughly 15 m^2/ha, similar to that found in historical stands. Owing to access limitations and soils concerns, all thinning treatments were yarded by helicopter. Thinning treatments were completed by spring 2003. Burn treatments were partially accomplished during spring 2004. Because of unusual weather, only four of the six units scheduled for burning were ignited, and they burned lightly or patchily. One of the burn-only units and one of the thin-burn units did not receive their burn treatment. The resulting unbalanced design with four control (Crow 3, Pendleton, Sand 19, Sand 2), four thin (Crow 1, Crow 6, Ruby, Slawson), two burn (Poison, Spromberg), and two thin-burn (Camas, Tripp) units was used in the first posttreatment sampling during 2004–2005. Data on the economics of harvesting dry eastern Cascade forests, using a variety of yarding techniques, were also gathered as part of the FFS project, and will be reported separately.

The resulting balanced design has three control units, three thin units, and three thin-burn units. This design is anticipated to remain stable and serve as the template for future remeasurement of the Mission Creek FFS site.

In 2006, the design was balanced by the addition of two more burns that occurred in spring 2006. The resulting balanced design has three control units (Crow 3, Sand 19, Sand 2), three thin units (Crow 1, Ruby, Slawson), three burn units (Pendleton, Poison, Spromberg), and three thin-burn units (Camas, Crow 6, Tripp). Immediate remeasurement of these units was not possible owing to lack of funding. It is anticipated that this design will remain stable and serve as the template for future remeasurement of the Mission Creek FFS site.

Contents

Chapter 1: Introduction

James K. Agee and John F. Lehmkuhl [1]

Background

Concerns about declining forest health in the dry forests of the Western United States have generated multiple efforts to restore fire-resilient forest structures across the dry forests of the West. Programs to deal with the problem of larger and more severe fires in the West have been initiated at the state level (e.g., Western Governor's Association 2003: Policy Resolution 03-18, September 15, 2003, at Big Sky, Montana) and the federal level (e.g., Healthy Forests Restoration Act of 2003). Focus has been both on values (the wildland-urban interface) and hazards (fuel reduction), including timing (Allen et al. 2002) and treatment type (Agee and Skinner 2005, Brown et al. 2004). A national research program, known as the Joint Fire Science Program, has also been enacted to provide information necessary to assist fuel reduction while protecting and restoring ecological values.

The largest project funded by the Joint Fire Science Program in its early years was the Fire and Fire Surrogate (FFS) project (http://frames.nbii.gov/ffs). The FFS is a long-term study to assess the effectiveness and ecological impacts of burning and fire "surrogates" such as cuttings and mechanical fuel treatments that are used instead of fire, or in combination with fire. For many of the sites, the long-term goal of these treatments is to restore historical ecosystem structure and function, but the consequences for nutrient cycling, seed scarification, plant diversity, disease and insect abundance, and wildlife are mostly unknown. Similarly, although combining managed fire with silvicultural treatments adds the critical effects of combustion, we know little about ecological effects, economics, and fire hazard reduction of these methods. There are 12 FFS sites with the full complement of fire and fire surrogate treatments, plus an additional satellite site with fire treatments only (table 1-1). Each of these sites had a historically short-interval, low- to moderate-severity fire regime. Eight of the western sites share ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) as an important tree component. One of the 12 complete sites is the Northeastern Cascades site at Mission Creek on the Okanogan-Wenatchee National Forest (fig. 1-1). For this report, we refer to this site as "Mission Creek."

The FFS is a long-term study to assess the effectiveness and ecological impacts of burning and fire "surrogates" such as cuttings and mechanical fuel treatments. This report summarizes the pretreatment conditions of the Mission Creek site.

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Table 1-1—The Fire and Fire Surrogate study national network

Site	Location
Northeastern Cascades	Mission Creek, north-central Washington
Blue Mountains	Wallowa-Whitman National Forest, Oregon
Northern Rocky Mountains	Lubrecht Experimental Forest, western Montana
Southern Cascades	Klamath National Forest, northeastern California
Central Sierra Nevada	Blodgett Forest, northern California
Southern Sierra Nevada[a]	Sequoia and Kings Canyon National Parks, California
Southwestern Plateau	Coconino and Kaibab National Forests, northern Arizona
Southern Rocky Mountains	Santa Fe National Forest, northern New Mexico
Central Appalachian Plateau	Ohio Division of Forestry and Mead Corporation, Ohio
Southeastern Piedmont	Clemson Experimental Forest, northwestern South Carolina
Florida Coastal Plain	Myakka River State Park, southwest Florida
Gulf Coastal Plain	Solon Dixon, southern Alabama
Southern Appalachian Mountains	Green River, western North Carolina

[a] No thinning treatments applied here.

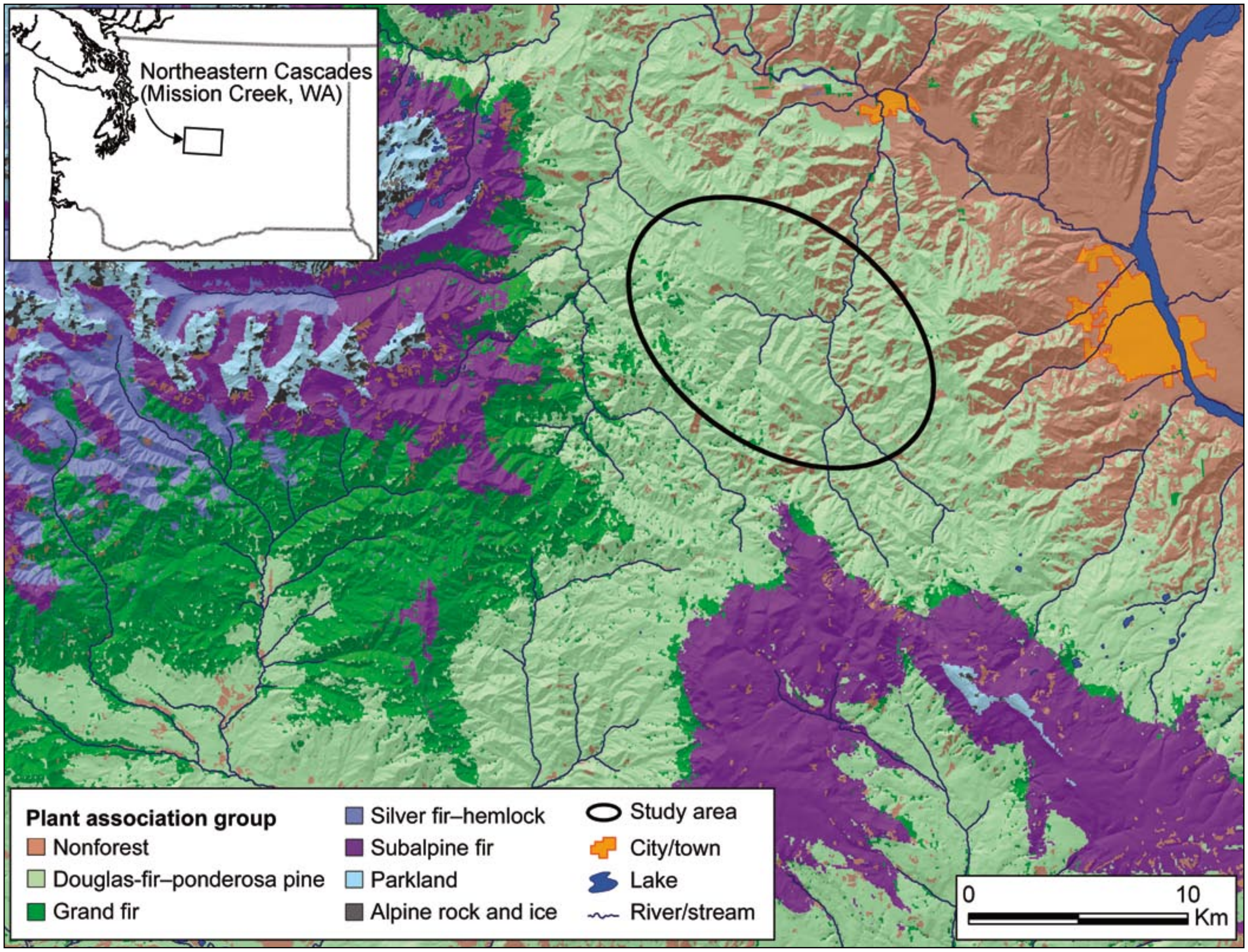


Figure 1-1—Location of Mission Creek study site in the northeastern Cascades of Washington.

Our goal with this paper is to document the physical and biological elements of dry forest found at the Mission Creek site. The study area includes 12 forested stands that encompass a representative range of dry forest conditions in the northeastern Cascade Range. In this paper, we describe site histories and environmental settings, experimental design, and field methods, and quantify the pretreatment composition and structure of vegetation, fuels, soils and soil biota, entomology and pathology, birds, and small mammals.

Study Site

The forests of the Mission Creek study site are representative of larger dry forest landscapes of the West.

The forests of the Mission Creek study site are representative of larger dry forest landscapes of the West. However, even within the dry forests, there is considerable ecological variability in terms of physical environment and biota. Mission Creek is within Ecological Subregion 11 (ESR 11) (fig. 1-2) of the interior Columbia River basin (Hessburg et al. 2000a), the primary scope of inference for the study results. The subregion extends along the lower elevation forested lands of the eastern Cascades of Washington and Oregon, and on an easterly arc to the Kettle Range

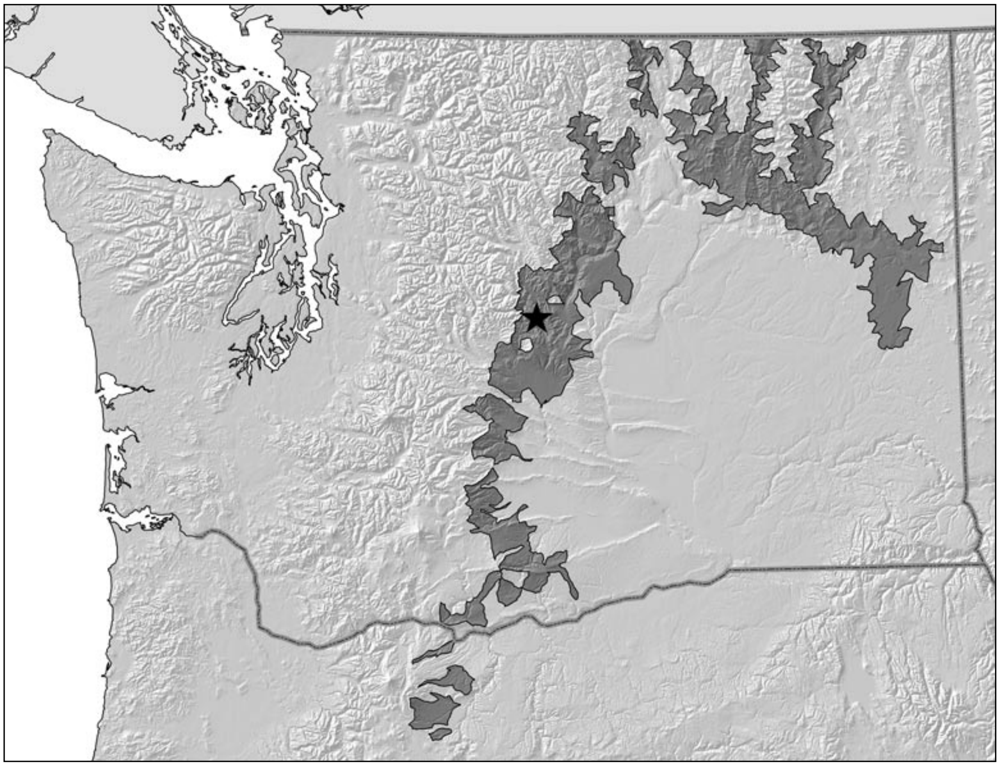


Figure 1-2—The primary scope of inference of the Mission Creek Fire and Fire Surrogate study (star) is best defined by dry forest Ecological Subregion 11 (dark grey) of the interior Columbia River basin (Hessburg et al. 2000a). Ecological Subregion 11 characterizes the lower montane forests of northeastern Washington and the east slope of the northern Cascade Range in Washington and parts of Oregon.

in northeastern Washington. The Mission Creek results combined with those from the Lubrecht FFS site in the northern Rocky Mountains of northeastern Montana and the Hungry Bob FFS site in the Blue Mountains of northeastern Oregon expand the FFS scope of inference to much of the dry forest in the interior Columbia River basin.

The northeastern Cascades of Washington have the most maritime climate within the interior Columbia River basin because of the proximity to Pacific Coast air masses (Hessburg et al. 2000a). Climatic data from Wenatchee (station 459074, 1931–2005) and Leavenworth 3S (station 454572, 1948–2005) east and west of Mission Creek for the last 60 years show average annual precipitation of 22.5 to 64.9 cm, most of which falls as snow between November and April, average maximum air temperatures of 16.3 to 16.9 °C, and average minimum air temperatures of 4.9 to 1.4 °C (Western Regional Climate Center online historical summaries: http://www.wrcc.dri.edu). Regional winds are generally westerly, and precipitation declines to the east and at lower elevation. Soil parent material is primarily nonglaciated sandstone intermixed with some shale and conglomerate (Tabor et al. 1982). Typical soil types (soil taxonomy great groups) found in the area include Haploxerepts, Haploxerolls, Argixerolls, and Haploxeralfs (USDA FS 1995).

The historical vegetation of these dry forests was dominated by ponderosa pine and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) (Agee 1993, 1994). In the lower Teanaway Valley, 25 km to the south, reconstructions of historical tree populations (ca. 1870–1890) from General Land Office records indicated that tree density and basal area were relatively low (Wright and Agee 2004) (table 1-2). Ponderosa pine was the dominant species in both the dry Douglas-fir and dry grand fir plant associations (sensu Daubenmire 1968). Other species, such as grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.), western larch (*Larix occidentalis* Nutt.), lodgepole pine (*Pinus contorta* Dougl. ex Loud.), western redcedar (*Thuja plicata* Donn ex. D. Don), and various hardwood species were minor components of the vegetation. Common understory species included pinegrass (*Calamagrostis rubescens* Buckl.), elk sedge (*Carex geyeri* Boott), bitterbrush (*Purshia tridentata*

Table 1-2—Historical vegetation of the lower Teanaway Valley, 25 km south of the Mission Creek units

			Relative density				Relative basal area			
Plant association group	**Density**	**Basal area**	**PIPO**	**PSME**	**ABGR**	**LAOC**	**PIPO**	**PSME**	**ABGR**	**LAOC**
	Trees/ha	*m^2/ha*	- - - - - - - - - - -	- - - -	- - - -	- - - - *Percent*	- - - -	- - - -	- - - -	- - - -
Dry Douglas-fir	73.3	15.3	76.5	22.1	1.4	—	72.9	26.4	0.7	—
Dry grand fir	49.3	13.4	61.4	35.4	—	2.5	70.5	28.2	—	1.2

PIPO = *Pinus ponderosa*, PSME = *Pseudotsuga menziesii*, ABGR = *Abies grandis*, LAOC = *Larix occidentalis*, — = not present.
Source: Wright and Agee 2004.

(Pursh) DC.), snowberry (*Symphoricarpos albus* (L.) Blake), spirea (*Spiraea betulifolia* Pall. var. *lucida* (Douglas ex Greene) C.L. Hitchc.), wood rose (*Rosa gymnocarpa* Nutt.), and ocean spray (*Holodiscus discolor* (Pursh) Maxim.). As on the Teanaway, Harrod et al. (1999) found a similar pre-1850 tree density (50 trees/ha) and basal area (16 m^2/ha) in Mission Creek with a higher relative density of ponderosa pine (90 percent).

Reconstructions of forest structure suggest that early successional stages covered a small proportion of the historical dry forest landscape. Hessburg et al. (2000b) estimated for a large study area north of Mission Creek that 11.3 percent of the landscape was in "stand initiation" structure, and for another large study area that encompassed Mission Creek, Agee (2003) estimated a range of 6.3 to 15 percent in seedling and sapling stages. Older forest totaled 41.2 percent in the Hessburg et al. (2000b) study and 38.4 to 63.5 percent for medium tree and larger structural stages in the Agee (2003) study. These proportions of larger, older structures would likely be higher if just the dry forest portions of those larger landscapes were considered.

Many studies in dry forests of the West have suggested that tree distributions tend to be clustered or aggregated rather than randomly or uniformly distributed (Cooper 1961, Thomas and Agee 1986). Locally this appears to be true, as Harrod et al. (1999) evaluated historical ponderosa pine forests in Mission Creek and found aggregated distribution of historical trees in patch sizes up to 15 m.

Low-severity fires had a major influence on the species composition and structure of historical dry forests across the West.

Low-severity fires had a major influence on the species composition and structure of historical dry forests across the West (Agee 1993). In the eastern Cascades near Mission Creek, Everett et al. (2000) found a mean fire-free interval of 6.6 to 7 years on two similar sites within 60 km of Mission Creek. These were composite fire intervals over roughly 582- to 644-ha sites. Fire-return intervals in the Swauk Valley, 25 km to the southwest of Mission Creek, averaged 6.2 years within polygons averaging 1096 ha (Schellhaas et al. 2007). In the lower Teanaway Valley, 40 km to the southwest of Mission Creek, Wright and Agee (2004) found an 18.8- to 20.6-year mean fire-free interval for dry Douglas-fir and dry grand fir plant associations, using a much smaller polygon size (0.5 to 2.5 ha). Most fires were late-season fires, some were above 10,000 ha, and larger fires were associated with drought years. Because of the high proportion of surviving trees with fire scars, most of these fires are assumed to have been of low severity (Hessburg et al. 2005).

Modern-era dry forests have been affected by many changes from the pre-Euro-American era (Hessburg and Agee 2003). Change agents acted to exclude fires, directly advance secondary succession, directly suppress fires, or some combination of these effects (Hessburg et al. 2005). Domestic livestock grazing beginning in the

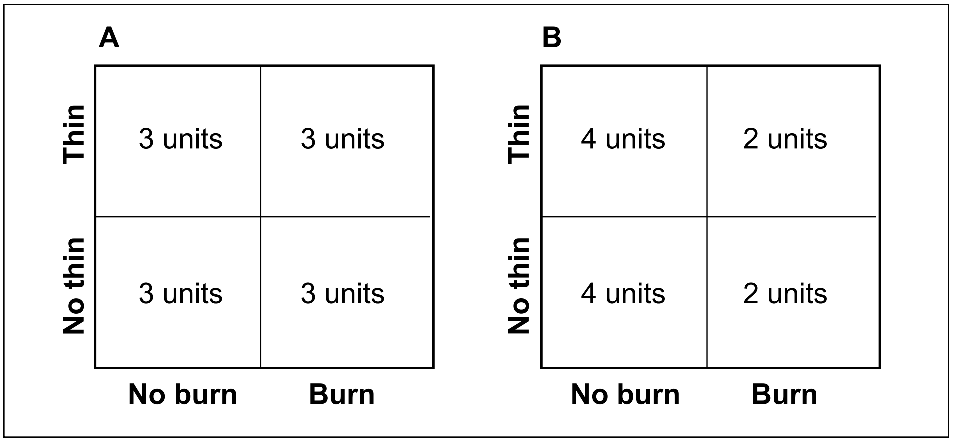
1870s acted to reduce flashy fuels that could carry fire across the landscape (Belsky and Blumenthal 1997). In Mission Creek, intensive sheep grazing was present by the 1890s and continued through the 1920s (USDA 1938). This watershed was a sheep driveway in spring and fall, and an estimated 30,000 sheep passed through the drainage twice a year. By the 1950s, the number of grazing animals had been reduced considerably (Holstine 1992). Currently, there is no active grazing by livestock on public lands in the watershed. Selective timber harvest began in Mission Creek in the late 1800s for railroad construction, mining, homesteading, and to supply boxes for an expanding fruit industry (Lolley 2005). From 1928 to 1938, the mill at Cashmere at the base of Mission Creek processed over 200 million board feet (472,000 cubic meters) a year (USDA SCS 1938). Selective harvest of the largest and best-form (clear bole) ponderosa pine occurred through the most accessible portions of the landscape. Active fire suppression during this period dramatically increased fire-free intervals (Everett et al. 2000, Wright and Agee 2004).

The resulting changes in forest structure are summarized by Hessburg et al. (2000b) and Hessburg and Agee (2003). More shade-tolerant species that once would have been removed by frequent, low-intensity fires filled in natural canopy gaps and gaps created by selective removal of large pines, average tree size declined, and multilayered canopies became common. In these dry forests, mixed- to high-severity fire regimes replaced the historical low-severity fire regime. The Rat Creek Fire (1994) and Fischer Fire (2004), directly adjacent to Mission Creek, are examples of this regime change.

The experimental design of the FFS project is a balanced, two-way design with three replications of each main factor: prescribed fire and low thinning.

General Methods

The experimental design of the FFS project is a balanced, two-way design with three replications of each main factor (fig. 1-3A). The first main factor is prescribed fire. The second is a surrogate for fire, which in our case was a low thinning. A total of 12 units out of a possible 30 were randomly chosen in the watersheds of Mission and Peshastin Creeks (fig. 1-4). The 30 units were selected with the criteria that the majority of the unit was in the *Pseudotsuga menziesii* series (Lillybridge et al. 1995), at least 90 percent of the unit was forested, none of the plant or animal species of concern were present (except Survey and Manage species under the Northwest Forest Plan), average slope was less than 50 percent, and the unit consisted of a relatively square or rectangular area of 10 ha or more. Treatments were randomly assigned, except that one unit initially designated a burn-only unit (Sand 19) was switched with a designated control unit (Spromberg) because of fire control concerns by ranger district fire personnel, who were responsible for conducting



A

	No burn	Burn
Thin	3 units	3 units
No thin	3 units	3 units

B

	No burn	Burn
Thin	4 units	2 units
No thin	4 units	2 units

Figure 1-3—Experimental design of the Fire and Fire Surrogate project. Each experimental treatment (prescribed fire or no prescribed fire, thinning or no thinning) was replicated three times in the initial (A) design. An unbalanced design (B) resulted when two burn units did not receive the prescribed fire treatment. This was resolved in 2006.

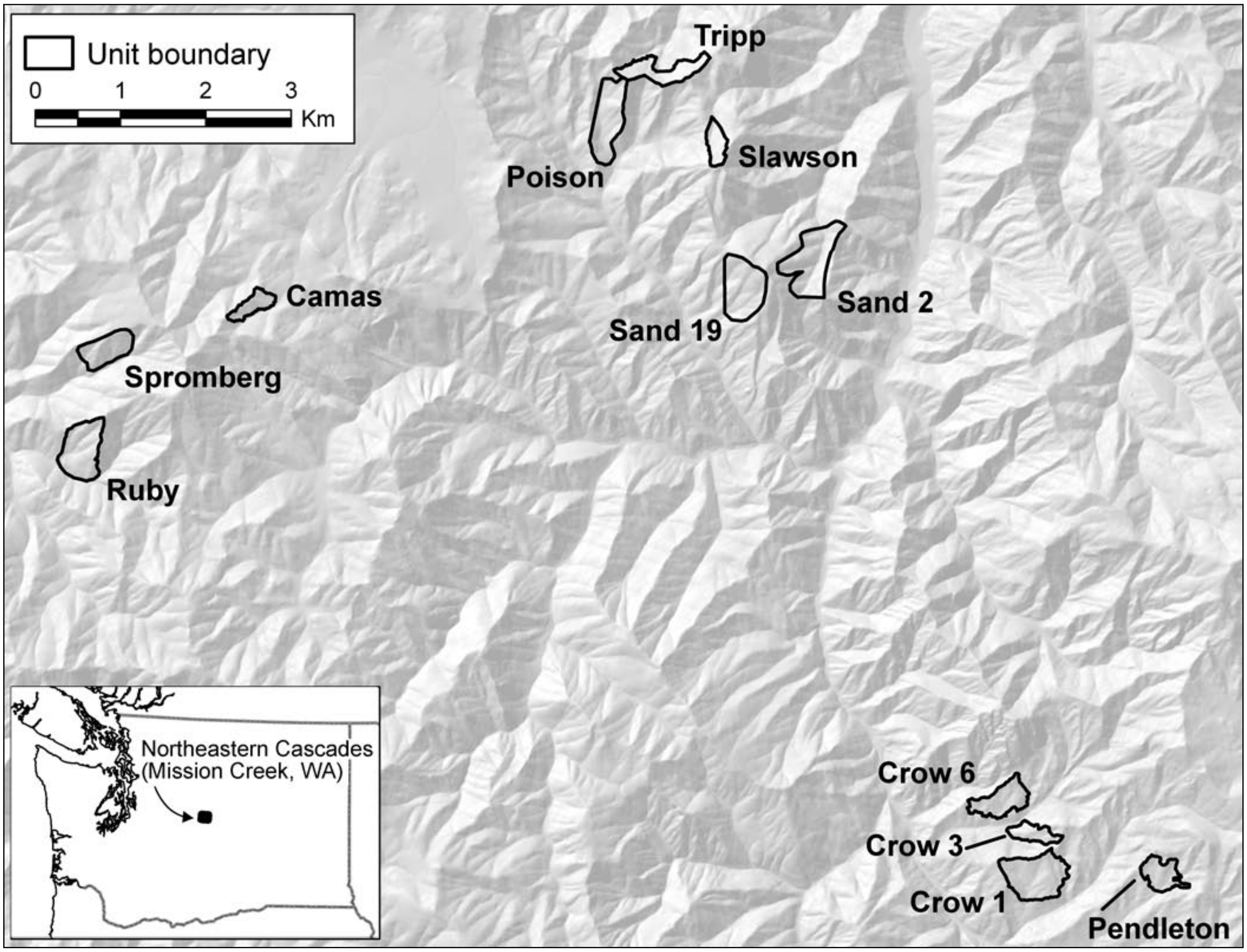


Figure 1-4—Location of the 12 experimental units at the Mission Creek study site.

the prescribed fires. Because of unanticipated events, two of the scheduled burns did not occur before the first posttreatment measurement, resulting initially in an unbalanced experimental design (fig. 1-3B). Burns were completed during 2006, rebalancing the design.

Each discipline developed unique methods to assess pretreatment conditions and the effect of treatment (Youngblood et al. 2005). A 40- by 40-m grid was established over each experimental unit. The fuels, soils, and bird sampling used different grid intersections across the entire matrix as plot centers to avoid concentrated disturbance at selected locations. The small mammal sampling was done on a 6- by 6-point interior portion of the grid, and the vegetation group used six 20- by 50-m rectangular "Daubenmire" plots to do the vegetation sampling in completely forested locations, augmented by four 20- by 20-m plots placed in nonforested locations. The entomology and pathology group used a 100-percent survey of trees across a 10-ha area of each unit.

This report summarizes the pretreatment conditions of the Mission Creek site. Although some units described herein are designated as control, thin, burn, or thin-burn units, all measurements reported were taken and analyzed before treatments were applied. Any "treatment" comparisons are for units aggregated by the final 2006 balanced design (fig. 1-3). Control units are Crow 3, Sand 19, and Sand 2. Thin units are Crow 1, Ruby, and Slawson. Burn units are Pendleton, Poison, and Spromberg. Thin-burn units are Camas, Crow 6, and Tripp.

Implementation of Treatments

Planning for the fire and surrogate (thinning) treatments began in 2000. The thin treatment was designed as a low thinning to reduce basal area so the residual stand would have a clumped distribution of roughly 15 m^2/ha, similar to that found in historical stands (Harrod et al. 1999). Because of access limitations and soils concerns, all thinning treatments were yarded by helicopter. Thin treatments were completed by spring 2003. Burn treatments were scheduled for autumn 2003, but an unusual, intense early rain so dampened the fuels that burning was delayed, and normal autumn rains then came shortly thereafter, postponing any burning until spring 2004. Unusual weather again prevailed. A late dry winter allowed substantial herbaceous growth ("greenup"), and when the spring prescribed fire season occurred, only four of the six units scheduled for burning were ignited and they burned lightly or patchily. One of the burn-only units (Sand 2) and one of the thin-burn units (Ruby) did not receive the planned burn treatment.

The resulting unbalanced design (fig. 1-3B) with four control (Crow 3, Pendleton, Sand 19, Sand 2), four thin (Crow 1, Crow 6, Ruby, Slawson), two burn (Poison, Spromberg), and two thin-burn (Camas, Tripp) units was used in the first posttreatment sampling during 2004-2005. Published results to date include effects of the first cycle of treatment on vegetation (Harrod et al. 2007) and fuels (Agee and Lolley 2006), and other disciplinary results are in preparation. Data on the economics of harvesting dry eastern Cascade forests, using a variety of yarding techniques, were also gathered as part of the FFS project, and will be reported separately.

The design was balanced by the addition of two more burns in spring 2006. A few changes in treatment allocations among stands were necessary to accommodate endangered species (northern spotted owl [*Strix occidentalis caurina*]) and prescribed fire control issues. The Pendleton unit was switched from a control unit to a burn-only unit, and Sand 2 from burn-only to control because of fire control and logistical issues in Sand 2. Crow 6 was randomly selected from the thin-only units to switch from a thin unit to a thin-burn unit, and the Ruby treatment was changed from thin-burn to thin-only because of spotted owl regulatory issues near Ruby. The resulting balanced design has three control units (Crow 3, Sand 19, Sand 2), three thin units (Crow 1, Ruby, Slawson), three burn units (Pendleton, Poison, Spromberg), and three thin-burn units (Camas, Crow 6, Tripp). Immediate remeasurement of these units was not possible owing to lack of funding. It is anticipated that this design will remain stable and serve as the template for future remeasurement of the Mission Creek FFS site.

The resulting balanced design has three control units, three thin units, and three thin-burn units. This design is anticipated to remain stable and serve as the template for future remeasurement of the Mission Creek FFS site.

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Chapter 2: Pretreatment Variation in Overstory and Understory Vegetation

Richy J. Harrod, Erich K. Dodson, Nicholas A. Povak, and David W. Peterson[1]

Introduction

Changes in forest vegetation as a result of fire exclusion, sheep and cattle grazing, and large-tree logging have been well documented for western forests (Agee 1998; Arno et al. 1997; Covington and Moore 1994; Everett et al. 2000; Harrod et al. 1999; Hessburg and Agee 2003; Hessburg et al. 2000, 2005). These changes in forest structure and composition have increased the hazard of large, high-severity wildfires (Agee 1994, Peterson et al. 2005, Scott and Reinhardt 2001, Weatherspoon and Skinner 1996), which lead to degraded water quality and costly losses of wildlife habitat, wood products, and residential buildings (Armstrong and Cumming 2003). In addition, current forests have increased vulnerability to many insects and diseases (Hessburg et al. 2005); have altered understory diversity, composition, and abundance (Covington et al. 1997, Hall 1977, Smith and Arno 1999); and generally have an overall deterioration in forest ecosystem integrity (Everett et al. 1996). A reduction in the current risks of uncharacteristically severe wildfires and insect outbreaks is desirable for local communities and many land managers. Before restoration activities are planned, it is important to understand the current variability in vegetation so that treatment effects can be appropriately interpreted.

Before restoration activities are planned, it is important to understand the current variability in vegetation so that treatment effects can be appropriately interpreted.

Current dry forest landscapes have become homogeneous in their composition and structure (Hessburg et al. 2005), but variable vegetation patterns at small scales still exist. Restoration treatments are designed to incorporate natural or historical fire regimes, which promote heterogeneous landscapes, reduce stand density, raise height to live crown, and lead to tree clumping at fine scales (Agee 1993, Harrod et al. 1999). However, restoration treatments are often implemented with minimal information of how current, local vegetation patterns and composition will respond to treatment. Thus, better information is needed about the pretreatment structural and compositional elements of dry forests, so that treatment effects can better be distinguished from vegetation variability. The national Fire and Fire Surrogates (FFS) study was designed to be a long-term research project to evaluate treatment alternatives, but it also provides an opportunity to quantify pretreatment structure and composition of forest vegetation.

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In this paper, we describe the pretreatment overstory and understory vegetation at the Mission Creek site, with the specific purpose of determining potential differences in vegetation variables among units that are assigned to specific treatments. The specific objectives for the pretreatment overstory measurements were to:

1. Quantify the pretreatment forest structure within treatment groups and the variability among treatment units.
2. Determine how biophysical factors affected initial forest characteristics.
3. Determine potential effects of pretreatment structural variability on subsequent analyses.

The specific objectives for the pretreatment understory measurements were to:

1. Describe the understory plant community, including variability among and within units, and test for significant pretreatment differences among units assigned to specific treatments.
2. Identify environmental variables that are associated with the understory community.
3. Examine the relationship between overstory trees and understory vegetation.
4. Compare pretreatment species cover and richness between sampling years (2000 or 2001).

Sampling of overstory and understory vegetation was conducted on a series of fixed-area plots that were permanently established within each treatment unit.

Methods

Sampling Design

Sampling of overstory and understory vegetation was conducted on a series of fixed-area plots that were permanently established within each treatment unit. Based on a preliminary analysis of an adjacent site, it was determined that six, 20- by 50-m plots would adequately capture the within-unit variability and provide an accurate estimate of vegetation characteristics. In summer 2000, the sample plots were nonrandomly located in areas of continuous forest vegetation and were stratified to include the most dominant plant associations within each unit. Sampling of understory and overstory vegetation occurred in 2000 and 2001.

Overstory Sampling and Analysis

Within each sample plot, all live and standing dead trees >7.62 cm diameter at breast height (d.b.h.) were identified; permanently numbered; and measured for diameter, total height, height to crown base (height from tree base to the intersection of the lowest live limb at the tree bole), bole scarring, and crown condition (e.g. Hawksworth 1977, Keen 1943). Diameters were measured using a diameter tape, and heights were estimated using a clinometer. Saplings (height >1.37 m, diameter

<7.62 cm) were tallied by species at the plot level. Canopy closure was estimated at the four cardinal directions from the center of the plot using a Lemmon Spherical Densiometer, Model-A. Canopy cover was then calculated by averaging the four measurements, multiplying its inverse by 1.04 and subtracting it from 100 (Lemmon 1956). Slope was measured at each plot using a clinometer, and aspect was measured using a compass.

Data were averaged up from the plot level to the experimental-unit level for all analyses unless otherwise noted (Hurlbert 1984). Live trees, snags, and saplings were analyzed separately. Folded aspects were calculated for each following plot, where aspect values range from 0 (NE slopes) to π (SW slopes). Heat load index (HLI) was calculated for each plot using equation 3 of McCune and Keon (2002), where slope, aspect, and latitude are used to estimate incident radiation. Values for HLI range from 0 to 1 with 0 representing cool NE-facing, shallow slopes and larger numbers representing warmer, SW-facing steep slopes. Canopy bulk density, canopy base height, and canopy fuel loading were calculated using the Crown Mass model included in the Fire Management Analyst (FMA) software package. Data were entered into the program at the plot level for these analyses.

A two-factorial analysis of variance was used to identify initial differences in forest structural components among treatments prior to the experiment. Unlike the understory analysis (see below), the experimental design was a balanced completely randomized design. Factors were burn (two levels: burn or no burn) and thin (two levels: thin or no thin). Least square means were used to compare significance between treatments, and the experiment-wise error rate was controlled using the Tukey-Kramer adjustment for multiple comparisons. Two plots were removed from all overstory analyses because they were outside of the fire line and were mistreated during the prescribed burn in 2004. For purposes of future integration of these data with subsequent analyses, these two plots were removed from the current analyses. A significance level of $P < 0.10$ was set prior to analyses.

Correlation analysis was conducted to determine significant relationships between overstory structural characteristics and environmental factors. Data were averaged to the plot level ($n = 70$), and Pearson's correlation coefficients were computed for each combination of variables. Significant correlations were based on $P < 0.10$.

Understory Sampling and Analysis

The 72 permanent 20- by 50-m plots used to sample trees were also used for sampling understory vegetation. Ten nested 5- by 10-m subplots placed in a continuous 10- by 50-m strip in the center of each plot were used to collect shrub cover.

Cover was ocularly estimated to the nearest percentage point. Also, twenty 1- by 1-m quadrats were nested in each plot in a stratified random fashion for sampling herbaceous vegetation. In each quadrat, all herbaceous species were inventoried and their cover ocularly estimated to the nearest percentage point.

Cover for each shrub and herbaceous species (forbs and graminoids) were averaged to the plot level for each species (n = 72). Average cover and richness (number of species on each plot) were then calculated for each site. Species were grouped into shrubs, forbs, and graminoids (Collins et al. 2007, Metlen and Fielder 2006). Understory variability in cover and richness was partitioned into within- and among-site components using a Hierarchical Linear Model in the SAS (Littell et al. 1996) Proc Mixed procedure (Singer 1998). The simplest Unconditional Means Model was fit with the overall mean as the fixed effect and two variance components—one within units and one among units (Raudenbush and Bryk 2002, Singer 1998). Variability in understory plant cover and richness was partitioned into within- and among-site components for each life form (shrub, forb, and graminoid) as well as overall cover and richness.

Pretreatment differences in units assigned to treatments were evaluated using a general linear model in SPSS (2001). Prior to analysis, all data were averaged to the site level where the treatments were to be applied later (n = 12 units). Thinning (yes or no) and burning (yes or no) were treated as fixed factors in the model applied in factorial combination. As posttreatment understory data were collected in 2005 and two units were not burned until 2006, the analysis was unbalanced in regard to burning (eight units unburned, four units burned), but balanced in regard to thinning. Separate tests were performed for richness and cover of each life form and overall richness and cover of all understory species. Marginal means were used to estimate the mean value and standard error for each of the four treatments with each variable for interpretation of the statistical tests.

The influence of environmental variables on community composition was assessed using nonmetric multidimensional scaling (NMS) (McCune and Grace 2002). We used the slow and thorough autopilot mode in PC-ORD (McCune and Mefford 1999) with a stepdown from six dimensions, with a random starting configuration and Sorenson distance measure. A Monte Carlo test with 250 randomizations was performed to determine the likelihood of obtaining as good a solution by chance. All species (including those that were uncommon) were included in the analysis. The final NMS solution had three axes, and correlations with environmental variables were calculated for each axis. Those with an r^2 greater than 0.2 were plotted on the graphs with site scores.

Relationships between overstory and understory variables were assessed using Pearson correlations in SPSS (2001). All data were averaged to the plot level (n = 72) for these analyses. Four measures of overstory trees, basal area, trees per hectare, canopy cover, and percentage of trees per hectare that are ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) were correlated with understory richness and cover for each life form and overall.

Total understory cover and richness were averaged to the site level for each year of pretreatment sampling (2000 or 2001) to determine if obvious trends between years were evident. Means and standard errors for cover and richness were calculated for each site in each year to look for systematic year biases. One site (Sand 19) had three plots sampled in each year. Therefore, the three plots sampled in each year were considered separately.

Diameter distributions for each unit were generally bell-shaped indicating an even-sized structure.

Results and Discussion

Overstory Structure and Composition

Forest density and stocking were similar among the study units prior to the experiment (table 2-1). Live tree density averaged 589 (standard error [SE] = 52) trees/ha, and basal area averaged 33.2 (SE = 1.5) m^2/ha over all treatment units. Diameter distributions for each unit were generally bell-shaped indicating an even-sized structure; however, the distribution of tree diameters on higher elevations

Table 2-1—Pretreatment structural characteristics for each treatment combination at the Mission Creek study site

	Control		Burn		Thin		Thin-Burn	
Structural characteristic	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**
Live tree density (trees/ha)	593	104	478	79	627	125	659	135
Basal area (m^2/ha)	33.5	0.5	31.9	5.9	34.5	2.9	33.1	2.1
Stand density index	616	16	572	97	643	42	626	43
Quadratic mean diameter (cm)	28.8	1.5	29.9	0.1	27.7	2.5	26.2	1.7
Overstory ponderosa pine (% of trees)	22.2	15.2	56.5	20.0	43.8	26.4	40.7	28.2
Canopy cover (%)	87.6	1.8	79.1	4.4	82.4	7.9	87.7	6.3
Canopy height (m)	18.7	0.7	18.9	0.5	19.1	0.1	17.8	0.1
Canopy base height (m)	3.9	0.6	3.8	0.7	4.6	0.6	4.2	0.7
Canopy bulk density (kg/m^3)	0.077	0.002	0.067	0.008	0.080	0.007	0.078	0.006
Snag Density (snags/ha)	58.3	11.1	49.2	13.8	51.1	17.5	30.6	11.3
Snag basal area (m^2/ha)*	1.4ab	0.4	1.8ab	0.3	2.5b	0.3	0.8a	0.4

SE = standard error, n = 3.

* No significant differences were found between treatments for forest structural components, with the exception of snag basal area. For this variable, cells with different letters are significantly different ($p < 0.1$), with analyses based on post hoc pairwise comparisons.

units tended to be slightly irregular (i.e., Ruby and Camas). Units dominated by Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) had higher densities of understory or subcanopy trees, particularly in the 10- to 20-cm d.b.h. class (fig. 2-1). Density, basal area, and stand density index (SDI) were somewhat lower on burn-only units as compared to the other treatments, but the differences were not significant. Tree density was higher on cooler, NE-facing units and on steeper slopes (tables 2-2 and 2-3). These sites are generally less water-limited, and dense understory tree layers often form under these conditions. Basal area was not correlated with either aspect or

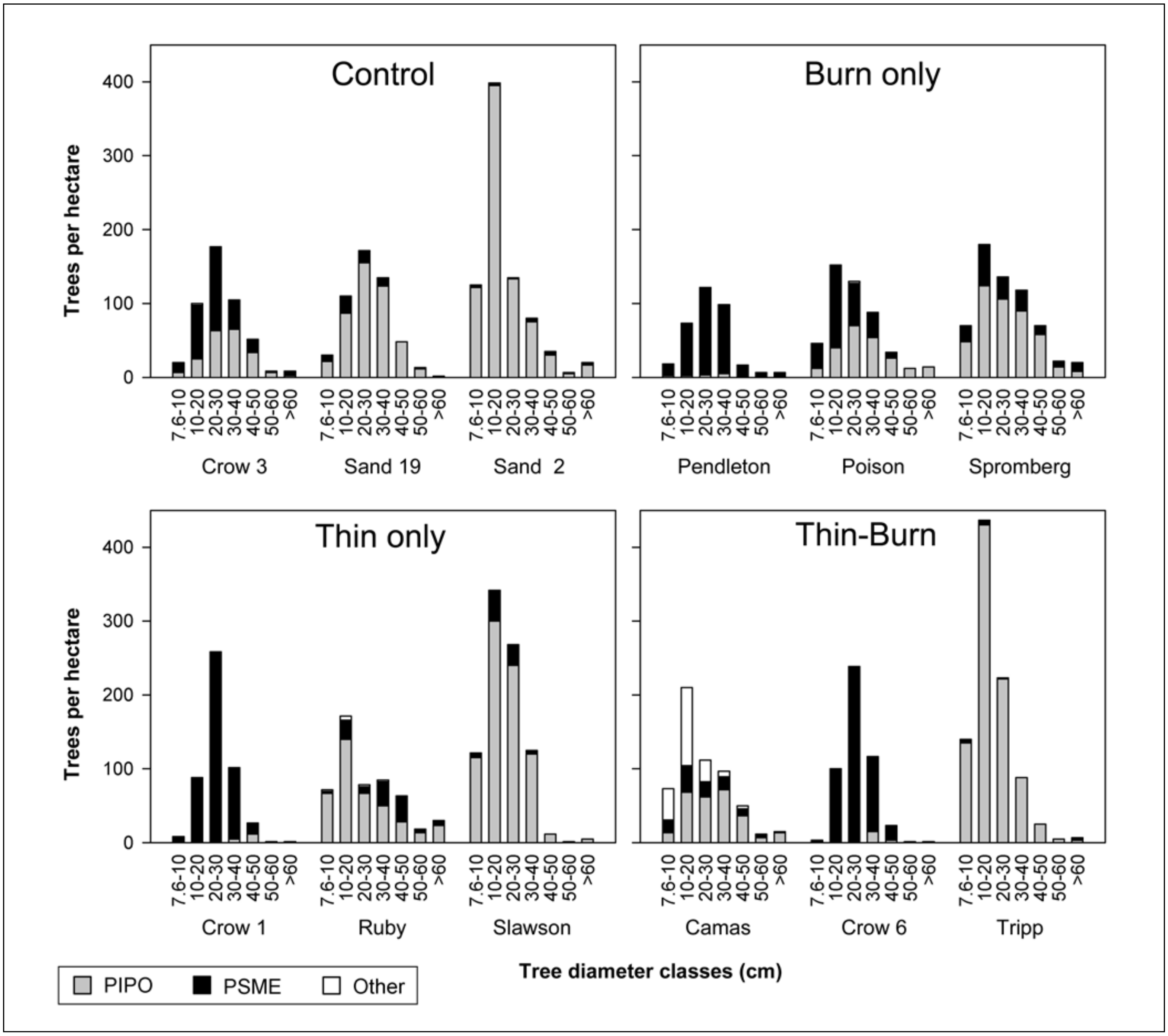

Figure 2-1—Diameter distributions for live trees located on treatment units within the Mission Creek study site. Units are arranged by treatment based on a balanced design with three replications per treatment combination. PIPO = *Pinus ponderosa*; PSME = *Pseudotsuga menziesii*.

HLI, most likely because the large number of small-diameter trees on the cooler sites does not contribute appreciably to basal area. Quadratic mean diameter was similar among treatments and averaged 28.1 (SE = 0.8) cm. Average diameters were larger on drier units where densities of small understory trees were lower.

Douglas-fir represented over half of the trees on experimental units and ponderosa pine made up >40 percent (fig. 2-1). Ponderosa pine dominance was somewhat lower on control units and higher on thin-only units, but there were no significant differences between treatments. Although there was much variability in species

Douglas-fir represented over half of the trees on experimental units and ponderosa pine made up >40 percent.

Table 2-2—Treatment unit descriptions, including treatment applied (planned treatment in parentheses), topographic attributes, and forest structural characteristics for the Mission Creek study sites

Treatment unit	Treatment description	Mean elevation	Mean slope	Folded aspect	Heat load index	Tree density	Tree basal area
		Meters	*Percent*			*Trees/ha*	*(m^2/ha)*
Crow 3	Control	747	38	2.17	0.87	470 (48)	32.5 (1.9)
Sand 19	Control	780	43	2.46	0.93	510 (93)	34.0 (3.1)
Sand 2	Control	683	58	1.40	0.70	800 (129)	34.1 (1.6)
Pendleton	Control (burn only)	841	16	2.59	0.91	342 (70)	22.7 (2.3)
Poison	Burn only	768	40	2.50	0.90	476 (77)	30.2 (2.8)
Spromberg	Burn only	848	57	2.36	0.90	616 (48)	42.7 (3.5)
Crow 1	Thin	738	21	2.18	0.90	487 (54)	29.0 (2.3)
Ruby	Thin	975	43	2.60	0.94	518 (104)	38.7 (4.9)
Slawson	Thin	838	35	1.39	0.81	875 (132)	44.6 (3.4)
Crow 6	Thin (thin-burn)	718	26	2.04	0.84	568 (98)	34.1 (4.9)
Camas	Thin-burn	1097	43	1.30	0.73	485 (38)	29.0 (2.4)
Tripp	Thin-burn	765	67	0.79	0.57	925 (116)	36.1 (2.6)

Note: Numbers in parentheses for tree density and basal area are standard errors.

Table 2-3—Pearson's correlation coefficients (r) for basic overstory structural and compositional components and biophysical environment

Variable	Slope (percent)	Folded aspect	Heat load index	Tree density (trees/ha)	Pine percentage of trees	Basal area (m^2/ha)	Mean diameter (cm)	Canopy cover (percent)	Sapling density (trees/ha)	Snag density (snags/ha)
Slope (percent)	1.00	**-0.31**	**-0.46**	**0.28**	**-0.64**	0.21	-0.15	**0.26**	-0.14	0.08
Folded aspect	**-0.31**	1.00	**0.92**	**-0.47**	**0.37**	0.00	**0.55**	**-0.33**	**-0.32**	-0.19
Heat load index	**-0.46**	**0.92**	1.00	**-0.39**	**0.38**	0.07	**0.50**	**-0.29**	-0.23	-0.11
Tree density (trees/ha)	**0.28**	**-0.47**	**-0.39**	1.00	**-0.44**	**0.50**	**-0.76**	**0.39**	**0.46**	**0.44**
Overstory ponderosa pine (percentage of trees)	**-0.64**	**0.37**	**0.38**	**-0.44**	1.00	**-0.44**	0.10	**-0.59**	-0.18	-0.18
Basal area (m^2/ha)	0.21	0.00	0.07	**0.50**	**-0.44**	1.00	0.09	**0.44**	0.07	**0.36**
Mean diameter (cm)	-0.15	**0.55**	**0.50**	**-0.76**	0.10	0.09	1.00	-0.10	**-0.39**	**-0.29**
Canopy cover (percent)	**0.26**	**-0.33**	**-0.29**	**0.39**	**-0.59**	**0.44**	-0.10	1.00	0.09	0.24
Sapling density (trees/ha)	-0.14	**-0.32**	-0.23	**0.46**	-0.18	0.07	**-0.39**	0.09	1.00	0.05
Snag density (snags/ha)	0.08	-0.19	-0.11	**0.44**	-0.18	**0.36**	**-0.29**	0.24	0.05	1.00

Note: Correlation analyses were conducted at the plot level. Bold-faced type indicates statistically significant correlations ($p < 0.05$).

composition, there were no significant differences between treatment combinations (table 2-1). The lack of significance is most likely due to the large standard errors associated with ponderosa pine dominance, which represents high within-treatment variability (table 2-1). In general, each treatment combination included one unit dominated by ponderosa pine, one by Douglas-fir, and one by mixed conifers (fig. 2-1). Douglas-fir dominance was higher on cooler, NE-facing units and on steeper slopes, and ponderosa pine dominance increased on warmer, SW-facing slopes. Other species observed in the experiment include grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.) and western larch (*Larix occidentalis* Nutt.), which generally occurred at higher elevations (>900 m).

The overstory composition at the Mission Creek site is somewhat different than other dry forest sites in the FFS network. The Southwest Plateau site in northern Arizona is more dominated by ponderosa pine than Mission Creek, largely owing to warmer and drier conditions (Fulé et al. 2001b). The Blodgett site in the southern Sierra Nevada is more overstory-species rich with higher tree densities than Mission Creek (Stephens and Moghaddas 2005) and the Lubrecht site in western Montana has a lodgepole pine (*Pinus contorta* Dougl. ex Loud.) component (Metlen and Fiedler 2006), which is missing from Mission Creek. The overstory at our site is probably most similar to Hungry Bob in eastern Oregon, although our climate is slightly more maritime as opposed to continental conditions in eastern Oregon (Youngblood et al. 2006). These differences in overstory structure make the Mission Creek site important to the overall understanding of treatment responses in a range of dry forest types.

Canopy cover averaged >80 percent across all treatment units.

Canopy cover averaged >80 percent across all treatment units, and there were no significant differences between treatments. Canopy cover was higher on Douglas-fir-dominated units and increased with higher tree densities. Canopy bulk density (CBD) averaged 0.071 (0.005) kg/m^3 and was statistically similar among treatments. Canopy base height (CBH), a measure of the vertical distance from the forest floor to the lowest point in the canopy where CBD is equal to 0.011 kg/m^3 (Reinhardt and Crookston 2003), averaged ~4 m on all treatment units and was not significantly different among treatments. These values are comparable to those reported by Stephens and Moghaddas (2005) for mixed-conifer forests of the Sierra Nevada, which were found to be highly susceptible to crown fires modeled under 90th- and 97th-percentile weather conditions. Similar results were found in ponderosa pine forests of northern Arizona prior to restoration treatments (Fulé et al. 2001a). Further, recent large-scale high-severity fires have burned in the forests surrounding Mission Creek, which suggests that our study area is susceptible to such a disturbance.

Sapling densities were also similar across all treatments prior to the experiment and averaged 235 per ha (range 32 to 405 per ha). There were no significant differences in sapling densities between treatments; however, sapling densities were highly variable within treatment with densities increasing on cooler, moister units. On average, about two-thirds of the saplings were Douglas-fir (range 9 to 97 percent). Grand fir saplings were present on higher elevation units, particularly at the highest elevation site where more than 300 grand fir saplings were tallied per hectare. The sapling layer at Mission Creek was most likely not present historically as it was suppressed by frequent, low-intensity surface fires (Harrod et al. 1999). This layer adds structural complexity to the forest by increasing canopy layering and lowering CBH; both of which increase crown fire susceptibility (Peterson et al. 2005).

Snag density (snags per hectare), but not snag basal area, was statistically similar across units prior to treatments.

Snag density (snags per hectare), but not snag basal area, was statistically similar across units prior to treatments (table 2-1, fig. 2-2). Snag density was higher in denser stands, and most snags were <20 cm d.b.h. The majority of snags were ponderosa pine on most units, with the exception of the higher elevation units and units heavily dominated by Douglas-fir (fig. 2-2). Snag basal area was significantly higher on thin-only units compared to thin-burn units, which had one-third the snag basal area of thin-only units.

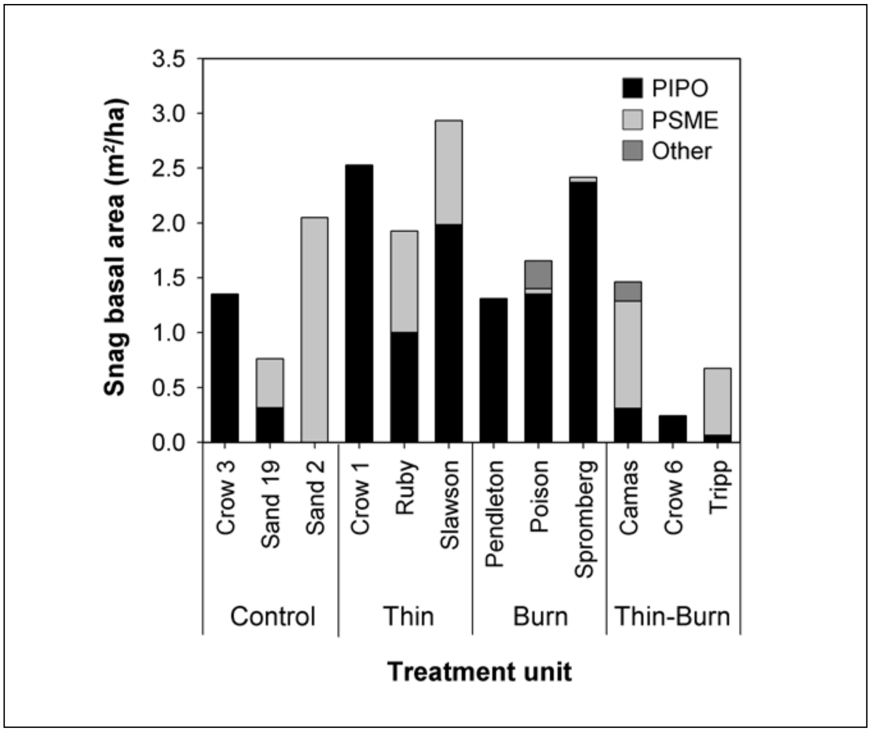

Figure 2-2—Basal area of snags on treatment units within the Mission Creek study site. PIPO = *Pinus ponderosa*; PSME = *Pseudotsuga menziesii*.

Understory Structure and Composition

A total of 124 species were identified at our 12 units, but the majority of these species were not abundant.

The majority of species found at our study site had low cover and frequency prior to any treatment application, which is similar to other forest ecosystems (Dodson et al. 2007, Peterson and Reich 2008, Stohlgren et al. 2005). A total of 124 species were identified at our 12 units, but the majority of these species were not abundant. The five most frequent species were all graminoids and shrubs; *Amelanchier alnifolia* (Nutt.) Nutt. ex M. Roem., *Carex geyeri* Boott, *Calamagrostis rubescens* Buckley, *Rosa* spp., and *Symphoricarpos albus* (L.) S.F. Blake. The most frequent forb was *Osmorhiza berteroi* DC., which was the sixth most frequent species overall, and the forb with the highest cover was *Arnica cordifolia* Hook., which had the ninth highest cover among all species. Only 10 species had more than 1 percent cover each and made up over 78 percent of total understory cover. Eighty-three species had less than 0.1 percent cover each, making up just over 3 percent total understory cover. A similar pattern was also apparent with frequency. There were only 15 species that occurred on more than half the plots, and 67 species occurred on less than one-tenth of the 72 plots.

Shrubs and graminoids made up the majority of understory cover across all units, whereas forbs made up the majority of understory richness (fig. 2-3). There was considerable variability among units in total cover and cover for each life form (fig. 2-3a). Average vascular plant cover for the 12 units ranged from 23.7 to 78.7

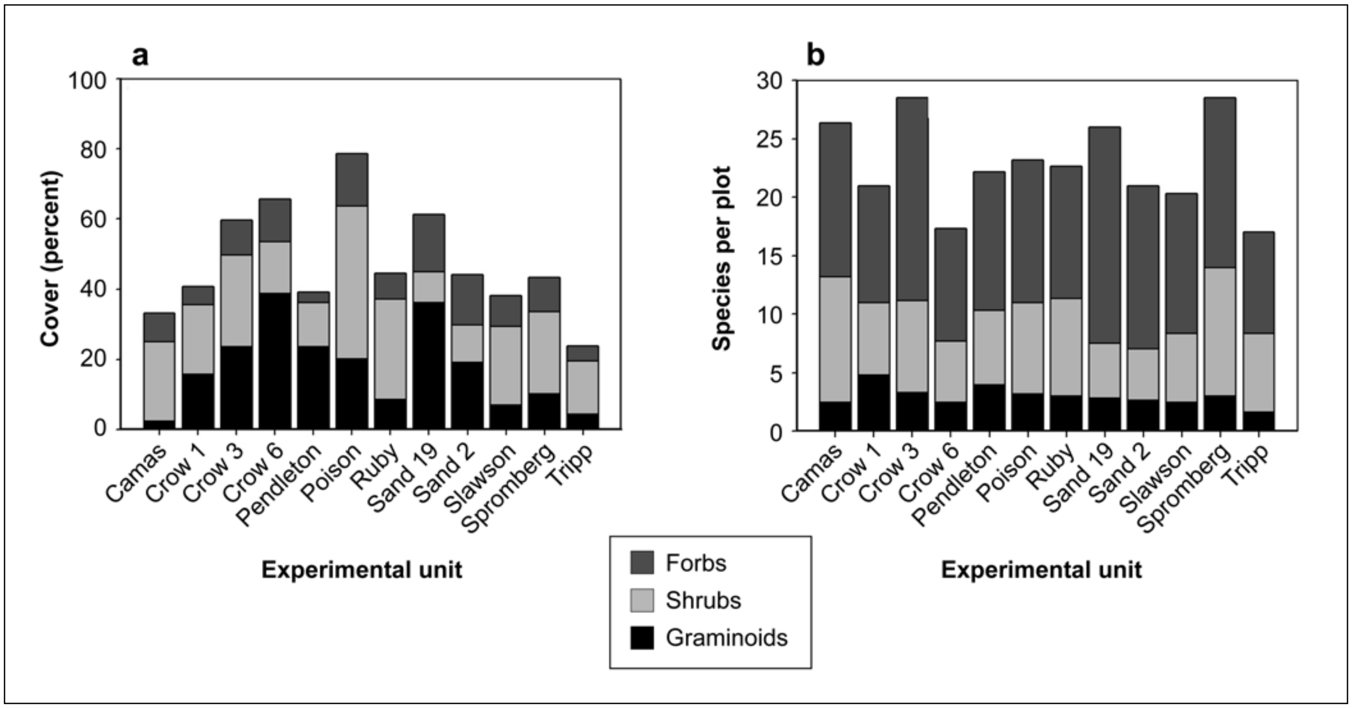

Figure 2-3—Mean cover (a) and richness (b) for each life form on the 12 Mission Creek treatment units.

percent. Each life form also differed among units, ranging from 2.4 to 38.7 percent cover for graminoids, from 3 to 16.3 percent cover for forbs, and from 8.8 to 43.5 percent shrub cover. Richness also showed considerable variability among units, although the variability was considerably less than for cover (fig. 2-3b). Among the 12 units, total richness per plot ranged from 17.3 to 29.8, graminoid richness from 1.7 to 4.8, forb richness from 8.6 to 18.5, and shrub richness from 4.3 to 11.

Despite the considerable variability among units in cover and richness for each life form, much of the variability was still within units. Partitioning the variance in cover into within- and among-site components revealed that most of the variability in cover for shrubs and forbs was within units (fig. 2-4). Only graminoid cover varied more among units than within units. Richness was about roughly equally divided into within and among unit components for each of the three life forms (fig. 2-4).

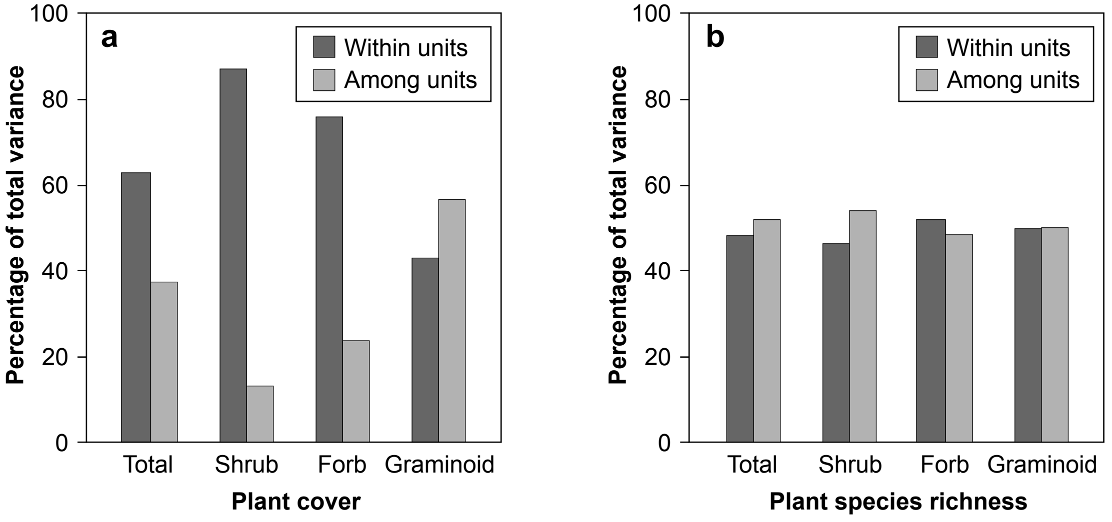

Figure 2-4—Diversity partitioning into within- and among-unit components for (a) cover and (b) richness of the Mission Creek site.

The understory vegetation at the Mission Creek units has notable similarities with and differences from the understory in other dry forests of the Western United States. Understory cover at our study site was dominated by shrubs and graminoids, whereas forbs and shrubs made up the majority of cover in a dry Montana mixed-conifer forest (Metlen and Fiedler 2006). Graminoids and forbs are more dominant in ponderosa pine forests in the Blue Mountains of eastern Oregon (Metlen et al. 2004, Youngblood et al. 2006) and in northern Arizona (Bailey and Covington

2002). As on our site, high forb richness is a common pattern throughout dry forests of the Western United States (Metlen and Fiedler 2006, Wayman and North 2007).

There were numerous significant pretreatment differences in understory vegetation among treatment units.

Comparisons among assigned treatments—

There were numerous significant pretreatment differences in understory vegetation among treatment units (table 2-4). Total understory cover averaged 56.5 percent in the unthinned units (burn-only and control) and was significantly greater than the 38.0 percent total cover in the thinned units (thin-burn and thin-only) (P = 0.0562). There was also a significant difference in species richness between the thinned and unthinned units (P = 0.0879), with the unthinned units averaging almost five more species per plot than thinned units (table 2-4).

Table 2-4—Means (and standard errors) of understory vegetation conditions within and among treatment groups prior to treatment

	Mean treatment value				Pretreatment effect significance		
Variable	**Control**	**Burn-only**	**Thin-only**	**Thin-burn**	**Thin**	**Burn**	**Thin × Burn**
Total understory cover (%)	51.44 (5.52)	61.62 (17.09)	47.38 (6.23)	28.69 (5.00)	0.056	NS	NS
Total understory richness	25.37 (1.78)	27.00 (2.83)	21.13 (1.15)	22.17 (4.83)	0.088	NS	NS
Shrub cover (%)	14.50 (3.93)	33.34 (10.13)	21.32 (2.9)	18.88 (3.68)	NS	NS	0.063
Shrub richness	5.79 (0.81)	9.42 (1.58)	6.38 (0.69)	8.67 (2.00)	NS	0.029	NS
Forb cover (%)	10.89 (2.93)	12.39 (2.61)	8.41 (1.44)	6.14 (2.09)	NS	NS	NS
Forb richness	15.42 (1.53)	13.33 (1.17)	10.75 (0.55)	10.92 (2.25)	0.040	NS	NS
Graminoid cover (%)	25.59 (3.66)	15.18 (4.94)	17.5 (7.33)	3.42 (1.00)	NS	0.089	NS
Graminoid richness	3.21 (0.30)	3.08 (0.08)	3.21 (0.55)	2.08 (0.42)	NS	NS	NS

Pretreatment conditions analysis of variance (using unbalanced design produced by actual, not planned, treatments).
Note: NS = No statistically significant differences among treatment groups (P > 0.1)

There were also pretreatment differences among assigned treatment units for each life form (table 2-4). The interaction of thinning and burning treatment assignments was significant for shrub cover (P = 0.0630), suggesting that units assigned to each treatment had different values for shrub cover (table 2-4). Shrub richness was significantly higher at the units that were slated to receive burn treatments than the units not designated for a burning treatment (P = 0.0291). Forb cover did not differ significantly among units assigned to treatments, but forb richness was significantly lower on thinned units than on unthinned units (P = 0.040). Similarly, graminoid cover was significantly lower in the burned units than in the unburned units (P = 0.089; table 2-4) reflecting especially high graminoid cover in control units and low graminoid cover in thin-burn units.

High variability among sites has also been observed in southwestern dry conifer forests (Abella and Covington 2006, Fulé et al. 2005, Gildar et al. 2004). The ubiquity of pretreatment differences in understory vegetation suggests that site characteristics, including the biophysical environment and the disturbance history of the site, may significantly influence vegetation composition. This highlights the need for pretreatment data to draw reasonable conclusions in these highly variable forests. Partitioning the variance into within- and among-unit components also revealed that much of the variability in these forests occurs at smaller spatial scales (i.e., much of the variabilities in cover and richness were within the 10-ha units). This high pretreatment variability among and within units contributes to the biodiversity of the Mission Creek dry forest ecosystem.

Site characteristics, including the biophysical environment and the disturbance history of the site, may significantly influence vegetation composition.

Environmental influences on understory—
Units differed considerably prior to treatment application in their understory communities as evidenced by the separation in ordination space (fig. 2-5). The nonmetric multidimensional scaling (NMS) ordination resulted in three axes that explained 83 percent of the variation in the original data set. The final solution with three dimensions was much better than expected by chance ($P = 0.004$). The third axis explained by far the most variation in the understory community (52 percent), and was positively correlated with trees per hectare and elevation while negatively correlated with the percentage of trees that were ponderosa pine and graminoid cover. The first axis explained 16 percent of the variation in the original data set, and was positively correlated with shrub cover (fig. 2-5a). The second axis explained 15 percent of the variation and was not significantly correlated with any measured environmental variables. Pretreatment canopy cover and basal area were not strongly correlated ($r^2 < 0.2$) with any of the axes in this study.

Understory vegetation showed significant correlations with overstory variables (table 2-5). In general, trees per hectare, basal area, and/or canopy cover were negatively correlated with understory cover and richness. However, tree density had a much stronger negative effect on understory vegetation than basal area or canopy cover (table 2-5). Graminoids had more significant correlation with overstory structure variables than forbs and shrubs. Despite being negatively correlated with tree density, basal area, and canopy cover, graminoid cover and richness were positively correlated with a ponderosa pine overstory. Shrubs were not significantly correlated with a ponderosa pine overstory, but forb richness was reduced with an increased presence of ponderosa pine in the overstory (table 2-5).

The understory vegetation in this study was influenced by environmental conditions and overstory trees. Elevation played an important role in dictating understory

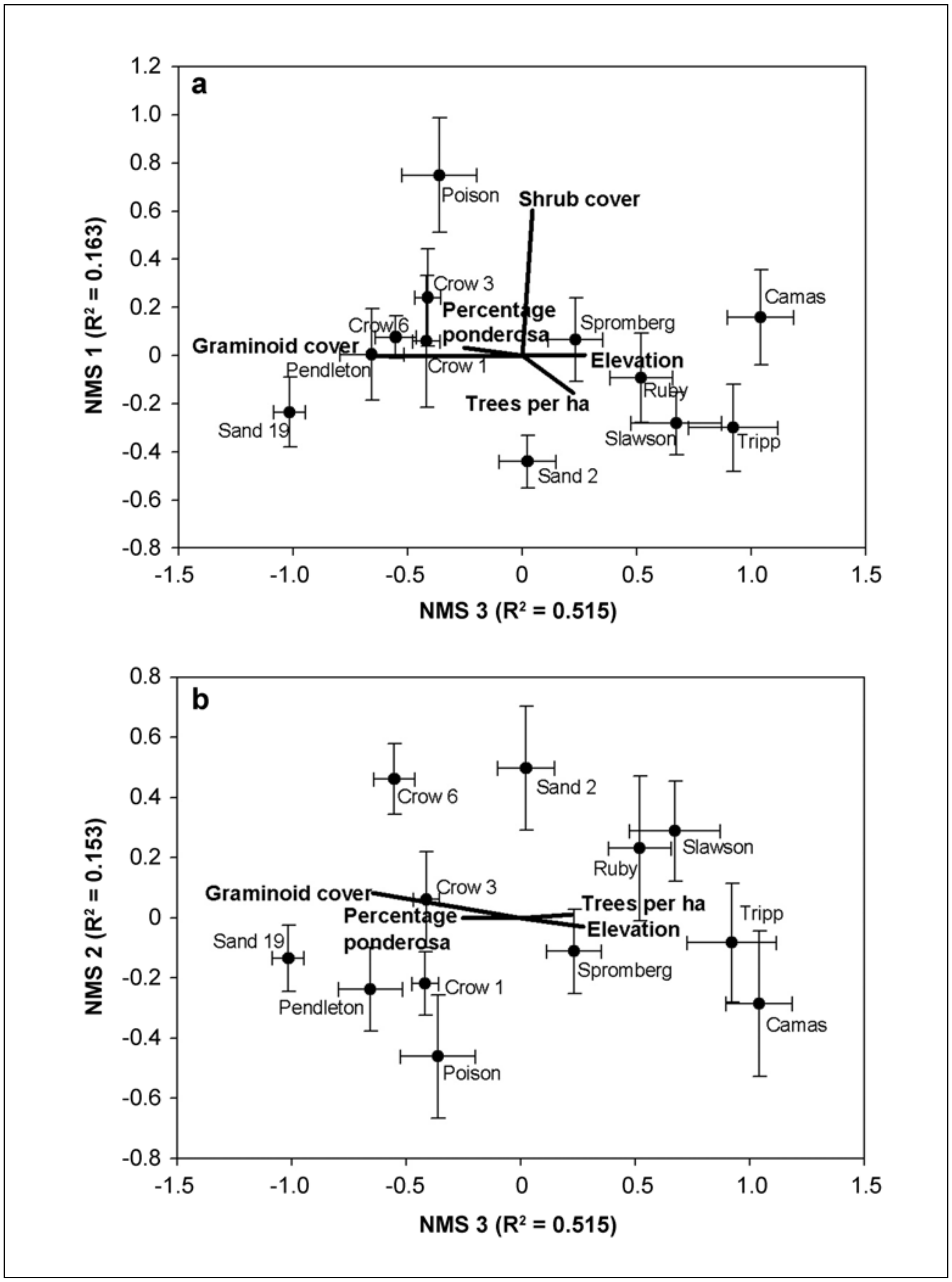

Figure 2-5—Nonmetric multidimensional scaling (NMS) ordination of Mission Creek understory species data prior to treatment with environmental variables with >0.2 correlation with axes 1 and 3 (a) and 2 and 3 (b).

Table 2-5—Relationships between overstory stand structure and understory vegetation characteristics prior to treatment, as indicated by Pearson correlation coefficients (r) and associated significance levels (P) in parentheses

	Tree basal area (m^2/ha)		Tree density (trees/ha)		Tree canopy cover (percent)		Ponderosa pine (percent)	
Understory vegetation characteristic	**r**	**(P)**	**r**	**(P)**	**r**	**(P)**	**r**	**(P)**
Understory cover (percent)	**-0.24**	**(0.04)**	**-0.45**	**(< 0.001)**	-0.10	(0.44)	**0.20**	**(0.09)**
Understory richness	-0.09	(0.46)	**-0.38**	**(0.001)**	**-0.22**	**(0.09)**	-0.13	(0.26)
Shrub cover (percent)	-0.02	(0.86)	**-0.24**	**(0.04)**	-0.02	(0.89)	-0.05	(0.66)
Shrub richness	**0.20**	**(0.09)**	-0.14	(0.24)	-0.08	(0.54)	-0.13	(0.29)
Forb cover (percent)	-0.15	(0.22)	-0.17	(0.16)	0.01	(0.97)	-0.07	(0.56)
Forb richness	-0.14	(0.23)	**-0.26**	**(0.03)**	-0.06	(0.63)	**-0.26**	**(0.03)**
Graminoid cover (percent)	**-0.27**	**(0.02)**	**-0.30**	**(0.01)**	-0.13	(0.31)	**0.41**	**(< 0.001)**
Graminoid richness	**-0.32**	**(0.01)**	**-0.37**	**(0.002)**	**-0.50**	**(< 0.001)**	**0.52**	**(< 0.001)**

Note: Ponderosa pine (percent) is the percentage of overstory trees that are ponderosa pine (tree density basis). Bold-faced type indicates statistically significant correlations ($P < 0.10$).

composition (fig. 2-5), which has also been documented in dry coniferous forests of the Southwestern United States (Fischer and Fulé 2004). In general, overstory trees had negative effects on understory species (table 2-5), a pattern that has been well documented for dry coniferous forests (Riegel et al. 1992, 1995). Interestingly, graminoid cover and richness were positively associated with stands that had higher proportions of ponderosa pine. Tree density had much stronger negative effects on the understory community than basal area or canopy cover. In dry forests, trees may reduce understory growth more by competing for belowground nutrients than by shading (Riegel et al. 1992). Units with higher tree densities were the result of many trees in smaller size classes (fig. 2-1), which may have a larger effect on understory species than basal area or canopy cover. This suggests that treatments should focus more on reducing stand density (trees per hectare) than on basal area to benefit understory species.

Treatments should focus more on reducing stand density than on basal area to benefit understory species.

Effects of sample year—

There was a large amount of unit-to-unit variability in understory cover within both pretreatment sample years. On average, units sampled in 2000 had 9.1 percent more cover than units sampled in 2001 (fig. 2-6). Much of the difference between years may be due to one unit with very high cover sampled in 2000 (Poison). The understory cover in Poison was more than 13 percent higher than the second highest unit in either pretreatment sampling year. Also, Sand 2, which had plots sampled in both years, actually had higher average cover on the plots that were sampled in 2001, contradicting the overall trend. Overall, there appears to be no obvious trend between the 2 years of pretreatment sampling for understory cover.

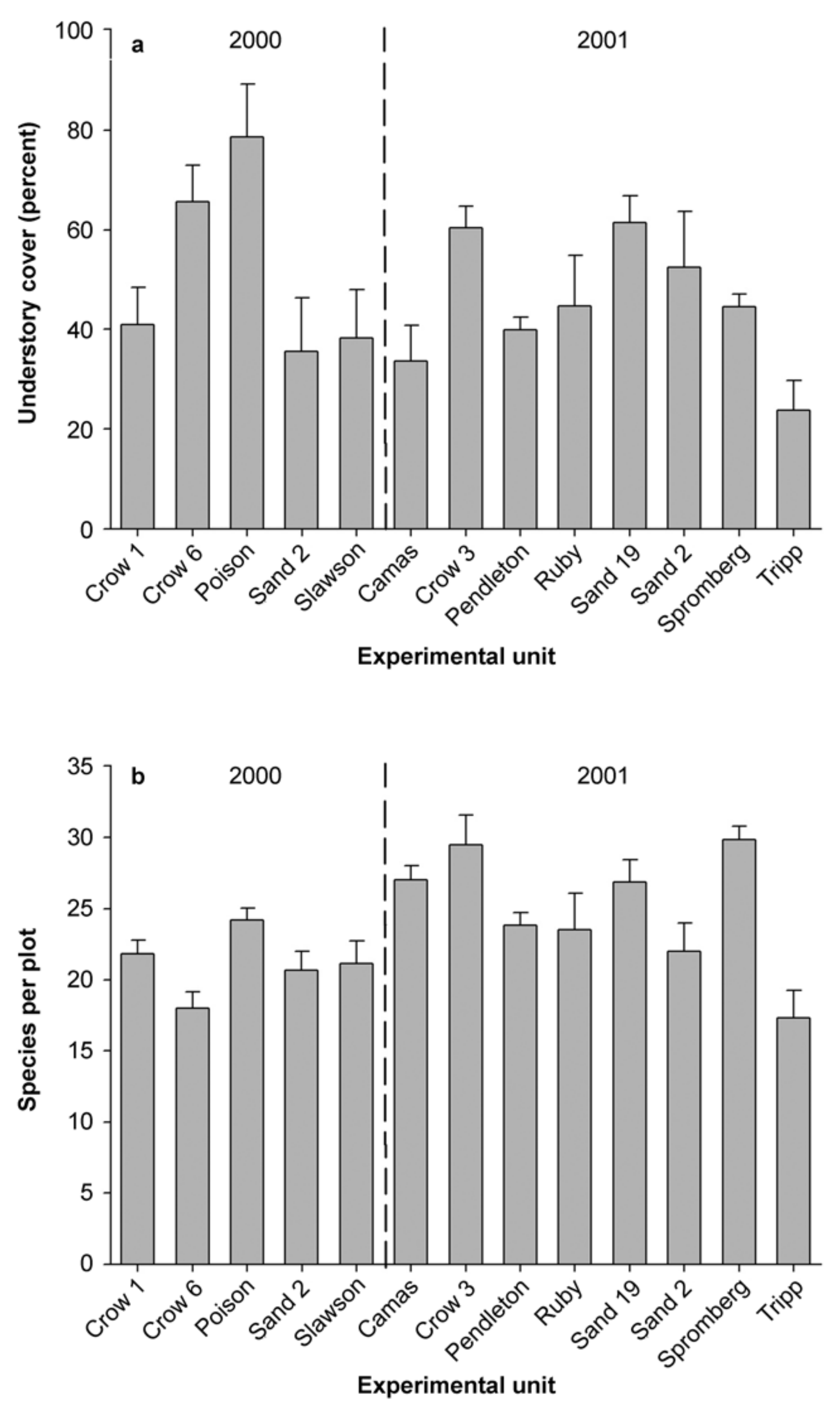

Figure 2-6—Site means for (a) cover and (b) richness (+ standard error) for the two pretreatment sample years at the Mission Creek site.

There was high variability among units, even within a given year, for understory richness. Richness was higher in 2001 than in 2000 by an average of about four species per plot, which was opposite the pattern for cover (fig. 2-6). Sand 2 had very little difference in species richness between plots sampled in 2000 and those sampled in 2001. Overall, there was little evidence of a strong year effect that may bias later results.

Conclusions

Overstory vegetation at the Mission Creek site was generally similar across all pretreatment units, but understory vegetation was highly variable. Tree density, snag density, live tree basal area, stand density index, sapling density, species composition, canopy cover, canopy bulk density, and canopy base height were statistically similar among the four assigned treatments. Only snag basal area was significantly higher on thin-only units compared to thin-burn units. Overstory vegetation at Mission Creek is relatively unique compared to most other sites in the FFS network.

Understory vegetation was variable among treatment units, but much more variable within units. The Mission Creek site is fairly species rich with 124 species, mostly dominated by graminoids and shrubs, but most species had low cover and frequency prior to treatment application. Species richness and cover was significantly different among assigned treatments, highlighting the importance of pretreatment measurements rather than the use of just control units. Understory community structure also differed among treatment units, and understory vegetation was correlated with physical and biological elements, such as elevation and overstory tree density. These correlations suggest that restoration treatments may be targeted to have positive effects on understory vegetation. Finally, there was a large amount of unit-to-unit variability in understory cover and richness between sample years, but there was little evidence of a strong year effect that may bias later results.

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Chapter 3: Fuels and Fire Behavior

James K. Agee and M. Reese Lolley[1]

Introduction

One of the major goals of silvicultural treatment of dry forest stands is to reduce the behavior of potential wildfires by reducing fuels. Thinning and burning treatments have the potential to either increase or decrease dead fuel loading, so it is important to measure fuels and fire behavior effects of such treatments.

Methods

Dead and down fuels, herb and shrub biomass, and canopy fuel parameters were estimated at 30 to 35 sample points in each of the 12 experimental units (see chapter 1). This allowed 1200 to 1400 m of line intersect transect per unit, sufficient to estimate average biomass of coarse woody debris within 20 percent of the mean 90 percent of the time in similar mixed-conifer forests (Bate et al. 2004). Beginning at the upslope row of gridpoints, each separated by 40 m in columns and rows, alternate gridpoints were sampled (with an 80-m separation), with the samples in the next row offset one column from the previous ones. Both line intersect and plot samples were used for fuels. Where the soils grid and sampling were already established, the fuels grid used alternate gridpoints to avoid excessive impact by either group.

Using 1200 to 1400 m of line intersect transect per unit is sufficient to estimate average biomass of coarse woody debris within 20 percent of the mean 90 percent of the time.

Dead and down woody load was measured using Brown's (1974) fuel transect method. Two 20-m transects were deployed at random azimuths from sampled grid points. The second transect had to be separated by at least 90° from the first. Each transect began 5 m away from the gridpoint to avoid excessive ground disturbance to the sampled area. Fuels of different timelag classes (Rothermel 1983) were measured in different ways. One-hr timelag fuels (0 to 0.62 cm diameter) were tallied along the first 2 m of the transect; 10-hr timelag fuels (0.62 to 2.54 cm diameter) were tallied along the first 3 m; and 100-hr timelag fuels (2.55 to 7.62 cm diameter) were tallied along the first 5 m of the transect. Dead and down fuels larger than 7.62 cm diameter (1,000-hr+ timelag) were individually measured along the entire 20-m length of the transect in one of five decay classes (Brown 1974), with specific gravities ranging from 0.48 to 0.30. Forest floor depth was measured at three locations (10, 15, and 20 m), and separate depths were recorded for litter (recognizable

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organic material, Oi), and duff (intermediate to highly decomposed organic material, Oe). Fuelbed depth was measured as the height of the tallest dead woody fuel particle along three 1-m-long, closely spaced segments of the transect (9 to 10, 10 to 11, and 11 to 12 m segments).

A regression to develop a depth-mass relationship for the forest floor was determined locally on the Ruby, Camas, and Pendleton experimental units, representing the geographical range of the units. Litter (n = 108) and duff (n = 21) samples were collected on 100-cm^2 plots using a square board and cutting through litter and duff along its edges. Samples were removed down to mineral soil, obvious mineral fraction was discarded, and the remainder was bagged for laboratory analysis. Depth of both layers was recorded along four edges of each sample and averaged. In the lab, each sample was dried at 70 °C until a constant mass was obtained, and then a 3-cm^3 subsample was ashed to determine mineral content. The mineral content in excess of 4 percent (Agee et al. 2002) was subtracted from the total forest floor biomass in calculating forest floor biomass for each sample, as it was assumed to be soil contamination. The mineral fraction determined from samples was 14.6 percent for litter and 21.6 percent for duff. Simple linear regression was used to develop general models for litter and duff mass from depth measurements obtained from the transect sampling.

Herbaceous and shrub fuel loading was estimated on a 400-m^2 plot centered on each sampled gridpoint by an ocular estimation of cover, height, density, and dominant type based on procedures of Burgan and Rothermel (1984). Herbaceous fuels were placed in one of two classes: type 1, a fine herb like cheatgrass (*Bromus tectorum* L.), or type 2, a coarser herb such as pinegrass (*Calamagrostis rubescens* Buckl.) or elk sedge (*Carex geyeri* Boott). One of three average density classes (1 [low], 3, or 5 [high]) was assigned from the Burgan and Rothermel photo key, as were cover classes: 1 (10 to 40 percent cover), 3 (40 to 70 percent cover, and 5 (70 to 100 percent cover). Depth was recorded as 70 percent of the maximum leaf or stalk height (Burgan and Rothermel 1984). Three shrub types were locally identified based on stem diameter and leaf thickness: type 1 (thin stems and leaves, such as shinyleaf spirea [*Spiraea betulifolia* Pall.] or dwarf rose [*Rosa gymnocarpa* Nutt.]); type 2 (thick stems but thin leaves, such as antelope bitterbrush [*Purshia tridentata* (Pursh) DC.] or ocean spray [*Holodiscus discolor* (Pursh) Maxim.]); and type 3, (thick stems and thick leaves, such as snowbrush ceanothus [*Ceanothus velutinus* Dougl.]). Density, cover, and depth were also recorded for the shrub layer. Using these data, biomass of shrubs and herbs were calculated using the program FUELCALC (Reinhardt et al. 2006).

Canopy fuel profiles were created by establishing square plots at each grid point where surface fuel transects were sampled. Plots were of variable size to include at least 10 trees per plot. Usually this was achieved with 400 m^2 plots, but occasionally plots were as small as 25 m^2 or as large as 3600 m^2. Data for all trees 1.4 m and taller included species, diameter at breast height, height to dead crown, height to live crown, and total height. Canopy closure was measured with a spherical densiometer at four directions from plot center, beginning in the upslope direction, and separated by 90°. Data were entered into the Forest Vegetation Simulator-Fire and Fuels Extension (FVS-FFE) (Reinhardt and Crookston 2003) to calculate canopy base height (CBH) and canopy bulk density (CBD), variables useful to predict crown fire hazard for each stand surrounding a sampled gridpoint.

Two types of analyses were applied to the fuels data. A canonical correspondence analysis (CCA) was used to evaluate the similarity of the entire fuels profile among units, and a nonparametric analysis of variance (ANOVA) was used to evaluate differences between individual fuel categories between units. Fuels and selected canopy parameters on experimental units (all variables in tables 3-1 and 3-2) were described by CCA using PC-ORD v. 4.0 (McCune and Mefford 1999). Environmental variables chosen for comparison in the ordination included unit elevation, average slope, percentage of unit in southerly aspect, percentage ponderosa pine (*Pinus ponderosa* Dougl.), and past logging history (represented by dummy variables: 0 = no logging; 1 = logging up to 1930s only; 2 = logging since 1970). Fuel differences between experimental units were tested using a nonparametric ANOVA by ranks (Kruskal-Wallis H test) using SPSS (2002), as sampled fuel populations rarely were normally distributed or had equal variances (Zar 1999).

Table 3-1–Biomass (and standard errors) of dead and down surface fuels by experimental unit

Unit	1-hr fuel load	10-hr fuel load	100-hr fuel load	1,000-hr+ sound fuel load	1,000-hr+ rotten fuel load	Forest floor load
			Mg/ha			
Camas	1.9 (0.4)	2.3 (0.3)	6.4 (1.0)	11.6 (2.4)	4.5 (1.3)	23.0 (2.3)
Crow 1	0.3 (0.0)	1.5 (0.3)	5.7 (0.7)	11.9 (1.7)	7.8 (1.6)	45.6 (2.8)
Crow 3	0.7 (0.1)	1.6 (0.2)	2.8 (0.6)	6.7 (2.4)	2.3 (1.0)	39.5 (2.9)
Crow 6	0.3 (0.0)	1.3 (0.2)	4.4 (0.6)	8.8 (1.9)	3.2 (0.8)	43.1 (3.3)
Pendleton	0.3 (0.1)	2.4 (0.3)	5.2 (0.9)	16.5 (3.2)	2.1 (1.0)	53.3 (3.0)
Poison	1.5 (0.3)	1.7 (0.2)	3.0 (0.7)	5.1 (1.2)	9.7 (4.3)	37.0 (3.1)
Ruby	1.5 (0.3)	1.3 (0.2)	6.6 (1.5)	17.1 (4.1)	16.4 (4.6)	40.7 (4.3)
Sand 19	1.6 (0.2)	1.5 (0.2)	2.0 (0.5)	3.0 (1.8)	3.0 (1.7)	34.4 (2.9)
Sand 2	1.7 (0.3)	2.6 (0.4)	4.0 (0.8)	6.8 (1.5)	4.9 (1.4)	32.8 (2.3)
Slawson	1.3 (0.1)	0.8 (0.1)	4.2 (0.8)	8.1 (3.3)	10.2 (2.3)	25.1 (2.0)
Spromberg	1.4 (0.2)	2.3 (0.4)	6.9 (1.2)	12.3 (3.4)	9.6 (2.3)	38.7 (3.9)
Tripp	1.7 (0.4)	2.3 (0.4)	5.0 (1.2)	17.7 (5.0)	8.8 (3.0)	32.6 (2.3)

Table 3-2—Vegetation characteristics (and standard errors) useful in fuel and fire modeling on experimental units

Unit	Canopy closure	Canopy base height[a]	Canopy bulk density[a]	Herb biomass	Shrub biomass	Fuelbed depth
	Percent	*Meters*	*Kg/m^3*	*Mg/ha*	*Mg/ha*	*Centimeters*
Camas	63 (5)	1.2	0.056	0.1 (0.0)	2.5 (0.8)	11.2 (2.4)
Crow 1	65 (3)	4.6	0.043	1.2 (0.1)	6.3 (1.3)	9.2 (1.7)
Crow 3	65 (5)	2.7	0.056	0.4 (0.1)	6.3 (2.4)	7.7 (1.4)
Crow 6	55 (4)	4.3	0.044	1.1 (0.1)	3.3 (0.7)	10.1 (2.5)
Pendleton	71 (4)	3.0	0.052	0.5 (0.1)	2.0 (0.8)	11.8 (2.1)
Poison	64 (5)	2.4	0.052	0.3 (0.1)	8.8 (2.3)	10.0 (1.4)
Ruby	75 (3)	1.5	0.051	0.3 (0.1)	3.8 (1.0)	15.8 (3.6)
Sand 19	67 (5)	3.0	0.062	0.5 (0.1)	2.3 (1.4)	8.3 (1.6)
Sand 2	76 (5)	2.4	0.072	0.6 (0.1)	5.7 (1.6)	13.2 (2.7)
Slawson	80 (3)	1.2	0.116	0.5 (0.1)	4.3 (1.8)	6.3 (0.8)
Spromberg	75 (5)	1.2	0.060	0.1 (0.0)	5.8 (1.1)	9.9 (2.0)
Tripp	73 (4)	1.5	0.082	0.3 (0.1)	5.8 (2.0)	18.2 (3.6)

[a] Forest Vegetation Simulator—Fire and Fuels Extension (FVS-FFE) does not calculate standard errors.

Differences between groups ($P < 0.05$) were analyzed with the Dunn test, a nonparametric multiple comparison with unequal sample sizes statistic that takes into account the number of comparisons made by using a Bonferroni adjustment. This adjustment uses a smaller alpha value in pairwise comparisons to keep the actual probability of family-wide type I error at or below the desired level (the significance level here was set at alpha = 0.05).

Surface fuel models were assigned using the Northern Forest Fire Laboratory (NFFL) models available in 2000 (Albini 1976, Anderson 1982). On a 400-m^2 plot centered on a sampled gridpoint, the proportion of the area in any of the 13 NFFL fuel models (minimum 10 percent cover) was recorded. Custom fuel models for each pretreatment unit were developed from the fuel transect data, with 0.5 cm of forest floor added to the 1-hr timelag fuel load. Worst-case fire weather was taken from Agee and Lolley (2006): 1-hr, 10-hr, and 100-hr fuel moisture were set at 3, 4, and 6 percent; open windspeed was set at 36 kph, and slope was held constant at 45 percent, the average slope for all units at the site. Fuelbed depth was defined in two ways: (1) by the average height of the tallest dead fuel particle on each experimental unit, and (2) by average shrub height on each unit (based on 70 percent of measured shrub height) to account for shrub contribution to the fuelbed depth. Fire behavior simulations were run in NEXUS (Scott 1999) with the most common NFFL models (2, 5, 8, 9, and 10) and for each custom unit model using the two fuelbed depths. Results were expressed by the type of fire predicted by NEXUS: surface, a fire restricted to the surface fuel profile; passive, a fire that will transition from a surface fire to torching fire behavior and return to a surface fire; active, a fire that will torch and maintain itself as an active, running crown fire; and conditional, a rare situation where the canopy bulk density is high enough to maintain active crown fire, given the predicted rate of spread, but one that is not supported by surface fire flame lengths or torching.

Results and Discussion

The CCA analysis suggested considerable homogeneity among sites for fuel variables.

Dead and down fuel, herb and shrub, and canopy variables showed moderate variability across experimental units (tables 3-1, 3-2). The CCA analysis (fig. 3-1) indicated that the total variance ("inertia") was a very low 0.0566, suggesting considerable homogeneity among sites for fuels variables. The cumulative variance explained by the first three axes totaled 55 percent (axis 1, 34.2 percent; axis 2, 14.1 percent; axis 3, 6.8 percent), and the correlation with the environmental variables and sample scores was 0.934 for axis 1 and 0.767 for axis 2. The environmental variables in figure 3-1 are shown as lines, with the direction indicating the correlation to axis and the line length indicating the strength of the relationship. The ordination separates the fuel characters of the units into three geographic groups: the first is at the left side of axis 1 and consists of the southerly group of units (Crow 1, Crow 3, Crow 6, and Pendleton). The second consists of the

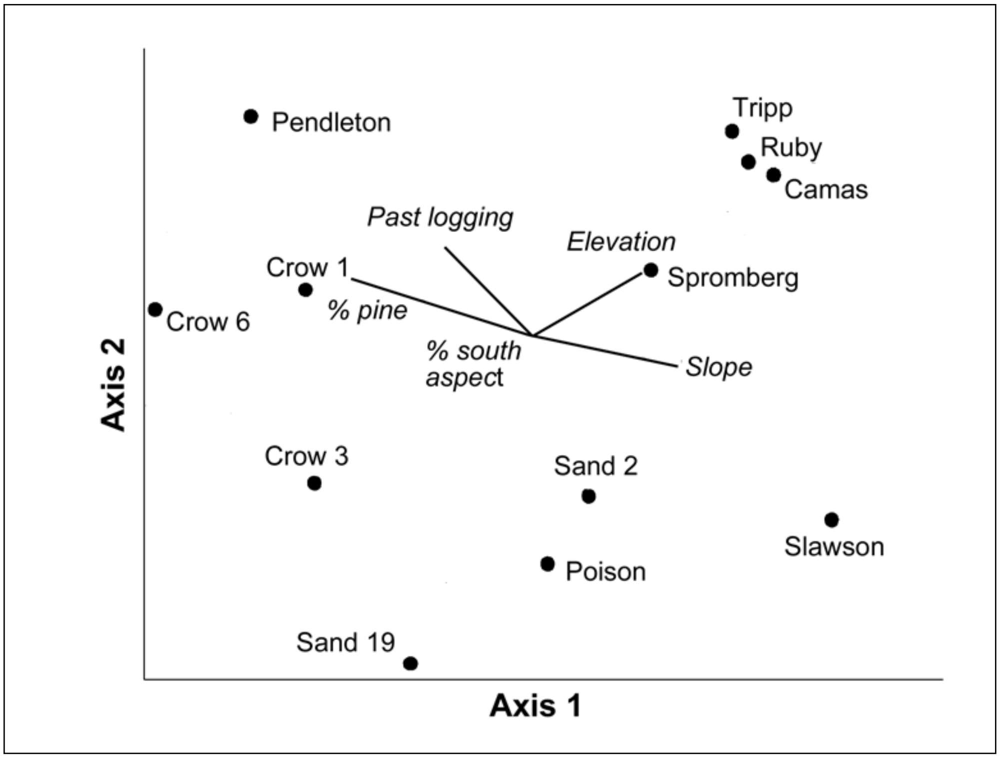

Figure 3-1—Canonical correspondence analysis of fuels data. Axis 1 was associated with increasing slope steepness (+) and percentage of forest in pine and percentage on south aspects (-). Axis 2 was less significantly associated with evidence of past logging (-) and elevation (+).

northeasterly units (Poison, Sand 19, Sand 2, and Slawson), and the third contains the westerly units (Camas, Ruby, Spromberg) plus Tripp (an anomalous primarily north aspect unit). The first group consists of units with a high proportion of southerly aspect, large basal area in ponderosa pine, and gentle slopes. The third group is at higher elevation, with more Douglas-fir (*Pseudotsuga menziesii* [Mirbel] Franco) basal area and steeper slopes, and the second group is intermediate to the first and third groups. These environmental variables are strongly related to axis 1. Of the environmental variables included, past logging history was most highly correlated to axis 2, followed by elevation and basal area in ponderosa pine.

Regression equations predicting forest floor mass from depth had coefficients of determination ranging from 0.75 to 0.86. Separate regressions for ponderosa pine-dominated litter, mixed Douglas-fir litter, a combined litter general model, and a duff general model are shown in figure 3-2.

The dead and down fuel loads are either in the range of or lower than other mixed-conifer forests.

The number of significant pairwise comparisons (of a possible total of 66 for each variable) between units was highest for the forest floor (23 for litter, 36 for duff), 1-hr timelag fuels (25), and herbaceous load (23) (Lolley 2005). The largest number of significant differences for all variables was between two groups of units: Crow 1-Crow 6-Pendleton versus Camas-Ruby-Slawson-Spromberg, and is fairly consistent with what the ordination showed. The dead and down fuel loads are either in the range of or lower than other mixed-conifer forests. The 1-hr and 10-hr fuel loads are in the range of southern Cascade and Sierra Nevada forests (Stephens and Finney 2002, Stephens and Moghaddas 2005, Thomas and Agee 1986), but 100-hr and 1,000-hr+ (both sound and rotten) fuel mass is 25 to 65 percent of the load in these other regions. Mission Creek loads of larger coarse woody debris are more comparable to those of northern Arizona (Fulé et al. 2002). Forest floor loads are well below the reported loads in these other areas, ranging from 20 to 50 percent of those loads. At Mission Creek, the duff (Oe) layer is generally poorly developed, and that is often where the bulk of the forest floor mass is found.

The canopy profiles appear fairly consistent with other mixed-conifer forests that have been protected from surface fires for many decades. Canopy closure values are high, averaging 69 percent, but typically canopy closure, as measured with densiometers, is about 20 percent higher than canopy cover as measured with densitometers (Ganey and Block 1994). Average canopy base heights (2.4 m) are low, representing the in-filling of shade-tolerant trees (Hessburg et al. 2005). Our canopy base heights are lower than those found by Harrod et al. (this volume) on selected 20- by 50-m plots in the same experimental units because we included trees 1 m and taller in our inventory (their smallest tree included was 7.62 cm). Average canopy bulk density (0.062 kg/m^3) is lower than estimated for other

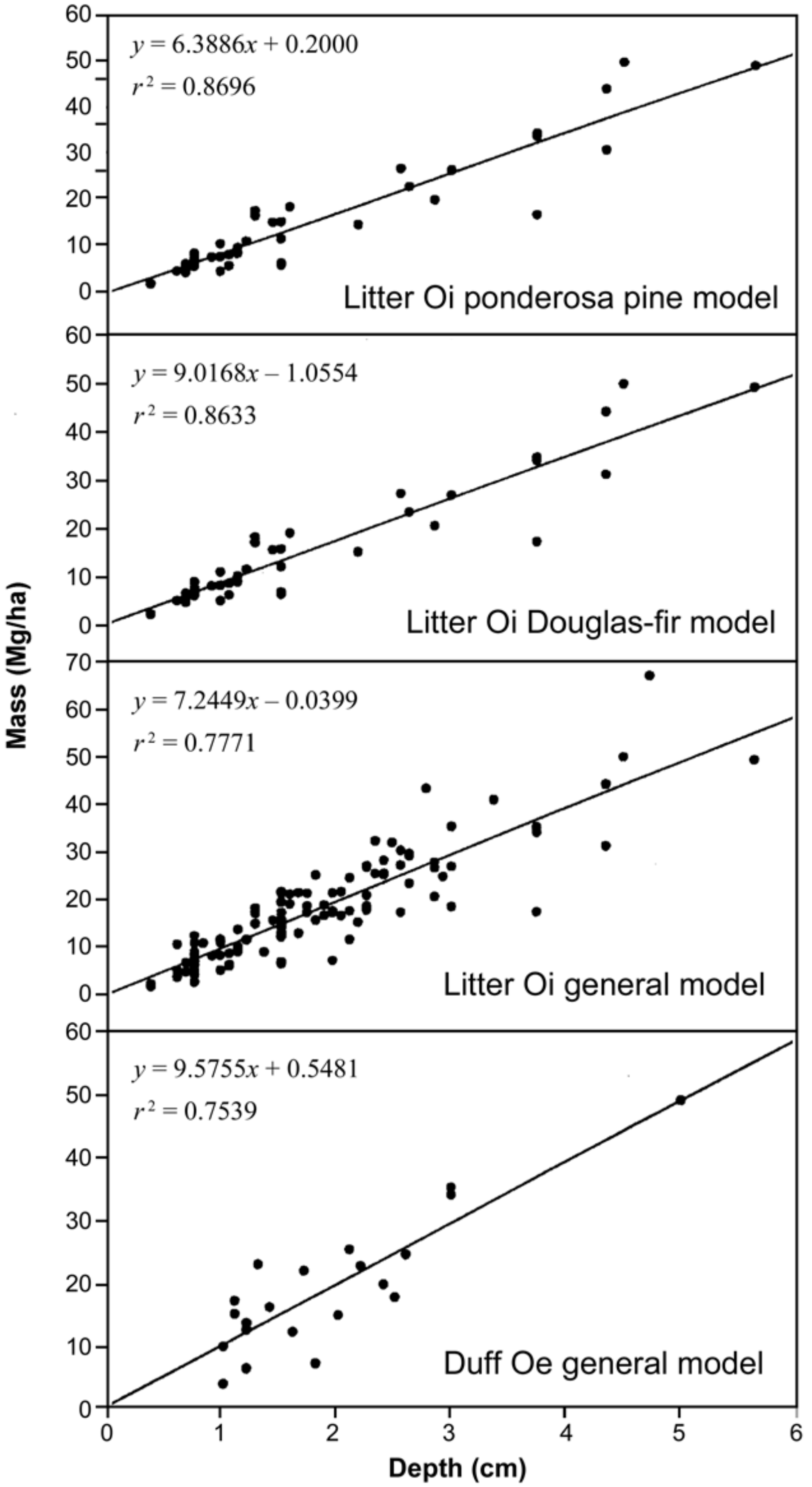

Figure 3-2—Depth/mass regressions for ponderosa pine, Douglas-fir, and mixed litter and for duff.

Cascade and Sierra Nevada mixed-conifer forests (Stephens and Moghaddas 2005) but comparable to ponderosa pine forests of northern Arizona (Fulé et al. 2002) and what Harrod et al. (this volume) found on forested portions of the same experimental units.

Surface fire behavior models (fig. 3-3) assigned to experimental units were weighted toward NFFL 2 (a grass-dominated fuel type) and NFFL 5 (a shrub-dominated fuel type). Fuel types dominated by timber litter (NFFL 8, 9, and 10) did not cover a majority of the area of any unit. The average fuel load across all experimental units does not have a reasonable fit to any of the new set of standard surface fuel models (Scott and Burgan 2005).

Fire behavior simulations using closest-fit NFFL models and worst-case (97 percentile) weather (fig. 3-4, left) showed active crown fire to be associated with most units, which were dominated by NFFL models 2 and 5 (Albini 1976). For this analysis, the output used in the pie chart was weighted to 75 percent models 2 and 5, and 25 percent allocated evenly between NFFL models 8, 9, and 10. Custom fuel models designed from the fuels data on each unit, and using the dead fuel particle height measured on the fuel transects, predicted more benign fire behavior with the same weather data (fig. 3-4, middle). Each unit represents one-twelfth of the total shown in the pie chart. Surface fire behavior dominated across the units, with only single units expressing passive (Camas) or active (Tripp) crown fire behavior. When fuelbed depth was defined by 70 percent of shrub depth on each unit, representative of the shrub component in the understory that was incorporated as part of the fuel load of the custom models, and maintaining the same fuel loads, the simulations produced results much more similar to the NFFL runs (fig. 3-4, right). The fire behavior in five units was predicted to be passive crown fire behavior (Crow 1, Crow 6, Pendleton, Poison, and Ruby), and the remaining seven units were predicted to have active crown fire behavior. Using the dead fuel particle height to represent fuelbed depth compresses the fuelbed to the extent that the packing ratio is too compact, so that the combustion properties of the fuelbed are not optimum. The increase in depth produces a more optimum packing ratio and makes fire behavior more intense. These results suggest that pretreatment conditions are conducive to severe fire behavior, and this has been borne out by empirical evidence on many recent wildfires in the area (Entiat 1970, Dinkelman 1988, Tyee 1994, Rat 1994, and Fischer 2004). The variability in predicted fire behavior between these fuel models argues for field calibration under average as well as worst-case fire behavior before they are used in fire management operations.

Results suggest that pretreatment conditions are conducive to severe fire behavior, and this has been borne out by empirical evidence.

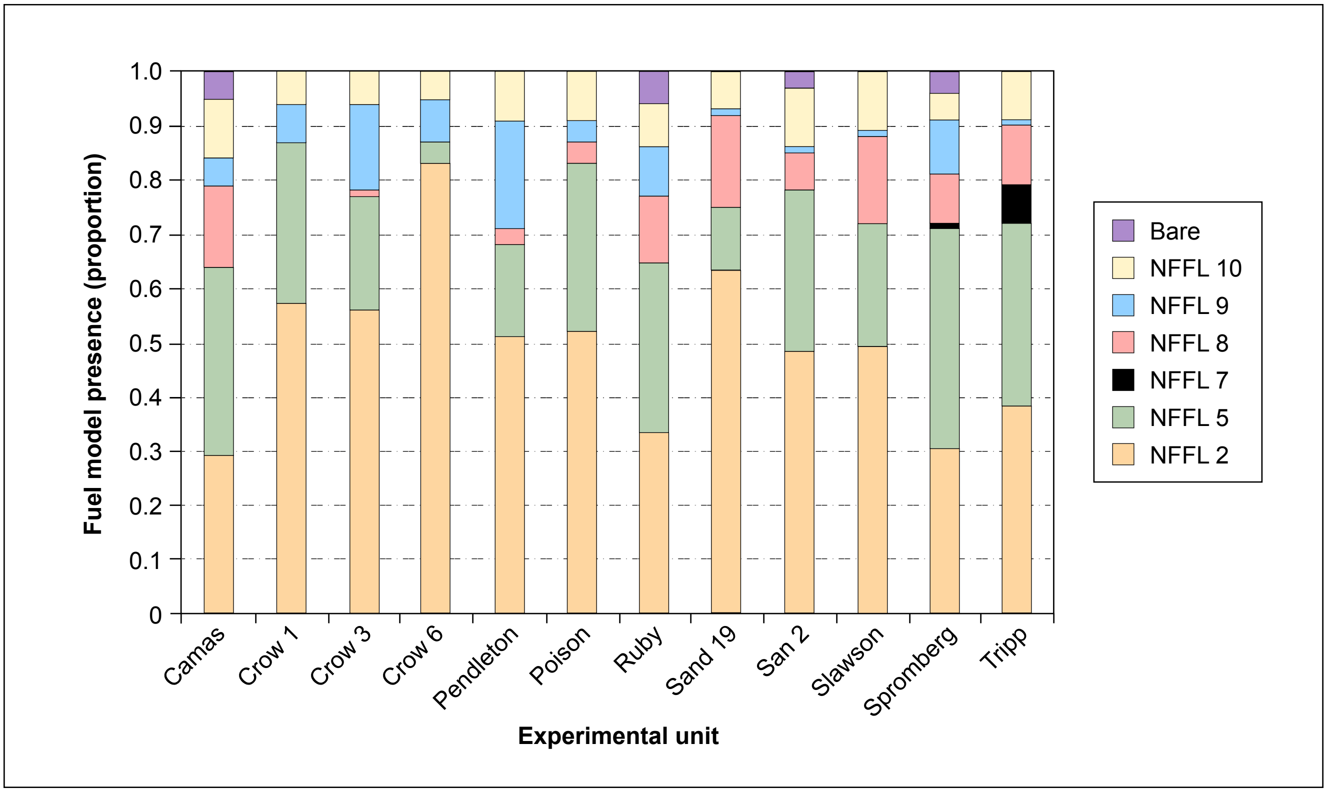

Figure 3-3—Closest-fit Northern Forest Fire Laboratory (NFFL) surface fuel models for the 12 experimental units. The NFFL models 2 (timber and grass) and 5 (low shrub) were the most common models assigned. The NFFL models 8 (compact timber litter), 9 (loose timber litter), and 10 (litter plus understory) were also present on most units.

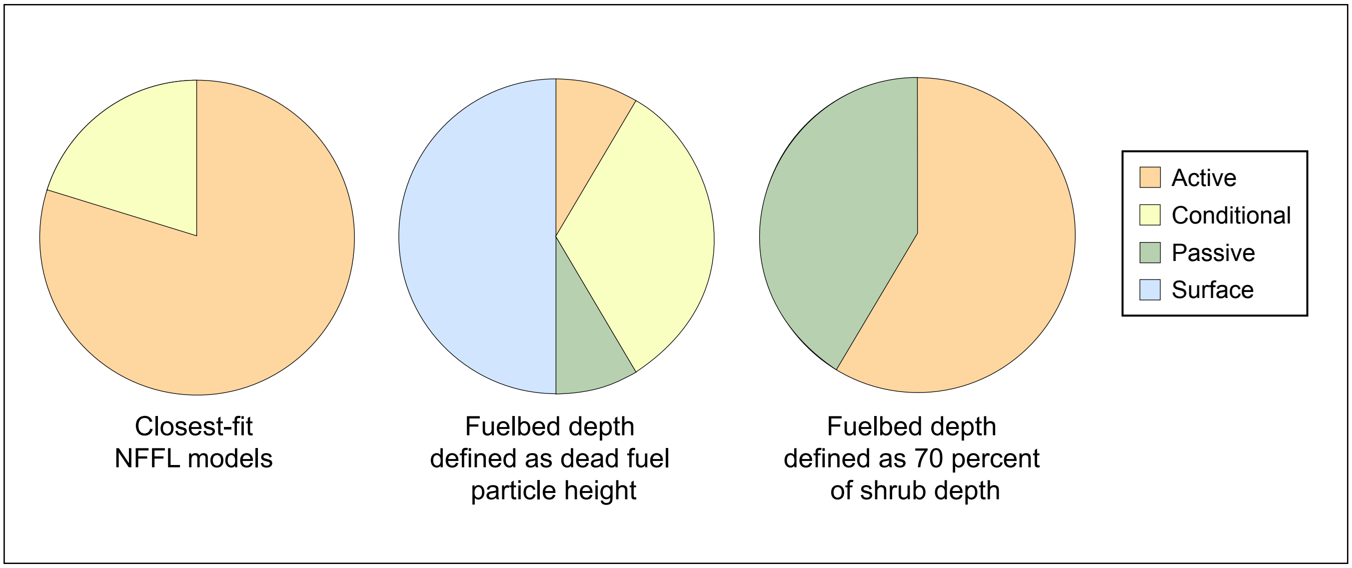

Figure 3-4—Proportion of active, conditional, passive, and surface crown fire at Mission Creek experimental units. Left: classification according to best-fit Northern Forest Fire Laboratory (NFFL) models; middle, classification using fuelbed depth defined by dead particle fuel height; right, classification using a fuelbed depth better representing the shrub component of the fuelbeds.

Fuels ordination and statistical comparisons suggest that these units are very similar and a randomized treatment design is justified.

The fuels ordination and statistical comparisons suggest that the similarity of these experimental units is high, so that a randomized design for assigning treatments (control, thin, burn, and thin-burn) is justified. When units were grouped by proposed treatment (the balanced design of 2006), no main factor variables showed significant differences among treatment for the 12 variables analyzed (see tables 3-1 and 3-2 for listing of variables). There was a single significant interaction for 1,000-hr rotten fuels, indicating biomass in that category was higher on units proposed for thinning and lower on units proposed for burning treatment.

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Chapter 4: Soils

Darlene Zabowski, Jeffrey Hatten, Elizabeth Dolan, and George Scherer[1]

Introduction

Soils in the Mission Creek study area occur on widely ranging slopes, aspects, elevations, parent materials, and topographic positions. Although the vegetation encompasses assemblages typical of ponderosa pine/Douglas-fir forests, it is also quite variable with some areas covered by thickets of young trees with sparse understory, whereas other areas have widely scattered trees with extensive grasses and shrubs. This variability could result in widely divergent soil types; therefore, a comprehensive study was made of soils in the area to insure that effects of prescribed burning could be determined despite soil variability. This report briefly summarizes the soils and soil properties found at the Mission Creek study site.

Methods

Soil Sampling

Soils were measured and sampled for physical and chemical properties during the summers of 2000 and 2001 using a grid with an 80-m spacing resulting in 15 to 21 sampling sites for each study plot (plots were irregularly shaped and not equally sized). Major genetic horizons were identified and horizon thickness measured. Soil samples were collected for laboratory analyses from O, A, and B horizons (either Bw or Bt) as present, to a mineral soil depth of 30 cm. Hydrophobicity of each O, A, and B horizon was measured in the field by dropping 1 ml of water and measuring the amount of time needed for the droplet to completely infiltrate the soil. Bulk density samples of O horizons were sampled by cutting out and removing a 150-cm^2 area of forest floor and measuring the depth at four locations along the edges to determine a volume; coarse woody debris >2 cm was not included in the O horizon samples. The O horizon material was taken to the lab and dried at 70 °C and weighed. The bulk density of each mineral soil horizon was determined on a representative soil core using a hammer corer with a 137 cm^3 core; samples were taken to the lab and dried at 105 °C to obtain dry weight and calculate bulk density.

Soil available N was measured once per year in spring (April-May) of 2000, 2001, and 2002. The buried bag method was used to determine net mineralizable nitrogen (N). Half (vertically divided) of a 137-cm^3 core (5.4 cm diameter by 6 cm depth) of O and A horizon were incubated at their respective depths in the field for

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4 weeks (Raison et al. 1987). The other half of the core was immediately returned to the lab and extracted using 2 M potassium chloride (KCl) for initial ammonium (NH_4) and nitrate (NO_3). Incubated samples were subsequently retrieved and also extracted using 2 M KCl for NH_4 and NO_3. Solution analysis was done using an autoanalyzer (Keeney and Nelson 1982). Net mineralization was the difference between final and initial extractable N.

In addition to sampling soil at each grid point, four soil pits were dug within each experimental unit to either a maximum of 1.5-m depth or to weathered bedrock for soil profile descriptions and additional sampling. Pit locations were stratified within each plot, with one pit located on a ridge, one in a valley, and one on a sideslope; the fourth pit was randomly located in an area bare of vegetation within each unit. Complete profile descriptions were done at each pit, and samples were collected from each horizon to determine soil classification.

Because the sideslope pit of each plot was considered to be the most representative of each plot (83 percent of the gridpoints were on sideslope or toeslope topographic positions), these pits were used to monitor soil moisture and temperature. Five fiberglass soil moisture/temperature blocks were installed in major genetic horizons (at the interface of the O and A horizons, and within the A, AB or Bw1, within the Bw2 or Bt, and within the Bt2 or BC horizons) of each treatment plot. Soil moisture and temperature were measured during the day (mid-morning to late afternoon) at monthly intervals over a 3-year period (2000 to 2003) using an electrical resistance meter with standard moisture calibration curves (Soil Moisture Corp.[2]) for fiberglass blocks. Measurements of air temperature were made using a shaded hand-held thermometer at 1 m above the soil surface each time soil temperature was measured. Snowpack depths were also measured with each site visit using a marked snow rod. Owing to the difficulty of accessing the sites in winter, only one reading was made in January 2001, which was during the first year of sampling.

Soil Analysis

Mineral soil samples were air-dried and sieved to 2 mm for chemical analyses. Subsamples from each mineral horizon and representative O horizon material were analyzed for pH using the saturated paste method (Van Miegroet et al. 1994). Six grid-point sampling sites were randomly selected for further chemical analysis of each horizon within each experimental unit. Subsamples of each horizon were finely ground and analyzed for total carbon (C) and nitrogen (N) using a Perkin-Elmer 2400 CHN analyzer. Available phosphorous (P) was extracted by shaking 10

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

g of soil with a 0.01M solution of calcium chloride ($CaCl_2$) for 30 minutes, filtering with a Whatman no. 42 filter paper, and analyzed using the molybdenum blue method of Olsen and Sommers (1982). Cation exchange capacity (CEC) was determined with a 12-hour extraction using unbuffered 2M ammonium chloride (NH_4Cl) and a mechanical extractor with a subsequent 2M KCl extraction (Skinner et al. 2001). Base cation saturation was determined on the NH_4Cl extracts using a Jarrel Ash Atomcomp ICP.

Results and Discussion

Soil Properties

The top 20 cm of the soil found in the Mission Creek area typically has horizons of O, A, and either a Bw or Bt horizon (table 4-1, fig. 4-1). The O horizons were predominantly Oi with some Oe, and almost no Oa; sometimes a very thin layer of charcoal separated the O horizon from the surface mineral horizon. The A-horizon depths were extremely variable, ranging from nonexistent in high-erosion areas, to an average thickness of 35 cm; in one valley location the A horizon reached 88 cm deep. In some cases, an AB horizon was also present. Although A horizons were variable, overall they were deep for a forest soil. Both Bw and Bt horizons were also quite variable, being absent in extremely steep eroded areas and greater than 75 cm in aggrading sites. BC or C horizons commonly occurred at depth, although in some cases, Crt horizons of weathered sandstone with inclusions of clay were present. Of the profiles shown in figure 4-1, three were found rarely, occurring only once (profile with O, A1, A2, C, 2BCb; and profile with O, A, Bw, C, CB, 2Bwb) or twice (profile with thin O over A, Cr) within the entire study area. The remaining five soil profile types dominated the soil landscape, but no one soil type consistently occurred at a particular topographic position.

The soil is generally well structured with a moderate bulk density in the upper horizons but higher bulk densities in the lower profile (table 4-1). The moderate to high bulk densities are partially explained by the texture, which is typically loamy or sandy loam in the upper profile, but is increasingly sandy or clayey at depth. This increase in sand or clay can increase bulk density despite good structure. In some cases, the BC or C horizon became massive resulting in even higher bulk densities (as high as 1.9 g/cm in a Crt horizon). Hydrophobicity was high in the O horizons, undoubtedly because of a high concentration of hydrophobic waxes and the extreme dryness of the O when hydrophobicity was measured during summer. There was a slightly slower water penetration rate in the A horizons than the B horizons, but no strong hydrophobic layer was evident.

Table 4-1—Common soil horizons and typical physical properties of soil within the Mission Creek study area

Horizon	Average thickness	Color	Texture	Structure	Hydro-phobicity	Bulk density (standard error)
	Centimeters				*Seconds*	*Mg/m^3*
O	7	—	Mostly Oi	—	40	0.08(0.01)
A or A1	25	10YR3/1 or 3/2	Loam, sandy loam	Weak, fine, granular to moderate medium subangular blocky	7	1.1 (.10)
AB or A2	25	10YR3/2	Loam, sandy clay loam	Weak to moderate medium subangular blocky	ND	1.1 (.05)
Bw	60	10YR4/3	Sandy loam, loamy sand	Weak to moderate, medium subangular blocky	3	1.2 (.05)
Bt	80	10YR4/4 to 5/6	Sandy clay loam, clay loam	Moderate, medium to, coarse angular blocky	ND	1.4 (.05)
BC, CB, C, Crt	0 to 73	10YR5/6 to 5/8	Loamy sand, sandy clay loam	Weak, fine granular to massive	ND	1.4 (.10)

ND = not determined.

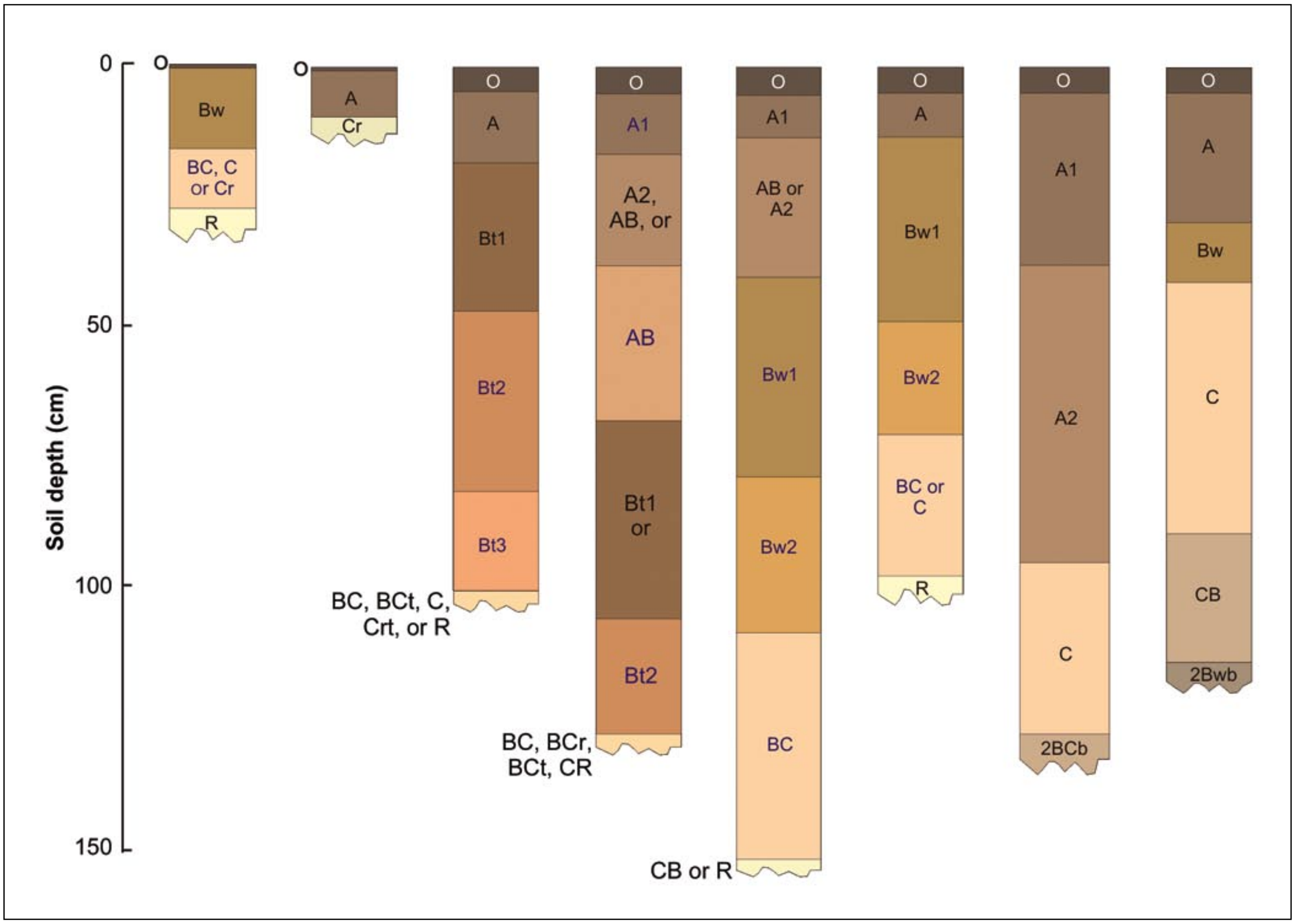

Figure 4-1—Typical soil profiles found within the Mission Creek study area showing relative horizon types and depths.

Soil temperature ranged from 0 to 15 °C in the O horizon with peak temperatures occurring in August (fig. 4-2). July, August, and September are the hottest months. The maximum recorded temperature in the A horizon was 14 °C. Average soil temperature at 50 cm depth was between 5 and 10 °C. The soil temperature regime is frigid. A snowpack generally began in December and left in March and was at least 18 cm thick, but a small snowpack was present in one November and April, and some snowpack persisted into May of one year. Typically, melting was complete in either late March or early April. When snow is present, soils are insulated and therefore little soil freezing occurs.

Soil moisture measurements indicate a xeric moisture regime, with the three drier months occurring in July, August, and September, the same months of maximum soil temperatures (fig. 4-2). Rewetting occurs in fall and winter through both rain and snowfall. Once the snowpack melts in March–April, the soil begins to rapidly dry in April, and soil moisture has already dropped to approximately half by June. Gravimetric water content ranged from 25 to 12 percent in the upper mineral horizons, and from 15 to 8 percent in the lower mineral horizons. This consistently lower moisture content in the deeper soil profile suggests that deeper soil horizons never completely wet; however, the drawdown in moisture during the summer does indicate that moisture in these horizons is used by vegetation.

Soil chemical properties of the Mission Creek area generally indicated fertile soils (table 4-2), with moderately high pH and organic carbon. Two notable properties were found. First, the total quantity of C and N in these soils is high. The concentration of N in the A and B horizons is moderately high for a forest soil, but these horizons are typically deep; therefore the total site N was high, averaging a total of 3.5 Mg/ha within the O, A, and B horizons, with a total soil profile N capital of 4.7 Mg/ha. Over 50 percent of this N is found in the A horizon. Similarly, total carbon was high with 82 Mg/ha within the combined O, A, and B horizons and 105 Mg/ha in the total soil profile. Total C and N in the grid sampling areas (table 4-2) were similar to that of the soil profiles but slightly less N (2.7 versus 3.5 Mg N/ha). The favorable N concentrations resulted in relatively low C/N ratios for all soil samples.

The total net N mineralized was moderately high (averaging 0.84 kg N/ha per **month** in spring) when compared to that of a more nutrient-poor ponderosa pine stand in eastern Oregon (2.0 to 6.4 kg N/ha per **year**) (Monleon et al. 1997). Overall, N availability does not appear to be the most limiting factor in growth of these forests. A fertilizer study in a ponderosa pine forest north of the Mission Creek area found no growth response by trees to N fertilization when total soil N was somewhat lower in the O horizon, but slightly higher in the mineral soil (Zabowski and Henry 1994).

Overall, N availability does not appear to be the most limiting factor in growth of these forests.

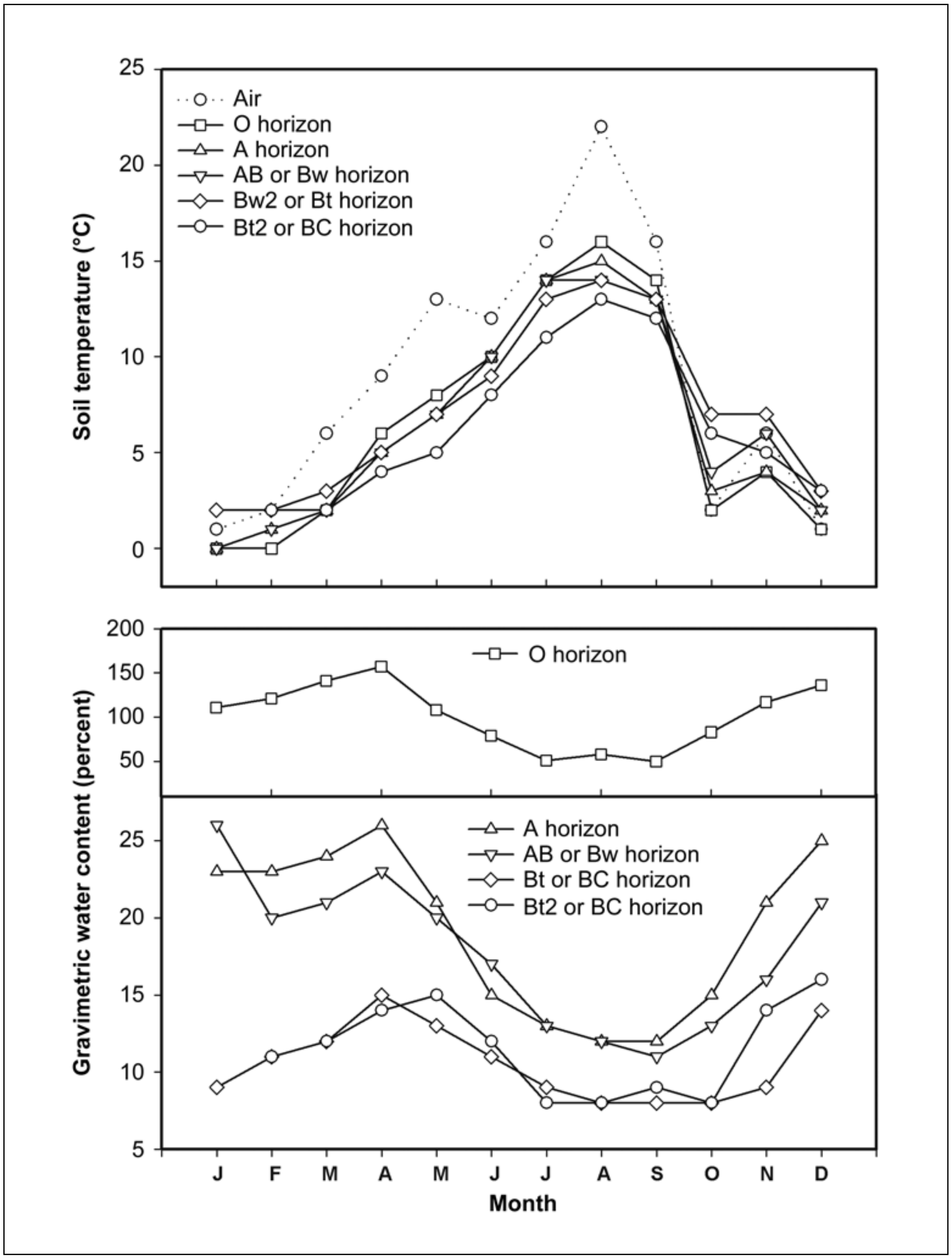

Figure 4-2—Average soil temperature and gravimetric soil water content by horizon for soils within the Mission Creek study area. Values are averages of 12 replicates from midslope soils taken over a 3-year period.

Table 4-2—Average and standard error of soil chemical properties of O, A, and B horizon grid samples in the Mission Creek study area of Wenatchee National Forest

Horizon	pH	Available phosphorus	Cation exchange capacity	Base saturation	Carbon	Nitrogen
		mg/kg	*cmol_c/kg*	- - - - - - - - - - - - *Percent* - - - - - - - - - - - -		
O	5.5 ± 0.1	ND	ND	ND	40 ± 2	1.50 ± 0.10
A	6.3 ± .2	1.11 ± 0.26	17 ± 2	90 ± 1	2.6 ± 0.6	.11 ± .05
B	6.2 ± .1	.61 ± .20	14 ± 2	90 ± 2	1.2 ± .2	.05 ± .01

	C/N	Extractable ammonium	Extractable nitrate	Net mineralizable nitrogen	Total carbon	Total nitrogen
		- - - - - - - - - - - - - - - *mg/kg* - - - - - - - - - - - - - - -			*Mg/ha*	*kg/ha*
O	29	6.70 ± 2.40	3.30 ± 3.90	1.20 ± 0.29	16 ± 1	550 ± 120
A	27	2.30 ± .56	.58 ± .19	.54 ± .16	47 ± 34	1300 ± 120
B	27	ND	ND	ND	21 ± 2	830 ± 80

ND = not determined, cmol_c = centimoles of charge.

Note: Values are averages of 8 to 18 samples collected within 12 plots (n = 12). Mineralizable N is that available within a 4-week period in spring determined by using buried bags.

The second notable soil chemical property found was the high base saturation. Base saturation averaged 90 percent, and was never less than 40 percent in the A horizon. Although CEC values are not exceptionally high, these base saturation values are high for a forest soil. Overall, nutrients do not appear to be limiting at the site when water is available for uptake. Water availability is probably the most limiting growth factor at this site. Zabowski and Henry (1994) also concluded that seasonal water limitations were probably more limiting to growth than nitrogen.

Water availability is probably the most limiting growth factor at this site.

Several areas within the study site had been burned by low-intensity wildfire within the last 27 years before sampling. The soils in the burned areas showed little effect from these contemporary fires compared to adjacent areas that had not burned since fire suppression began (Hatten et al. 2005). Soil variability appears to be dominated more by topographic factors, erosion, and vegetation (see Dolan 2002) than recent fire.

Soil Classification

A preliminary soil survey done in this area found Haploxerolls, Argixerolls, and Haploxeralfs throughout the area (McColley 1995). Soils data from the Mission Creek study site found that Haploxerolls predominate, with some Argixerolls, Haploxeralfs, Haploxerepts, and Entisols (a few Xeropsamments or Xerorthents on steep, eroded slopes) also present. The common deep A horizons with good physical properties and high base saturation have resulted in Mollisols being common despite the presence of a mixed coniferous forest. Several soil-forming factors probably contribute to this. First, although the bedrock is nonglaciated sandstone intermixed with limited amounts of shale and conglomerate from the Chumstick Formation (Tabor et al. 1982), loess and volcanic ash deposits are common in the area providing young, fine-textured parent materials in the upper profile. The total precipitation is about 60 cm per year (NRCS 2005), which does not strongly leach cations, but is adequate for grasses to be a common understory species contributing fine-root organic matter to keep well-developed A horizons prevalent. The low precipitation also favors frequent low-severity fires, which can replenish base cations without removing mineral soil organic matter. This effect has persisted despite fire exclusion in the last 100 years. Despite prevalent steep slopes, the understory vegetation and O horizon depth appear adequate to limit erosion to the steepest sites with the coarsest soil textures and the poorest soil structure (mostly the areas with shale and conglomerate).

Although soil variability is extensive within the plots, almost all of the soils present were found to have physical and chemical properties favorable for forest growth.

Conclusions

Soils within the Mission Creek Study area were found to be largely Mollisols with some Alfisols, Inceptisols, and Entisols. Although soil variability is extensive within the plots, almost all of the soils present were found to have physical and chemical properties favorable for forest growth. Overall, the soils appear to be fertile containing high amounts of C, N, and base cations. Nitrogen availability does not appear to be the primary limitation to tree growth. The most limiting growth factor is undoubtedly availability of soil water, which declines beginning in June.

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Chapter 5: Soil Enzyme Activity

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Introduction

Fire suppression and selective logging over the past 100 years in eastern Washington have changed the vegetation from primarily low-density forests dominated by ponderosa pine (*Pinus ponderosa* Laws) to high-density forests of mixed ponderosa pine, Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and other species (Agee 1993). There is now considerable interest in returning fire to these ecosystems as a management tool and making these ecosystems less likely to burn catastrophically. As the use of fire grows as a restoration tool, so has the need for a robust, quantitative metric of fire effects reflecting soil health and functioning. Enzymes produced by soil microbes are sensitive indicators of stress on ecosystems and can provide robust measures of the ecosystem health and sustainability (Boerner and Brinkman 2003, Boerner et al. 2005, Dick 1994). However, there are few data on soil enzyme activities available from the dry forests of eastern Washington.

Enzymes produced by soil microbes can provide robust measures of ecosystem health and sustainability.

We selected four extracellular soil enzymes to study: β-glucosidase, acid phosphatase, N-acetyl-glucosaminidase (chitinase), and polyphenoloxidase. β-glucosidase aids in accelerating litter decomposition by breaking down labile cellulose and other carbohydrate polymers (Burns 1978). Acid phosphatase is responsible for the mineralization of organic phosphorus (P) to inorganic P (Burns 1978, Ho 1979). High levels of both acid phosphatase and β-glucosidase indicate high organic matter quality, rapid mineralization, and high nutrient availability (Sinsabaugh and Moorhead 1997). Chitinase degrades chitin, and releases low molecular weight carbon (C)- and nitrogen (N)-rich compounds (Skujiņš 1978). Polyphenoloxidase is one of several enzymes involved in the degradation of lignin in plant litters and is in higher abundance where litter quality is low (Skujiņš 1978). Organisms most effective at lignin degradation are adapted to low-nutrient conditions (Killham 1994).

The objectives of this study were to determine (1) soil conditions (i.e., bulk density, moisture content, and mineral soil pH) in forests at the Mission Creek site on the east slopes of the Cascade mountains of Washington, (2) mineral soil enzyme activities across the landscape in different experimental units and

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landscape positions (ridge, slope, bare areas, and valley), and (3) whether enzyme activities are related to soil and stand conditions. We hypothesized that (1) soil enzyme activity would be highest in valleys and lowest in bare areas, (2) soil enzyme activity would be positively correlated with soil moisture, and (3) soil enzyme activity would be related to stem density and forest floor depth (indicators of organic matter inputs and decomposition rates).

Methods

Sampling Design

Four landscape positions were delineated on each of the 12 experimental units for soil enzyme analysis: (1) ridge—highest elevation area within plot, closed canopy, (2) slope—midslope, closed canopy, (3) bare—midslope in an overstory canopy gap, and (4) valley—lowest elevation area, closed canopy.

Field Sampling

Mineral soil samples were taken in August 2001. Soil was sampled with a 2.5-cm-diameter bulk density corer at 48 sampling locations (at each of the four landscape positions—ridge, slope, bare areas, and valley—on the 12 experimental units). Each soil core was taken to a depth of 8 cm below the forest floor, and the corer was sterilized with 95-percent ethanol between each use to reduce cross contamination. Soil cores were brought on ice to the laboratory at the College of Forest Resources, University of Washington, where soil was placed in a refrigerator at 3 °C.

Lab Analyses

Moisture content and bulk density were determined for each soil core. Soil was dried at a temperature of 105 °C for 24 hours. Soil pH was determined using a Radiometer pH meter (pHM-92), a pHC2051-8 combined pH electrode, and a 1:1 (volume:volume) soil to water paste.

For enzyme analyses, 3 g (wet weight) of soil from each sample core (< 2 mm fraction) were sent overnight on ice to Dr. Robert Sinsabaugh at the University of New Mexico, Albuquerque. Activities of β-glucosidase, acid phosphatase, chitinase, and polyphenol oxidase were determined. Enzyme assays were conducted using methods adopted from Saiya-Cork et al. (2002). Briefly, assays were conducted using 0.5 g of soil in sodium acetate buffer (50 mmol/L, pH 5.0) with either methyl-umbelliferyl enzyme substrates for β-glucosidase, acid phosphatase, and chitinase, or L-3, 4-dihydroxyphenylalanine (L-DOPA, 10 mmol/L, [Sigma-Aldrich, St. Louis, Missouri, USA]) as the substrate for polyphenoloxidase. Four analytical replicates per sample were used, and assays were conducted at 20 °C for 1 to 2 hours, using both substrate and sample controls. Enzyme activity was expressed as mmol • kg^{-1} • h^{-1} of dry weight mineral soil for each sample.

Statistical Analyses

Two-way Analysis of Variance (ANOVA) was used to determine if there were differences in soil enzyme activity relative to site and landscape feature (bare, ridge, slope, valley) using SPSS (2004). Two-way ANOVA was also used to determine if soil moisture content, soil bulk density, and soil pH were different by site and landscape position. Data were checked with respect to variance and normality. Soil enzyme data were natural-log transformed, and soil moisture data were square-root transformed. Tukey's Honestly Significant Difference (HSD) test was used to determine differences among means. Linear regression (SPSS 2004) was used to determine if enzyme activity was related to soil and stand variables. Significance levels were interpreted at $\alpha = 0.05$.

Results

Soil Moisture, Bulk Density, and pH

Soil characteristics were not significantly different among the 12 experimental units.

Soil characteristics (bulk density, moisture content, and pH) were not significantly different among the 12 experimental units ($p > 0.05$) (table 5-1). Bulk density ranged from 0.87 to 1.18 Mg/m^3 (average 1.05 Mg/m^3). Soil moisture ranged from 4.0 to 10.8 percent (average 7.6 percent) and pH ranged from 5.97 to 6.64 (average 6.16). There were also no significant differences in soil characteristics by landscape

Table 5-1—Soil moisture in August 2001, bulk density, and pH in the 0- to 8-cm mineral soil layer, by experimental unit at Mission Creek

Experimental unit	Gravimetric moisture	Bulk density	pH_{water}
	Percent	*Mg/m^3*	
Camas	8.9 (3.6)	0.99 (0.10)	5.98 (0.35)
Crow 1	8.5 (4.0)	1.08 (0.31)	6.14 (0.28)
Crow 3	7.1 (1.6)	1.11 (0.26)	6.04 (0.26)
Crow 6	6.9 (1.3)	1.17 (0.06)	6.15 (0.49)
Pendleton	6.3 (2.1)	1.18 (0.11)	5.97 (0.34)
Poison	7.3 (2.6)	1.01 (0.10)	6.18 (0.33)
Ruby	6.1 (2.3)	1.19 (0.11)	5.97 (0.23)
Sand 2	4.0 (1.7)	1.01 (0.04)	6.47 (0.45)
Sand 19	6.5 (1.8)	1.18 (0.20)	6.64 (0.78)
Slawson	8.5 (2.9)	0.87 (0.13)	6.26 (0.13)
Spromberg	10.8 (5.3)	0.89 (0.21)	6.04 (0.86)
Tripp	9.5 (2.7)	0.88 (0.07)	6.11 (0.22)
Average	7.6 (3.0)	1.05 (0.19)	6.16 (0.44)

Note: No significant differences in moisture, bulk density, and pH among experimental units ($p < 0.05$). Standard deviations are in parentheses ($n = 4$).

position except that soil moisture was significantly higher in the valleys than in bare canopy gaps ($p > 0.05$) (table 5-2). The lack of differences in bulk density and pH probably reflects the high variability in the data and low sample numbers. There was a tendency for lower bulk density and pH in valleys and higher bulk density and pH in bare canopy gaps (table 5-2).

Table 5-2—Mineral soil moisture in August 2001, bulk density, and pH in the top 8 cm by landscape position

Landscape position	Gravimetric moisture	Bulk density	pH_{water}
	Percent	*Mg/m^3*	
Bare	6.0 (1.9)a	1.10 (0.15)a	6.43 (0.56)a
Ridge	8.1 (3.8)ab	1.10 (0.23)a	6.06 (0.34)a
Slope	7.2 (1.8)ab	1.01 (0.18)a	6.15 (0.42)a
Valley	9.2 (3.3)b	0.97 (0.15)a	6.00 (0.32)a
Average (all)	7.6 (3.0)	1.05 (0.19)	6.16 (0.44)

Note: Values with different letters are significantly different among landscape positions ($p < 0.05$). Standard deviations are in parentheses ($n = 12$).

Soil Enzyme Activities

β-glucosidase and chitinase activities were significantly different among the 12 experimental units (table 5-3). β-glucosidase ranged from 172 to 823 mmol • kg^{-1} • h^{-1}, and chitinase ranged from 41 to 362 mmol • kg^{-1} • h^{-1}. The Camas site had the highest activities of both β-glucosidase and chitinase, and Crow 3 had the lowest activity. Acid phosphatase followed a trend similar to both β-glucosidase and chitinase, but activities were not significantly different among units and ranged from 85 to 328 mmol • kg^{-1} • h^{-1}. Polyphenoloxidase activity ranged from 0 to 317 mmol • kg^{-1} • h^{-1} (table 5-3); it was not detected in 8 of the 12 experimental units (table 5-3). Enzyme activity was not significantly different by landscape position for any of the four enzymes (table 5-4). There was a tendency for β-glucosidase, acid phosphatase, and chitinase to be highest in valleys, but not polyphenoloxidase.

Activity levels of β-glucosidase, acid phosphatase, and chitinase were negatively correlated with bulk density and were positively correlated with moisture content.

Relationships Among Soil Enzyme Activities and Soil and Stand Conditions

Enzyme activities were related to soil bulk density and moisture content (table 5-5), but not pH (data not shown in table 5-5). Activity levels of β-glucosidase, acid phosphatase, and chitinase were negatively correlated with bulk density (table 5-5), which explained < 30 percent of the variance in enzyme activity. β-glucosidase, acid phosphatase, and chitinase activities were positively correlated with moisture

Table 5-3—Enzyme activity in August 2001 by experimental unit at Mission Creek

Experimental unit	β-glucosidase	Acid phosphatase	N-acetyl-glucosaminidase (chitinase)	Polyphenol oxidase
		mmol • kg^{-1} • h^{-1}		
Camas	823 (326)a	245 (182)a	362 (205)a	0a
Crow 1	230 (68)bc	122 (96)a	53 (25)bc	0a
Crow 3	172 (61)c	85 (36)a	41 (30)c	0a
Crow 6	230 (57)bc	123 (28)a	88 (45)abc	0a
Pendleton	354 (195)abc	159 (73)a	78 (41)abc	0a
Poison	394 (340)abc	169 (81)a	91 (103)abc	0a
Ruby	422 (159)abc	158 (25)a	115 (96)abc	317 (634)a
Sand 2	409 (336)abc	118 (62)a	142 (118)abc	102 (205)a
Sand 19	419 (81)abc	259 (18)a	188 (107)abc	0a
Slawson	627 (108)abc	275 (165)a	187 (91)abc	141 (282)a
Spromberg	619 (200)abc	328 (298)a	240 (135)ab	0a
Tripp	702 (212)ab	242 (61)a	221 (92)ab	2 (3)a
Average	450 (203)	190 (77)	151 (93)	47 (97)

Note: Values with different letters in columns are significantly different ($p < 0.05$). Standard deviations are in parentheses (n=4).

Table 5-4—Enzyme activity in August 2001 by landscape position

Landscape position	β-glucosidase	Acid phosphatase	N-acetyl-glucosaminidase (chitinase)	Polyphenol oxidase
		mmol • kg^{-1} • h^{-1}		
Bare	472 (270)	150 (80)	117 (99)	106 (366)
Ridge	408 (245)	217 (183)	147 (141)	47 (163)
Slope	467 (307)	158 (79)	153 (134)	1 (2)
Valley	472 (272)	235 (144)	185 (140)	34 (118)
Average (all)	455 (267)	190 (130)	151 (128)	47 (205)

Note: No significant differences in any enzyme activity measured among Landscape Positions at $p < 0.05$.
Standard deviations are in parentheses (n = 12).

Table 5-5—Pearson correlation coefficients for relationships among soil enzyme activities and soil bulk density, soil moisture, stem density, and forest floor depth in experimental units at Mission Creek

	Bulk density[b]		Gravimetric moisture[b]		Stem density[c]		Forest floor depth[c]	
Enzyme[a]	*r*	*p*	*r*	*p*	*r*	*p*	*r*	*p*
β-glucosidase	-0.525	<0.001	+0.347	0.016	+0.751	0.005	-0.742	0.006
Acid phosphatase	-0.548	<0.001	+0.699	<0.001	+0.530	0.077	-0.540	0.077
Chitinase	-0.458	0.001	+0.439	0.002	+0.521	0.082	-0.761	0.004

[a] Phenol oxidase activity was not related to any variable measured.
[b] n = 48.
[c] n = 12.

content (table 5-5). Polyphenoloxidase activity was not related to any soil variable, but only four of the units had polyphenoloxidase activity.

Linear regression was also used to determine whether enzyme activity was related to slope, aspect, elevation, forest floor depth, canopy cover, tree basal area, and stem density. Enzyme activity was only related to stem density and forest floor depth (table 5-5). β-glucosidase activity was positively related to stem density and negatively related to forest floor depth (table 5-5). There were similar trends for acid phosphatase and chitinase activities, but the relationship was only significant for chitinase and forest floor depth (table 5-5). Polyphenoloxidase activity was not related to any stand variable. More of the variance in β-glucosidase activity was explained by stand variables than by soil variables, e.g., 56.4 percent for stem density and 55.1 percent for forest floor depth. Also 57.9 percent of chitinase activity was explained by forest floor depth. However, no single variable explained more than 60 percent of the variance in enzyme activity.

Discussion

Soil Moisture, Bulk Density, and pH

There was little variation in mineral soil bulk density, moisture content, and pH among the 12 experimental units, despite differences in soil types across the study area (Dolan 2002). Variations in slope, elevation, and aspect did not strongly influence the soil environmental factors. There were no significant differences relative to landscape position except for soil moisture, which was significantly higher in the valleys than in bare areas on slopes in canopy gaps. The time of year of sampling (summer) may have reduced differences in soil moisture content among study units. Soil moisture contents in this study were similar to those in a mixed-conifer dry forest at the Teakettle site in the Sierra National Forest, California, in summer (Marra and Edmonds 2005). Bulk densities at Teakettle, however, had a slightly greater range (0.27 g/cm^3 under canopies to 1.03 g/cm^3 in canopy gaps) than those at Mission Creek (0.97 g/cm^3 in valleys to 1.10 g/cm^3 in canopy gaps and ridges). Parent material at the Mission Creek site is largely sedimentary compared to granite at Teakettle. Soil pH values were similar to those in unburned forest plots in the Entiat Experimental Forest located just north of our study area (Grier 1975). However, they were lower than those in dry western juniper and ponderosa pine forests in eastern Oregon (Ho 1979).

Soil Enzyme Activities

Soil enzyme activities were based on soil mass rather than organic matter mass, so as to provide a robust ecosystem-level measure of soil health and productivity (Boerner et al. 2005). We found that β-glucosidase had the highest average activity in the top 8 cm of mineral soil (455 ± 267 [standard deviation] mmol • kg^{-1} • h^{-1}) followed by acid phosphatase (190 ± 130 mmol • kg^{-1} • h^{-1}), chitinase (151 ± 128 mmol • kg^{-1} • h^{-1}) and polyphenoloxidase (47 ± 205 mmol • kg^{-1} • h^{-1}). There are few studies of soil enzyme activity in the dry interior forest ecosystems of the Pacific Northwest with which to compare our data. Ho (1979) investigated acid phosphatase activity in *Juniperus occidentalis* (Hook.) and *Pinus ponderosa* forests in eastern Oregon and found activity levels of 114 and 119 mmol • kg^{-1} • h^{-1}, respectively, slightly less than the average of 190 mmol • kg^{-1} • h^{-1} we found. More recently, slightly lower β-glucosidase, and similar chitinase activities, were reported from soils in ponderosa pine forests in northern Arizona (Boyle et al. 2005).

Activity levels of all enzymes in our study were lower than those found in moister temperate deciduous and coniferous forests.

Overall, activity levels of all enzymes in our study were lower than those found in moister temperate deciduous and coniferous forests in Australia, Spain, Canada, Missouri, Ohio, and Michigan (Boerner and Brinkman 2003, Eivazi and Bayan 1996, Grierson and Adams 2000, Hassett and Zak 2005, Saa et al. 1993, Staddon et al. 1998, Waldrop et al. 2004). Low summer soil moisture in the Pacific Northwest greatly reduces enzyme activities. In comparison, Boerner et al. (2005) found higher activities in August for some enzymes in a moist oak forest soil in Ohio: acid phosphatase (1000 mmol • kg^{-1} • h^{-1} versus 190 at Mission Creek) and polyphenoloxidase (1500 mmol • kg^{-1} • h^{-1} versus 47 at Mission Creek). On the other hand, chitinase activity was similar in the oak forest (100 mmol • kg^{-1} • h^{-1}) to that at Mission Creek (151 mmol • kg^{-1} • h^{-1}). Grierson and Adams (2000) reported much higher acid phosphatase activities in a eucalyptus forest in southwestern Western Australia (9250 mmol • kg^{-1} • h^{-1} in the 0- to 10-cm layer), but also noted that activity varied with depth, being twice as high in the 0- to 5-cm layer than the 5- to 10-cm layer.

Tropical forests also have high enzyme activity. Olander and Vitousek (2000) found phosphatase levels of 7820 to 9686 mmol • kg^{-1} • h^{-1} and chitinase levels of 380 to 2760 mmol • kg^{-1} • h^{-1} in Hawaii along a soil chronosequence. Phosphatase was high at all their sites, but chitinase decreased as N availability increased. This difference in response of chitinase and phosphatase suggested that the N supply in older soils is high enough to trigger feedback and inhibit enzyme production, whereas the P supply was never high enough to trigger inhibition of phosphatase.

We were particularly interested in the variability in mineral soil enzyme activities across the 12 experimental units. Although we found few differences in soil environmental conditions among units, there were differences in some soil enzyme activities. There were significant differences in β-glucosidase and chitinase activity among the 12 experimental units, but not in acid phosphatase or polyphenoloxidase activity. This might have been expected considering the range in elevation, aspect, forest characteristics, and soil types at Mission Creek. The lack of significant differences in polyphenoloxidase and phosphatase activities among experimental units was probably due to the high variability in activities. For example, only four of the units had detectable activity levels of polyphenoloxidase.

Our findings suggest that Douglas-fir-dominated units had higher decomposition and mineralization rates compared to ponderosa pine.

Tree species composition may also influence enzyme activity (Grierson and Adams 2000). We found that units dominated by ponderosa pine (e.g., Crow 1, 3, and 6) tended to have lower activities of β-glucosidase, acid phosphatase, and chitinase compared to sites with high proportions of Douglas-fir (e.g., Camas) (table 5-3). This suggests that Douglas-fir-dominated units had higher decomposition and mineralization rates compared to ponderosa pine. β-glucosidase activity levels were positively related to chitinase activity. This was unexpected, as the two are typically negatively correlated with one another (Sinsabaugh et al. 1999). High levels of β-glucosidase activity are indicative of high organic matter quality, high nutrient availability, and increased mineralization (Burns 1978, Killham 1994). In contrast, high levels of chitinase activity are indicative of low N availability and poor organic matter quality. High chitinase in soils suggests the site supports large soil microbial communities of fungi, actinomycetes, and arthropods that would immobilize available nutrients, making them unavailable to plants until they die and decompose (Decker et al. 1999, Killham 1994). The Douglas-fir-dominated sites had higher activities of both enzymes, suggesting they had higher organic matter quality and a larger fungal population than the ponderosa pine-dominated site (Burns 1978). Even polyphenoloxidase activities tended to be higher in the Douglas-fir sites, and this enzyme was not detected in sites dominated by ponderosa pine. High polyphenoloxidase activity suggests that sites have low-quality litter and slow nutrient mineralization rates, where lignin makes up a larger portion of decomposable materials (Decker et al. 1999), but this seems to be the opposite of what we found.

In contrast to individual experimental units, there were no significant differences in enzyme activities relative to landscape position. This is contrary to our hypothesis that enzyme activity would be highest in valley areas and lowest in bare areas. There was considerable variation in the data, and we only examined soil enzyme activities in the summer when soils were dry. Differences may have been detected in other seasons. Other investigators have studied seasonal variability in soil enzymes. Boerner et al. (2005) found little seasonal effect on acid phosphatase and polyphenoloxidase activities in oak-dominated forests in Ohio in a humid climate. On the other hand, Grierson and Adams (2000) found seasonal differences in acid phosphatase in soil in a eucalyptus forest in Australia, and Boerner et al. (2005) noted high chitinase activity in spring in Ohio oak forest soils. Seasonal differences in enzyme activities at Mission Creek should be examined in the future.

Relationships Between Soil Enzyme Activities and Soil and Stand Conditions

We hypothesized that soil moisture would be a major factor contributing to variation in soil enzyme activity. Ross et al. (1984) found that temporal fluctuations in soil enzyme activity were mostly the result of soil moisture and were independent of soil organic C and N. β-glucosidase, acid phosphatase, and chitinase activities were positively correlated with moisture content; soil moisture explained 48.9 percent of the variance in acid phosphatase. Grierson and Adams (2000) and Boerner et al. (2005) also showed that soil moisture has a strong influence on activity of some soil enzymes, particularly chitinase. Similarly, Boyle et al. (2005) found that soil moisture was correlated with β-glucosidase and chitinase activities in a Southwestern U.S. ponderosa pine forest during the summer monsoon period.

Soil enzyme activity was negatively related to bulk density, which explained < 30 percent of the variation in enzyme activity. Dick et al. (1988) also found soil enzyme activity to be negatively correlated with soil bulk density on compacted soils in a 4-year-old clearcut in western Oregon. Many of our experimental units have been subjected to management activities in the past, which may have affected bulk density. Low bulk density favors soil microbes and invertebrates and is usually associated with a relatively structured soil profile with developed systems of pore channels and spaces and a soil organic matter content higher than that of soils with high bulk densities (Marra and Edmonds 2005). There was no relationship between soil enzyme activity and soil pH probably because of the narrow range of soil pH at Mission Creek (6.00 to 6.43).

We examined potential relationships among soil enzyme activities and stand variables, including slope, aspect, elevation, forest floor depth, canopy cover, tree basal area, and stem density. The experimental units differed considerably in slope, aspect, and elevation. However, enzyme activities were only significantly related to stem density and forest floor depth. We hypothesized that units with high stem densities (No. trees/ha) would have increased enzyme activities. Higher tree density would provide more litterfall, therefore stimulating soil microbes and overall enzymatic activity. β-glucosidase activity was positively related to stem density, supporting our hypothesis. There were similar trends for acid phosphatase and chitinase activities. Polyphenoloxidase activity, however, was not related to any stand variable.

Deeper forest floors indicate slower litter decomposition, greater litter input, or both. We hypothesized that soil enzyme activity would be negatively related to forest floor depth. β-glucosidase and chitinase activities were significantly negatively related to forest floor depth, and there was a trend for a similar relationship with acid phosphatase, supporting our hypothesis. These data support the concept that β-glucosidase, chitinase, and acid phosphatase aid in accelerating litter decomposition and releasing N and P in soils (Burns 1978, Ho 1979, Sinsabaugh and Moorhead 1997, Skujiņš 1978). Overall, more of the variation in enzyme activity was explained by stand variables than by soil variables.

In conclusion, we found a wide range of variability in the activity of the four enzymes in the top 8 cm of mineral soil at Mission Creek. Activity levels were generally low compared to those in other temperate forest ecosystems. β-glucosidase had the highest activity across experimental units averaging 450 mmol • kg^{-1} • h^{-1}. Acid phosphatase was next highest (190 mmol • kg^{-1} • h^{-1}), followed by chitinase (151 mmol • kg^{-1} • h^{-1}), and polyphenoloxidase (47 mmol • kg^{-1} • h^{-1}). Chitinase and β-glucosidase activities were significantly different among the 12 units, but acid phosphatase and polyphenoloxidase were not. Units with a higher proportion of Douglas-fir tended to have higher enzyme activity than those dominated by ponderosa pine. Enzyme activity was not significantly different by landscape position. Soil characteristics (bulk density, soil moisture, and pH) were not significantly different among units or by landscape position, except that soil moisture was significantly higher in the valleys than in bare areas in canopy gaps. β-glucosidase, acid phosphatase, and chitinase activities were negatively correlated with soil bulk density and positively correlated with soil moisture. β-glucosidase activity was also positively correlated with stem density and negatively correlated with forest floor depth. Polyphenoloxidase activity was not correlated with any measured variable. No single variable explained more than 60 perecent of the variance in a given enzyme activity.

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Chapter 6: Soil Microarthropods

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Introduction

Selective logging and fire suppression over the past century have changed the tree species in the eastern Washington Cascade Mountains as well as increasing tree density and fuel loading. Fires that occur now tend to burn more catastrophically than historical, low-intensity, pre-European-settlement fires. There is currently much interest in using fire and thinning in these forests as management tools to reduce the incidence of high-intensity fires.

The influence of fire and thinning treatments on soil organisms, such as soil microarthropods, is not well understood. Soil microarthropods are sensitive bio-indicators of environmental change because of the fine grain at which they occupy the soil and their rapid reproductive rates (Moldenke and Lattin 1990). They are important contributors to soil functioning (Seastedt 1984, Seastedt and Crossley 1987). Through their interactions with soil microbes, they regulate decomposition and nutrient mineralization processes and increase soil structure and porosity (Asquith et al. 1990, Killham 1994, Stork and Eggleton 1992). In particular, soil microarthropods mobilize nutrients, provide food for soil micro-organisms through their feces, and transport bacteria and fungi (Behan-Pelletier 1999, Moldenke 1990, Moldenke et al. 2000, Norton and Behan-Pelletier 1991).

Soil microarthropods mobilize nutrients, provide food for soil micro-organisms, and transport bacteria and fungi.

Mites and Collembola dominate in conifer ecosystems (Marra and Edmonds 1998, 2005). Landscape position can strongly influence them, particularly in mountainous terrain where vegetation may be patchy. For example, Marra and Edmonds (2005), working in the Sierra Nevada Mountains of California, found differences in mite and Collembola taxa and densities under closed-canopied forests compared to those in canopy gaps. In the eastern Washington Cascades, there is considerable topographic variation in the landscape. Steep slopes, sandy soils, rock outcroppings, and low precipitation create patchy vegetation cover (Dolan 2002, Lillybridge et al. 1995). Little information is available on soil microarthropods in the dry forests of eastern Washington, especially in relation to landscape position.

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Data on forest soil arthropod taxonomic richness in different landscape positions and habitats can provide understanding of how ecosystems respond to forest management practices and ecological change (Moldenke et al. 2000). The study described here provides data on soil microarthropods at the Mission Creek site of the National Fire and Fire Surrogate Study, which was implemented to examine the ecological consequences of fire and thinning treatments.

The specific objectives of our study were to determine (1) pretreatment soil environmental conditions (i.e., soil moisture, bulk density, and pH), (2) pretreatment soil microarthropod density and taxonomic richness, (3) the effect of soil environmental conditions on soil microarthropod density and taxonomic richness, (4) the influence of landscape position (ridge, slope, open canopy gap, and valley) on soil microarthropod density and taxonomic richness, and (5) the influence of soil depth (0 to 4 and 5 to 9 cm) on soil microarthropod density and taxonomic richness.

Materials and Methods

Sampling Design

In May 2001, the 12 Mission Creek experimental units were sampled, but only 9 of the units were processed for invertebrates (Camas, Crow 1, Crow 3, Crow 6, Pendleton, Poison, Sand 2, Sand 19, and Spromberg). These units ranged in area from 9.5 to 20 ha and ranged in elevation from 640 to 1219 m. Each unit had four landscape positions: (1) ridge, (2) slope, (3) open canopy gap, and (4) valley. Slope positions were located between the ridges and valleys on the side of a mountain, and canopy gaps were defined by the absence of overstory vegetation in an area of at least 0.25 ha. Two samples were collected at each landscape position at each of the nine experimental units adjacent to the unit soil pit, for a total of 72 samples. One sample from each landscape position in each experimental unit was used to determine soil moisture, bulk density, and pH (36 samples), and the other sample at each landscape position was used for microarthropod extraction (36 samples). All sampling locations were mapped to follow ecosystem responses through time.

Soil Sampling

Soil was collected with a soil corer (5.4-cm diameter) at each of the 72 sampling locations. Each soil core was taken from the top of the forest floor to a depth of 10 cm in the mineral soil and divided into two samples, 0 to 4 cm and 5 to 9 cm depths. The 4-5 and 9-10-cm soil was not used. The volume of each soil sample was 91.6 cm^3. Soil cores were brought on ice to the laboratory in the College of Forest Resources, University of Washington, Seattle, Washington, and stored at 3 °C.

Lab Analyses

Microarthropods were extracted using a modified high-gradient Berlese method (Moldenke 1994). Soil samples were left in the arthropod extractors for 14 days. Arthropods fell into antifreeze (polyethylene glycol) and were stored until identified. Soil samples were processed for soil moisture, bulk density, pH, and arthropods within 4 days of field sampling. Oribatid mites were identified to genus, if possible, and then to morphospecies (Oliver and Beatie 1996). All other arthropods and annelids were identified to class, order, or family, depending on local identification expertise. Bulk specimens were sorted and identified using a dissection scope boosted to a magnification of up to 120×. A reference collection was established for each species. Some morphospecies identified with a dissection scope were found to contain more than one genus or species when viewed with the higher magnification compound scope. Since it was not possible to mount and identify all specimens with a compound scope, some species groups listed contain more than one genus or species. For data analysis, taxa were composited across the 9 experimental units by depth and landscape position.

Moisture content and bulk density samples were dried at 105 ºC and both were expressed as Mg/m^3 on a dry mass basis. Soil pH was determined using a Radiometer pH meter (pHM-92), a pHC2051-8 combined pH electrode, and a 1:1 (volume: volume) soil to water paste.

Statistical Analyses

One-way analyses of variance (ANOVA) were used to compare differences among number of taxa and density (number/m^3) of oribatid mites and Collembola for different soil depths and landscape positions using SPSS (2004). One-way ANOVAs were also used to compare differences in soil moisture content, bulk density, and pH among landscape positions. Tukey's Honestly Significant Difference (HSD) test was used to determine differences among means. Linear regression was used to determine relationships among oribatid mite and Collembola density and taxonomic richness and soil moisture, bulk density, and pH using SPPS (2004). Pearson's correlation coefficients were calculated. Significance was determined at $\alpha \leq 0.05$.

Results

Soil Moisture, Bulk Density, and pH

Soil moisture content was not significantly different among landscape positions (table 6-1). Bulk density of the soil was significantly higher in the open canopy gaps (0.88 Mg/m^3) than in valleys (0.55 Mg/m^3) ($p \leq 0.05$), while slope and ridge

Table 6-1—Mean soil moisture, bulk density, and pH for different landscape positions, May 2001

Landscape position	Soil moisture	Bulk density	pH_{water}
	Mg/m^3	*Mg/m*^{3}	
Ridge	0.10 (0.03)a	0.65 (0.18)ab	6.40 (0.61)a
Slope	0.10 (0.04)a	0.65 (0.14)ab	6.02 (0.38)a
Open canopy gap	0.10 (0.04)a	0.88 (0.23)a	6.22 (0.46)a
Valley	0.13 (0.05)a	0.55 (0.22)b	5.98 (0.32)a

Note: Values with different letters in the same column are significantly different ($p \leq 0.05$). Standard deviations are in parentheses ($n = 9$).

bulk densities were intermediate. Bulk densities were low because forest floor material was included. Soil pH ranged from 5.98 to 6.40 and did not differ significantly among landscape positions.

Microarthropod Taxa and Influence of Landscape Position

Mites (Acarida), including Mesostigmata, Oribatida, and Prostigmata were the dominant microarthropods present along with Collembola. Oribatid mites had the highest number of adult taxa (69). Taxa of oribatid mites and Collembola sampled in May 2001 are shown in table 6-2. Insects sampled were Coleoptera (beetles), Diptera (flies), Hemiptera (true bugs), Hymenoptera (ants, bees, and wasps), Lepidotpera (moths and butterflies), Psocoptera (barklice), Thysanoptera (thrips), and Thysanura (silverfish). Other invertebrates present were Diplura (two-pronged bristletails), Araneae (spiders), Chilopoda (centipedes), Diplopoda (millipedes), Annelida (segmented worms), and Pauropoda (pauropods). Mites were the most numerically dominant group, making up 84 percent of the total soil arthropods extracted. Collembola made up 7 percent of the total, and all other invertebrates made up 9 percent.

Taxonomic richness of oribatid mites and Collembola was not significantly different among soil depth and landscape positions

Taxonomic richness of the oribatid mites was not significantly different among soil depth and landscape positions (table 6-3). The average number of taxa per sample ranged from 5.0 in the open canopy gaps at the 5- to 9-cm depth to 11.8 in the valleys at the 0- to 4-cm depth. Average oribatid mite densities ranged from 327,511/m^3 in ridges at the 5- to 9-cm depth to 1,146,288/m^3 in the open canopy gaps at the 0- to 4-cm depth (table 6-3) and were not significantly different among landscape positions. However, densities were significantly higher in the 0- to 4-cm depth than the 5- to 9-cm depth ($p \leq 0.05$).

Collembola were only identified to family. Taxonomic richness of Collembola was not significantly different between soil depths and among landscape positions (table 6-4). The average number of taxa per sample ranged from 0.7 in the ridges at the 0- to 4- and 5- to 9-cm depths and in the open canopy gaps at the 5- to 9-cm

Table 6-2—Taxa of oribatid mites (genera and superfamilies) and Collembola (families) found at the Mission Creek site

Taxonomic level	Genera or family name	Taxonomic level	Genera or family name
Oribatid genera	*Achiptera* (1 species)	Oribatid genera	*Maerkelotritia* (3 species)
	Achiptera immature		*Phauloppia*
	Anachipteria (3 species)		*Quadroppia*
	Camisia		*Scheloribates* (4 species)
	Carabodes (2 species)		*Sphaerochthonius*
	Caenobelba		*Suctobelbella*
	Ceratoppia		*Tectocepheus*
	Ceratozetes		Unknown genera (20 species)
	Cosmochthonius		
	Cultroribula	Oribatid superfamily	*Crotonioidea*
	Ctenacarus		*Damaeoidea* (3 taxa)
	Eniochthonius		*Damaeoidea* immature (2 taxa)
	Eporibatula		*Gustavioidea* immature (2 taxa)
	Eremaeus (4 species)		*Phenopelopoidea* immature
	Eremaeus immature (1 species)		*Plateremaeoidea* (2 taxa)
	Hermaniella		
	Liacarus (4 species)	Collembola family	*Entomobryidae*
	Neoribates		*Hypogastrura*
	Odontodamaeus (2 species)		*Isotomidae*
	Oppia		*Onychiuridae*
	Oppiella		*Sminthuridae*
	Oribatella (2 species)		

Table 6-3—Mean oribatid mite richness and density relative to soil depth and landscape position

	Ridge		Slope		Open canopy gap		Valley	
Metric	0–4 cm	5–9 cm	0–4 cm	5–9 cm	0–4 cm	5–9 cm	0–4 cm	5–9 cm
Richness (number of taxa)	7.9	5.2	9.3	6.9	8.0	5.0	11.8	7.3
(standard deviation)	(6.6)	(4.8)	(6.3)	(5.3)	(5.9)	(4.9)	(5.8)	(5.3)
Density (number/m^3)	906,114	327,511	1,048,035	698,690	1,146,288	425,764	1,102,620	665,939
(standard deviation)	(873,362)	(393,013)	(917,031)	(807,860)	(1,189,956)	(502,183)	(545,852)	(753,275)

Note: Average number of taxa per sample unit was not significantly different among landscape positions and depths ($p \leq 0.05$). Individual sample unit = 91.6 cm^3 (n = 9).

Average density was not significantly different among landscape positions, but was significantly higher in 0–4-cm depth than 5–9-cm depth ($p \leq 0.05$).

Table 6-4—Collembola richness and density relative to soil depth and landscape position

	Ridge		Slope		Open canopy gap		Valley	
Metric	0–4 cm	5–9 cm	0–4 cm	5–9 cm	0–4 cm	5–9 cm	0–4 cm	5–9 cm
Richness (number of taxa)	0.7	0.7	1.1	0.9	0.8	0.7	1.9	1.6
(standard deviation)	(0.9)	(0.9)	(1.3)	(0.8)	(1.4)	(0.9)	(1.6)	(1.0)
Density (number/m^3)	120,087	65,502	32,751	87,336	21,834	10,917	98,253	98,253
(standard deviation)	(316,594)	(152,838)	(43,668)	(163,755)	(54,585)	(21,834)	(87,336)	(120,087)

Average number of taxa per sample unit and density were not significantly different among landscape position and depths ($p \leq 0.05$). Individual sample unit = 91.6 cm^3 (n = 9).

depth to 1.9 in the valleys at the 0- to 4-cm depth. Average Collembola densities ranged from 10,917/m^3 in gaps at the 5- to 9-cm depth to 120,087/m^3 on ridges at the 0- to 4-cm depth (table 6-4), but were not significantly different by depth and landscape position because of the large variance.

Relationships Between Oribatid Mite and Collembola Taxa Richness and Density and Soil Moisture, Bulk Density and pH

There were no significant relationships between oribatid mite taxa richness or density in the 5- to 9-cm layer and soil environmental variables except that mite density and pH were correlated (r = -0.97, p = 0.03, n = 4) as were Collembola density and soil bulk density (r = -0.97, p = 0.03, n = 4).

Discussion

Oribatid mites and Collembola were the most numerous soil microarthropods at Mission Creek. This is similar to reports for the dry soils of the Teakettle Experimental Forest in the southern Sierra Nevada Mountains of California (Marra and Edmonds 2005) and for the moist soils on the Olympic Peninsula, Washington (Marra and Edmonds 1998). There were more mites (84 percent of invertebrates) than Collembola (7 percent) at Mission Creek. Similarly, Marra and Edmonds (2005) found that mites made up 78 percent of invertebrates sampled in spring in the top 8 cm of soil at Teakettle, whereas Collembola made up 13 percent. Studies in other conifer (mediterranean pine, mixed-conifer) and hardwood (birch, red oak, red maple) forest types also found greater numbers of mites than Collembola in the soil (Ojala and Huhta 2001, Peck and Niwa 2005, Reynolds et al. 2003, Tsiafouli et al. 2005).

Mites tend to dominate undisturbed habitats and forest soils, whereas Collembola are more dominant in managed grasslands and arable lands.

Mites tend to dominate undisturbed habitats and forest soils, whereas Collembola are more dominant in managed grasslands and arable lands (Filser 2002). The high incidence of mites in forest soils across the landscape could be due to the lack of mite predators such as thrips and predatory mites, but there are few studies examining forest soil predator-prey relationships (Anderson 1978, Halliday 2005, Naher et al. 2005). The thick cuticles and leg protection mechanisms of adult oribatid mites probably decrease the probability of predation (Norton 1994).

Sixty-nine total taxa of mature oribatid mites occurred at Mission Creek, only slightly fewer than the number of taxa found in the moist soils of the Olympic Mountains of Washington, which involved a similar sampling method (Marra and Edmonds 1998). The higher number of taxa at the Olympic Peninsula site may be due to the milder climate on the Olympic Peninsula compared to that at the Mission

Creek site, which is drier and has more extreme temperature fluctuations. It could also be because the taxa at Mission Creek are adapted to unique habitat niches, represented by the different landscape positions we sampled.

Similar or lower total oribatid mite taxa have been reported from other forest soils in North America and Europe. For instance, 22 oribatid mite taxa were found in an oak forest in Greece (Stamou and Sgardelis 1989), 51 in a mixed-conifer forest in California (Marra and Edmonds 2005), 61 in a hardwood forest in New York (Minor and Cianciolo 2007), 71 in the organic layer of spruce stands in Germany (Zaitsev et al. 2002), and 73 in a hardwood forest in North Carolina (Johnston and Crossley 1993).

Landscape position at Mission Creek did not significantly influence the taxonomic richness of mites. However, there was a tendency toward lower richness of oribatid mites on ridges and in open canopy gaps, suggesting that conditions in these environments were less favorable for mites than those in slopes and valleys. Landscape position also did not significantly influence Collembola taxonomic richness, but the largest average numbers were in the valley position. Collembola have close associations with roots and plant communities (Hiol et al. 1994). Thus Collembola richness patterns could be related to higher root production and biological activity occurring within rhizospheres and rhizoplanes in valley positions (Curl et al. 1988, Hiol et al. 1994, Maire et al. 1999). Collembola can feed on a wide variety of organisms, ranging from algae to invertebrates to living plant matter (Rusek 1998, Thimm and Larink 1995). Many species of Collembola are generalists, but some have been shown to be highly selective on fungi in soils (Jorgensen et al. 2005, Marshall 1978, Scheu and Falca 2000, Thimm and Larink 1995).

Because Collembola are sensitive to drought, their taxonomic richness has been related to vegetation type in areas with dry soils (Ehrnsberger et al. 1997; Hagvar 1982; Materna 2004; Rusek 1993, 1994). Soil moisture may influence Collembola because some species lack true tracheal systems and are prone to desiccation (Joosse 1981, Tsiafouli et al. 2005). However, we found that Collembola taxa and soil moisture were not strongly related, and soil moisture was not significantly different by landscape position.

Bulk density also influences oribatid mites and Collembola. Low bulk density tends to be favorable to many soil arthropods because soils with low bulk density usually have more pore spaces and higher soil organic matter content than soils with high bulk densities (Marra and Edmonds 2005). Pore size is positively correlated with microarthropod abundance (Vreeken-Buijs et al. 1998). Many Collembolan species feed on fungi, and soil with greater pore sizes allows for greater fungal growth and sporulation. Furthermore, the presence of small invertebrate species may serve as food sources for larger invertebrates (Anderson et al. 1984, Visser 1985, Vreeken-Buijs et al. 1998). Marra and Edmonds (2005) found that bulk density explained a high amount of the variance in invertebrate populations in the California Sierra Nevada Mountains. We also found a significant negative relationship between Collembola density and soil bulk density at the Mission Creek site.

There was little variation in soil pH across our experimental units (5.98 to 6.40), and pH did not strongly influence Collembola communities. However, we did find a significant relationship between oribatid mite density and soil pH, with higher densities in the more acidic environments.

In conclusion, we found soil microinvertebrates in all landscape positions (ridges, slopes, open canopy gaps, and valleys), with oribatid mites and Collembola being the most abundant. Oribatid mites had a higher number of taxa and higher densities than Collembola. There were no significant differences among landscape positions with respect to mite or Collembola taxonomic richness or density, except that oribatid mite density was significantly higher in the 0- to 4-cm layer than the 5- to 9-cm layer. Collembola density was negatively correlated with bulk density. No soil environmental variables were significantly correlated with oribatid mite taxa richness and density except pH, which was negatively correlated with mite density.

Collembola density was negatively correlated with bulk density and pH was negatively correlated with mite density.

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Chapter 7: Pretreatment Variability in Bark Beetle, Root Disease, and Dwarf Mistletoe Incidence and Severity

Paul F. Hessburg, R. Brion Salter, and Anne C. Conway[1]

Introduction

A primary objective of dry forest stand silvicultural treatments is to reduce the overstory and understory tree density while favoring large-diameter trees and fire-tolerant species. Thinning and burning treatments have the potential to either increase or decrease root disease and bark beetle mortality and dwarf mistletoe incidence. For example, after treatments, residual trees with larger diameters may occur in clumps where basal area per hectare may be higher, which may be attractive to bark beetles.

Thinning and burning treatments have the potential to either increase or decrease root disease and bark beetle mortality and dwarf mistletoe incidence.

Similarly, dwarf mistletoe infections in dry mixed-conifer stands may show considerable latency when stands are relatively dense, not fully expressing the presence of all infections via visible mistletoe plants or significant witches brooms until stands have been released by thinning or burning. In such cases, the residual trees and the infections both tend to respond significantly to release. Thus, it is important to measure and compare pre- and posttreatment incidence and severity to determine whether treatments are producing favorable effects on native pathogen and insect disturbances.

Methods

In this segment of the study, 10 contiguous hectares of each experimental unit were systematically surveyed; all trees showing signs, symptoms, or evidence of tree decline or mortality were evaluated for evidence of attack by native root pathogens or bark beetles. A 10-ha survey area was chosen because the smallest experimental units (Camas, Crow 3, and Slawson) were approximately 10 ha (fig. 7-1). All dead and symptomatic trees of each experimental unit were recorded on 1:900-scale orthophotograph field maps.

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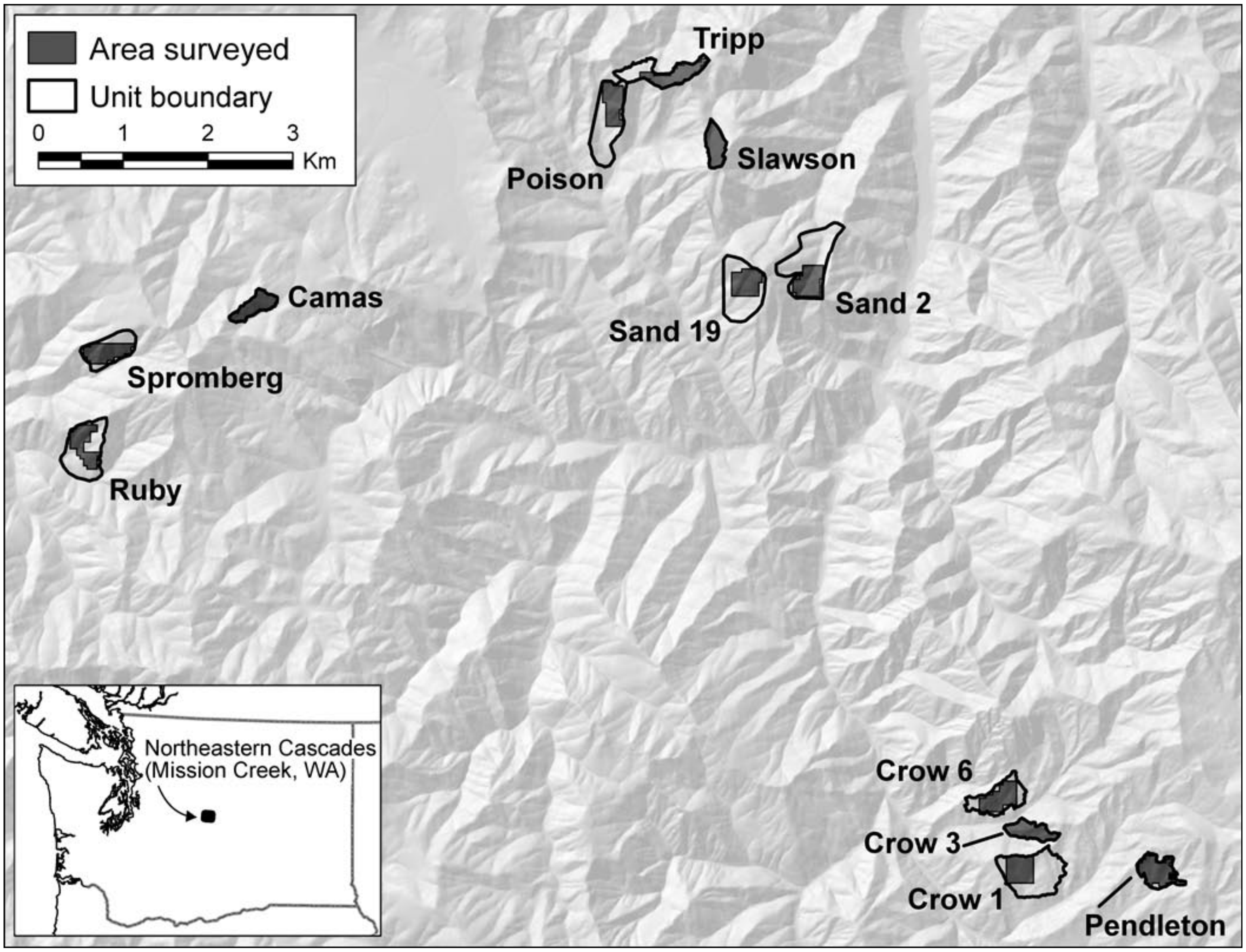

Figure 7-1—Vicinity map of the Mission Creek Fire and Fire Surrogate Study Area showing experimental unit boundaries and sampled areas.

Field maps were generated in a geographic information system (GIS) by overlaying a high-resolution (0.2-m ground resolution) color digital orthophotograph with a 40-m sampling grid and 40-ft (12-m) elevation contours. The sampling grid spatial layer was created with the ArcInfo (Environmental Systems Research Institute 2006) GENERATE command and was anchored on universal transverse mercator (UTM) coordinates that were evenly divisible by 40 m to ease navigation intervals in the field.

We generated the color digital orthophotos by georeferencing high-resolution (15 µm) digitally scanned images of standard resource aerial photograph negatives, which we then digitally flipped to photo-positives. The aerial photo scans were acquired from the USDA Farm Services Agency (FSA) Aerial Photography Field Office (APFO) and had an original photo scale of 1:15,840. Orthorectification was conducted with ERDAS Imagine[2] (Leica Geosystems 2006) software, and the high-resolution orthophotos were horizontally georeferenced to the 1998 USDA Forest Service, 1-m digital orthoquadrangles, and vertically to USGS 7.5-minute digital elevation models (10-m DEM). The field-mapped trees were then digitized on-screen with ArcView software (Environmental Systems Research Institute 2006) over the same digital orthophoto and 40-m grid as the field maps. This constituted the initial spatial data layer of tree locations; the orthophotos functioned as high-resolution aerial photographs and georeferenced maps of tree locations such that little time was needed in the field to georeference tree data. We simply needed to circle the relevant trees on the photos when we recorded data on them.

The dead and symptomatic trees ≥ 12.7 cm (5.0 in) diameter at breast height (d.b.h.) were identified in each experimental unit and mapped directly on the orthophotos in the field, and site and tree attribute data were recorded. Site and tree data included:

1. Plant association
2. Elevation
3. Aspect
4. Experimental unit number
5. UTM grid cell alphanumeric code (letters-rows, numbers-columns)
6. Tree number (restarted numbering each UTM grid cell)
7. Tree height (m)
8. Tree d.b.h (cm)
9. Tree species
10. Tree status (dead, symptomatic, nonsymptomatic)
11. Damaging agent(s)
12. Tree live crown ratio (LCR)
13. Tree radial growth (last 10 yr growth measured in 20ths of an inch, e.g., 13 = 13/20 in)
14. Bark beetle species present
15. Extent of bark beetle activity

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

16. Presence/absence of woodborers
17. Extent of any woodpecker foraging activity
18. Type of woodpecker foraging (**hits**—deep round holes, **scaling**—flaked-away outer bark)
19. Identity of any sap-rotting fungi present
20. Extent of sapwood decay

Attribute data were entered into a database, and this database was joined with the spatial data of tree locations based on a unique identifier, which included site, grid cell, and grid-cell-specific tree number codes (e.g., 11-C8-1).

For pathology surveys, we implemented a 100-percent field survey of 10 ha of each experimental unit.

Identifying Tree-Killing Root Pathogens

The primary purpose of the national pathology protocols for all installations was to survey for root disease and treatment effects on root diseases. For the pathology pre- and posttreatment surveys, we implemented a 100-percent field survey of 10 ha of each experimental unit because we anticipated that we would find little to no root disease on these dry plant associations. We surveyed and mapped all existing root-disease-affected trees. Root-disease-affected trees and mortality centers were georeferenced in the field to high-resolution orthorectified 1:900 resource aerial photographs by means the 40-m UTM grid and a Mylar® overlay. Individual diseased and symptomatic or killed trees were mapped to the full extent of the drip line of each affected tree.

During the pre- and posttreatment surveys, we identified and mapped all trees showing new symptoms or mortality associated with any native root disease pathogen. Root pathogens of interest were *Phellinus weirii* (PHWE), the cause of laminated root rot, *Heterobasidion annosum* (HEAN) (both S- and P-groups), the cause of annosum root disease, *Armillaria ostoyae* (AROS), the cause of Armillaria root disease, and *Leptographium wageneri* (LEWA), the cause of black stain root disease.

Minimally invasive root collar excavations were used to locate and identify possible root pathogens in symptomatic trees. We exposed a major lateral root to a distance of 1 m from the root collar and looked for signs of infection such as presence of rhizomorphs on or around the root collar or root, laminated decay with pitting on one or both sides, setal hyphae, mycelial fans beneath the bark, resin-soaked straw-colored decay with characteristic flecking, basal resinosus, or resin-soaked bark, and evidence of mushrooms or other more conk-like basidiomes in stumps, under or on roots, or at the root collar.

Specimens were collected from the root of symptomatic trees by cutting a wedge, with bark intact, using a hand pruning saw. The specimen was then wrapped in tree flagging to secure the bark and placed in a sealable plastic sandwich bag. The root was then recovered with soil. Putative rhizomorphs were collected and placed in plastic drinking straws, the straw ends were crimped and flame-sealed. All specimens were marked with date, species code, and unique identifier showing the site number, the grid cell number, and the tree number of that grid cell. Specimens were placed in cold storage until weekly shipments in coolers on dry ice were made to the Moscow, Idaho, Forestry Sciences Laboratory for identification.

Isolations from infected root samples and rhizomorphs were conducted using standard protocols and cultural media (Hoff 2002, Hoff et al. 2004a). Any putative *Armillaria* spp. isolates were identified to genet and North American Biological Species by our cooperator Dr. Ned Klopfenstein or his students using established tester cultures and amplified restriction fragment length polymorphism molecular assays (Hoff 2002; Hoff et al. 2004a, 2004b). We also isolated and identified any fungal endophytes that were present in roots (Hoff 2002; Hoff et al. 2004a, 2004b).

In addition to collecting samples from visibly symptomatic trees, trees without visible symptoms of root disease or other damaging agents were surveyed in a systematic random sample for HEAN and *Armillaria* spp. because these diseases could be present at low levels or as yet unexpressed. Nonsymptomatic trees were selected in every other grid cell, forming a checkerboard pattern. The first apparently healthy tree encountered north of the SW corner of each sampled grid cell was used as a nonsymptomatic sample tree. We then recorded the site and tree data, as described above, examined one major lateral root to a distance of 1 m from the root collar, and removed and placed a root core in a soda straw; the straw was sealed and labeled with a unique identifier, cold stored, and shipped as previously described for later identification.

Identifying Tree-Killing Bark Beetles

For the entomology pre- and posttreatment surveys, we implemented a 100-percent field survey of a 10-ha area of each experimental unit. We surveyed and mapped all existing bark-beetle affected-trees. Bark-beetle-affected trees and mortality centers were likewise georeferenced in the field to high-resolution orthorectified 1:900 resource aerial photographs by means of the 40-m UTM grid and a Mylar® photo overlay. Individual bark-beetle-infested or killed trees were mapped as the full extent of the drip line of each affected tree.

Bark beetles of primary interest in Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) were *Dendroctonus pseudotsugae* (DEPS), the Douglas-fir beetle, and *Scolytus unispinosus* (SCUN), the Douglas-fir engraver. In ponderosa pine (*Pinus ponderosa* Dougl. ex Laws), the bark beetles of interest were *D. ponderosae* (DEPO), the mountain pine beetle; *D. brevicomis* (DEBR), the western pine beetle; *D. valens* (DEVA), the red turpentine beetle; and *Ips* spp. In addition, we surveyed for *Scolytus ventralis* (SCVE), the fir engraver, on grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.), which occurred in only four units (Camas, Crow 3, Ruby, and Poison). Bark beetles were identified by the host species attacked, the characteristic egg and larval gallery patterns, the presence and color of boring frass, and by the presence and number of gallery ventilation holes.

Survey crews were trained in the field to identify dry forest pathogen and insect damages by their signs and symptoms. They were also given line drawings, photos, and keys depicting these diagnostic criteria. Diagnostic aids were used to link the field training with the field identification procedure, and to establish positive identifications. Galleries were examined in the field by removing a section of bark with a Pulaski or cruiser's axe in the vicinity of pitch tubes, pitch streaming, woodpecker activity, and/or boring frass.

Recording Woodpecker Foraging

We recorded the vertical extent of the bole with visible signs of woodpecker foraging by identifying the section(s) that displayed evidence of woodpecker foraging. There were two types of foraging recognized: subsurface **scaling** or flaking of the bark, and **hits**, a narrow horizontal cone of bark removal to the cambium that was required to remove deeply buried woodborer or bark beetle larvae.

Scaling only occurred on ponderosa pine that had been mass attacked by the western pine beetle.

Scaling only occurred on ponderosa pine that had been mass attacked by the western pine beetle, where woodpeckers would find late instar larvae migrating outward in the outer bark. Teneral (callow) adults ultimately emerge from a region just beneath the outer bark surface. **Hits** were associated with woodpecker foraging for bark beetles and woodborers that complete their development in the cambial region between the inner bark and secondary xylem.

Woodpeckers were found **hit** foraging on most Scolytid (*Ips* spp., *Dendroctonus* spp., *Scolytus* spp.) bark beetles (excluding the western pine beetle, *Dendroctonus brevicomis*) and most woodborers (Coleoptera/Cerambycidae, long-horned borers; Coleoptera/Buprestidae, metallic woodborers). Furthermore, woodpeckers tended to produce a large hit-foraging excavation hole to remove woodborer larvae, which tend to be much larger at maturity than bark beetle larvae. Each bole segment displaying woodpecker **hit** or **scaling** foraging behavior was recorded by entering the starting and ending vertical heights of the bole segments displaying such foraging. The lengths of bole segment(s) were ocularly estimated to the nearest 1.5 m. To calibrate the observer's eye, ocular estimates were checked each day against clinometer measurements of the same.

We also recorded the presence and extent of saprot basidiomes (fruiting bodies) such as that associated with *Cryptoporus volvatus*, the pouch fungus, and *Hirschioporus abietinum*, the purple conk. The presence of the pouch fungus is highly indicative of prior bark beetle mass or strip attack episodes.

Recognizing Dwarf Mistletoe Incidence and Severity

Because the characterization of pretreatment dwarf mistletoe conditions is not addressed in the national protocols, collection of the pre- and posttreatment dwarf mistletoe data was optional in our study plan. We collected dwarf mistletoe data because field experience taught us that there would be no appreciable root disease identified in dry forest stands such as these. Preliminary surveys showed that dwarf mistletoe (*Arceuthobium* spp.) infection centers were relatively well distributed in many experimental units; thus, we deemed it impractical to conduct a complete census of the infected trees. We adopted a variable-plot sampling method to capture the overstory tree mistletoe data. This involved placing a single variable-prism or Relaskop plot (BAF 20) in the center of every other 40-m UTM grid cell (random start), as indicated on the rectified aerial photo, until 32 plots had been measured. The grid cells sampled were overlaid on the same area (64 grid cells covered 10 ha) sampled for bark beetles.

For each tree that fell within the variable radius prism plot, we collected:

1. Tree d.b.h.
2. Tree species
3. Live crown ratio
4. Crown class
5. Dwarf mistletoe rating (Hawksworth 6-class system)

Tree height, age, and radial growth were recorded for the first tree of each species and size class in each prism plot. We tallied all seedlings and saplings below breast height in a 1/100-acre microplot using the same plot center.

Each tree was inspected for signs and symptoms of dwarf mistletoe infection. If mistletoe was present, the damage and severity were listed with the individual tree information. Dwarf mistletoe infections were identified by the presence of mistletoe-induced witches' brooms in the host, the presence of basal cups or branch swelling, and by the presence of the male (staminate—pollen producing) and female (pistillate—seed producing) plants of the parasite. Infection severity was recorded by estimating a Hawksworth dwarf mistletoe rating (DMR) for that tree (Hawksworth 1977). For reference, the individual tree DMRs were obtained by dividing the stacked live crown vertically into equal thirds, and to each third assigning a rating of 0, 1, or 2 as follows:

0 = no visible branch infections (mistletoe plants) or witches' brooms located anywhere in the third.
1 = ≤ ½ of the branches in the third are infected or with brooms.
2 = > ½ of the branches in the third are infected or with brooms.

In ponderosa pine, a single large witches' broom (i.e., the broom is the dominant feature) in any crown third is sufficient to give that crown third a rating of 2. In Douglas-fir, mistletoe plants were often too small to see from the ground, and individual tree DMR ratings were based almost entirely on visible brooms in infected branches. The maximum DMR score was 6 for any tree; the minimum score was 0.

Analysis

Dwarf mistletoe and root disease incidence and severity, bark beetle mortality, and woodpecker foraging evidence were recorded in all experimental units. Root disease infection and mortality and bark beetle mortality were surveyed in a complete census owing to our recent advances with developing high-resolution orthophoto field maps. Such georeferencing saved about 80 percent of the time ordinarily needed in the field to georeference data, thereby allowing a complete census rather than sampling, and it freed up resources to conduct the dwarf mistletoe survey.

Owing to the large number of infected trees, dwarf mistletoe incidence and severity were sampled by using variable and fixed plots. Data were averaged up from the plot level to the experimental unit level for all analyses, or they were summed in the case of the total census.

Bark beetle impact on the site conditions was summarized by experimental unit and reported by diameter class for trees per hectare (fig. 7-2) and basal area per hectare (m^2/ha, fig. 7-3) killed by each bark beetle species. Likewise, dwarf mistletoe impact to the baseline conditions was summarized by experimental unit and reported by diameter distributions of mistletoe-infected trees per hectare (fig. 7-4) and infected basal area per hectare (m^2/ha, fig. 7-5) by host species.

In addition to showing tree stocking and basal area effects, we summarized average dwarf mistletoe infection severity (average DMR) for ponderosa pine and Douglas-fir host diameter classes (fig. 7-6).

Finally, we summarized baseline conditions with respect to snags and available woodpecker foraging habitat. To complete this analysis, we computed the percentage of the total trees greater than 20 cm d.b.h. that were snags with evidence of any woodpecker foraging. These results are shown in figure 7-7.

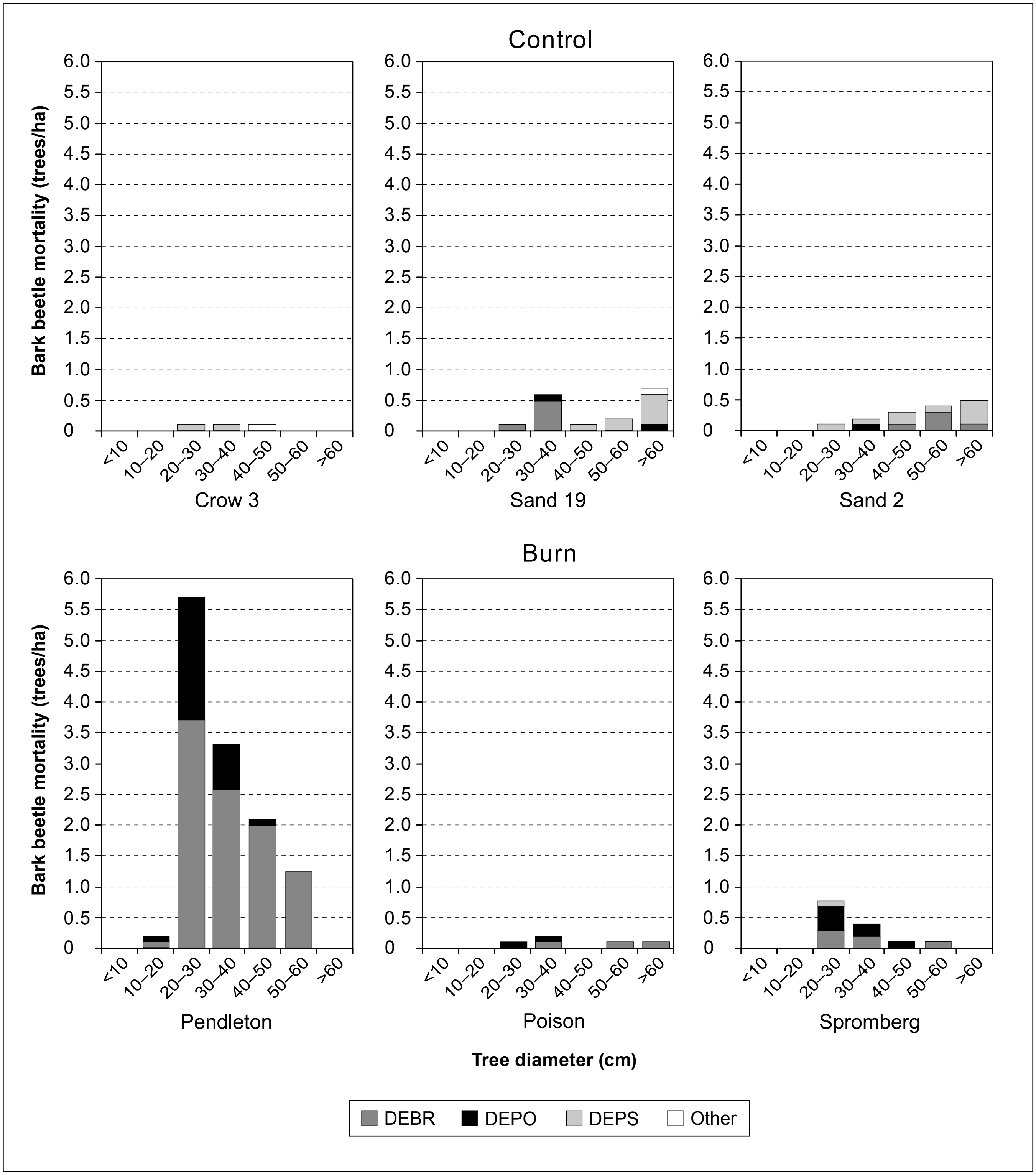

Figure 7-2—Diameter distributions of trees killed by bark beetles in experimental units within the Mission Creek Fire and Fire Surrogate Study. Units are arranged by treatment based on a balanced design with three replicates per treatment or treatment combination. Abbreviations are: DEBR—*Dendroctonus brevicomis*, the western pine beetle; DEPO—*Dendroctonus ponderosae*, the mountain pine beetle; DEPS—*Dendroctonus pseudotsugae*, the Douglas-fir beetle; Other—*Scolytus ventralis*, the fir-engraver beetle, *Pseudohylesinus nebulosus*, the Douglas-fir engraver beetle, *Dendroctonus valens*, the red turpentine beetle, and *Ips* spp., pine engraver beetles.

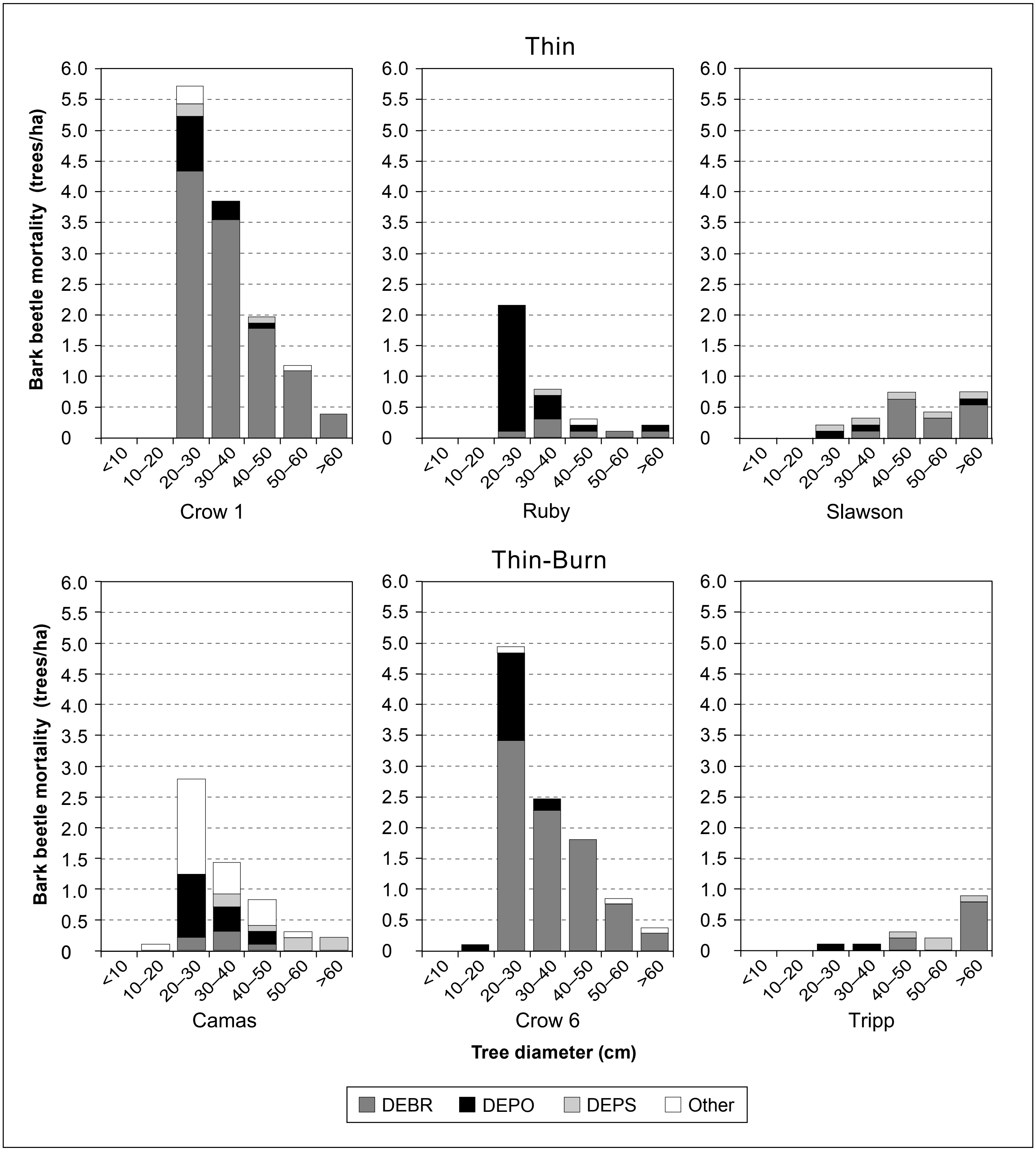

Figure 7-2—Diameter distributions of trees killed by bark beetles in experimental units within the Mission Creek Fire and Fire Surrogate Study (continued).

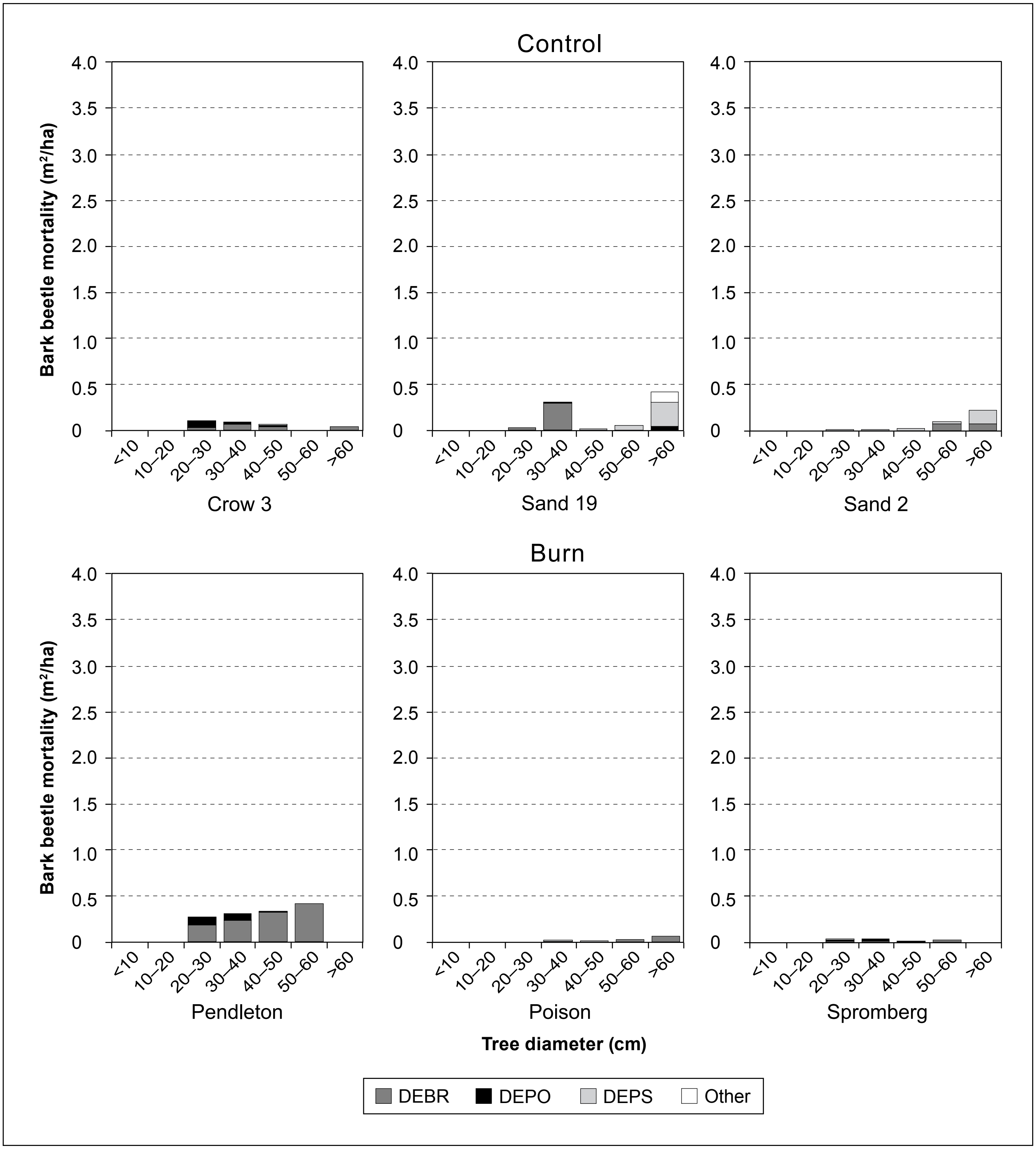

Figure 7-3—Diameter distributions of basal area killed by bark beetles in experimental units within the Mission Creek Fire and Fire Surrogate Study. Units are arranged by treatment based on a balanced design with three replicates per treatment or treatment combination. Abbreviations are: DEBR—*Dendroctonus brevicomis*, the western pine beetle; DEPO—*Dendroctonus ponderosae*, the mountain pine beetle; DEPS—*Dendroctonus pseudotsugae*, the Douglas-fir beetle; Other—*Scolytus ventralis*, the fir-engraver beetle, *Pseudohylesinus nebulosus*, the Douglas-fir engraver beetle, *Dendroctonus valens*, the red turpentine beetle, and *Ips* spp., pine engraver beetles.

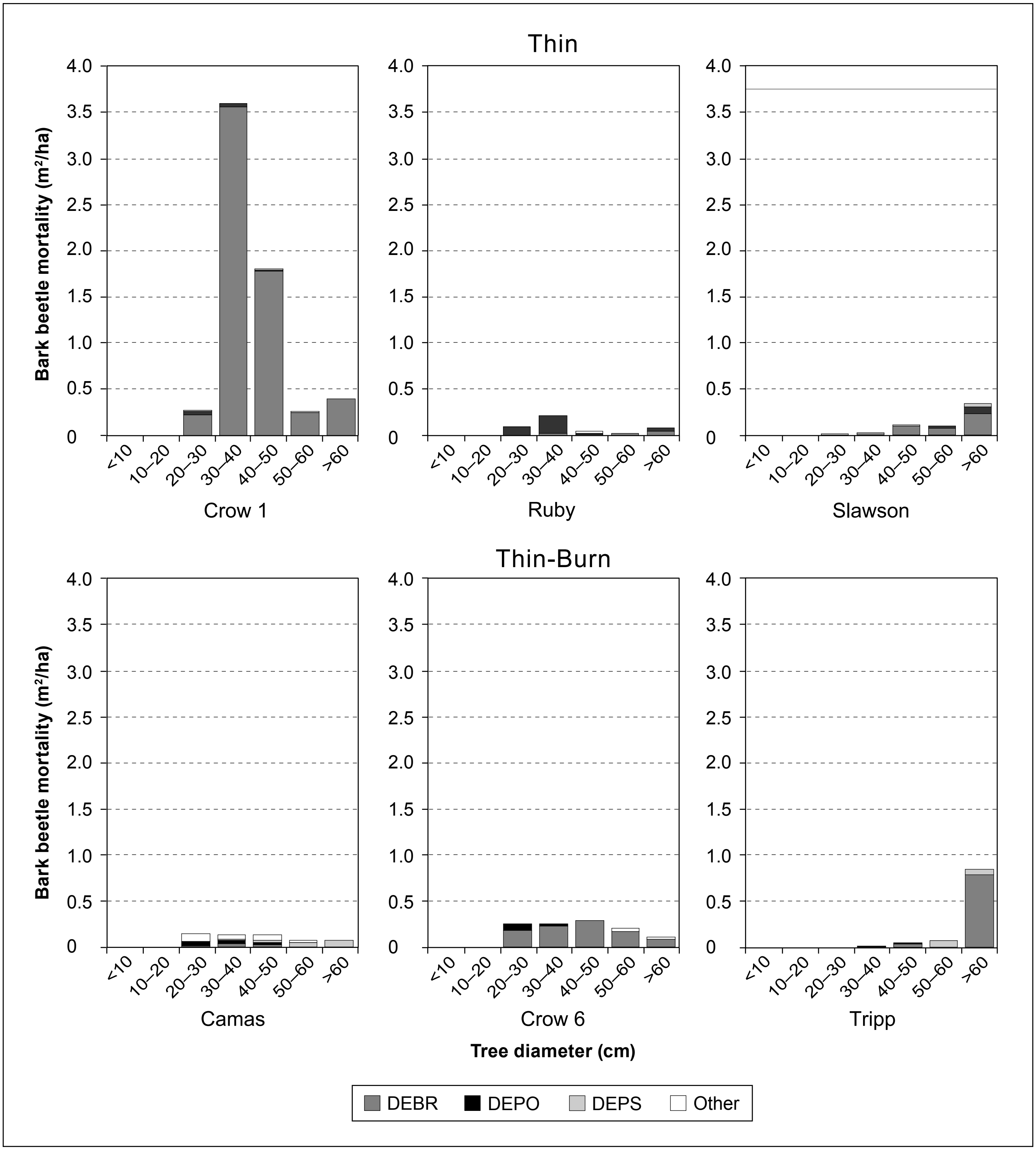

Figure 7-3—Diameter distributions of basal area killed by bark beetles in experimental units within the Mission Creek Fire and Fire Surrogate Study (continued).

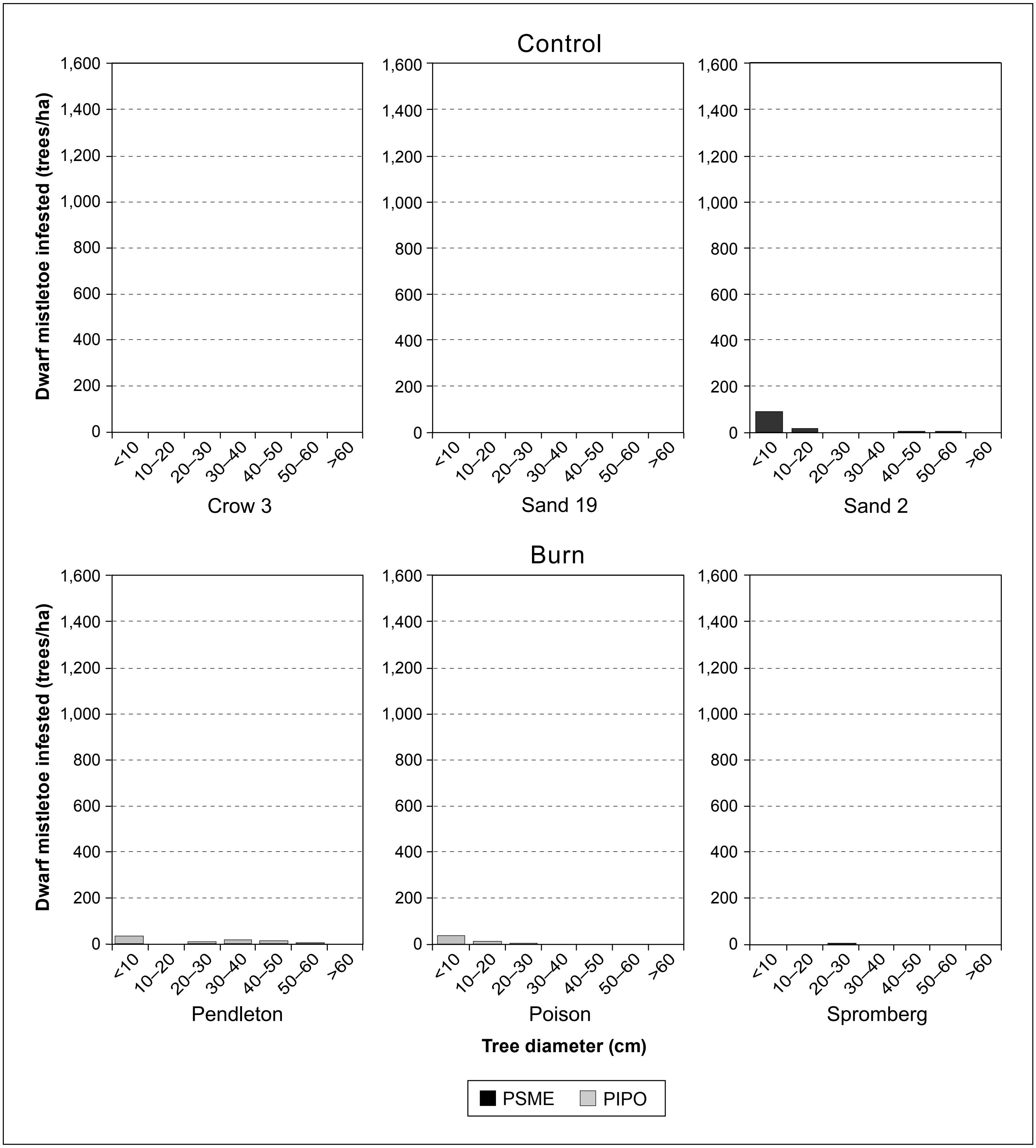

Figure 7-4—Diameter distributions of trees infected by dwarf mistletoes in experimental units within the Mission Creek Fire and Fire Surrogate Study. Units are arranged by treatment based on a balanced design with three replicates per treatment or treatment combination. Abbreviations are: PSME—*Pseudotsuga menziesii*, which is infected by the host specialized mistletoe—*Arceuthobium douglasii*, and PIPO—*Pinus ponderosa*, which is infected by the host-specialized mistletoe—*Arceuthobium campylopodum*.

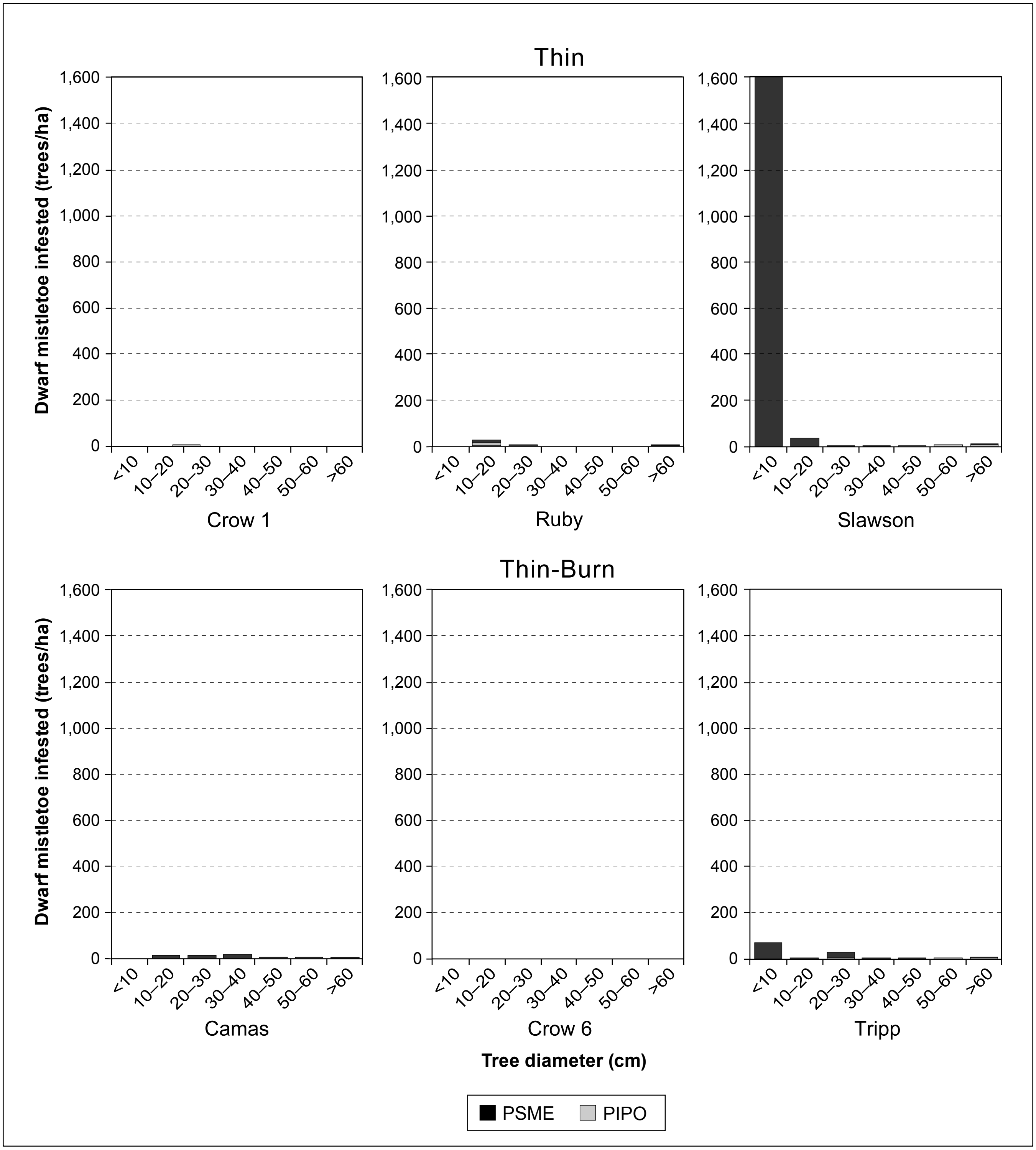

Figure 7-4—Diameter distributions of trees infected by dwarf mistletoes in experimental units within the Mission Creek Fire and Fire Surrogate Study (continued).

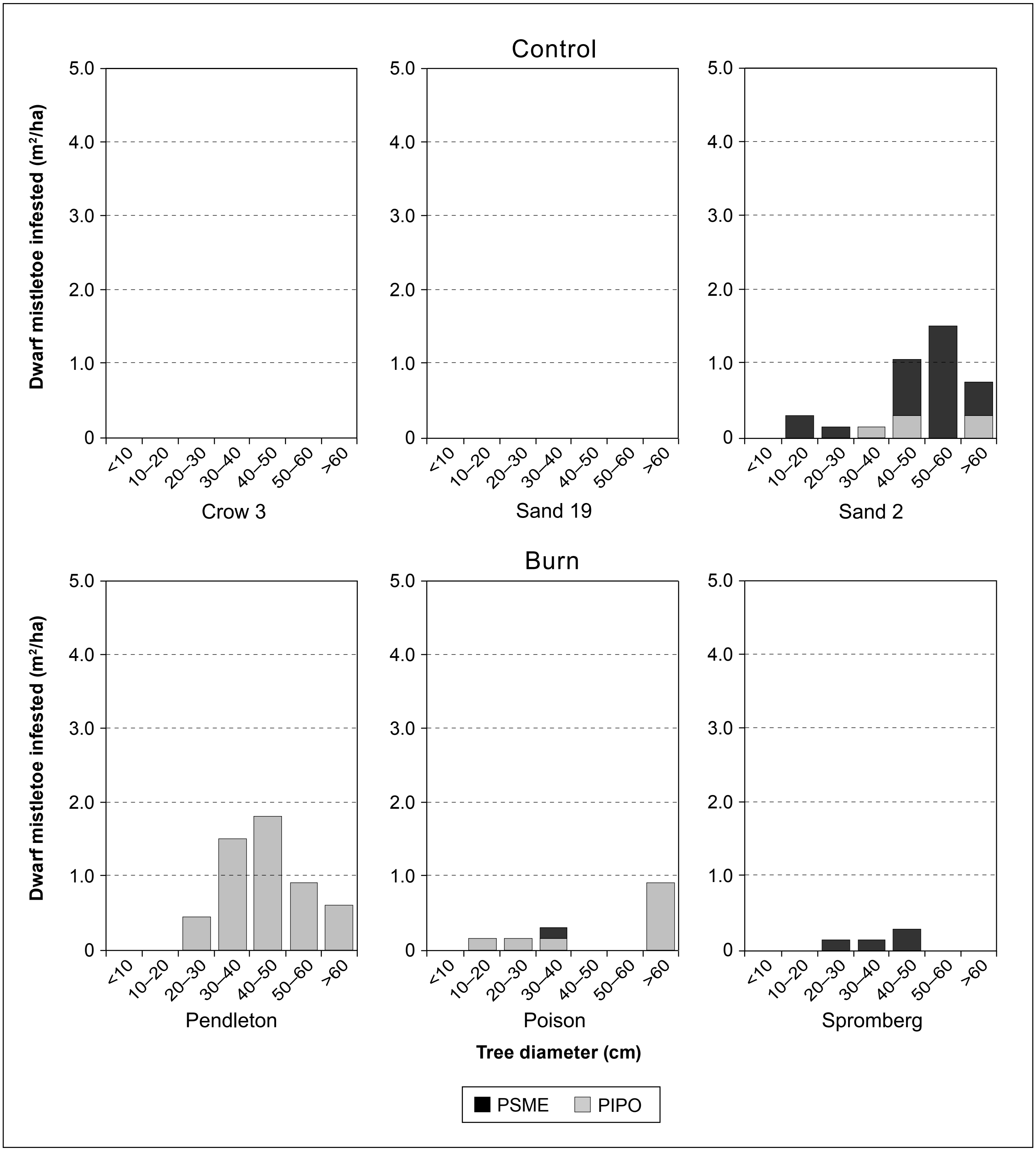

Figure 7-5—Diameter distributions of basal area infected by dwarf mistletoes in experimental units within the Mission Creek Fire and Fire Surrogate Study. Units are arranged by treatment based on a balanced design with three replicates per treatment or treatment combination. Abbreviations are: PSME—*Pseudotsuga menziesii*, which is infected by the host specialized mistletoe—*Arceuthobium douglasii*, and PIPO—*Pinus ponderosa*, which is infected by the host-specialized mistletoe—*Arceuthobium campylopodum*.

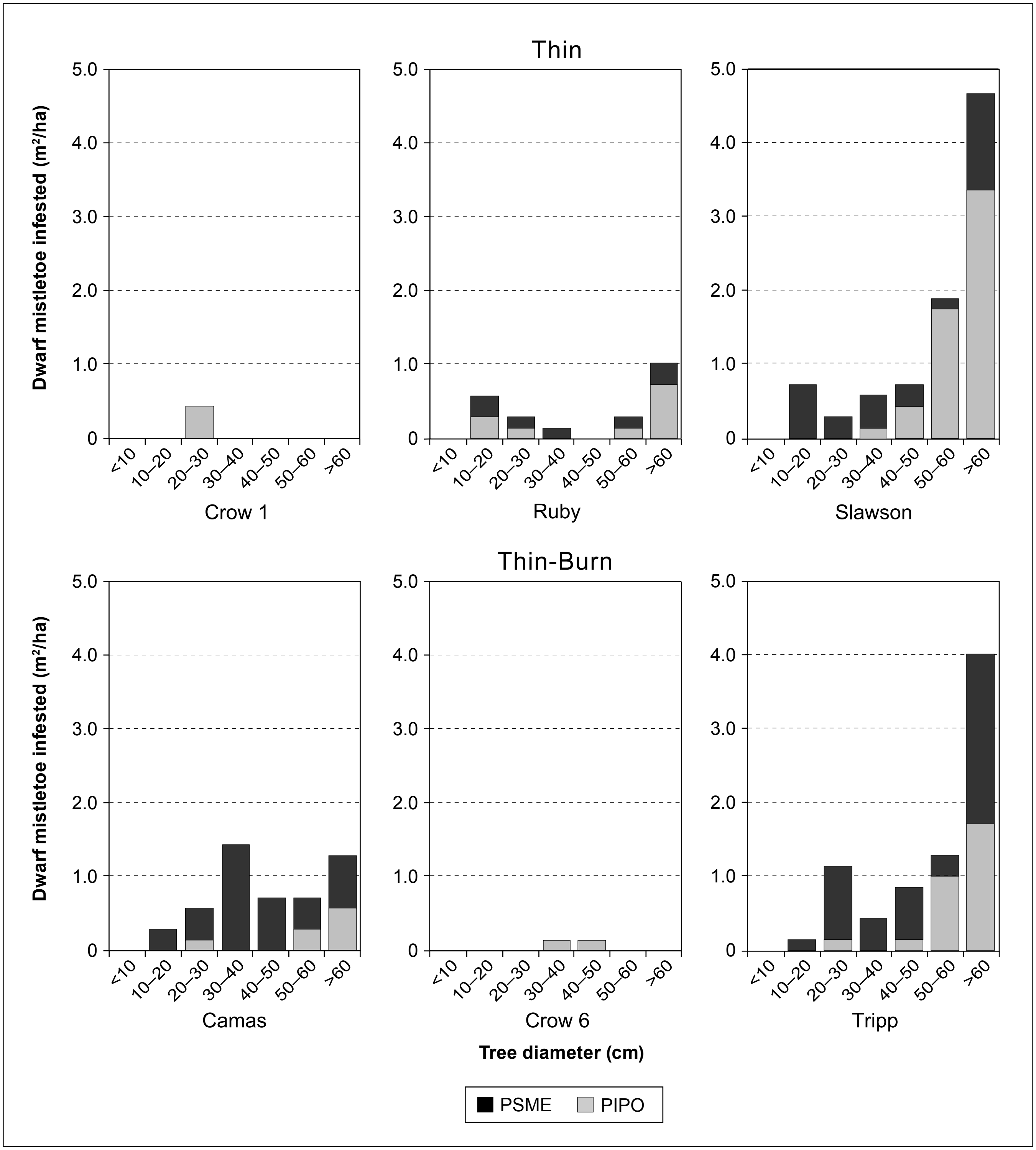

Figure 7-5—Diameter distributions of basal area infected by dwarf mistletoes in experimental units within the Mission Creek Fire and Fire Surrogate Study (continued).

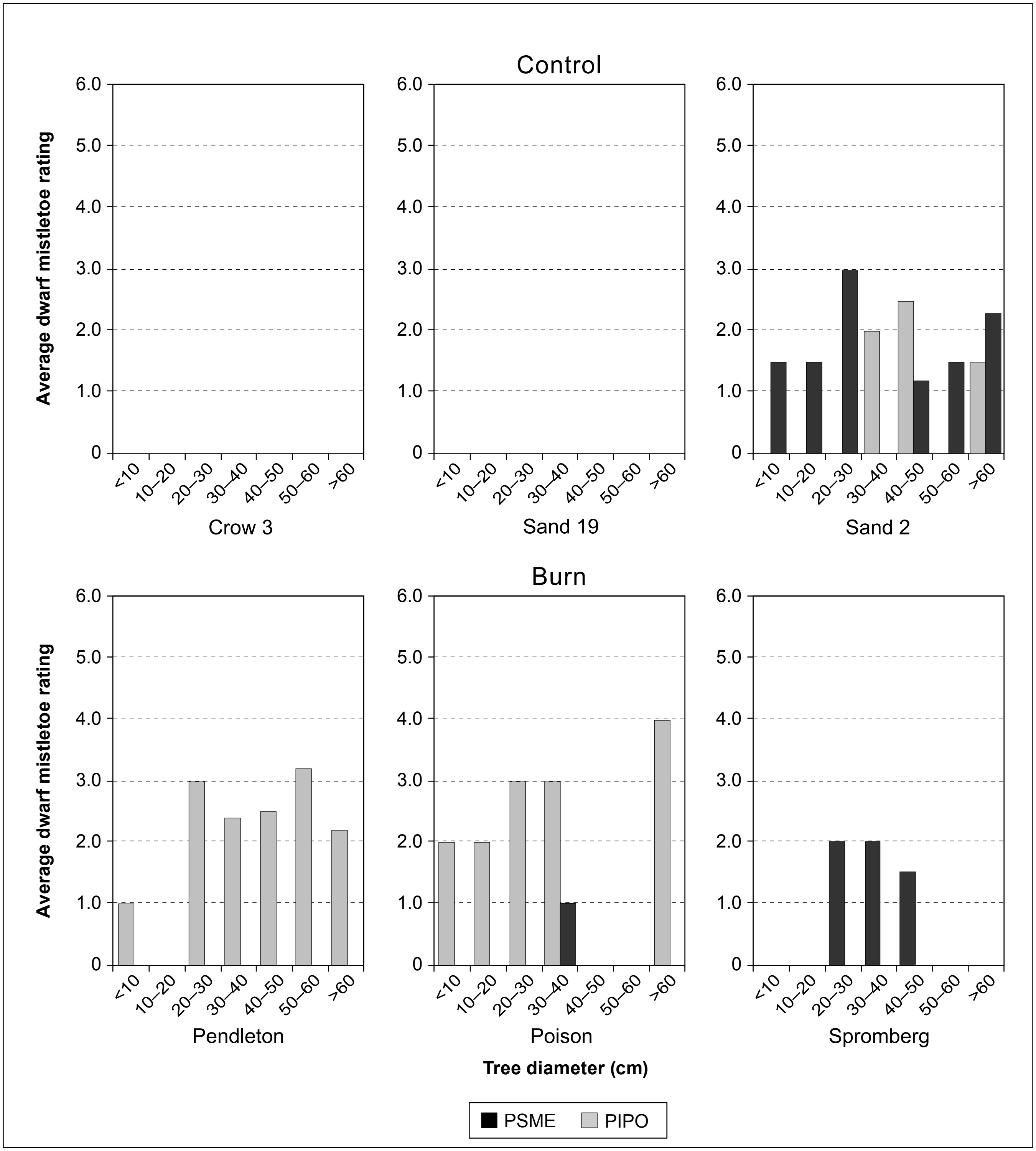

Figure 7-6—Diameter distributions and average dwarf mistletoe rating of trees infected by dwarf mistletoes in experimental units within the Mission Creek Fire and Fire Surrogate Study. Units are arranged by treatment based on a balanced design with three replicates per treatment or treatment combination. Abbreviations are: PSME—*Pseudotsuga menziesii*, which is infected by the host specialized mistletoe—*Arceuthobium douglasii*, and PIPO—*Pinus ponderosa*, which is infected by the host-specialized mistletoe—*Arceuthobium campylopodum*.

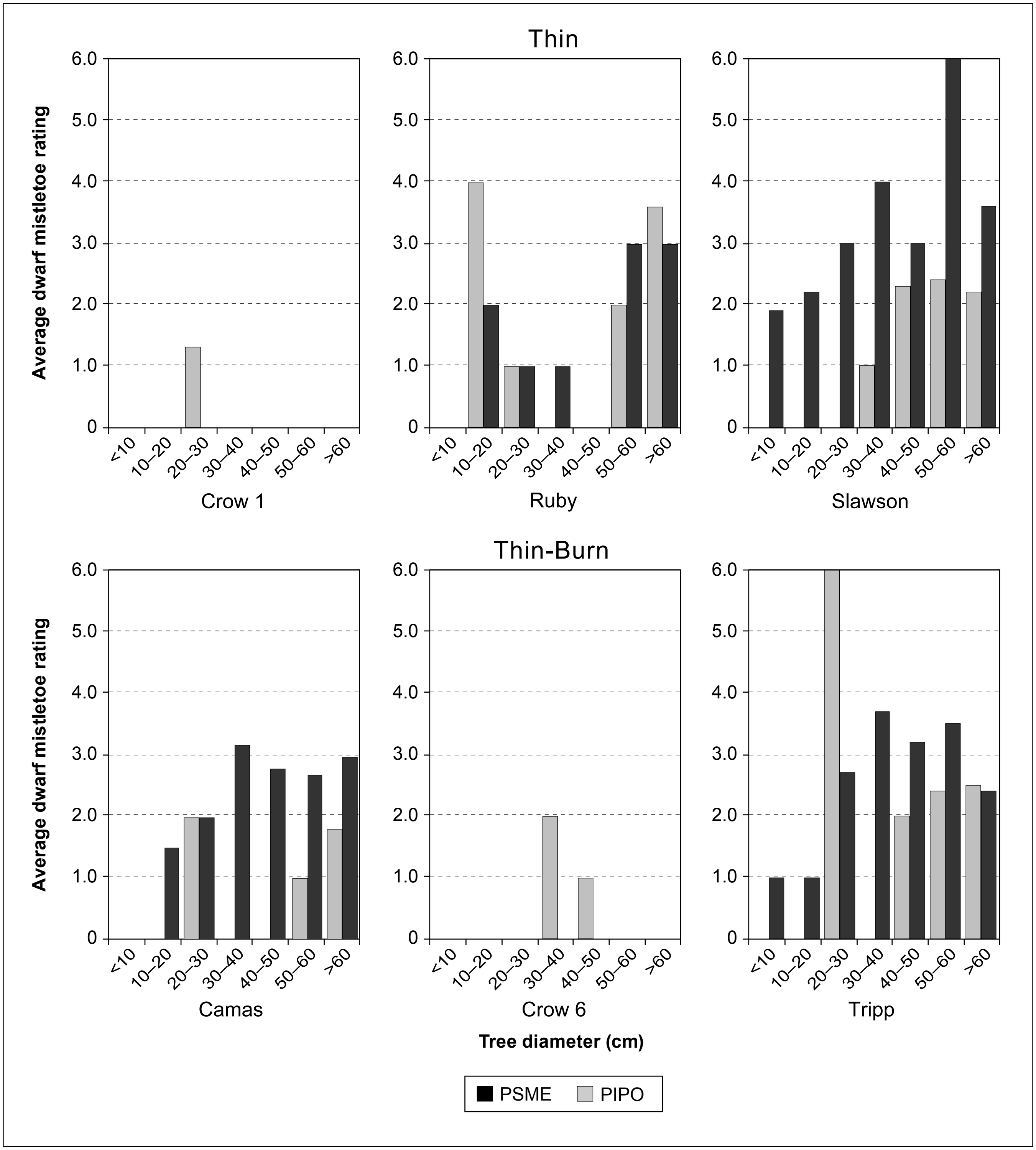

Figure 7-6—Diameter distributions and average dwarf mistletoe rating of trees infected by dwarf mistletoes in experimental units within the Mission Creek Fire and Fire Surrogate Study (continued).

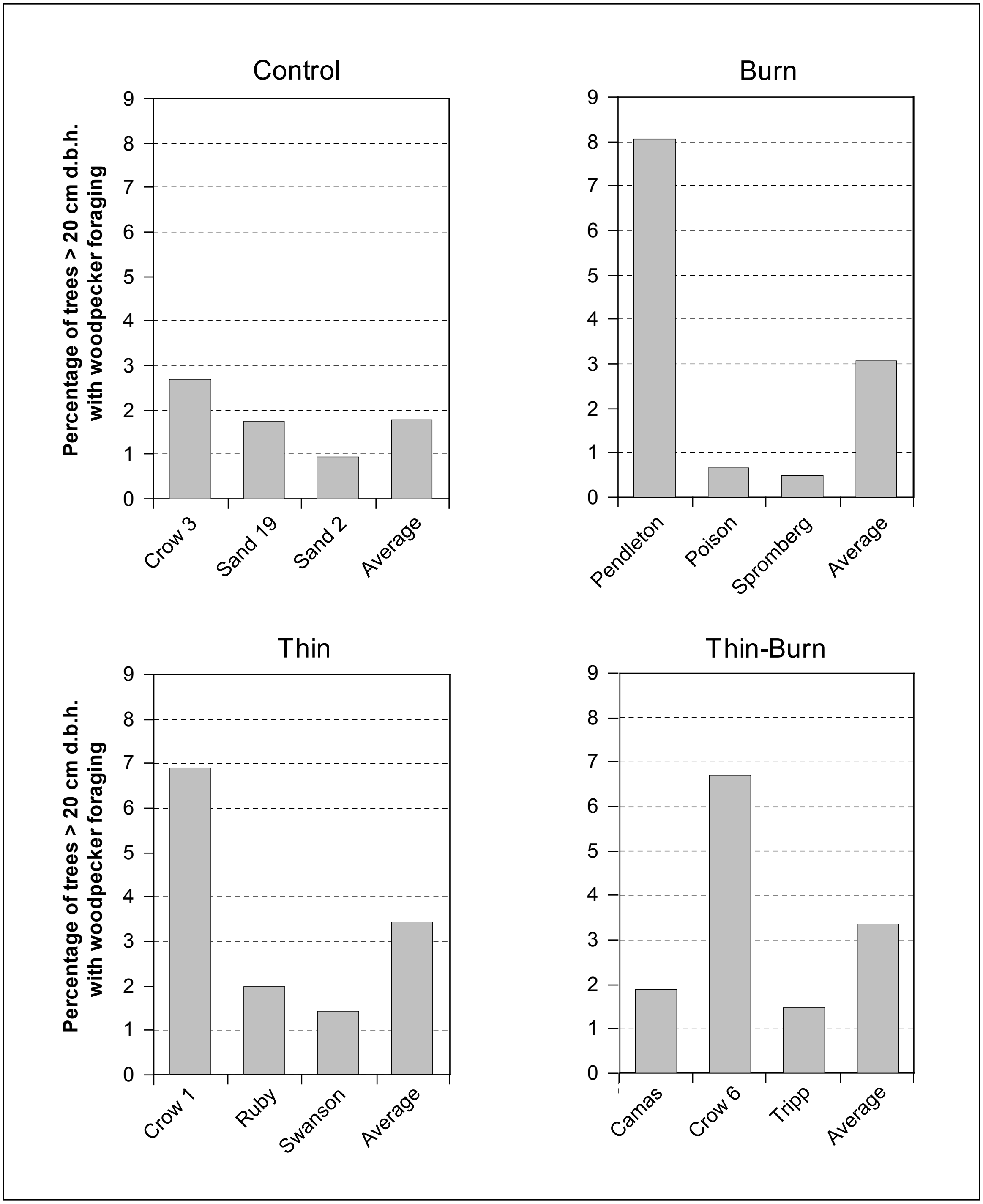

Figure 7-7—Percentages of trees greater than 20 cm diameter at breast height (d.b.h.) that were snags with evidence of woodpecker foraging. Units are arranged by treatment based on a balanced design with three replicates per treatment or treatment combination. Evidence of woodpecker foraging was found in trees attacked by all bark beetles species surveyed, and in those attacked by woodborers (Coleoptera/Cerambycidae and Buprestidae).

Results and Discussion

In all, 1,170 trees were mapped and processed for evidence of root pathogens and bark beetle attacks or mortality. Of this total, 398 were nonsymptomatic trees, 595 were attacked by bark beetles, and 177 were either symptomatic or dead from causes other than root pathogens or known bark beetles. In these latter instances, the bark beetle gallery signatures in the inner bark had long since decomposed, and egg gallery engravings had sap-rotted such that a positive identification of the attacking bark beetle species was impossible. However, blue staining of the sapwood was in evidence so that a positive identification of an unknown bark beetle attack was possible. Other causes of symptomatic or dead trees included drought-related mortality, dwarf mistletoe infection, mechanical injury, or top killing from historical insect defoliation (western spruce budworm [*Choristoneura occidentalis* Freeman]) or comandra rust (*Cronartium comandrae*) infections.

Of 1,170 trees, examined, 398 were nonsymptomatic, 595 were attacked by bark beetles, and 177 were symptomatic or dead from other causes. There was only one confirmed case of a root pathogen.

Root Pathogens

There was only one confirmed case of a root pathogen (HEAN), which occurred in the Pendleton experimental unit. Interestingly, this root pathogen was cultured from an apparently clean root sample taken from an asymptomatic tree. There were no confirmed cases of root disease mortality caused by any of the surveyed pathogens. No further studies beyond those reported here are anticipated for root pathogens.

Fungal Endophytes

The fungal community inhabiting large woody roots of healthy conifers has not been well documented. To provide more information about such communities, we conducted an exploratory survey of the fungal endophyte community of dry forests using the same increment cores extracted from the woody roots of the asymptomatic Douglas-fir and ponderosa pine that were sampled for latent root disease. Fungal isolates were cultured on standard media, and then were identified using a combination of molecular and morphological methods (Hoff et al. 2004b). Fungal genera and species identified in this study provide baseline data for future surveys of fungal endophytes. A complete write-up and explanation of the ecological roles of endophytes, methods for isolation, and a discussion of their importance may be found in Hoff 2002, and Hoff et al. 2004b.

A total of 289 root cores were evaluated for the presence of endophytes, and fungi were isolated from woody roots of 138 (47.8 percent) asymptomatic ponderosa pine and Douglas-fir (table 7-1). Averaged across all experimental plots, fungal isolates were obtained from 50 percent of the ponderosa pine roots and 45 percent of the Douglas-fir roots. Twenty-seven cores (9.3 percent) yielded multiple fungal

Table 7-1—Summary of fungal species isolated from roots of asymptomatic ponderosa pine and Douglas-fir hosts growing in dry forest environments of the Mission Creek study area

	Ponderosa pine host		Douglas-fir host		
	PIPO plant association	**PSME plant association**	**PIPO plant association**	**PSME plant association**	**Total**
			Number (percent)		
Total number of trees	48	112	1	128	289
Trees with isolated fungal endophyte	22 (45.8)	58 (51.8)	0 (0)	58 (45.3)	138 (47.8)
Trees with no isolated fungal endophytes	26 (54.2)	54 (48.2)	1 (100)	70 (54.7)	151 (52.2)
Trees with multiple fungal endophytes	6 (12.5)	8 (7.1)	0 (0)	13 (10.2)	27 (9.3)
Byssochlamys sp.	5 (10.4)	22 (19.6)	0	32 (25.0)	59 (20.4)
Umbelopsis spp.	4 (8.3)	16 (14.3)	0	10 (7.8)	30 (10.4)
Mucor sp.	1 (2.1)	8 (7.1)	0	9 (7.0)	18 (6.2)
Penicillium sp.	1 (2.1)	3 (2.7)	0	6 (4.7)	10 (3.5)
Verticillium sp.	2 (4.2)	0 (0)	0	2 (1.6)	4 (1.4)
Cladosporium sp.	1 (2.1)	1 (0.9)	0	1 (0.8)	3 (1.0)
Tremella sp.	1 (2.1)	1 (0.9)	0	0 (0)	2 (0.7)
Epicoccum sp.	1 (2.1)	0 (0)	0	0 (0)	1 (0.3)
Phialophora sp.	1 (2.1)	0 (0)	0	0 (0)	1 (0.3)
Alternaria sp.	0 (0)	1 (0.9)	0	0 (0)	1 (0.3)
Ulocladium sp.	1 (2.1)	0 (0)	0	0 (0)	1 (0.3)
Rhinocladiella sp.	0 (0)	1 (0.9)	0	0 (0)	1 (0.3)
Heterobasidion sp.	0 (0)	1 (0.9)	0	0 (0)	1 (0.3)
Eupenicillium sp.	1 (2.1)	0 (0)	0	0 (0)	1 (0.3)
Merimbla sp.	0 (0)	0 (0)	0	1 (0.8)	1 (0.3)
Hormonema sp.	0 (0)	0 (0)	0	1 (0.8)	1 (0.3)
Pesotum sp.	0 (0)	0 (0)	0	1 (0.8)	1 (0.3)
Unidentified	7 (14.6)	11 (9.8)	0	9 (7.0)	27 (9.3)

Note: For this study, PIPO typically included PIPO/PUTR, PIPO/CARU, and PIPO/AGSP plant associations; PSME included PSME/SPBEL, PSME/PUTR, PSME/CARU, PSME/SYAL, and PSME/AGSP plant associations (see text; Lillybridge et al. 1995, Williams and Smith 1991). AGSP, *Agropyron spicatum* Pursh.; CARU, *Calamagrostis rubescens* Buckler; PIPO, *Pinus ponderosa* C. Lawson; PSME, *Pseudotsuga menziesii* (Mirb.) Franco; PUTR, *Purshia tridentata* (Pursh) DC.; SPBEL, *Spiraea betulifolia* var. *lucida* Pall.; SYAL, *Symphoricarpos albus* (L.) S.F. Blake. Table excerpted from Hoff et al. 2004a.

genera, 111 (38.4 percent) yielded growth of one fungal genus, and 151 (52.2 percent) yielded no fungi (table 7-1). A wide diversity of fungi was recovered from root cores, comprising 3 phyla, 11 orders, 13 families, and 17 genera. Fourteen genera were isolated from ponderosa pine, and nine genera were isolated from Douglas-fir. Although a medium selective for basidiomycetous fungi was used, the only basidiomycetes isolated were *Tremella* spp. (two isolates) and *Heterobasidion* spp. (one isolate). Most of the fungi isolated were ascomycetes and zygomycetes (table 7-2). *Byssochlamys nivea* Westling, *Umbelopsis* spp., and *Mucor* spp. were the fungi most frequently recovered from ponderosa pine (16.9, 12.5, and 5.6 percent of cores, respectively) and Douglas-fir (24.8, 7.8, and 7.0 percent of cores, respectively).

Results revealed that a relatively small number of cores from woody roots of Douglas-fir and ponderosa pine were colonized by various other fungal genera (e.g., *Mucor* spp., *Penicillium* spp., *Verticillium* spp., *Cladosporium* spp., and *Tremella* spp.). Many of the taxa isolated infrequently in this study have been isolated in studies of root-colonizing fungi of other conifers (Fisher et al. 1991, Holdenrieder and Sieber 1992). Fisher et al. (1991) speculated that the presence of common soil and organic-debris-colonizing fungi, such as *Cladosporium*, *Penicillium*, *Mucor*, and *Umbelopsis* (as *Mortierella*), in woody roots may indicate that many different saprotrophic fungi have some limited ability to live in living root tissue.

The diversity of fungi obtained from healthy, large, woody roots of two conifer species growing in a relatively narrow range of dry-forest settings suggests that an even broader flora of fungal endophytes might be found across diverse forest environments. This may be fertile ground for future research. It will be of further interest to study how fungi occupying this recently discovered niche might also influence the dynamics of fungi, perhaps most especially pathogenic fungi, in the rhizosphere of forest tree roots.

The majority of bark-beetle-killed trees were killed by the western and mountain pine beetles.

Bark Beetles

Across all 12 experimental units, the majority of bark-beetle-killed trees were killed by the western (378) and mountain pine beetles (144). We found very few instances of Douglas-fir beetle (41), Douglas-fir engraver (6), fir engraver (22), red turpentine beetle (4) and *Ips* spp. (6), considering that all dead and declining trees were censused across 12 overstocked, 10-ha experimental units. Table 7-2 shows the number of trees killed by pooled bark beetle species, experimental unit, and overstory tree size class. In each unit, the overstory is dominated by small (22.7- to 37.9-cm d.b.h.) and medium sized (38.0- to 75.9-cm d.b.h.) trees, and most beetle mortality occurred in these same size classes. Three units, Crow 1, Crow 6, and

Table 7-2—Pooled bark beetle mortality by overstory tree size class

	Control			Burn			Thin			Thin-burn		
Overstory tree size class (cm)	**Crow 3**	**Sand 19**	**Sand 2**	**Pendleton**	**Poison**	**Spromberg**	**Crow 1**	**Ruby**	**Slawson**	**Camas**	**Crow 6**	**Tripp**
12.7–22.6 cm (poles)	2	0	0	23	0	4	14	9	0	4	14	1
22.7–37.9 cm (small trees)	19	1	3	70	2	8	73	19	6	16	63	1
38.0–75.9 cm (medium trees)	7	5	10	38	4	2	43	6	16	9	34	8
≥ 76.0 cm (large trees)	0	10	2	1	1	0	0	1	2	0	0	6

Note: Units are arranged by treatment based on a balanced design with three replicates per treatment or treatment combination.

Pendleton, had large bark beetle mortality centers (fig. 7-8, and table 7-2), but when compared to the number of available host trees, the percentage of affected trees was relatively low (see fig. 7-2, and chapter 2, fig. 2-1). In each of these three experimental units, the cover type was clearly dominated by ponderosa pine, where pine cover exceeded 90 or 95 percent in each instance. Crow 1, the thinning experimental unit with the highest percentage of bark-beetle-killed trees, had a pine infestation rate of about 3.6 percent when averaged across trees of all size classes. All of the experimental units displayed a Douglas-fir beetle mortality rate of less than 1 percent of the total trees.

The basal area of bark-beetle-infested trees told a similar story. Mortality expressed in terms of basal area per hectare (m^2/ha) was relatively low in all units, highest in thinning experimental unit Crow 1, and much lower elsewhere (fig. 7-3).

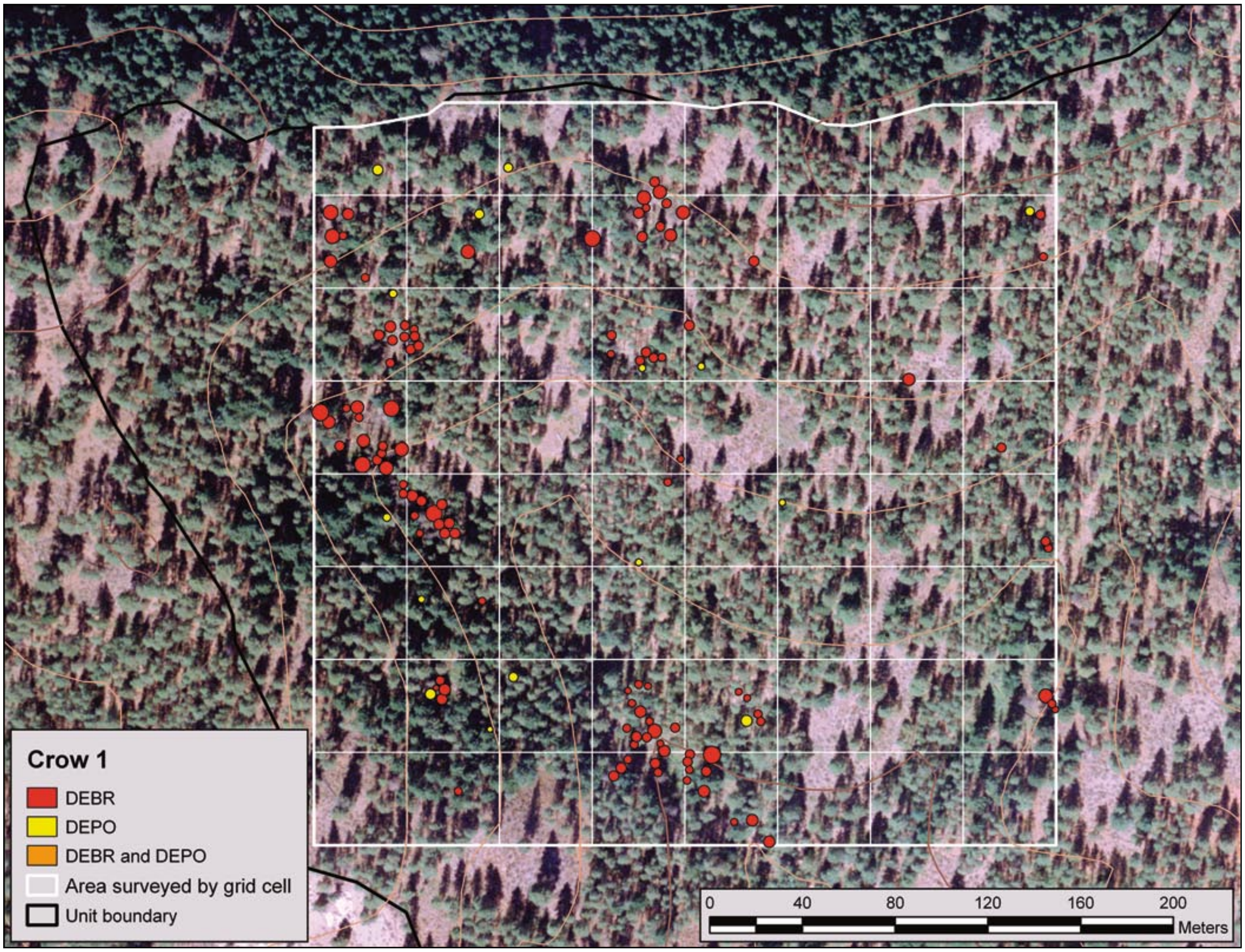

Figure 7-8—Western and mountain pine beetle mortality spatial distributions in experimental units Crow 1, Crow 6, and Pendleton. DEBR—*Dendroctonus brevicomis*, the western pine beetle; DEPO—*Dendroctonus ponderosae*, the mountain pine beetle.

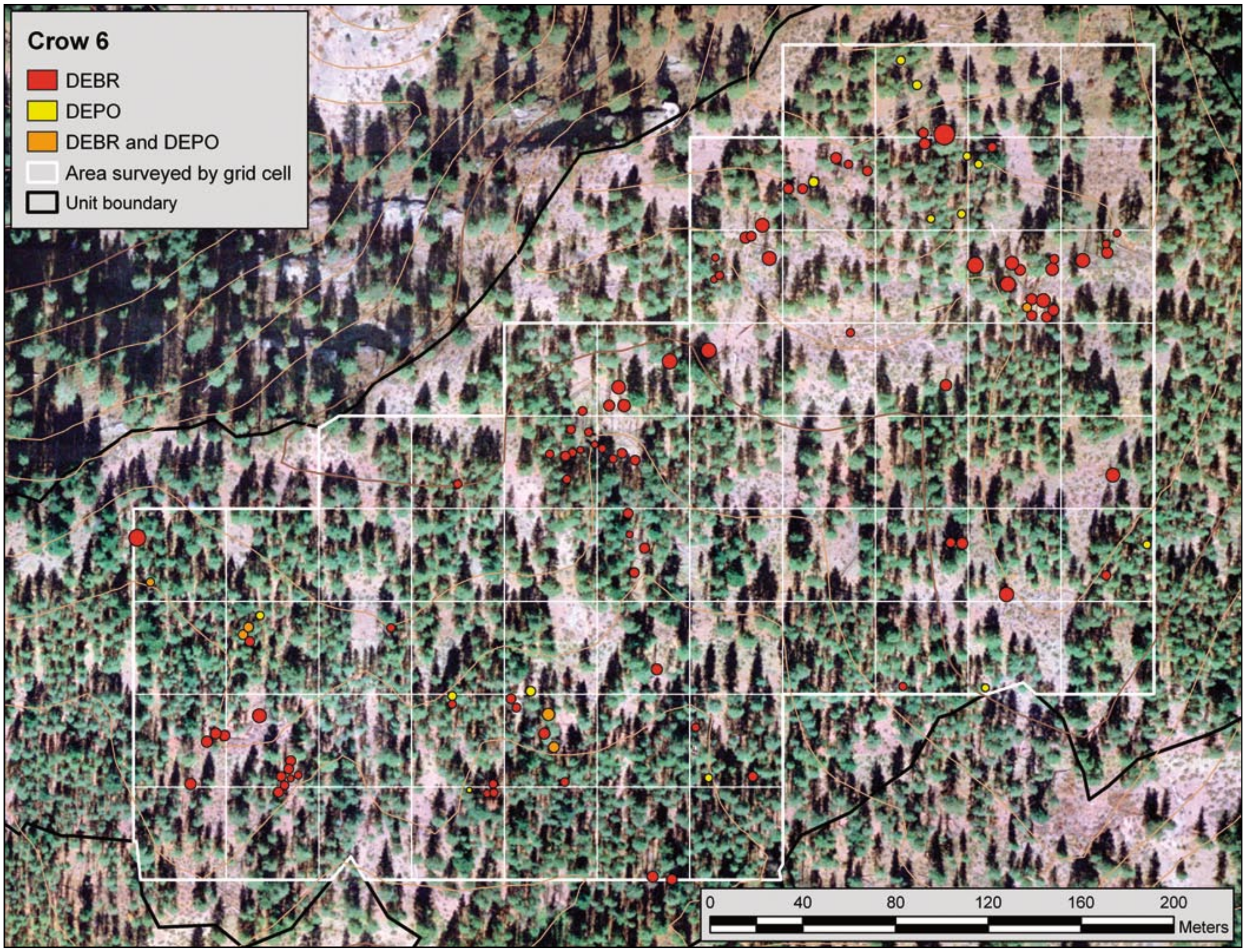

Figure 7-8—Western and mountain pine beetle mortality spatial distributions in experimental units Crow 1, Crow 6, and Pendleton. DEBR—*Dendroctonus brevicomis*, the western pine beetle; DEPO—*Dendroctonus ponderosae*, the mountain pine beetle.

Mistletoe

Dwarf mistletoe was found on all but two control units (Crow 3 and Sand 19), and incidence in Crow 6 was exceedingly low in ponderosa pine. In the Slawson thinning unit (fig. 7-4), mistletoe effects on stocking were greatest in Douglas-fir, but the greatest effects on basal area were to ponderosa pine. Douglas-fir and ponderosa pine were both infected in every overstory size class, although infection levels expressed in trees per hectare were low. These overstory infections had, over time, produced relatively high levels of infection in understory Douglas-fir seedlings and saplings (fig. 7-4) throughout the stand. Mistletoe effects on basal area per hectare were also greatest in Slawson, but most of the infected basal area occurred in the largest diameter trees, which were ponderosa pine (fig. 7-5). The story was

Dwarf mistletoe was found on all but two units, but infection levels were low.

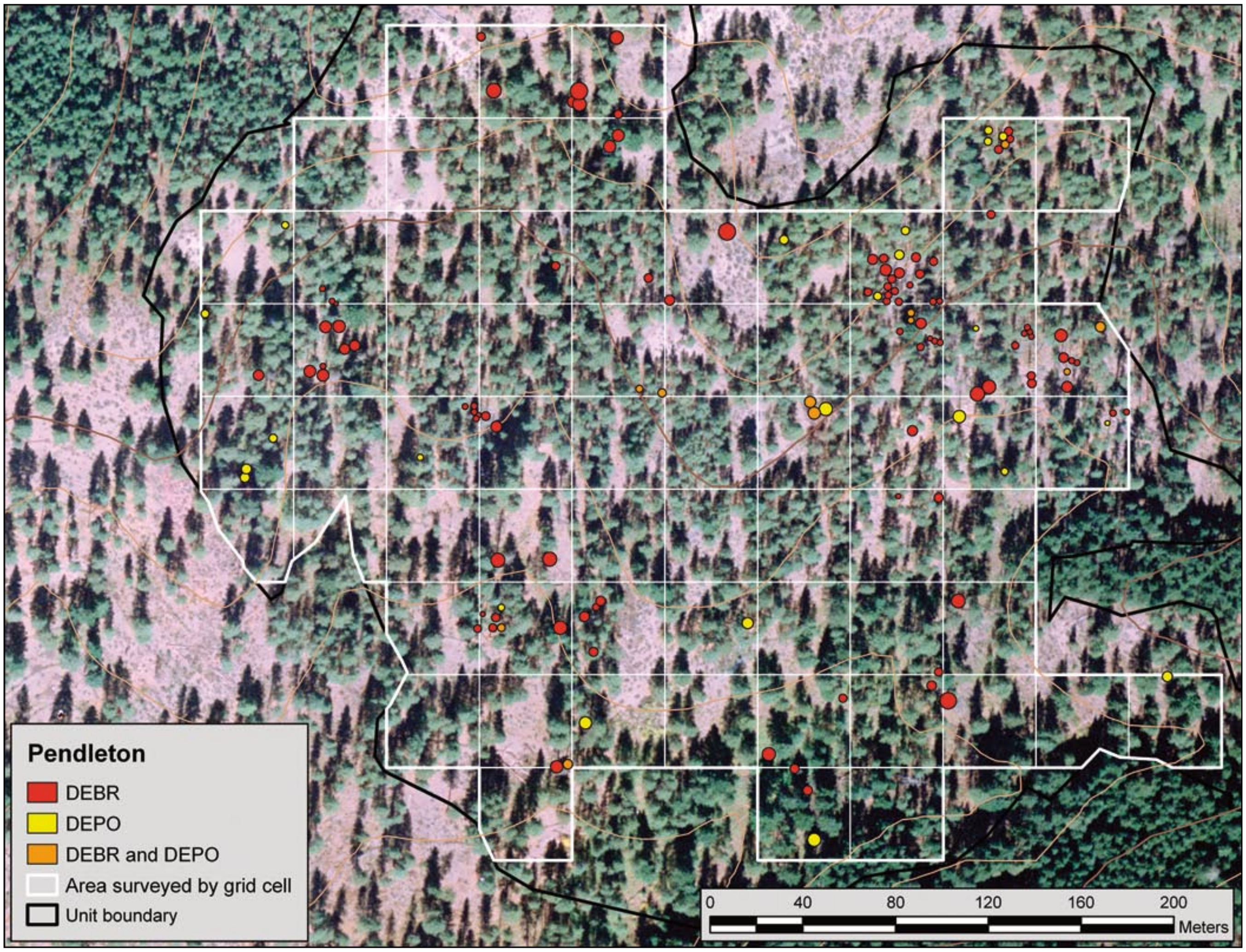

Figure 7-8—Western and mountain pine beetle mortality spatial distributions in experimental units Crow 1, Crow 6, and Pendleton. DEBR—*Dendroctonus brevicomis*, the western pine beetle; DEPO—*Dendroctonus ponderosae*, the mountain pine beetle.

quite similar in the thin and burn experimental unit Tripp, although the number of Douglas-fir seedlings and saplings infected was much lower.

Overall, dwarf mistletoe infection levels were low, affecting as little as 0 percent (Crow 3, Sand 19) and as much as 7.9 (Tripp) to 8.8 percent (Slawson) of the total basal area per hectare (fig. 7-5). In Sand 2, Spromberg, Tripp, and Camas, overstory Douglas-fir made up the primary affected basal area. In Pendleton, Poison, Ruby, and Slawson, the primary infection was in ponderosa pine (fig. 7-5). Mistletoe effects on trees per hectare were trivial in all units but Slawson (fig. 7-4).

The distribution of mistletoe by severity rating (DMR) told a somewhat different story (fig. 7-6). In Slawson, dwarf mistletoe severity in Douglas-fir was nearly twice that in ponderosa pine, and the largest overstory size classes generally supported the worst mistletoe infections. The average DMR in the 50- to 60-cm size class was 6, denoting infection top to bottom. Ponderosa pine infection levels were much lower, and we suspect that thinning and combined thin-burn treatments could drastically reduce average infection levels. Tripp was quite similar to Slawson, but the highest average DMRs were in 20- to 30-cm ponderosa pine, which carried an average DMR severity level of 6. Above this size class, infection severity in overstory Douglas-fir and ponderosa pine was generally evenly distributed, and DMR severity ranged between 2.5 and 3.5.

Dwarf mistletoe infection severity was evenly distributed among most size classes of ponderosa pine in the Pendleton unit, but generally increased with increasing size class in the Poison unit. Infection severity was lowest in ponderosa pine in the Crow 1 and Crow 6 units, which had relatively low incidence to begin with. It is likely that with carefully designed silvicultural prescriptions, dwarf mistletoe may be maintained at low levels indefinitely in these two units.

In naturally occurring stands, dwarf mistletoe infection severity tends to be highest in the oldest and largest trees and lowest in the smaller and younger trees, even though effects on stocking may be highest in the smaller size classes. The distribution of infection severity in the Ruby and Sand 2 units suggests that prior thinning, partial removal, or selection cutting entries targeted the most severely infected ponderosa pine and Douglas-fir. In Ruby, these occurred in the intermediate size classes. In Sand 2, these occurred in the upper size classes (fig. 7-6).

Woodpecker Foraging

On average, approximately 2 to 3 percent of the trees > 20 cm were snags with evidence of woodpecker foraging.

At the treatment level, there was little difference in the percentage of trees > 20 cm with woodpecker foraging. On average, approximately 2 to 3 percent of the trees > 20 cm were snags with evidence of woodpecker foraging. However, in the burn (Pendleton), thin (Crow 1), and thin-burn (Crow 6) treatments, higher than average snag abundance and evidence of woodpecker foraging was evident in at least one unit, but no difference was significant ($p \leq 0.05$) owing to small sample size.

We opted to display the amount of foraging in trees greater than 20 cm (7.9 in) because many tree-killing bark beetles ordinarily have difficulty successfully rearing broods in small trees owing to their extremely thin secondary phloem (living inner bark), and the most prevalent bark beetles in these units (namely: *Dendroctonus brevicomis*, the western pine beetle; *D. ponderosae*, the mountain pine beetle; *D. pseudotsugae*, the Douglas-fir beetle; and *Scolytus ventralis*, the fir engraver) generally attack larger trees.

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Chapter 8: Avian Community Composition, Nesting Ecology, and Cavity-Nester Foraging Ecology

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Introduction

The effects of fire exclusion on forests of the interior Western United States include changes to forest structure and composition at both the stand and landscape scales (Agee 2003, Camp et al. 1997, Everett et al. 2000, Harrod et al. 1999, Hessburg et al. 2000, 2007). It is expected that these changes result in corresponding changes to wildlife communities, with species generally associated with closed-canopy forests being favored over species associated with open-canopy forests (Lyon et al. 2000). Fire exclusion has been identified as one factor contributing to habitat loss for some key avian focal species (Germaine and Germaine 2002, Wisdom et al. 2000). However, limited information is available concerning the structure and composition of avian communities in dry interior western forests that have changed dramatically as a result of fire-exclusion policies. In this paper, we present the results of 2 years of monitoring avian species in forest stands that have experienced several decades of fire exclusion. In addition, we hypothesize about the effects that fire and fire surrogate restoration treatments may have on avian communities and some key focal species.

Fire exclusion has been identified as one factor contributing to habitat loss for some key avian focal species.

A growing number of studies have investigated the effects of restorative treatments in dry forests on avian species and communities (Gaines et al. 2007, Germaine and Germaine 2002, Greenberg et al. 2007, Wightman and Germaine 2006, Zebehazy et al. 2004). Generally, these treatments have little effect on overall species abundance, richness, or evenness (Gaines et al. 2007, Greenberg et al. 2007, Zebehazy et al. 2004). However, the response of individual avian species varies. For example, the western bluebird (*Sialia mexicana*) and white-headed woodpecker may respond favorably to these treatments (Gaines et al. 2007, Wightman and

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Germaine 2006), whereas the red-breasted nuthatch may occur at lower densities in treated stands (Gaines et al. 2007). (See table 8-1 for scientific names of birds.) As Germaine and Germaine (2002) pointed out, the effects of these kinds of restoration treatments on wildlife need to be understood because it is imperative that forest restoration not focus solely on forest structural attributes without consideration of impacts to native wildlife.

Our study objectives were to determine (1) the composition of the avian community, (2) the foraging behavior and habitat of cavity-nesting birds, and (3) the nesting success and nesting substrate of as many species within the bird community as possible.

We monitored birds because (1) they are the most easily and inexpensively detected and identified vertebrate animals, (2) a single survey method can cover many species, and (3) accounting for and maintaining many species with different requirements promotes conservation strategies at the landscape scale (Hutto 1998, Hutto and Young 2002). In this paper we present results of pretreatment monitoring for the avian portion of the Fire and Fire Surrogate Study. We anticipate these results will be useful when interpreting the posttreatment monitoring results. In addition, they provide valuable insights into the ecology of avian species in dry forest stands that have experienced several decades of fire exclusion, a condition that is predominant across dry forest landscapes (Agee 1993, Harrod et al. 1999, Hessburg et al. 2005). Our study objectives were to determine (1) the composition of the avian community, (2) the foraging behavior and habitat of cavity-nesting birds, and (3) the nesting success and nesting substrate of as many species within the bird community as possible.

Methods

Avian Community Composition

Point counts are a relatively standardized method for estimating the relative abundance and diversity of avian species (Ralph et al. 1993). Point counts were conducted at 4 to 6 points per stand for four visits to each of the 12 study stands during May and June. Point count centers were located 200 m apart and 100 m from the stand edge. Each point count began within ½-hr of the official sunrise and was completed no later than 10 a.m. PST. Once at the point, the observer waited at least 2 min., then identified and counted birds for 10 min. Birds were identified by song, call, or visual observations. Detections of birds were recorded at 10-m increments out to 80 m using horizontal distance. Flyovers were documented but not used in the analyses. Standard four-letter codes for each bird species were recorded.

A survey of all stands was completed before the next survey was initiated. Logistical constraints prevented a random sequencing of sampling. Some stands were grouped together (Crow 1, Crow 3, Crow 6) and sampled on a single day to improve sampling efficiency. All points used for counts were georeferenced using global positioning system.

Nest Productivity and Substrate

Standardized methods have been developed to assess avian productivity (Martin and Geupel 1993, Ralph et al. 1993). Nest searches provide the most direct measurement of nest success in specific habitats. They also allow identification of important habitat features associated with successful nests and insight into important habitat requirements and species coexistence. We conducted nest searching and monitoring from early May until mid July.

Nest searches were conducted in two replicates of each treatment (including controls), and nests were monitored until the fate (fledging young or failure) was determined. Stands were thoroughly searched for nests following routes that traversed all parts of the stands. Once a nest was found, flagging was used (10 to 15 m away) to indicate the species and nest number. Detailed instructions with a drawing were made so that the nest could be relocated for subsequent monitoring.

Nests were monitored from a distance and all efforts were made to minimize disturbance. Nests were checked every 2 to 4 days, keeping careful track of the stage of each nest. Species-specific literature on clutch sizes, incubation, and nestling periods was used to estimate when incubation, hatching, or fledging was likely to occur, and more frequent visits were made during these times. Dead-end paths were avoided when checking nests by entering along one path and exiting along another so that predators had difficulty determining the exact nest location. Active nests were not visited if predators were observed nearby.

Once the nesting season was over in 2001 (the nest substrate protocol was not available in 2000), nesting habitat data were collected for the most common species (>9 nests). At each nest we recorded the height of the nesting substrate (m), nest substrate diameter at breast height (d.b.h.) (cm) (if applicable), nest height (m), percentage cover at the nest, and the orientation of the nest (N, S, E,W).

Cavity-Nester Foraging Behavior and Habitat

We used focal animal sampling (Martin and Bateson 1986) to quantify foraging activities of cavity-nesting bird species such as nuthatches and chickadees. Birds were located within each stand by walking a random transect for a 2-hr observation session, for a total of 24 hr of observation per stand per year. Upon visual detection of a cavity-nesting species, we observed and recorded the first foraging behavior demonstrated by the bird (Hejl et al. 1990) after a 10-sec delay to eliminate conspicuous behavior. To avoid collecting multiple observations of the same individual and to insure independence between observations, we collected data on individuals of the same species only if they were detected >120 m apart during a sampling session (Bell et al. 1990, Hurlbert 1984, Weikel and Hayes 1999). We then recorded habitat information for the substrate upon which they were feeding at the observed moment. Habitat variables were tree species, tree height (m), d.b.h. (cm), status (alive or snag), beetle presence, fire effects (evidence of past fire), percentage of bark present, snag condition (based on Parks et al. 1997), and vertical and horizontal strata of foraging location on the tree.

Data for assessing "available" habitats were collected for all trees greater than 1.3 m tall in three 20- by 50-m Whitaker plots per study stand (see Harrod et al. this volume). Habitat variables recorded were tree species, tree height (m), tree d.b.h. (cm), and status (alive or snag). Canopy cover for each study stand was also calculated using a spherical densiometer. The available habitat data set consisted of 5,338 trees, from which 360 trees were randomly selected to describe habitat "available" to cavity-nesting birds. We used a subset of the 5,338 trees to provide a ratio of used:available habitat appropriate for our habitat selection modeling approach (Max, T. 2003. Personal communication, statistician, Pacific Northwest Research Station, Portland, OR 97208).

Statistical Analyses

We examined the compositional similarity of the avian communities among the treatments using a nonparametric multivariate analysis in PC-ORD4 software (McCune and Mefford 1999). We chose this method over univariate measures because it summarizes all the information on the bird community simultaneously, allowing for an assessment of the community-level response to the treatments (Hanowski et al. 2003). For the community-level analyses, we used data out to 50 m from point centers combining pretreatment (2000, 2001) years to calculate an index of relative abundance for species with >1 detection. Examination of the detection probabilities for several species using program DISTANCE (Buckland et al. 2001) indicated that they were nearly equal out to 60 to 70 m; therefore, we chose 50 m to be conservative. We used multiresponse permutation procedures (MRPP) to test the hypothesis of no difference in avian species composition between the treatments in pretreatment avian communities (Biondini et al. 1988, Zimmerman et al. 1985). This produced an A statistic that measured the grouping "effect size," or distinctiveness of groups, on a scale of 0 to 1. Values of A >0.3 are considered fairly high. Monte Carlo permutations calculated probabilities for differences between the treatments. To address the issue of multiplicity, we compared those probabilities with Bonferroni-adjusted P-values obtained by dividing the experiment-wise $P \leq 0.05$ by three comparisons.

We used indicator species analysis (ISA), in PC-ORD4, to identify characteristic species found mostly in a single type (treatment) and present in the majority of the sites (replicate stands) belonging to that type (Dufrene and Legendre 1997). The ISA combined information on both species relative abundance and constancy to estimate indicator values for each species in each group. The maximum indicator value for a species within the treatment types was tested for statistical significance against random expectation calculated by Monte Carlo permutation.

We used program DISTANCE to estimate bird density and assess detection probabilities of several species (Buckland et al. 2001). Again, data were pooled across the pretreatment (2000, 2001) sample years to increase sample sizes and the number of species for which we could calculate density estimates. We calculated mean density and 95-percent confidence intervals (CI) using data from the replicates of each treatment. We used the density estimates to compare stands among the assigned treatments. Abundance indices and ISA were used to corroborate the density estimate results. We did not use the abundance indices or the ISA alone to evaluate differences among assigned treatments because of difficulties in accounting for differing detection probabilities (Buckland et al. 2001).

We calculated daily survival rates, Mayfield estimates (Mayfield 1961, 1975), and standard error (SE) for species, or for a nesting guild (ground nester, foliage nester, cavity nester), with >20 nests (Hensler and Nichols 1981).

We analyzed selection of foraging habitat for cavity-nesting birds by comparing characteristics of foraged trees to randomly available trees within the study stands. We examined variables that would explain the variability inherent in habitat selection and attempted to create models that would be useful to habitat managers in predicting foraging habitat selection of cavity-nesting birds. A detection rate for cavity-nesting species was also calculated as the number of observations per 24-hr period. Additionally, we described bird foraging behavior and characteristics of selected substrates to supplement modeling results.

We designated five, a priori, binary linear logistic regression models of foraging tree selection that included data collected for both used and available trees. These model variables were chosen because (1) previous research indicated they are influential in habitat selection, (2) they are quantifiable components that can affect foraging habitat selection, and (3) they may be altered through forest management. We used SPSS Base Version 10.1 (SPSS, Inc. 2001) to model habitat selection. We evaluated each of the models separately using the "Enter" method, where all variables are entered in a single step. Our evaluation continued by applying

Akaike's Information Criteria (AIC) (Anderson et al. 2001, Burnham and Anderson 2002), adjusted for small sample sizes (AICc), to determine which of the models, or combination of models, best approximated the structure of our data set. The relative importance of each model variable was also calculated as described by Burnham and Anderson (2002: 168). This method provides a sum of the Akaike weights (w_i) for each model where the variable of interest is present. The most "important" variables therefore have the largest total Akaike weights.

Results

Avian Community Composition

We recorded a total of 5,999 detections representing 84 avian species in our surveys. We calculated abundance indices for 49 species (table 8-1) and had sufficient data to calculate density estimates for 9 species (table 8-2). The most commonly detected species included the chipping sparrow, dark-eyed junco, mountain chickadee, Nashville warbler, pine siskin, red-breasted nuthatch, Townsend's warbler, western tanager, and yellow-rumped warbler. The foraging guilds that composed the avian community included low understory/ground insectivores (36.2 percent), tree foliage insectivores (33.0 percent), bark insectivores (11.1 percent), tree seedeaters (10.5 percent), aerial insectivores (8.1 percent), omnivore/scavengers (0.8 percent), and nectivores (0.5 percent). Of the species for which we calculated abundance indices, 48 percent were neotropical migrants, making up 49.5 percent of the total detections.

None of the avian species assemblages differed among assigned treatments, but there were some differences in individual species densities and detections.

None of the avian species assemblages differed among assigned treatments based on the pretreatment data (MRPP A = 0.065, P = 0.901). However, there were some differences in individual species densities and detections among the assigned treatments (table 8-2). The chipping sparrow occurred at a higher density in the assigned thin-only stands compared to the assigned thin-burn stands (table 8-2). The Nashville warbler occurred at higher densities in the assigned burn-only, and thin-burn stands compared to the control (table 8-2). Red-breasted nuthatch was found in higher densities in the assigned thin-only and thin-burn stands compared to the control (table 8-2). Finally, the yellow-rumped warbler had higher densities in the assigned burn-only stands compared to the control stands (table 8-2). In addition to these differences, Wilson's warbler and white-throated swift were only detected in the assigned burn-only stands (tables 8-1, 8-3).

Table 8-1—Mean avian abundance and 95-percent confidence interval (95% CI) by treatment for species with >1 detection for the pretreatment sampling years 2000–2001, Northeastern Cascades Fire and Fire Surrogate Study site

		Thin		Thin-burn		Burn		Control	
Scientific name	Common name	Mean	±95% CI	Mean	±95% CI	Mean	±95% CI	Mean	±95% CI
		Detections per point per visit							
Low understory ground insectivores:									
Carpodacus purpureus	Purple finch	0.25	0.12	0.18	0.07	0.28	0.17	0.26	0.24
Catharus guttatus	Hermit thrush[a b]	0.05	0.06	0.16	0.18	0.06	0.02	0.05	0.08
Catharus ustulatus	Swainson thrush[b]	0		0.04	0.04	0.03	0.06	0.03	0.04
Junco hyemalis	Dark-eyed junco	0.99	0.36	1.12	0.18	0.89	0.22	1.07	0.30
Molothrus ater	Brown-headed cowbird[b]	0.06	0.06	0.11	0.16	0.16	0.09	0.07	0.04
Myadestes townsendi	Townsend's solitaire	0.30	0.14	0.31	0.07	0.48	0.09	0.36	0.19
Oporornis tolmiei	MacGillivray's warbler[b c]	0.24	0.17	0.20	0.16	0.35	0.25	0.22	0.20
Pheucticus melanocephalus	Black-headed grosbeak[b]	0.02	0.02	0.15	0.26	0.08	0.14	0.05	0.04
Pipilo maculatus	Spotted towhee	0.17	0.09	0.42	0.35	0.31	0.26	0.48	0.14
Spizella passerina	Chipping sparrow[a b]	0.33	0.27	0.37	0.35	0.44	0.19	0.40	0.43
Troglodytes aedon	House wren[b]	0		0.05	0.10	0.01	0.02	0	
Troglodytes troglodytes	Winter wren	0.02	0.02	0.01	0.02	0		0.01	0.02
Turdus migratorius	American robin	0.09	0.07	0.24	0.29	0.15	0.10	0.22	0.16
Vermivora ruficapilla	Nashville warbler[b]	0.54	0.50	0.78	0.92	0.77	0.67	0.44	0.20
Tree foliage insectivores:									
Catharus fuscescens	Veery[b]	0		0.05	0.10	0		0.02	0.03
Dendroica coronata	Yellow-rumped warbler[b]	0.74	0.19	0.71	0.26	0.80	0.28	0.57	0.49
Dendroica petechia	Yellow warbler[b]	0.02	0.04	0.05	0.08	0.03	0.05	0	
Dendroica townsendi	Townsend's warbler[b]	0.81	0.80	1.05	1.19	0.64	0.36	0.39	0.48
Passerina amoena	Lazuli bunting[b]	0		0		0.03	0.01	0.04	0.04
Piranga ludoviciana	Western tanager[b d]	0.94	0.45	0.73	0.36	0.92	0.27	0.92	0.41
Poecile gambeli	Mountain chickadee	0.51	0.25	0.71	0.31	0.75	0.07	0.55	0.09
Poecile rufescens	Chestnut-backed chickadee	0.01	0.02	0.06	0.11	0		0.02	0.03
Regulus satrapa	Golden-crowned kinglet	0.08	0.05	0.19	0.24	0.07	0.12	0.01	0.01
Wilsonia pusilla	Wilson's warbler[b]	0		0		0		0.02	0.01
Vireo cassinii	Cassin's vireo[b]	0.22	0.18	0.15	0.04	0.33	0.13	0.29	0.22
Vireo gilvus	Warbling vireo[b]	0.01	0.01	0.03	0.05	0.06	0.12	0.01	0.03
Zenaida macroura	Mourning dove	0		0.01	0.01	0		0.01	0.01
Bark insectivores:									
Certhia americana	Brown creeper[a]	0.05	0.04	0.03	0.03	0.09	0.09	0.07	0.06
Colaptes auratus	Northern flicker	0.06	0.07	0.06	0.06	0.05	0.04	0.07	0.07
Picoides albolarvatus	White-headed woodpecker[c]	0.02	0.02	0.01	0.01	0.01	0.02	0.03	0.03
Picoides villosus	Hairy woodpecker	0.08	0.06	0.09	0.13	0.15	0.22	0	
Sitta canadensis	Red-breasted nuthatch[d]	0.92	0.20	0.97	0.09	0.78	0.10	0.81	0.28
Sitta carolinensis	White-breasted nuthatch	0.04	0.05	0.06	0.09	0.07	0.14	0.02	0.03
Sitta pygmaea	Pygmy nuthatch[a]	0.01	0.02	0.01	0.02	0		0	
Aerial insectivores:									
Aeronautes saxatalis	White-throated swift[b]	0		0		0		0.01	0.03
Empindonax difficilis	Pacific-slope flycatcher[b]	0.01	0.01	0		0.03	0.03	0.08	0.13
Empindonax oberholseri/ hammondii	Dusky/Hammond's flycatcher[b]	0.63	0.68	0.61	0.54	1.35	0.62	0.58	0.58
Tachycineta thalassina	Violet-green swallow[b]	0		0.01	0.01	0		0.01	0.03

Table 8-1—Mean avian abundance and 95-percent confidence interval (95% CI) by treatment for species with >1 detection for the pretreatment sampling years 2000–2001, Northeastern Cascades Fire and Fire Surrogate Study site (continued)

		Thin		Thin-burn		Burn		Control	
Scientific name	**Common name**	**Mean**	**±95% CI**	**Mean**	**±95% CI**	**Mean**	**±95% CI**	**Mean**	**±95% CI**
		Detections per point per visit							
Tree seedeaters:									
Carduelis pinus	Pine siskin	0.56	0.40	0.66	0.20	0.59	0.30	0.54	0.23
Carpodacus cassinii	Cassin's finch[c]	0.28	0.17	0.27	0.23	0.21	0.17	0.35	0.30
Coccothraustes vespertinus	Evening grosbeak	0.01	0.01	0.02	0.04	0.03	0.04	0.01	0.02
Loxia curvirostra	Red crossbill	0.20	0.06	0.18	0.05	0.10	0.04	0.14	0.11
Nucifraga columbiana	Clark's nutcracker	0.02	0.02	0.06	0.04	0.05	0.05	0.03	0.01
Omnivore-scavengers:									
Corvus corax	Common raven	0.02	0.02	0.05	0.07	0.01	0.01	0.02	0.04
Cyanocitta stelleri	Steller's jay	0.03	0.05	0.03	0.06	0.07	0.13	0.02	0.04
Perisoreus canadensis	Gray jay	0	0	0.05	0.07	0.01	0.02	0.01	0.03
Nectivores:									
Selasphorus rufus	Rufous hummingbird[b]	0.01	0.01	0.01	0.01	0.01	0.01	0	
Stellula calliope	Calliope hummingbird[b]	0.04	0.04	0.06	0.07	0.03	0.04	0	
	Unknown hummingbird	0		0.01	0.01	0.01	0.01	0.01	0.02

[a] Focal species in Altman 2000
[b] Neotropical migratory species
[c] Pacific Northwest Region Focal species (USDA FS 2006)
[d] Indicator species in Gaines et al. 2007

Table 8-2—Density (and confidence interval [CI]) estimates of avian species during the pretreatment sampling years of 2000 and 2001, Northeastern Cascades Fire and Fire Surrogate study site

	Control		Burn		Thin		Thin-burn	
Species	**Density**	**95% CI**	**Density**	**95% CI**	**Density**	**95% CI**	**Density**	**95% CI**
	Number per hectare							
Chipping sparrow[a]	0.84	0.41–1.74	0.68	0.41–1.15	1.83	0.94–3.57	0.30	0.13–0.70
Dark-eyed junco	2.62	1.77–3.89	3.54	2.28–5.52	5.51	3.55–8.56	4.08	2.64–6.29
Mountain chickadee	0.53	0.37–0.75	0.69	0.48–0.99	0.87	0.47–1.60	0.69	0.32–1.50
Nashville warbler	0.36	0.17–0.78	4.10	0.97–17.42	1.47	0.34–6.39	3.20	0.94–10.96
Pine siskin	0.75	0.39–1.47	0.34	0.16–.074	0.31	0.18–0.53	0.93	0.37–2.34
Red-breasted nuthatch[b]	0.50	0.37–0.69	0.76	0.54–1.09	1.02	0.78–1.34	1.37	1.00–1.87
Townsend's warbler	0.21	0.08–0.54	0.84	0.40–1.79	0.58	0.22–1.56	1.25	0.45–3.48
Western tanager[b]	0.56	0.39–0.80	0.90	0.56–1.46	0.56	0.35–0.89	0.60	0.33–1.07
Yellow-rumped warbler	0.49	0.25–0.96	2.27	1.00–5.15	0.54	0.43–0.67	0.53	0.28–1.01

[a] Focal species in Altman 2000
[b] Indicator species in Gaines et al. 2007

Table 8-3—Avian indicator species values by treatment for the pretreatment sampling years of 2000 and 2001, Northeastern Cascades Fire and Fire Surrogate study site

	Indicator species values				
Species	**Control**	**Thin**	**Thin-Burn**	**Burn**	**P-value**
Low understory ground insectivores:					
Chipping sparrow	21	24	29	26	0.946
Dark-eyed junco	24	27	22	26	0.701
Townsend’s solitaire	21	21	33	25	0.090
American robin	13	35	21	31	0.827
Brown-headed cowbird	14	27	41	18	0.243
House wren	0	83	17	0	1
Nashville warbler	21	31	30	17	0.907
Black-headed grosbeak	7	50	27	17	0.843
Spotted towhee	13	31	23	34	0.462
Hermit thrush	17	48	18	18	0.407
MacGillivray’s warbler	24	20	35	22	0.536
Purple finch	26	19	29	27	0.935
Swainson thrush	0	39	32	29	0.673
Winter wren	50	25	0	25	0.614
Tree foliage insectivores:					
Yellow-rumped warbler	26	25	28	20	0.766
Western tanager	27	21	26	26	0.960
Mountain chickadee	20	28	30	22	0.464
Townsend’s warbler	28	36	22	14	0.679
Cassin’s vireo	22	15	33	30	0.581
Chestnut-backed chickadee	12	68	0	20	1
Golden-crowned kinglet	22	55	20	3	0.291
Veery	0	75	0	25	1
Wilson’s warbler	0	0	0	100	0.022
Lazuli bunting	0	0	36	64	0.266
Warbling vireo	6	24	58	12	1
Mourning dove	0	33	0	67	0.496
Yellow warbler	19	47	31	3	0.526
Bark insectivores:					
Red-breasted nuthatch	26	28	22	23	0.408
Hairy woodpecker	25	28	45	1	0.441
White-breasted nuthatch	22	33	36	9	0.952
Pygmy nuthatch	50	50	0	0	1
White-headed woodpecker	33	10	14	43	0.730
Brown creeper	18	11	41	30	0.869
Northern flicker	26	26	20	29	0.896
Aerial insectivores:					
Pacific-slope flycatcher	6	0	24	71	0.489
Dusky/Hammond’s flycatcher	20	19	43	18	0.148
Violet-green swallow	0	33	0	67	1
White-throated swift	0	0	0	100	1

Table 8-3—Avian indicator species values by treatment for the pretreatment sampling years of 2000 and 2001, Northeastern Cascades Fire and Fire Surrogate study site (continued)

	Indicator species values				
Species	Control	Thin	Thin-Burn	Burn	P-value
Tree seedeaters:					
Cassin’s finch	25	24	19	31	0.754
Pine siskin	24	28	25	23	0.841
Red crossbill	32	29	16	23	0.472
Evening grosbeak	11	32	42	16	0.884
Clark’s nutcracker	13	38	29	21	0.381
Omnivore/scavengers:					
Gray jay	4	67	13	17	0.486
Steller’s jay	18	20	45	16	1
Common raven	22	48	7	22	0.791
Nectivores:					
Calliope hummingbird	30	44	23	2	0.719
Rufous hummingbird	43	29	29	0	0.728
Unknown hummingbird	0	29	29	43	1

Nest Productivity and Substrate

Overall nest success was highest for cavity-nesters, followed by ground-nesters, and then foliage-nesters.

We located and monitored a total of 198 nests representing 25 avian species (table 8-4). The most common species that we found nests for included the dark-eyed junco (n = 53), a ground-nester, and red-breasted nuthatch (n = 27), a cavity-nester. Overall nest success was highest for cavity-nesters (70 percent, n = 39), followed by ground-nesters (48 percent, n = 75), and then foliage-nesters (43 percent, n = 67). We did not detect any differences in daily survival rates (DSR) for ground-nesting and cavity-nesting species among the assigned treatments (table 8-5). However, DSRs were higher in the assigned control stands compared to the assigned burn-only and thin-only stands for foliage-nesters. We observed three incidents of cowbird nest parasitism, one each on MacGillivray’s warbler, Nashville warbler, and dusky flycatcher. We were able to calculate Mayfield estimates (not by treatment) for two species. Mayfield survival estimate for the dark-eyed junco was 27 percent (n = 53, 16.2 to 45.8, 95-percent CI) and for the red-breasted nuthatch was 59 percent (n = 27, 41.2 to 83.9, 95-percent CI).

We initiated the nest substrate measurement protocol in 2001 and gathered data on a total of 65 nests from four species; two cavity-nesters (n = 28), one foliage-nester (n = 9), and one ground-nester (n = 28). The average tree height used by cavity-nesters was 3.1 m for the mountain chickadee and 13.6 m for the red-breasted nuthatch, shorter than average height of trees available within the study stands, which was 16.9 m. The average tree height used by foliage-nesters was 18.6 m for the chipping sparrow, which was similar to available trees (18.6 m). Tree d.b.h. for cavity-nesters averaged 46.9 cm for mountain chickadees and 40.4 cm for red-breasted nuthatches. The foliage-nesting chipping sparrow used trees that averaged 47.1 cm. All three avian species used trees much larger for nesting than available in the study stands (mean d.b.h. = 22.5 cm). The amount of cover at the nest averaged 52.5 percent for the chipping sparrow, a foliage-nester, and 74.1 percent for the dark-eyed junco, a ground-nester.

Table 8-4—Number of nests/treatment and nest success (percentage) for the pretreatment sampling years of 2000 and 2001, Northeastern Cascades Fire and Fire Surrogate study site

	Burn		Control		Thin		Thin-burn	
Species	**Total**	**Successful**	**Total**	**Successful**	**Total**	**Successful**	**Total**	**Successful**
Ground-nesting species:								
Common nighthawk	1	0	0	—	3	1	0	—
Dark-eyed junco	12	4	17	9	13	7	11	6
Nashville warbler	3	2	1	1	0	—	0	—
Spotted towhee	2	1	0	—	2	2	1	0
Townsend's solitaire	2	1	0	—	5	2	2	0
Total	20	8 (40%)	18	10 (55%)	23	12 (52%)	14	6 (43%)
Foliage-nesting species:								
Cassin's vireo	0	—	1	0	1	1	1	1
Chipping sparrow	3	3	3	2	7	4	1	1
Dusky/Hammond's flycatcher	10	2	2	2	7	2	4	0
Hermit thrush	0	—	0	—	0	—	1	0
MacGillivray's warbler	1	0	2	1	0	—	0	—
American robin	0	—	0	—	4	3	0	—
Cassin's finch	1	0	0	—	0	—	0	—
Calliope hummingbird	0	—	0	—	1	0	1	1
Townsend's warbler	0	—	0	—	0	—	1	1
Western tanager	2	0	4	3	4	0	2	1
Yellow-rumped warbler	1	1	0	—	1	—	1	0
Total	18	6 (33%)	12	8 (67%)	25	10 (40%)	12	5 (42%)
Cavity-nesting species:								
American kestrel	0	—	1	1	0	—	0	—
Chestnut-backed chickadee	1	1	0	—	0	—	1	0
Hairy woodpecker	0	—	0	—	3	2	0	—
House wren	0	—	0	—	3	2	0	—
Mountain chickadee	6	6	5	3	3	2	1	1
Northern flicker	0	—	0	—	2	2	0	—
Pygmy nuthatch	0	—	0	—	1	0	1	0
Red-breasted nuthatch	8	6	5	3	10	6	4	3
White-breasted nuthatch	0	—	0	—	1	1	0	—
Total	15	13 (87%)	11	7 (64%)	23	15 (65%)	7	4 (57%)
All species	53	27 (51%)	41	25 (61%)	71	37 (52%)	33	15 (45%)

Table 8-5—Daily nest survival rates (DSR) and standard error (SE) by nesting guild during pretreatment breeding seasons, 2000–2001, Northeastern Cascades Fire and Fire Surrogate study site

Functional group	Number of nests	Number of observation days	DSR ± SE
Ground-nesting species:			
Control	18	193.5	0.964 + 0.013
Burn	20	228.5	0.952 + 0.014
Thin	23	294	0.963 + 0.011
Thin-burn	14	162.5	0.957 + 0.016
Foliage-nesting species:			
Control	12	112	0.982 + 0.013
Burn	18	221	0.950 + 0.015
Thin	25	252.5	0.945 + 0.014
Thin-burn	12	171.5	0.965 + 0.014
Cavity-nesting species:			
Control	11	239	0.983 + 0.008
Burn	15	327.5	0.994 + 0.004
Thin	23	427	0.984 + 0.006
Thin-burn	7	126.5	0.976 + 0.014

Cavity-Nester Foraging Behavior and Habitat

The available habitat data set consisted of 360 trees to describe habitat available to cavity-nesting birds. Approximately 50 percent of the trees available were ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) and 48 percent were Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). Canopy cover in the plots ranged from 66 to 95 percent (mean = 79 percent) closed canopy. Mean d.b.h. was 22.5 cm ± 0.7, and mean tree height was 16.9 m ± 0.4 (table 8-6).

Table 8-6—Mean (± standard error) diameter at breast height and height of trees used for foraging by cavity-nesting birds, and trees available within the pretreatment study stands, Northeastern Cascades Fire and Fire Surrogate study site

Bird species and habitat availability	Diameter	Height
	Centimeters	*Meters*
Hairy woodpecker	40.9 ± 2.51	17.5 ± 1.15
Mountain chickadee	38.6 ± 1.65	15.9 ± 0.62
Red-breasted nuthatch	43.4 ± 1.97	18.5 ± 0.63
Available habitat	22.5 ± 0.65	16.9 ± 0.37

In all, 253 foraging bouts were detected for all woodpeckers and cavity-nesting species. Observed species were black-capped chickadee (*Poecile atricapillus*), chestnut-backed chickadee, hairy woodpecker, mountain chickadee, northern flicker, pygmy nuthatch, red-breasted nuthatch, white-breasted nuthatch, and white-headed woodpecker. We conducted subsequent habitat selection analyses on the three species detected most frequently (88 percent of detections): hairy woodpecker (n = 32), mountain chickadee (n = 91), and red-breasted nuthatch (n = 99). In general, these species foraged on trees with diameters that were much greater than the average available trees within the study stands (table 8-6).

Hairy woodpeckers, mountain chickadees, and red-breasted nuthatches foraged on species with diameters much greater than the average available trees.

Logistic regression results suggested these three species exhibited selection with regard to foraging substrate (tables 8-7, 8-8). Although all five models were analyzed for all three bird species, we only retained those models with good support from the data, i.e., models with ΔAIC_c (Δ_i table 8-8) < 2. (Burnham and Anderson 2002). This resulted in two models for each species that most likely approximated the "true" model. Evidence ratios for the models not presented here were generally exponentially greater than the chosen models.

The global model was the most parsimonious model (i.e., lowest AIC_c value) for both hairy woodpecker and mountain chickadee (table 8-8). The second best mountain chickadee model, model 4, also had substantial empirical support. Model 4 was the most parsimonious model for the red-breasted nuthatch (table 8-8). The probability of the top-ranked models being "true," was much greater relative to the other models for the hairy woodpecker and red-breasted nuthatch. All models included the variables of tree species, diameter, height, and the diameter × height interaction term. The global model also included status.

All variables in each of the top-ranked models were relatively important ($\sum w_i \approx 1$) with the exception of status. Status was moderately important in the hairy woodpecker model ($\sum w_i = 0.68$) and the mountain chickadee model ($\sum w_i = 0.51$). Status was not as important in the red-breasted nuthatch model ($\sum w_i = 0.31$).

Table 8-7—Description of the models and variables tested to describe foraging selection of cavity-nesting birds in pretreatment sampling, Northeastern Cascades Fire and Fire Surrogate study site

Model	Model variations	Variable	Label	Definition
1	= tree spp.	x_1	tree spp.	Tree species (PIPO, PSME, ABGR)
2	= tree spp. + dbh	x_2	dbh	Tree diameter at breast height (cm)
3	= tree spp. + dbh + ht	x_3	ht	Tree height (m)
4	= tree spp. + dbh + ht + dbh × ht	x_4	dbh × ht	Interaction term, diameter × height
5	= tree spp. + dbh + ht + dbh × ht + status	x_5	status	Tree status, live or snag

Note: PIPO = *Pinus ponderosa*, PSME = *Pseudotsuga menziesii*, ABGR = *Abies grandis*, dbh = diameter at breast height.

Table 8-8—The best models for describing foraging habitat selection in cavity-nesting birds in pretreatment stands, Northeastern Cascades Fire and Fire Surrogate study site

Species	Model number	Model variables	*n* (used)	-2log(L)	K	AIC$_c$	ΔAIC	L(M_i/x)	Weight (w_i)
Hairy woodpecker	5	Tree spp. + dbh + ht + dbh × ht + status	31	114.84	7	133.71	0	1.0	0.68
	4	Tree spp. + dbh + ht + dbh × ht	31	120.26	6	135.76	2.05	0.36	0.24
Mountain chickadee	5	Tree spp. + dbh + ht + dbh × ht + status	91	230.04	7	245.39	0	1.0	0.51
	4	Tree spp. + dbh + ht + dbh × ht	91	232.50	6	245.50	0.11	0.95	0.49
Red-breasted nuthatch	4	Tree spp. + dbh + ht + dbh × ht	99	275.71	6	288.62	0	1.0	0.69
	5	Tree spp. + dbh + ht + dbh × ht + status	99	274.99	7	290.22	1.60	0.45	0.31

Note: K = number of parameters; AIC$_c$ = Akaike's Information Criterion adjusted for small sample sizes; ΔAIC = difference in AIC$_c$ between models; L(M_i/x) = evidence ratio; w_i = Akaike weights.

Tree size appeared to influence selection for all three bird species (table 8-9). Both diameter and height were useful in explaining habitat use variation. Tree diameter was influential in each of the "best" models and B_{dbh} was always positive and significant in models that included diameter (table 8-9). All three bird species selected for trees with a diameter at least 1.7 times as large as the average available diameter. The average size of the foraging trees used by hairy woodpecker was 40.9 cm (±2.51 cm SE), for the mountain chickadee it was 38.6 cm (±1.65 cm SE), and for the red-breasted nuthatch 43.4 cm (±1.97 cm SE). Most foraging observations occurred on 31- to 40 cm-diameter trees (fig. 8-1). Height was also influential, but

Table 8-9—Values of *β*, standard error (SE) and confidence intervals for influential foraging selection variables in the top-ranked model for each cavity-nesting bird species within the Mission Creek study stands

Bird species	Covariate	β ± SE	95% confidence interval for β
Hairy woodpecker	diameter	0.369 ± 0.092	1.208, 1.731
	height	0.050 ± 0.142[a]	0.796, 1.388
	dbh × height	-0.009 ± 0.004	0.983, 0.999
	status	1.559 ± 0.662	1.299, 17.386
Mountain chickadee	diameter	0.356 ± 0.043	1.313, 1.553
	height	-0.218 ± 0.058	0.718, 0.900
	dbh × height	-0.006 ± 0.697	0.991, 0.996
	status	1.055 ± 0.001[a]	0.733, 11.247
Red-breasted nuthatch	diameter	0.317 ± 0.047	1.253, 1.505
	height	-0.078 ± 0.071[a]	0.805, 1.062
	dbh × height	-0.006 ± 0.002	0.990, 0.997

[a] Covariate was not significant at $P < 0.05$.

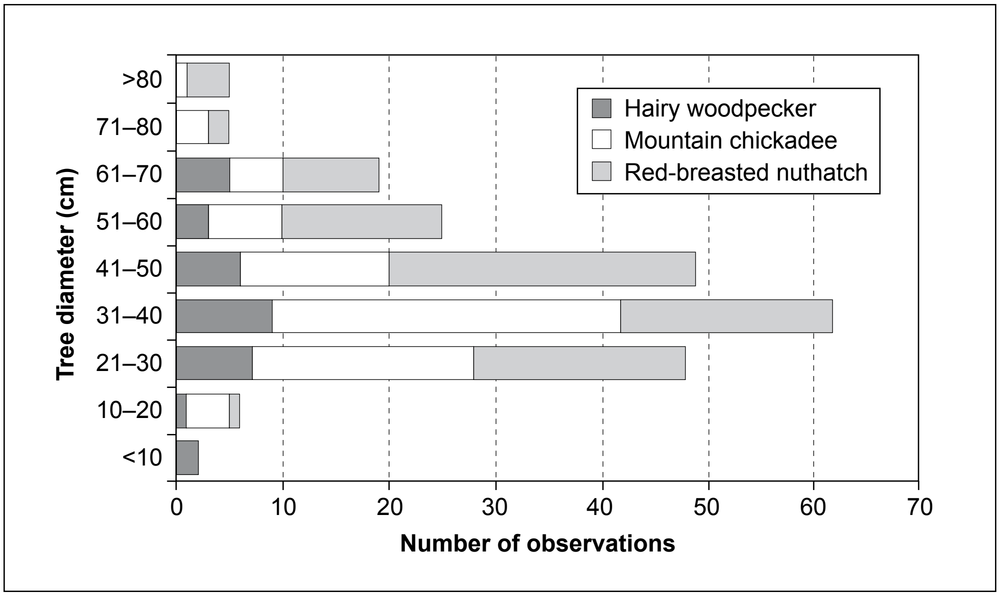

Figure 8-1—Diameter class of trees used as foraging substrate by cavity-nesting birds in the pretreatment stands, Northeastern Cascades Fire and Fire Surrogate study site.

the relationship, in terms of correlation and significance, differed among species. Values of β_{ht} and $\beta_{dbh \times ht}$ were negative for all species with the exception of β_{ht} for hairy woodpeckers. The interaction term was significant for all three species, whereas height was only significant for mountain chickadees. Most foraging observations occurred on trees 11 to 15 m tall (fig. 8-2).

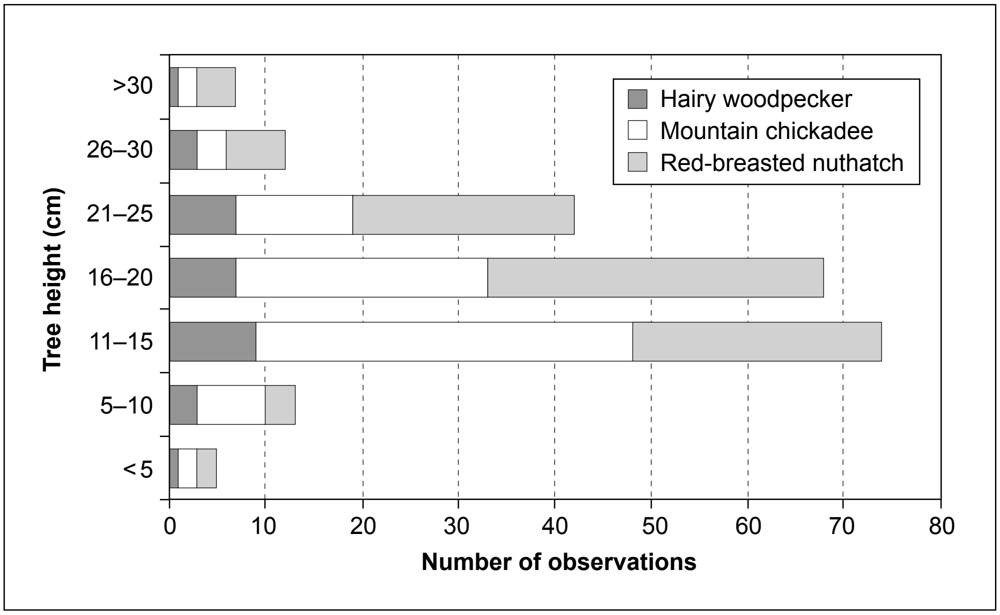

Figure 8-2—Height class of trees used as foraging substrate by cavity-nesting birds in the pretreatment stands, Northeastern Cascades Fire and Fire Surrogate study site.

Tree species did not influence selection. Birds were observed foraging more frequently on ponderosa pine (fig. 8-3); however, the relationship was not statistically significant relative to availability.

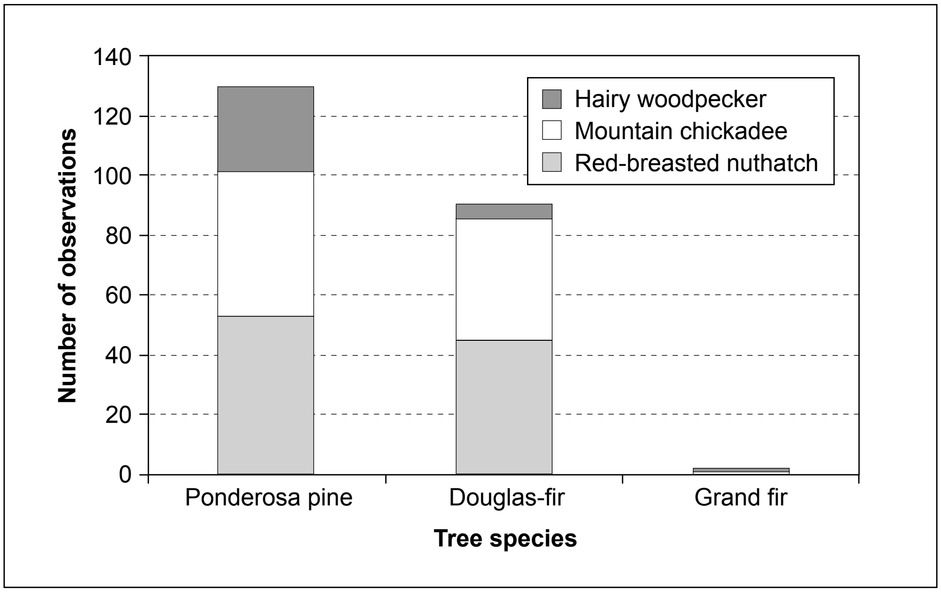

Figure 8-3—Tree species used as foraging substrate by cavity-nesting birds in the pretreatment stands, Northeastern Cascades Fire and Fire Surrogate study site.

Tree status indicated whether the tree upon which the bird foraged was alive or dead (snag). Status did not significantly influence selection with the exception of hairy woodpeckers, suggesting woodpeckers selected for snags. Woodpeckers accounted for 40 percent of overall snag use. Only approximately 8 to 9 percent of the trees available were snags.

We also examined foraging activity on different substrates (table 8-10). The majority of observations were located on living portions of live trees. With the exception of hairy woodpeckers, birds were seldom observed using snags or dead portions of trees as a foraging substrate. Mountain chickadees and red-breasted nuthatches predominantly foraged on distal or medial portions of branches. Hairy woodpeckers were usually observed foraging on proximal portions of branches or on the bole of the tree. The vertical locations of observations were similar for all three species. Although birds demonstrated a slight tendency to forage higher up on the tree, observations were distributed fairly evenly between low and high locations on the tree. This does not give an indication of actual tree height, but rather relative location on the tree. Woodpeckers were generally observed pecking at or excavating the substrate, whereas chickadees and nuthatches were observed gleaning insects from the bark surface or foliage.

Table 8-10—Percentage of bird detections relative to tree status and location on the tree for cavity-nesting birds observed foraging in pretreatment stands, Northeastern Cascades Fire and Fire Surrogate study site

	Status (of tree or branch)		Horizontal location		Vertical location	
Species	Dead	Alive	Proximal or bole	Distal or medial	High	Low
			Percent			
Hairy woodpecker	26	74	90	10	52	48
Mountain chickadee	6	94	18	82	58	42
Red-breasted nuthatch	11	89	36	64	60	40

Bole = main trunk of tree or snag.

High = upper half of tree/snag; low = lower half of tree/snag.

Proximal = proximal end of branch; medial = midway point on branch; distal = distal end of branch.

Detection rates differed among bird species. Detection rates for mountain chickadees and red-breasted nuthatches were at least three times that of hairy woodpecker rates. Hairy woodpeckers were observed approximately 1.3 times per hour. Mountain chickadees and red-breasted nuthatches were observed approximately 3.8 and 4.1 times per hour, respectively. Detection rates for all other species were less than 0.4 per hour.

Detection rates for mountain chickadees and red-breasted nuthatches were at least three times that of hairy woodpecker rates.

Discussion

Avian Community Composition

Dense dry forest stands provide habitat for a bird community different than found in presettlement forests, and represent a forest condition that is currently dominant on the dry forest landscape (Agee 1993, Harrod et al. 1999, Hessburg et al. 2005). We found no differences in the overall avian community composition among stands, and this may be explained in part by the similarity in stand structure among the stands. Harrod et al. (this volume) found no significant differences in tree density and stocking, basal area, species composition, and canopy closure among the study stands. Gaines et al. (2007) found differences in avian species assemblages among dry forest stands that were dense due to fire exclusion, compared to stands that had been treated with restorative treatments.

Neotropical birds made up 49.5 percent of the avian community in our study area.

The dominant foraging guilds (tree foliage insectivore, low understory/ground insectivore) within our study stands are similar to those reported by Piper (1996) for Douglas-fir forests within the same subbasin. Neotropical birds made up 49.5 percent of the avian community in our study area. This compares to 33 percent in mature Douglas-fir forests and 45 percent in young Douglas-fir forests of the southern Washington Cascade Range (Manuwal 1991), and 70 percent in the oak woodlands of south-central Washington (Manuwal 2003). Forest stands in our study provide important habitat for a number of Neotropical bird species.

We limited our discussion about individual species to those either identified as a USFS Pacific Northwest Region (Region 6) focal species (USDA FS 2006) or a focal species in the east-slope Cascade Mountains conservation strategy (Altman 2000). The chipping sparrow was identified as a focal species for ponderosa pine forests in the east-slope Cascade Mountains conservation strategy (Altman 2000). Their populations have been reported as declining in the Cascade Mountains physiographic province (Sauer et al. 1999). They nest in open woodlands with a relatively open overstory (10 to 30 percent canopy closure) (Altman 2000, Baicich and Harrison 1997). The densities of chipping sparrow in our pretreatment stands ranged from 0.3 to 1.83 birds/ha, which are similar to those reported for the control stands and much lower than densities in the treated stands described in Gaines et al. (2007). The control stands in Gaines et al. (2007) were similar in stand structure to the pretreatment stands in this study. Based on the findings of Gaines et al. (2007), we suspect that restorative treatments will result in a positive numerical response for chipping sparrow.

Hermit thrush is a focal species in the east-slope Cascade Mountains conservation strategy for multilayered, structurally diverse mixed-conifer forests (Altman 2000). We did not detect the hermit thrush in sufficient numbers to estimate pretreatment densities. Putnam (1983) found significantly fewer hermit thrush in precommercially thinned stands compared to unthinned stands of ponderosa pine. Their abundance was positively correlated with foliage volume of shrubs and small trees indicating a preference for areas with a dense understory (Putnam 1983). Gaines et al. (2007) did not detect any hermit thrush in stands that were the most heavily thinned. These results indicate that the hermit thrush is likely to respond negatively to the Fire and Fire Surrogate treatments being implemented in this study.

The MacGillivray’s warbler was selected as a Region 6 focal species for riparian forests (USDA FS 2006). Monitoring of this species on the Wenatchee National Forest has shown a significant decline over a period of 10 years, 1992-2001 (Nott et al. 2005). We did not detect the MacGillivray’s warbler in sufficient numbers to estimate pretreatment densities. The abundance of MacGillivray’s warbler was lower in stands treated with thinning and prescribed fire described in Gaines et al. (2007). Lehmkuhl et al. (2007) found MacGillivray’s warbler to be four times as abundant in dry forest riparian habitats compared to dry forest upland habitats. Thus, although dry forest restoration treatments may reduce the abundance of

MacGillivray's warbler in dry forest upland habitats, riparian protections provided by the Northwest Forest Plan are likely important to the viability of this species.

Other studies have reported either a positive numerical response or no response for western tanager to prescribed fire and light overstory removal silvicultural treatments (Anderson and Crompton 2002, Bock and Bock 1983, Finch et al. 1997, Hejl et al. 1995). However, Gaines et al. (2007) found lower densities in stands that were treated with thinning and prescribed fire. In northeastern Oregon (Sallabanks et al. 2002) and west-central Idaho (Medin 1985), western tanager was associated with high canopy closures, perhaps explaining their response to the treatments described in Gaines et al. (2007). The densities that we found in pretreatment stands in this study ranged from 0.56 to 0.9 birds/ha, similar to those reported for control stands in Gaines et al. (2007) but higher than densities found in treated stands (0.31 to 0.32 birds/ha). We anticipate a negative posttreatment response for western tanager.

Reported responses of red-breasted nuthatches to prescribed fire (Finch et al. 1997, Horton and Mannan 1988) and light overstory-removal silvicultural treatments (Anderson and Crompton 2002, Finch et al. 1997) have been mixed. Gaines et al. (2007) reported reduced densities for the red-breasted nuthatch in stands treated with restorative treatments. We found densities in our pretreatment stands that ranged from 0.5 to 1.37 birds/ha, somewhat higher that those reported for control stands (0.67 birds/ha), and much higher than densities in treated stands (0.19 to 0.32 birds/ha) in Gaines et al. (2007). The red-breasted nuthatch has been associated with high canopy closure in northeast Oregon (Mannan and Meslow 1984, Sallabanks et al. 2002) and in west-central Idaho (Medin 1985). We predict a negative numerical response to the treatments in our study.

Pygmy nuthatch was selected as a focal species in the land-bird conservation strategy for the east slope of the Cascade Mountains owing to their association with large ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) trees (Altman 2000). We could not estimate densities for this species because of low pretreatment sample sizes. A decline in source habitat for the pygmy nuthatch was reported by Wisdom et al. (2000). We detected pygmy nuthatches in low numbers, similar to those reported in Gaines et al. (2007). It is unknown if we'll be able to evaluate any posttreatment effects because of small samples sizes.

The white-headed woodpecker is a focal species for ponderosa pine forests in the east-slope Cascade Mountains conservation strategy (Altman 2000) and for Region 6 (USDA FS 2006). Broad-scale habitat declines have been reported for this species, especially in the North Cascades (Wisdom et al. 2000). Several researchers have suggested that active management, such as Fire and Fire Surrogate treatments, be used to restore habitat for this species (Gaines 2000, Marshall 1997, Wisdom et al. 2000). Although we did detect white-headed woodpeckers in our pretreatment study stands, they were low in abundance (0.01 to 0.03)—much lower than the abundance (0.09) reported in Gaines et al. (2007) for the low-retention thinning/prescribed fire restoration treatment. We anticipate a favorable response by white-headed woodpeckers to our restorative treatments.

The brown creeper is a focal species for large trees in mixed-conifer forests in the east Cascade Mountains land-bird conservation strategy (Altman 2000). Declines in both numbers (Sauer et al. 1999) and available habitat (Wisdom et al. 2000) have been reported for the brown creeper. Brown creeper is associated with old forests having high structural complexity (Anthony et al. 1996, Carey et al. 1991, Lundquist and Mariani 1991). Heavy thinning has been documented to reduce the abundance of brown creepers (Weikel 1997). We only detected the brown creeper in relatively low abundance, making it uncertain if we can measure any treatment response for this species.

The western bluebird is a Region 6 focal species for open dry forests (USDA FS 2006). Recent declines have been noted for the western bluebird in many parts of its range (Ehrlich et al. 1988), and Wisdom et al. (2000) reported strong declines in source habitat in the North Cascades. Fire exclusion leading to a decrease in open forest habitat has been cited as one reason for this decline (Guinan et al. 2000, Wisdom et al. 2000). Germaine and Germaine (2002) studied the effects of ponderosa pine forest restoration treatments on the nesting success of western bluebirds in Arizona and found a higher probability of successfully fledging young, but a greater risk of parasitic infections that reduced postfledgling success. Wightman and Germaine (2006) reported habitat variables associated with higher nest success included increased herbaceous and bare ground cover, and reduced ponderosa pine densities. Gaines et al. (2007) reported a positive numerical response in the abundance of western bluebirds to restorative treatments, as western bluebirds were only detected in stands that were treated with thinning and prescribed fire. We predict a similar response in our study, as no western bluebirds were detected during pretreatment sampling.

The Cassin's finch was identified as a focal species in Region 6 of the Forest Service (USDA FS 2006). Sallabanks et al. (2002) reported that Cassin's finch was a generalist and showed no strong associations with any forest structural class in the Blue Mountains of Oregon. Mannan and Meslow (1984) suggested that managed forest may provide more openings that can be used for foraging and nesting by Cassin's finch. Gaines et al. (2007) found higher densities of Cassin's finch in stands treated by thinning/prescribed fire. We expect similar responses to our Fire and Fire Surrogate treatments.

Nest Productivity and Substrate

We consistently found that cavity-nesters were more successful in completing nesting attempts compared to ground- or foliage-nesters. This result was also reported by Zebehazy et al. (2004) at another Fire and Fire Surrogate study site in South Carolina. In addition, survival estimates for dark-eyed junco, a ground-nester, were much lower than the survival estimates of the red-breasted nuthatch. This information exemplifies the advantages of a cavity in providing extra protection for young birds resulting in increased survival.

Based on the low pretreatment sample sizes for nesting birds, it is unlikely that we will be able to analyze for treatment effects on nest survival of any individual species. Such analyses could be done at a broader spatial scale using information from the FFS network. For example, Farris et al. (in press) used data from nine network sites to analyze for treatment effects of nest survival. They found that ground-nesters experienced an increase in daily nest survival in FFS treatments, shrub-nesters had lower nest survival in treatments, and snag and tree-nesters showed no response to treatments. The results were highly variable among study sites (Farris et al., in press). Other studies of the effects of silvicultural treatments on nest survival have shown that although the treatments may enhance habitat, they can also predispose nests to higher levels of nest predation (Martin 1992, 1993) or increase the risk of parasitic infections (Germaine and Germaine 2002). However, our study is not designed to specifically determine the causes of nest failure.

Many studies have reported the importance of large-diameter trees as nesting habitat for cavity nesters (Adams and Morrison 1993, Bevis and Martin 2002, Bunnell et al. 2002a, Dickson et al. 2004, Lundquist 1988, Lundquist and Manuwal 1990, Thomas et al. 1979, Weikel and Hayes 1999). Our findings concur, as cavity-nesters used trees that were much larger than available. In addition, we found the same importance of large trees as nesting substrate for foliage nesters.

Cavity-nesters used trees that were much larger than available. In addition, we found the same importance of large trees as nesting substrate for foliage nesters.

Cavity-Nester Foraging Behavior and Habitat

Hairy woodpeckers, mountain chickadees, and red-breasted nuthatches foraged on large trees and snags in dry conifer forests within the Mission Creek Watershed.

Hairy woodpeckers, mountain chickadees, and red-breasted nuthatches foraged on large trees and snags in dry conifer forests within the Mission Creek Watershed. In our study, most of the trees available for foraging were alive, and use of snags differed among bird species. Although they foraged primarily on live trees, hairy woodpeckers selected for snags relative to snag availability. Hessburg et al. (this volume) found similar levels of woodpecker foraging activity among study stands. Bent (1939) found hairy woodpeckers prefer to forage on snags, where they excavate and probe bark to obtain insects. This behavior was also observed in woodpeckers in the Blue Mountains (Thomas et al. 1979), the coastal range of Oregon

(Weikel and Hayes 1999), and southern Washington (Lundquist 1988, Lundquist and Manuwal 1990). However, hairy woodpeckers in our study area and other regions (Conner 1980, Imbeau and Desrochers 2002) do not strictly limit foraging to snags and often foraged on live trees. The density of snag habitat did not differ among study stands; however, basal area was higher in the thin-only stands compared to the thin-burn (Harrod et al. this volume)

Snags were not a preferred foraging substrate for chickadees and nuthatches. In general, tree status was not an important variable in these models. The diet of nuthatches and chickadees consists of both seeds and insects that they forage from foliage and the bark surface (Ghalambor and Martin 1999, Lundquist 1988, McCallum et al. 1999). We observed these behaviors primarily on live trees. Similarly, Lundquist and Manuwal (1990) found chickadees preferred live trees and nuthatches did not exhibit selection. Mannan and Meslow (1984) found nuthatches foraging on snags infrequently in Oregon. Woodpeckers, nuthatches, and chickadees all forage primarily on arthropods (Ghalambor and Martin 1999, Jackson et al. 2002, McCallum et al. 1999), and were most likely keying into the distribution and abundance of their prey, not necessarily the substrate. Analysis of the bark condition of foraged trees indicated most trees and snags had a great deal of intact bark such as found on condition class 1 and 2 snags (Parks et al. 1997). Generally, more than 90 percent of the bark was intact on live trees, and most snags had more than 75 percent intact bark. Hessburg et al. (this volume) found that about 15 percent of the trees sampled in the study stands had evidence of bark beetles. The primary species were western (*Dendroctonus valens*) and mountain (*D. brevicomis*) pine beetles. This information may also help explain why hairy woodpeckers frequently foraged on live trees. Imbeau and Desrochers (2002) found three-toed woodpeckers (*Picoides tridactylus*) foraging on large snags that were less deteriorated than the average available and on large live trees that were more deteriorated than available. In Arizona (Dickson et al. 2004), hairy woodpeckers and other bark-gleaning species were observed on more vigorous trees. Farris et al. (2002) found that woodpeckers concentrate their foraging on younger snags.

Tree size, particularly diameter, appeared to be an influential aspect of habitat selection. All three bird species selected for larger diameter trees. The importance of diameter in habitat selection of cavity-nesting birds is well documented (Adams and Morrison 1993, Bevis and Martin 2002, Bunnell et al. 2002a, Dickson et al. 2004, Lundquist 1988, Lundquist and Manuwal 1990, Thomas et al. 1979, Weikel and Hayes 1999). Cavity-nesting birds, especially woodpeckers, need large-diameter trees for nesting, to have cavities capable of housing large birds (Thomas et al. 1979). Foraging tree diameters are not directly dependent on bird size, but again appear to be related to prey. In the spring, cavity-nesters are busily foraging for themselves and their young. To maximize foraging efficiency, birds will forage where they are most likely to find large quantities of prey (Krebs 1994). Nuthatches and chickadees are considered "major predators of forest insect pests in the Pacific Northwest" (Torgersen et al. 1990: 19). In spring, a large prey base is readily available (Adams and Morrison 1993, Lundquist and Manuwal 1990), and arthropod abundance and distribution is influenced by bark characteristics (Adams and Morrison 1993). Older trees usually have a greater diameter, which provides for greater surface area, deeper furrows in the bark, and greater likelihood of decay, all of which increase the likelihood of being inhabited by insects and exploited by birds (Adams and Morrison 1993, Bull et al. 1997).

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The relative importance of diameter and height as predictor variables was calculated as the sum of w_i in all models where the variables were present (Burnham and Anderson 2002). The sum of w_i for models containing *dbh* and *ht* was at least 0.98 for all model sets, indicating these were very important covariates when predicting foraging selection of cavity-nesting birds. However, tree height was not a consistent factor. Although inclusion of the diameter × height interaction term improved the model, there was likely greater variability in tree height than can be explained by these models. Several studies found tree height was not a consistent factor in habitat selection (Bunnell et al. 2002b, Conner 1980, Jackson et al. 2002). Bevis and Martin (2002), however, did find that birds selected for taller snags. A great amount of literature exists to describe the importance of tree diameter in predicting foraging selection (Dickson et al. 2004, Weikel and Hayes 1999), nesting selection (Bull et al. 1997, Thomas et al. 1979, Welsh and Capen 1992) and bird abundance (Morrison et al. 1987), whereas height does not appear to have as great an influence and can differ greatly (Conner 1980, Jackson et al. 2002). With regard to foraging, this may be because tree diameter has a greater effect than height on the immediate foraging surface area, thereby increasing foraging efficiency. More than half of the foraging observations were detected in the upper parts of the tree, where the diameter is actually smaller and the bark is smoother. The discrepancy increased for mountain chickadees and red-breasted nuthatches. Woodpeckers, nuthatches, and chickadees use different strategies for catching their prey (i.e., excavating vs. gleaning) (Ghalambor and Martin 1999, Jackson et al. 2002, McCallum et al. 1999) so foraging here might be explained by two possibilities. First, nuthatches and chickadees may have been foraging on insects or seeds in the foliage (Weikel and Hayes 1999). Or secondly, these two species are not as well equipped as woodpeckers to forage in deep bark furrows (Ghalambor and Martin 1999, Jackson et al.

2002, McCallum et al. 1999), so they may have been trying to forage in the more accessible bark, higher up on the tree bole and on branches.

Management for cavity-nesting birds is usually based on managing for nesting habitat through protection of snags. Although focusing on snags may be beneficial to some species, our information suggested that large live trees also provide important foraging habitat in dry forest types for cavity-nesting birds such as woodpeckers, chickadees, and nuthatches. Several studies concur that nesting cavities are not a limiting factor when adequate numbers of snags are available (Waters et al. 1990, Welsh and Capen 1992). However, in managed stands, snag abundance is often below optimal nesting levels (Ohmann et al. 1994, Thomas et al. 1979). Additionally, birds need far more foraging sites than nesting sites (Bunnell et al. 2002b); thus management should include adequate foraging habitat as well as nesting habitat.

Although snags may be beneficial to some species, large live trees provide important foraging habitat, and birds need far more foraging sites than nesting sites.

Management Implications

There are three findings from our pretreatment analyses that are worthy of discussion in terms of management implications. First, we found that large trees were important for both nesting and foraging habitats. For nesting, this included both cavity-nesters and foliage-nesters. Second, management of nesting habitat for cavity-nesters is typically focused on providing adequate snag habitat. Our analyses show that although snags are important nesting habitat component, so are large live trees, and both components need to be considered in silvicultural treatments. Finally, the study design posed limitations for gaining sufficient sample sizes for some avian species that allow for the assessment of treatment effects. Examples include the relatively few species in which adequate pretreatment samples allowed for estimates of density (only 18 percent of those in which abundance indices were calculated). Another example includes the inability to calculate survival estimates for any species at the stand or treatment level. These limitations may restrict the inferences that managers will be able to draw on the effects of dry forest restorative treatments on birds, especially on their survival.

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Chapter 9: Small Mammals

John F. Lehmkuhl[1]

Introduction

The ecology of small mammals in the eastern Cascade Range is interesting and important for several reasons. The area is a biogeographic ecotone for species typical of wet forests of western Washington, for species of the dry shrub-steppe of eastern Washington, and for species with primary distributions in forests of the Rocky Mountain region (Johnson and Cassidy 1997) (table 9-1). As such, species occurrences and abundances in the eastern Cascade Range vary along temperature-moisture environmental gradients that affect vegetation pattern in a regional east-west direction and locally by topography. For example, presence and abundance of species typical of wet western Washington forests decline in an easterly direction from the Cascade crest as the environment becomes increasingly dryer, but species may occur locally at low elevations on cool-moist northerly aspects. The distribution of shrub-steppe species follows an opposite pattern for warm-dry conditions. The complexity of the environment and the fact that most species occur at the edge of their range, where they may be less adapted than at the core of their ranges (Brown and Lomolino 1998), makes predicting species occurrence, abundance, and response to management actions difficult.

Small mammals may be good indicators of ecosystem integrity, or the integrity of “biodiversity pathways” of forest development in response to natural or human disturbance.

Small mammals may be good indicators of ecosystem integrity, or the integrity of “biodiversity pathways” of forest development in response to natural or human disturbance (Carey and Harrington 2001, Carey and Johnson 1995). Small mammals are a vital prey base for a host of mammalian and avian carnivores. The varied life histories of small mammals encompass a wide range of habitat requirements for live and dead herbaceous and woody vegetation. Feeding behavior ranges from insectivory to granivory to mycophagy, which involves small mammals in many ecological webs. Their short lifespan and high productivity suggest they will respond quickly to management that impacts the environment of the forest floor.

In this chapter, I describe patterns of pretreatment occurrence and abundance of small mammals in dry ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests at the Mission Creek, Washington, site of the Fire and Fire Surrogate (FFS) study (Weatherspoon and McIver 2000). Insectivores, mice, voles, chipmunks, and squirrels are the primary focus of the small-mammal study. I also compare those results to other studies of small mammals in the local area.

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Table 9-1—Small mammal insectivores and rodents expected to occur in upland dry forests studied by the Fire and Fire Surrogates study in the northeastern Cascade Range, Washington

	Habitat		Distribution[a]	
Species[b]	Forest	Shrub-steppe	West	East
Trowbridge's shrew (*Sorex trowbridgii*)	x		x	
Coast mole (*Scapanus orarius*)	x		x	
American shrew-mole (*Neurotrichus gibbsii*)	x		x	
Douglas's squirrel (*Tamiasciurus douglasii*)	x		x	
Keen's mouse (*Peromyscus keeni*)	x		x	
Creeping vole (*Microtus oregoni*)	x		x	
Townsend's chipmunk (*Neotamias townsendii*)	x		x	
Cascade ground squirrel (*Spermophilus saturatus*)	x		x	
Yellow-pine chipmunk (*Neotamias amoenus*)	x			x
Western gray squirrel (*Sciurus griseus*)	x			x
Masked shrew (*Sorex cinereus*)	x		x	x
Montane shrew (*Sorex monticolus*)	x		x	x
Northern flying squirrel (*Glaucomys sabrinus*)	x		x	x
Southern red-backed vole (*Clerthrionomys gapperi*)	x		x	x
Long-tailed vole (*Microtus longicaudus*)	x		x	x
Northern pocket gopher (*Thomomys talpoides*)		x		x
Great Basin pocket mouse (*Perognathus parvus*)		x		x
Montane vole (*Microtus montanus*)		x		x
Vagrant shrew (*Sorex vagrans*)	x	x	x	x
Deer mouse (*Peromyscus maniculatus*)	x	x	x	x
Bushy-tailed woodrat (*Neotoma cinerea*)	x	x	x	x

Note: Species are sorted by habitat, distribution, and taxonomic order.

[a] Primary species distribution in Washington state relative to the crest of the Cascade Range. West indicates a range primarily in western Washington; east indicates a range primarily in eastern Washington.

[b] Taxonomy according to Baker et al. (2003).

Source: Johnson and Cassidy 1997.

Methods

I used three trapping methods to target small mammals of varied size, natural history, and behavior in an efficient and humane (ASM 2007) manner. I strived to safely sample relatively large (8+ ha) and remote (up to 1.5 hours walk from a road) treatment units, and to replicate sampling designs of other regional small mammal studies (Aubry et al. 1991, 1997; Hallett and O'Connell 1997). I sampled terrestrial small mammals with pitfall kill traps and snap traps during a 5-week spring period to effectively capture different species (McComb et al. 1991) and to minimize the exposure of crews to aerosol transmission of endemic hantavirus while handling animals. Snap traps also more efficiently captured target rodent species in this area than Sherman live traps.[2] Kill trapping is controversial (Sullivan et al. 2003), but

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[2] West, S.D. 2000. Personal communication. Professor wildlife science, College of Forest Resources, Box 352100, University of Washington, Seattle WA 98195.

our methods were humane (ASM 2007), effective, and efficient. Moreover, more reliable data are collected by kill trapping than live trapping. Necropsy of collected animals allowed species that are difficult to identify in the field (e.g., shrews, voles, two potential species of deer mice) to be positively identified, and reproductive condition and age to be determined with greater certainty than with live trapping. I sampled squirrels and chipmunks with Tomahawk live traps using guidelines recommended by Carey et al. (1991) and used in other regional (Carey 1995, Lehmkuhl et al. 1999) and local (Lehmkuhl et al. 2006b) studies of arboreal rodents.

Sampling was on a 6 by 6 grid with 40-m spacing within each unit.

I sampled on a single 6 by 6 grid with 40-m spacing within each experimental unit, per the national FFS small mammal protocol (Weatherspoon and McIver 2000). I built pitfall traps from two large (No. 2) coffee cans taped together with duct tape, buried traps to the lip of the can, and inserted a plastic margarine container with the bottom cut out to prevent animals from getting out. Each trap was filled with about 10 cm of water to operate them as humane kill (drowning) traps. When possible, I placed pitfall traps alongside logs or other natural runways of small mammals or at the base of a tree as close to the grid point as possible, but not more than 5 m away. I placed two Tomahawk live traps at each of the 36 grid points. One trap was placed on the ground and the other on a tree bole 1.5 m above the ground. In addition to the FFS protocol, I placed Museum Special snap traps on a 6 by 11 grid (20-m spacing on rows and 40-m spacing on columns) inset into the basic 6 by 6, 40-m grid.

I sampled units over a 5-week period from mid-May through June 2000 in two groups of six experimental units. I operated Tomahawk and snap traps for 8 days over a 12-day period, which conformed to a recommended trapping period for arboreal rodents (Carey et al. 1991, 1995; Lehmkuhl et al. 2006b). Traps were set on Monday, checked and closed on Friday, then reopened Monday for another four trap-nights. I concurrently operated pitfall traps in each stand for 11 continuous days. Analysis of data from another local study (Gaines and Lehmkuhl 2007) indicated that the combined pitfall and snap trap effort over 11 days likely would be adequate to sample most species known to occur in the area.

Abundance for each species is reported as individuals captured per 100 trap-nights, after adjusting trap-nights for sprung, stuck, or missing snap or Tomahawk traps (Nelson and Clark 1973). I assumed that capture probabilities for species were similar among stands owing to similar overstory, understory, and forest floor conditions (Harrod et al., this volume), and proximity of stands (15 km maximum); thus, I assumed heterogeneity in capture probabilities for individual species would be small to negligible. Weather during the relatively short 6-week spring period was consistently favorable, so time effects would be negligible. These assumptions were necessary because removal estimates of abundance that could correct for potential differences in capture probability were not feasible to estimate. Many species were absent or captured in small numbers in individual stands: only deer mice (*Peromyscus maniculatus*) and yellow-pine chipmunk (*Neotamias amoenus*) were captured in all sites and in numbers that might allow removal estimates of abundance. Nomenclature follows Baker et al. (2003).

I characterized mammal assemblages within and among units in three ways using algorithms in PCORD software (McCune and Mefford 1999). I used a frequency-abundance curve to show dominance relationships among species (McCune and Grace 2002). I used TWINSPAN cluster analysis (Gauch 1982) to examine how stands differ in mammal assemblages and to characterize those assemblages. The use of TWINSPAN was justified by a hypothetical single environmental gradient (temperature-moisture) represented by the sample stands (McCune and Grace 2002). Pseudospecies cut levels (recoding of a single species' data into several separate "pseudospecies" abundance categories for analysis) for TWINSPAN were: $1 = 0 < \times \leq 2$ captures/100 trap-nights; $2 = 2 < \times \leq 5$ captures; and, $3 = 5 < \times \leq 10$ captures. I did not consider grouping continuous capture data for TWINSPAN a disadvantage, vs. using original data with other clustering techniques (McCune and Grace 2002), because of the inherent variability in a sample of 12 experimental units and relatively few captures of many uncommon species. Finally, I used multiresponse permutation procedures (MRPP) to test the hypothesis of no difference in small mammal assemblages among assigned treatment groups of stands (Biondini et al. 1988, Zimmerman et al. 1985). The MRPP method estimates an A statistic that measures the distinctiveness of groups on a scale of ~0 to 1. Values of $A > 0.3$ are considered fairly high for ecological studies. Monte Carlo permutations calculated the probability of observed vs. expected differences among assigned treatment groups. Assigned treatment groups were control (Crow 3, Sand 19, Sand 2), thin-only (Crow 1, Ruby, Slawson), burn-only (Pendleton, Poison Spromberg,), and thin-burn (Crow 6, Camas, Tripp).

Sampling captured 413 animals of 12 species—an average 3.8 animals per 100 trap-nights. Deer mice and yellow-pine chipmunks were commonly found in all units.

Results

I collected 413 animals of 12 species (table 9-2) in pitfall and snap traps over 10,800 trap-nights, with an average of 3.8 animals captured per 100 trap-nights. Deer mice and yellow-pine chipmunks were commonly found in all units. Deer mice, however, dominated the small mammal community (fig. 9-1) accounting for 52 percent of the catch and more than twice the captures of yellow-pine chipmunks (21 percent of catch). These two species were co-dominant, however, if dominance was assessed in terms of species' biomasses (deer mice = 797 g/100 trap-nights; yellow-pine

Table 9-2—Captures per 100 trap-nights of small mammals in pitfall and snap traps during June 2000 on the Mission Creek sites of the Fire and Fire Surrogates study, Washington

Unit[a]	SOMO[b]	SOTR	SOVA	NEGI	NEAM	NETO	THTA	PEPA	PEMA	CLGA	MILO	MIOR	Total
Camas	0.3	0.3			1.3	0.2	0.3		7.1	0.2		0.3	9.7
Crow 1			0.2		0.6		0.3		1.3				2.3
Crow 3		2.4	0.6		1.4			0.3	1.7				6.3
Crow 6					3.2			0.6	1.3				5.0
Pendleton					4.2				1.2				5.3
Poison		0.8			1.9		0.3		2.7			0.2	4.3
Ruby	0.4	3.7		1.9	0.2	0.2			6.9			0.2	12.6
Sand 19			0.3		3.5			0.8	4.6				9.8
Sand 2					0.4			0.3	0.8				1.4
Slawson	0.6	1.1	0.8	0.6	1.7		0.6		5.8		0.3	1.8	12.8
Spromberg		2.9	0.3		0.8				9.4			0.2	13.6
Tripp		0.4		0.2	0.4		0.3	0.7	3.7			0.5	6.1
Total	1.2	11.4	2.0	1.8	18.6	0.4	1.5	2.5	45.8	0.2	0.3	3.9	88.7
Total (%)	1.3	12.9	2.2	2.0	21.0	0.5	1.7	2.8	51.6	0.2	0.3	4.4	
Units (%)	25	58	42	25	100	17	42	42	100	8	8	50	

[a] Survey mean 900 trap nights per stand.

[b] SOMO, Montane shrew (*Sorex monticolus*); SOTR, Trowbridge's shrew (*S. trowbridgii*); SOVA, vagrant shrew (*S. vagrans*); NEGI, American shrew-mole (*Neurotrichus gibbsii*); NEAM, yellow-pine chipmunk (*Neotamias amoenus*); NETO, Townsend's chipmunk (*N. townsendii*); THTA, Northern pocket gopher (*Thomomys talpoides*); PEPA, Great Basin pocket mouse (*Perognathus parvus*); PEMA, deer mouse (*Peromyscus maniculatus*); CLGA, southern red-backed vole (*Clethrionomys gapperi*); MILO, long-tailed vole (*Microtus longicaudus*); MIOR, creeping vole (*M. oregoni*).

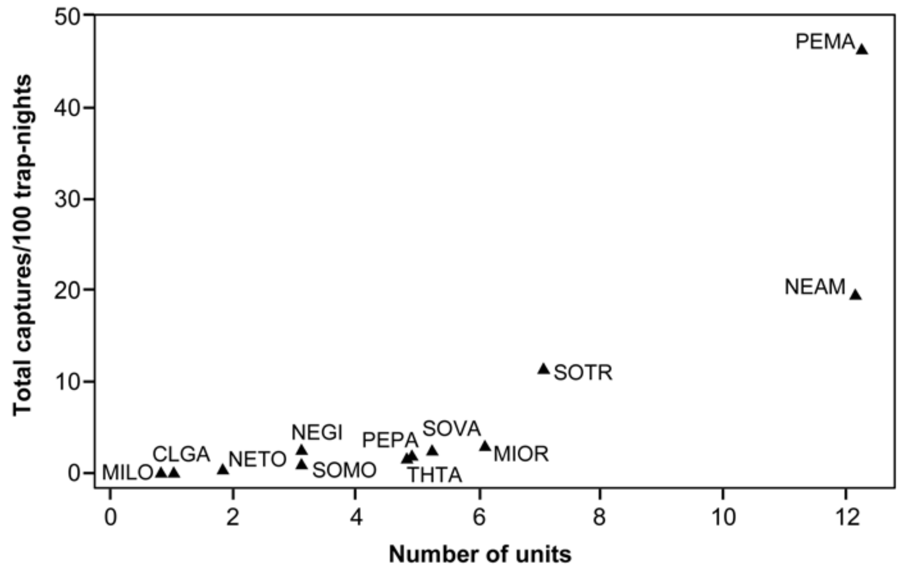

Figure 9-1—Dominance curve of small mammals captured during June 2000 with pitfall and snap traps in 12 units of the Mission Creek study site of the Fire and Fire Surrogate study. See Table 9-2 for explanation of species labels.

chipmunk = 807 g/100 trap-nights). Trowbridge's shrew (*Sorex trowbridgii*) was the third most abundant (13 percent of catch) and frequently caught (58 percent of units) species. The other nine species were a collection of rodents and insectivores that individually accounted for <5 percent of the total catch, and were captured in ≤50 percent of the units. Captures of northern pocket gophers (*Thomomys talpoides*) should be considered incidental because pitfall and snap traps poorly sample this fossorial species.

Pitfall and snap traps differed in effectiveness of capturing some species. Both trap types were equally efficient in capturing the dominant deer mice, which accounted for about 50 percent of the catch with both trap types (table 9-3). Relatively more insectivores were caught in pitfall traps. Captures of Trowbridge's shrew, in particular, were four times as many with pitfall as with snap traps. In contrast, the percentage of yellow-pine chipmunk captures with snap traps was six times that of captures with pitfall traps.

Table 9-3—A comparison of total captures/100 trap nights and percentage total catch of small mammals in pitfall and snap traps on the 12 Mission Creek sites of the Fire and Fire Surrogates study, Washington

	Pitfall traps		Snap traps	
Species	**Captures**	**Percent**	**Captures**	**Percent**
Montane shrew	0.8	2.3	0.4	0.7
Trowbridge's shrew	7.8	24.2	3.6	6.4
Vagrant shrew	1.8	5.5	0.2	0.4
American shrew-mole	0	0	1.8	3.2
Yellow-pine chipmunk	1.5	4.7	17.1	30.3
Townsend's chipmunk	0	0	0.4	0.7
Northern pocket gopher	1.5	4.7	0	0
Great Basin pocket mouse	2.3	7.0	0.2	0.4
Deer mouse	14.9	46.1	30.9	54.8
Southern red-backed vole	0	0	0.2	0.4
Long-tailed vole	0.3	0.8	0	0
Creeping vole	1.5	4.7	1.6	2.8
Total	32.3	100.0	56.3	100.0

Yellow-pine chipmunks dominated the catch from Tomahawk live traps (10.8 captures/100 trap-nights) (table 9-4). There were few captures of five other species, including northern flying squirrels (*Glaucomys sabrinus*) and bushy-tailed woodrats (*Neotoma cinerea*), which were of particular interest. Snowshoe hares (*Lepus americana*) captured were juveniles, and should be considered incidental because the Tomahawk traps were too small to properly sample adults.

Table 9-4—Captures per 100 trap-nights of small mammals in Tomahawk live traps during June 2000 on the Mission Creek sites of the Fire and Fire Surrogates study, Washington

Species[a]	LEAM[b]	SPSA	GLSA	NEAM	NETO	NECI	Total
Camas	0.2	0.4	0.5	1.2			5.5
Crow 1			0.2	0.4			3.2
Crow 3	0.4			0.7			3.2
Crow 6				0.9			2.6
Pendleton				2.5			8.0
Poison			0.2	0.5			4.1
Ruby	0.7			0.5	0.2		6.0
Sand 19			0.4	2.5			6.9
Sand 2				0.2			2.5
Slawson				0.7			3.3
Spromberg		0.2	0.5	0.7		0.2	5.3
Tripp			0.2				0.7
Total	1.2	0.5	1.9	10.8	0.2	0.2	51.2

[a] Survey mean 568 trap-nights per stand.

[b] LEAM, snowshoe hare (*Lepus americana*); SPSA, Cascade golden-mantled ground squirrel (*Spermophilus saturatus*); GLSA, northern flying squirrel (*Glacomys sabrinus*); NEAM, yellow-pine chipmunk (*Neotamias amoenus*); NETO, Townsend's chipmunk (*N. townsendii*); NECI, bushy-tailed woodrat (*Neotoma cinerea*).

Mammal assemblages differed among units, but two primary assemblages were apparent (table 9-5). Half the units, which were relatively mesic habitats, supported a richer assemblage of small mammals that included all the captured species in at least one unit of the group, compared to the other relatively species-poor dry units. Voles and insectivores, except for vagrant shrews (*Sorex vagrans*) were found mostly in the mesic units. Deer mice were found in all units, but were most abundant and dominated the catch in mesic units. In contrast, yellow-pine chipmunks likewise were caught in all units, but were more abundant in the dry units. The Great Basin pocket mouse (*Perognathus parvus*) was a third species associated with open patches in dry units. Mammal assemblages were similar among the assigned treatment groups of stands (MRPP A = -0.111, P = 0.899).

Units in relatively mesic habitats supported a richer assemblage of small mammals compared to dry units.

Table 9-5—TWINSPAN clustering of stands and species based on captures/100 trap-nights in pitfall and snap traps in the Mission Creek site of the Fire and Fire Surrogates Study, Washington

	Mesic stands[a]						Dry stands						
Species	**Camas**	**Ruby**	**Spromberg**	**Slawson**	**Poison**	**Tripp**	**Crow 1**	**Crow 3**	**Crow 6**	**Pendleton**	**Sand 19**	**Sand 2**	**Species group**
Southern red-backed vole	1[b]												000[c]
Long-tailed vole				1									000
Montane shrew	1	1		1									001
Townsend's chipmunk	1	1											001
Creeping vole	1	1	1	1	1	1							001
American shrew-mole		1		1		1							001
Deer mouse	3	3	3	3	2	2	1	1	1	1	2	1	01
Trowbridge's shrew	1	2	2	1	1	1		2					01
Northern pocket gopher	1			1	1	1	1						01
Great Basin pocket mouse						1		1	1		1	1	1
Vagrant shrew			1	1			1	1			1		1
Yellow-pine chipmunk	1	1	1	1	1	1	1	1	2	2	2	1	1
First-level stand groups[d]	0	0	0	0	0	0	1	1	1	1	1	1	
Second-level stand groups[d]	0	0	0	0	1	1	0	0	1	1	1	1	

[a] Stand type corresponds to first-level grouping at bottom of table.

[b] Captures/100 trap-nights recoded as pseudospecies with analysis categories (cut levels) of $1 = 0 < \times \leq 2$ captures; $2 = 2 < \times \leq 5$; $3 = 5 < \times \leq 10$.

[c] Sequence of 0 and 1 (in rightmost column) indicates membership in one of four species groups (indicated by a unique 0,1 sequence) typically found in similar environmental (stand) conditions.

[d] The sequence of 0,1 (in bottom rows) indicates membership in a stand group with similar species composition.

Discussion

Most (15 of 21) of the expected suite of species (table 9-1) were trapped in the FFS experimental units. Most of the expected species that were not collected in the FFS units, however, were trapped in low numbers by other nearby studies (table 9-6). The western gray squirrel (*Sciurus griseus*) has a very localized distribution north of the study area (Johnson and Cassidy 1997) and has not been detected in any dry forest studies near the FFS site. Small mammal assemblages caught in pitfall traps in the FFS study units were mostly similar in composition and richness to those captured in nearby dry forest areas, but the dominance of deer mice and Trowbridge's shrew differed among studies (table 9-6). Species similarly captured by autumn pitfall trapping for 28 days on 6 by 6 grids with 40-m spacing in the Peavine Canyon area just south of the FFS sites were similar to the FFS units, but deer mice dominated the Peavine catch with 72 percent of the captures (Gaines and Lehmkuhl 2007) vs. 46 percent in FFS units.

Table 9-6—Percentage composition of small mammals captured only with pitfall traps at the Mission Creek site of the Fire and Fire Surrogates (FFS) study compared to pitfall captures in nearby dry-forest areas from other studies

	FFS Mission Creek			Other studies		
Species	**Dry stands n = 6**	**Mesic stands n = 6**	**Total**	**Peavine[a] n = 12**	**Devil's Gulch[b] n = 8**	**Teanaway[b] n = 8**
Masked shrew				3		1
Montane shrew		3	2	3	3	7
Trowbridge's shrew	21	26	24	3	78	58
Vagrant shrew	8	4	5	5	2	2
American shrew-mole						3
Yellow-pine chipmunk	5	4	5	6		
Northern pocket gopher	3	6	5	2	1	
Great Basin pocket mouse	18	2	7	1		
Keen's mouse						13
Deer mouse	45	47	46	72	13	2
Southern red-backed vole				1		12
Long-tailed vole		1	1	5		2
Montane vole				2		
Creeping vole		7	5		3	2
Total	100	100	100	100	100	100

[a] Gaines and Lehmkuhl 2007. Peavine Canyon is in the Mission Creek drainage, and stands were similar to and immediately south of the FFS stands.

[b] Peffer 2001. Devil's Gulch sites were in the Mission Creek drainage and similar to FFS sites. Teanaway River sites were about 25 km east of Mission Creek and somewhat more mesic than Mission Creek sites. Data are from captures at distances >60 m from streams.

Peffer (2001) captured similar upland species in late spring on the two upland hillside (>60 m from a stream) rows of eight 5 by 6 pitfall grids (15-m spacing) sampled over 2 weeks in the nearby Devil's Gulch Fork of Mission Creek. In contrast to the FFS and Peavine sites, Trowbridge's shrew strongly dominated Peffer's catch with 78 percent of the captures, and deer mice were only 13 percent of captures. Slightly more mesic dry forest stands sampled by Peffer in the Teanaway River drainage, about 25 km west of Mission Creek, also were dominated by Trowbridge's shrew (58 percent of captures), but less so than the dryer sites in Devil's Gulch. The southern red-backed vole (*Clethrionomys gapperi*), Keen's mouse (*Peromyscus keeni*), and montane shrew (*Sorex monticolus*) were more abundant in the Teanaway than in the dryer FFS, Devil's Gulch, or Peavine stands (tables 9-5, 9-6).

As at the FFS site, uncommon species individually represented ≤5 percent of the captures at both the Peavine and Devil's Gulch sites. Some other species captured in those sites, but not in the FFS site, add to the list of dry forest small mammals. Masked shrews (*Sorex cinereus*) and montane voles (*Microtus montanus*) were captured in small numbers at the Peavine or Teanaway sites.

Peffer (2001) caught a few species in riparian areas that were not caught in the FFS or Peavine upland forest stands: water shrews (*Sorex palustris*) (2.7 percent of catch), Keen's mouse (2.3 percent), and western jumping mice (*Zapus princeps*) (0.9 percent). Peffer also had incidental captures of marsh shrews (*Sorex bendirii*) and coast moles (*Scapanus orarius*) in riparian or upland areas.

The dominance differences among Trowbridge's shrew and deer mice among studies is puzzling. The FFS, Peavine, and Peffer studies sampled in different years and seasons (spring 2000, fall 1996, spring 1997–98, respectively), so annual or seasonal differences in weather could have influenced the relative numbers of species. The strong dominance of Trowbridge's shrew in Peffer's study possibly could have been an artifact of differences in sampling design. Peffer's pitfall grid had 15-m spacing, whereas pitfall traps were spaced at 40-m intervals in the FFS and Peavine studies.

Trap type also influenced conclusions about terrestrial small mammal assemblages. Pitfall traps poorly sampled yellow-pine chipmunks, which both snap traps and Tomahawk live traps showed to be a dominant species in dry forest stands.

Other than yellow-pine chipmunks, unexpectedly few arboreal or semiarboreal rodents were captured. Lehmkuhl et al. (2006b) captured an average of 2.6 northern flying squirrels/100 trap-nights (range 1.9 to 3.0 across open- to closed-forest gradient) in similar dry forest in the Swauk Creek drainage 16 km southeast of the FFS sites, whereas, an average 0.32 squirrels/100 trap-nights were caught in FFS units. Lehmkuhl et al. (2006a) also captured bushy-tailed woodrats in all Swauk Creek study stands at an average rate of 0.5 woodrats/100 trap-nights (range 0.2 to 0.8 across years). In the FFS stands, woodrats were captured in only one unit at a capture rate of 0.2 woodrats/100 trap-nights.

The low numbers of arboreal rodents captured in FFS units compared to Swauk Creek could have resulted from differences in methods, season of capture, or inherent differences in stands. The 6 by 6 grid that barely fit in the relatively small FFS units was smaller than the 8 by 8 grid used in the Swauk study, which was the minimum size recommended by Carey (1991). An abundance of truffle and herbaceous food during the spring FFS trapping season compared to the autumn Swauk trapping season could have contributed to fewer captures.[3] The FFS stands may have been trapped during a low in populations, but winter weather prior to trapping was close to average, or at least not severe enough to reduce rodent numbers as observed by Lehmkuhl et al. (2006b). Finally, differences in the amount of

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[3] Carey, A.B. 1996. Personal communication. Chief research wildlife biologist (retired). U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, 3625 93rd Ave. SW, Olympia WA 98512.

cover habitat (snags, logs, mistletoe for both species) and food (truffles and lichens for flying squirrels) associated with both species (Lehmkuhl et al. 2006b) may have been lacking.

A final issue about comparing results among studies is the use of potentially biased capture indices instead of true abundance estimated by mark-recapture or removal modeling of capture probabilities. The use of captures/100 trap-nights as an index of abundance to compare stands or sites assumes that capture probabilities are equal across time and space within studies and across studies, which is considered an unlikely assumption (Williams et al. 2002). Nevertheless, there is some evidence that capture indices may be biased high or low, but can be highly correlated with mark-recapture abundance and useful for comparison of sites (Hopkins and Kennedy 2004, Lehmkuhl et al. 2006b, Ramsey et al. 2005, Ruscoe et al. 2001)

Despite the uncertainty of relative dominance among species from available studies, it is clear that Trowbridge's shrew dominates the insectivore group, deer mice dominate the small rodents, and yellow-pine chipmunks dominate among arboreal and semi-arboreal rodents. That general pattern and the contribution of other species to the small mammal community will depend on the geographical and landscape location of the stand.

Trowbridge's shrew dominates the insectivore group, deer mice dominate the small rodents, and yellow-pine chipmunks dominate among arboreal and semi-arboreal rodents.

The known distribution of species along the regional west-east temperature-moisture gradient (Johnson and Cassidy 1997) also was observed at smaller landscape scales. Species typical of mesic closed forests west of the Cascade crest (e.g., Trowbridge's shrew, creeping vole [*Microtus oregoni*]), or forest species with a wide distribution across Washington (e.g., southern red-backed vole) were more abundant closer to the crest (e.g., Keen's mouse in the Teanaway drainage). Those species also were locally more abundant in mesic than dry stands in the same landscape (this study, Peavine). Eastern forest species (e.g., yellow-pine chipmunk), shrub-steppe species (e.g., Great Basin pocket mouse [*Perognathus parvus*], northern pocket gopher [*Thomomys talpoides*]), or open forest generalists (e.g., deer mouse) were found in stands at the dry end of the gradient because of easterly longitude or topographic location.

Microhabitat patterns of understory vegetation, down wood, and climate in FFS units are the likely proximate basis for the observed mesic and dry forest small mammal assemblages (table 9-5). Dry units identified as having similar mammal assemblages are about 100 m lower in elevation, on warmer aspects, and have slightly lower tree density than mesic units (Harrod et al., this volume). Understory conditions in the FFS stands are most strongly differentiated by grass cover (Harrod et al., this volume), which in dry stands is three times that in mesic stands. Shrub cover in mesic units is 60 percent greater than in dry units, and is a primary driver of differences in understory among mesic stands. Mesic units also have larger amounts of small and large down wood than dry stands (Agee and Lolley, this volume), which adds to the structural complexity of small mammal cover and food resources. Biomass of fine woody debris (<0.62-cm diameter, 1-hour fuels) and large (>7.6-cm diameter, 1,000+ hour fuels) rotten down wood, which is cover and a potential source of invertebrate food, in mesic stands is 2 to 2.5 times that in dry units. Management practices that affect the temperature-moisture conditions by opening and drying a stand, reducing large down wood, and shifting understory dominance to grass will shift mammal species assemblages to favor species associated with the dry end of the gradient.

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English Equivalents

When you know:	Multiply by:	To find:
Microns (µ)	0.000394	Inches
Millimeters	0.0394	Inches
Centimeters (cm)	0.394	Inches
Meters (m)	1.094	Yards
Kilometers (km)	0.621	Miles
Hectares (ha)	2.47	Acres
Square centimeters (cm^2)	0.155	Square inches
Square meters (m^2)	10.76	Square feet
Square kilometers (km^2)	0.386	Square miles
Cubic centimeters (cm^3)	0.061	Cubic inches
Liter (l)	1.057	Quart
Grams (g)	0.036	Ounces
Grams	0.0022	Pounds
Kilograms (kg)	2.205	Pounds
Grams per cubic centimeter (g/cm^3)	0.036	Pounds per cubic inch
Kilogram per cubic meter (kg/m^3)	1.686	Pounds per cubic yard
Megagrams per cubic centimeter (Mg/cm^3)	180.6	Tons per cubic inch
Kilograms per hectare (kg/ha)	0.893	Pounds per acre
Megagrams per hectare (Mg/ha)	0.446	Tons per acre
Square meters per hectare (m^2/ha)	4.37	Square feet per acre
Trees per hectare	0.405	Trees per acre
Board feet (bf)	7	Cubic meters (m^3)
Degrees Celsius (C)	1.8 °C + 32	Degrees Fahrenheit

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