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Changes in stand structure and composition after restoration treatments in low elevation dry forests of northeastern Oregon

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Abstract

In many fire-dependent forests in the United States, changes occurring in the last century have resulted in overstory structures, conifer densities, down woody structure and understory plant communities that deviate from those described historically. With these changes, many forests are presumed to be unsustainable. Broad-scale treatments are proposed to promote stand development on trajectories toward more sustainable structures. Yet little research to date has identified the effects of these restoration treatments, especially in low elevation dry ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) forests of northeastern Oregon. We report changes in tree structure, coarse woody debris (logs), and understory composition from an operational-scale, replicated ($N = 4$), completely randomized experiment. Treatments included a single entry thin from below conducted in 1998, a late season burn conducted in 2000, a thin followed by burning (thin + burn), and a no action treatment which served as a control. Changes in live and dead tree structure and understory vascular plant community composition were compared between pre-treatment and 2004. Tree seedling density and composition and coarse woody debris structure were evaluated in 2004. Thin, burn, and thin + burn treatments reduced the density but not the basal area of live overstory trees. Thinning reduced the number of medium-diameter trees, and tended to decrease the abundance of shade tolerant, moist-site understory species yet increased the dominance of several rhizomatous species such as *Calamagrostis rubescens*, *Symphoricarpos albus*, and *Spiraea betulifolia*. Burning alone had little effect on large trees but reduced the number of small Douglas-fir and logs. Shade tolerant perennial species associated with fine textured soils such as *S. albus*, *Spiraea betulifolia*, *C. rubescens*, *Carex geyeri*, and *Arnica cordifolia* increased in frequency and average cover with burning. Conversely, cover of the bunch grass *Festuca idahoensis* was reduced while non-native invasive species establishment was little affected. Ordination scores suggested that burning increased the abundance of species representing greater shade tolerance and finer-textured soils. The thin + burn treatment left both ponderosa pine and Douglas-fir with modal or normal diameter distributions, and increased the abundance of understory species representing shallow, coarse texture soils. These results are discussed in the context of management options for restoration of ecosystem health in similar low elevation dry ponderosa pine and Douglas-fir forests.

Keywords: Stand structure; Ponderosa pine; Restoration treatments; Thinning; Prescribed burning; Douglas-fir

1. Introduction

Many fire-dependent forests in the United States – especially those with historically short-interval, low- to moderate-severity fire regimes – contain more small trees and fewer large trees, more down woody debris, and less diverse and vigorous understory plant communities compared to conditions under historical fire regimes (Kilgore and Taylor, 1979; Agee, 1993; Covington and Moore, 1994; Caprio and Swetnam, 1995; Arno et al., 1997; Taylor and Skinner, 1998). In low elevation dry forests of the Pacific Northwest, these shifts in forest structure and composition have been caused by fire exclusion, livestock grazing, timber harvests, and changes in climate (Steele et al., 1986; Dolph et al., 1995; Arno et al., 1997). Collectively, these altered structural conditions may contribute to increased probability of unnaturally severe wildfires, susceptibility to uncharacteristic insect outbreaks, and drought-related mortality (Wickman, 1992; Mutch et al., 1993; Stephens, 1998). Reports from the Interior Columbia Basin Ecosystem Management Project emphasized this problem (Quigley et al., 1996) and promoted large scale and strategically located thinning and burning to manage landscapes within the context of ecological

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processes (Hardy and Arno, 1996). Strategies for restoring forest structure and function include thinning live and dead trees to promote late seral structures and burning surface fuels to increase plant community diversity and vigor and reduce the risk of crown fires (Brown et al., 2004). Land managers implementing these treatments often lack a full understanding of the effects of these restoration treatments (Fiedler et al., 1992; Pollet and Omi, 2002; Noss et al., 2006).

Prior to the 20th century, low severity surface fires burned frequently in low elevation dry forests in the Pacific Northwest (Everett et al., 2000; Heyerdahl et al., 2002; Youngblood et al., 2004; Arabas et al., 2006). Ignitions were predominantly caused by lightning and coincided with the time of year when moisture content of fine fuels was lowest (Agee, 1993; Rorig and Ferguson, 1999). Under historical disturbance regimes, frequent surface fires controlled regeneration of fire-intolerant species, reduced density of small-diameter stems, consumed litter and down wood, opened the stands to increased sunlight, led to vertical stratification of fuels by eliminating fuel ladders between the forest floor and the overstory canopy (Agee, 1993), and maintained early seral plant associations. Crown fires occurred rarely under these natural disturbance regimes. Consequently, the structure in these stands generally consisted of open, predominantly widely spaced medium to large and old trees with continuous low herbaceous understory vegetation (Wickman, 1992; Agee, 1994).

There is broad agreement that some form of restoration treatment is necessary to move stands from their current structure and developmental trajectory to conditions that more closely incorporate natural disturbance regimes under pre-Euro-American influences. Restoring landscapes to some semblance of pre-Euro-American influence assumes that reference conditions can be quantified, that departures from reference conditions can be measured, and that the efficacy of treatments designed to move stands toward reference conditions can be measured (Landres et al., 1999). While reference conditions exist for low elevation dry forests along the east slope of the Cascade Range (Youngblood et al., 2004), similar work is lacking for other portions of the Pacific Northwest containing dry forests. Lacking definitive reference conditions, restoration treatments that incorporate thinning or burning may be designed to decrease crown density while retaining large live and dead trees of fire resistant species, to increase the height to live crown, to reduce surface fine fuels yet retain large woody debris, and to increase disturbance-adapted understory species underrepresented on the contemporary landscape (Agee and Skinner, 2005; Fulé et al., 2005).

This work is part of the Fire and Fire Surrogate study (Weatherspoon, 2000), a national network of 13 long-term study sites established to evaluate the ecological and economic consequences of treatments for restoring forest ecosystems. The effects of thinning and burning treatments on vegetation, fuels, wildfire hazard, soils, wildlife habitat and use, insect population dynamics, and ecosystem structure and process are being evaluated across 13 fire-dependent ecosystems. A randomized and replicated study design and a common set of response variables were used to facilitate comparisons among and between sites. Details of the network and links to individual sites are available at the web site http://www.fs.fed.us/ffs/.

This paper reports the direct effects of treatment and the first 4 years of change in forest structure at the Fire and Fire Surrogate study site in low elevation dry ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) forests of northeastern Oregon. Our overarching ecological hypothesis was that manipulative restoration treatments such as thinning and burning create distinctly different standing vegetation and down woody structure in the short-term (<5 years post-treatment) when compared to non-manipulated or control treatments. The following objectives are addressed in this paper: (1) Quantify changes in stand structural attributes (tree species composition, basal area, tree density, canopy height, crown distribution, or diameter distribution) that result from each treatment and how these changes differ between and among treatments. (2) Quantify the similarities and differences in down woody structure (log density, size, volume, or cover) resulting from each treatment. (3) Quantify the similarities and differences in understory vascular plant community composition and cover resulting from each treatment. (4) Relate this information to ecological restoration of ponderosa pine and Douglas-fir in low-elevation dry forests of the Pacific Northwest.

2. Methods

2.1. Study area

This study was conducted in the northern Blue Mountains of northeastern Oregon, USA. The study area (latitude 45°40′N, longitude 117°13′W) includes almost 9400 ha of plateaus, benches, and deeply incised drainages formed in highly fractured ancient Columbia River basalt. Soils are formed primarily from accumulated or redeposited volcanic ash and wind-blown loess; total soil depth varies with surface topography and often is greatest on north and east-facing slopes. Soils within the study area, from deep to shallow, include typic Vitrixerands (ashy over loamy-skeletal in the Olot series), vitrandic Argixerolls (fine-loamy in the Melhorn series and loamy-skeletal in the Larabee series), lithic ultic Haploxerolls (loamy-skeletal in the Fivebit series), and lithic Haploxerolls (loamy-skeletal in the Bocker series), and are highly variable within stands. North-facing plateau slopes, draws, and swales with relatively deep soils support grand fir (*Abies grandis*), Douglas-fir, ponderosa pine, and occasionally lodgepole pine (*P. contorta*). Warm and dry sites on gentle south-facing plateau slopes and ridge tops have thinner soils and support ponderosa pine and open, low shrub and grass-dominated communities. Discrete forest stands typically are ≤200 ha in size. Elevation ranges from about 1040 to 1480 m. The closest weather station is in Enterprise, OR (latitude 45°42′N, longitude 117°05′W) with continuous records since 1969. The climate is strongly continental; mean annual temperature is 7.4 °C (Fig. 1). Frost can occur any month of the year. Annual precipitation at Enterprise averages nearly

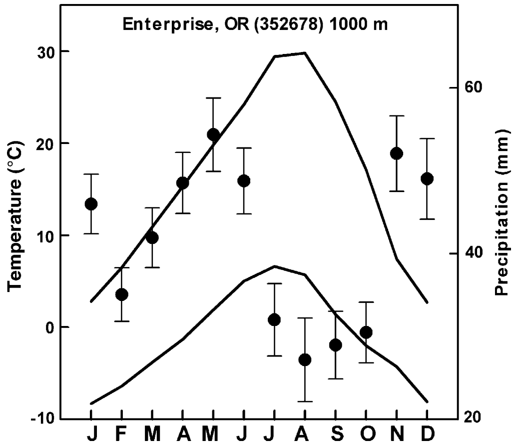

Fig. 1. Climatic features near the Fire and Fire Surrogate restoration study area in northeastern Oregon. Pattern of mean monthly maximum temperature and mean monthly minimum temperature (left axis and solid lines), with the pattern of mean monthly precipitation (right axis, dots with standard error bars) for the period 1969–2004 at Enterprise, OR.

500 mm. Two distinct periods of precipitation occur, the first as snow in November and the second as rain in March. Mean annual snowfall is 66 cm. Moisture within the growing season results from highly variable convection storms. Precipitation in this portion of the Blue Mountains is influenced at a regional scale by moisture moving eastward along the Columbia River and at a temporal scale in response to fluctuations in El Niño-Southern Oscillation (Heyerdahl et al., 2002). The Enterprise station is 100–400 m below the study site and may be in a rain shadow formed by adjacent high ridges of the Wallowa Mountains. Therefore, it probably receives less precipitation than the study area.

Forest composition and structure in the study area is representative of warm and dry biophysical environments common throughout the upper Columbia Basin of Washington and Oregon. Prior to the 20th century, ponderosa pine and Douglas-fir stands were dominated by late successional and old-growth structure in a mosaic pattern of large-diameter, single stratum, even-aged groups interspersed with non-forest communities across the landscape. Disturbance mechanisms during the 18th and 19th centuries were dominated by frequent, low-intensity fires burning at intervals of $\leq$20 years (Heyerdahl et al., 2001). Under this historical disturbance regime, surface fires resulting from lightning ignitions thinned stands from below by killing small-diameter stems through a combination of cambial and root damage and crown scorch, consumed litter and down wood, opened the stands to increased sunlight, and led to vertical stratification of fuels by eliminating ladder fuels represented by saplings of fire-sensitive conifers such as grand fir. In addition, occasional stand-replacing fires may have occurred during prolonged dry spells (Pierce et al., 2004). Other disturbances were caused by endemic levels of bark beetles (*Dendroctonus brevicomis* and *D. ponderosae*). Human-caused disturbances, especially past timber-cutting practices and exclusion of wildfire, have resulted in most of the study area characterized as the stem exclusion-closed canopy structural stage of development (as in O'Hara et al., 1996). These stands mainly occur as 70–100-year-old even-aged ponderosa pine and Douglas-fir pole and small-diameter sawtimber that regenerated after extensive partial cutting. Overtopping this cohort is a stratum of scattered large and old remnants of the previous stand that were not harvested earlier. At least two entries within the past several decades reduced densities in some stands. During the 1990s, the study area was affected by increased levels of bark beetles, especially *D. ponderosae* and *Ips pini*.

2.2. Field methods

2.2.1. Treatment implementation

We used aerial photos, maps, and reconnaissance visits to select potential experimental units from a set of 44 stands previously marked for fuels reduction. From a subset of 37 potential experimental units with the most similar topographic features and stand structure, we randomly selected 16 units and assigned experimental units and treatments for an operational experiment with a completely randomized design consisting of four treatments replicated four times. Experimental units ranged from 10 to 20 ha in size. Treatments included: (1) thin—a single entry thin from below, (2) burn—a single underburn, (3) thin + burn—a single entry thin from below followed by an underburn, and (4) control—untreated or no action. As part of the Fire and Fire Surrogate network, the primary objective of the treatments was to modify stand structure such that 80% of the dominant and co-dominant trees in the post-treatment stand would survive a wildfire modeled under 80th percentile weather conditions (Weatherspoon, 2000). Active treatments were designed to reduce total basal area from about 26 to about 16 $m^2\ ha^{-1}$.

Thinned treatment units were cut-tree marked, leaving dominant and codominant crown classes and accepting the often wide distribution in spacing to account for natural clumps. Overall, ponderosa pine was favored for retention over other species. All live trees $\geq$53 cm in diameter at breast height (dbh; 1.37 m above the ground) representing late and old seral structural characteristics were retained and competing conifers within 9 m of dominant ponderosa pine were removed. The thin treatment employed a cut-to-length harvesting system featuring a single-grip harvester and forwarder to remove merchantable live and standing dead and down material. Forwarders used the same trail system as the harvesters, operating on top of the accumulated limbs and tree tops within trails that ran up or down slope within each unit. All thinning occurred during the summer of 1998 after pre-treatment sampling.

Experimental units assigned the burn treatment had individual burn plans developed to account for differences in site physical features and fuel accumulations. Mortality targets for trees between 20 and 51 cm dbh were $\leq$30% for ponderosa pine, $\leq$40% for Douglas-fir, and $\leq$70% of the grand fir. Basal area of ponderosa pine $\geq$51 cm dbh was to be reduced by $\leq$20%, with similarly large Douglas-fir reduced by $\leq$30% and grand fir $\leq$50%. Burns were planned for fall 1999 to allow limbs and tops to cure, but were postponed until mid-September

Table 1
Fuel conditions and fire behavior for a set of restoration treatments in northeastern Oregon

Unit	Treatment	Ambient temperature (°C)	Relative humidity (%)	Wind speed (m s^{-1})	10-h fuel moisture[a] (%)	1000-h fuel moisture[b] (%)	Mean flame length (m)
8B	Burn	20–24	33–48	2.2	13.7 ± 0.3	63.0 ± 6.2	0.9
10B	Burn	20–24	33–48	2.2	12.8 ± 0.7	40.9 ± 7.2	0.9
21	Burn	12–23	23–50	0.8–2.8	15.0 ± 1.1	55.9 ± 8.1	0.6
24	Burn	13–21	25–49	0.8–2.2	12.2 ± 1.1	48.6 ± 7.0	0.5
6B	Thin + burn	12–21	24–76	0.8–3.6	12.9 ± 0.7	28.6 ± 2.7	0.9
8A	Thin + burn	20–24	33–48	2.2	17.8 ± 1.1	45.5 ± 3.4	0.9
10A	Thin + burn	20–24	33–48	2.2	17.6 ± 1.7	31.2 ± 4.0	0.9
1112	Thin + burn	12–23	23–50	0.8–2.8	16.4 ± 2.4	37.7 ± 4.5	0.9

[a] Fuel moisture in dead fuels from 0.6 to 2.5 cm in diameter, mean and S.E. from five samples; data on file, LaGrande Forest and Range Science Lab, LaGrande, OR.
[b] Fuel moisture in dead fuels from 7.6 to 20.3 cm in diameter, mean and standard error from 21 to 31 samples; data on file, LaGrande Forest and Range Science Lab, LaGrande, OR.

2000 because of weather. Burns were conducted as both backing and strip head fires under conditions conducive to low flame heights (Table 1). Ignition was by hand-carried drip torches, generally beginning in the early afternoon. Burns were complete by 1800 h.

Experimental units assigned the thin + burn treatment first were thinned from below (as described above) in 1998 and then were burned (as described above) in 2000. While the same targets for residual structure were applied, actual on-site application of fire required careful management of fire behavior by modifying the width of burn strips to treat accumulations of cured tree branches and tops.

If the characteristics of all units were identical before treatments were randomly assigned, the no-action treatment would have represented a true control. Differences in site and disturbance history, however, resulted in differences in stand structure that were recognized before treatments were applied. The term ‘‘control’’ is used here for ease of interpretation.

2.2.2. Plot establishment and sampling

Experimental units (treatment units) were whole, discrete stands or portions of larger stands, all having irregular boundaries. Before treatments were applied, a systematic grid of sample points was established along compass lines within each unit, with sample points 50 m apart and ≥50 m from stand boundaries (Table 2). A total of 380 sample points was established. This intensity of grid points represented 1 sample per 0.5 ha. Grid points were permanently marked and geo-referenced by GPS. At each point, aspect was estimated to the nearest 1° azimuth by using a compass, slope was estimated to the nearest 1% inclination by using a clinometer, and elevation was estimated to the nearest 15 m from USGS 7.5’ series contour maps. We used the method of McCune and Keon (2002) to calculate a heat load index for each grid point, using a folded aspect about the NE–SW line, slope, and latitude, with 0.0 being the coolest slope (northeast) and 1.0 being the warmest slope (southwest).

Table 2
Environmental setting and site characteristics of experimental units across four restoration treatments in northeastern Oregon

Unit	Treatments	Size (ha)	Elevation (m)	Aspect (°)	Slope (%)	Heat load index[a]	Number of plots	Soil series	Plant association[b]
6A	Thin	14.4	1361	283	11	0.9078	26	Fivebit	Pseudotsuga menziesii/Calamagrostis rubescens
7	Thin	11.9	1305	292	21	0.8952	25	Fivebit	Pseudotsuga menziesii/Symphoricarpos albus
9	Thin	9.1	1235	51	12	0.8258	23	Melhorn	Pinus ponderosa/Agropyron spicatum
22	Thin	8.8	1380	286	8	0.9049	28	Larabee	Pseudotsuga menziesii/Spirea betulifolia
8B	Burn	10.1	1170	82	8	0.8645	23	Olot	P. ponderosa/Festuca idahoensis
10B	Burn	8.4	1920	50	8	0.8733	21	Bocker	P. ponderosa/F. idahoensis
21	Burn	10.4	1373	230	18	0.9503	30	Fivebit	Pseudotsuga menziesii/C. rubescens
24	Burn	8.4	1260	84	9	0.8579	23	Bocker	P. ponderosa/F. idahoensis
6B	Thin + burn	12.1	1388	221	15	0.9219	29	Fivebit	Pseudotsuga menziesii/C. rubescens
8A	Thin + burn	10.4	1174	297	9	0.8955	23	Fivebit	P. ponderosa/F. idahoensis
10A	Thin + burn	15.9	1186	297	13	0.8903	24	Bocker	P. ponderosa/F. idahoensis
1112	Thin + burn	11.0	1185	274	9	0.9136	27	Bocker	P. ponderosa/F. idahoensis
245	Control	13.8	1286	218	11	0.8928	21	Fivebit	P. ponderosa/S. albus
15	Control	15.1	1113	108	18	0.8257	10	Melhorn	Pseudotsuga menziesii/C. rubescens
18	Control	7.5	1333	296	26	0.8822	19	Melhorn	Pseudotsuga menziesii/C. rubescens
23	Control	12.0	1379	345	10	0.8714	28	Olot	Pseudotsuga menziesii/C. rubescens

[a] Heat load index (McCune and Keon, 2002) based on aspect, slope, and latitude, with 1.0 being the warmest slope (southwest).
[b] A relatively stable plant community of definite composition growing in a uniform habitat; see Johnson and Simon (1987).

Pre-treatment measurements of vegetation were taken in 1998 with circular 0.02-ha plots centered at each grid point. Overstory canopy cover was estimated within each plot by taking measurements at five regularly spaced points (at plot center and 2.0 m from the plot center at the four cardinal directions) with a GRS densitometer (Ganey and Block, 1994). On each plot, all standing trees $\geq$1.37 m in height were inventoried by species, noted as live or dead, and their height measured to the nearest 0.1 m using either a telescoping height pole or clinometer. Diameter was measured at 1.37 m above the ground with a diameter tape. Crown ratio, defined as the percent of bole with live foliage, was ocularly estimated to the nearest 5% for all live trees. In order to characterize treatment unit vegetation and classify plant communities into plant associations (Johnson and Simon, 1987), cover of dominant vascular plant species on each plot was ocularly estimated to the nearest 1% for values up to 10% and to the nearest 5% for all values >10%. We assigned a cover value of 0.3% to those species present in trace amounts.

Post-treatment vegetation measurements reported here were taken in 2004, the sixth growing season after the thin treatment and the fourth growing season after the burn treatment. Understory vegetation responses immediately post-treatment (2001) were previously reported in Metlen et al. (2004). Circular 0.04-ha plots were centered on each grid point; this larger plot conformed to Fire and Fire Surrogate guidelines for sampling intensity and resulted in sampling of 8.5% of the total area contained in treatment units. Overstory strata measurements and tree measurements were made as before. Seedlings ($\geq$15 cm in height) were counted within the plot by species. In 2004, seedlings within several meters of each plot in control and thin units 2004 were selected across the range of seedling heights, measured for height, and then excavated for later processing in the lab to determine the year of establishment. Cover of all vascular plants was estimated as before, with the objective of assessing the full species richness of each plot. Nomenclature follows the Plants Database (USDA NRCS, 2004). Voucher specimens for all species were collected and filed at the Eastern Oregon University Herbarium (EOSC).

Coarse woody debris sampling techniques followed the strip-plot method (Bate et al., 2004). Log density, large-end diameter, mean length, total or cumulative length, volume, and cover of logs were determined with sampling in 2004. The middle of a 4 m $\times$ 40 m strip plot was centered over each of the grid points, with the strip oriented at a bearing determined randomly. Only logs that had a large end diameter $\geq$15 cm and a length $\geq$1 m were considered. Length and large and small diameters of logs wholly or partially within the strip plot were measured. Log length was constrained by either the strip plot boundary or a minimum small end diameter of 8 cm. We recorded whether or not the center of the log at the large end fell within the strip plot. Finally, the decomposition class (Maser et al., 1979) of each log was recorded.

2.3. Statistical methods

2.3.1. Family-wise error rate

Forest ecologists and silviculturists commonly measure a suite of response variables when assessing changes in stand structure and development. There is as yet no single and concise parameter that captures all the complexity of structure and composition from the forest floor to the canopy (McElhinny et al., 2005). In this study, we described structure and composition by many attributes, and conducted separate tests for treatment differences with each structural attribute. The number of separate tests easily met the criterion for a multiplicity problem with a potential for an increase in family-wise error rate (Westfall et al., 1999). If left uncontrolled, the result would be a high probability of making at least one Type I error.

Three provisions were used to control the family-wise Type I error rate. We first restricted tests to one of three families of inferences. Each family of inferences was defined by the collection of hypotheses addressing questions of natural interest and considered relevant in decision-making (Westfall et al., 1999; Quinn and Keough, 2002). The three families were (1) all attributes for standing trees, (2) all attributes for coarse woody debris, and (3) all attributes for understory vegetation. Partitioning the structure and composition of forest stands into tree structure, coarse woody debris, and understory vegetation provided a means of better understanding the dynamics of manipulated stands.

At the same time, there is a need to fully consider treatment effects across the full suite of response variables represented by the three families of inferences. The family-wise error rate was controlled within these three families. Fine structure differences among treatments were examined by using *a priori* single degree-of-freedom tests within the context of the analysis of variance (ANOVA; Statistix version 8, Analytical Software, 2003). The five contrasts were specified as (1) the difference between the control and the three active treatments; (2) the difference between the mean of the thin and burn treatments and the single thin + burn treatment (thus assessing the interaction effect of the combined treatments); (3) the difference between the thin treatment and the burn treatment; (4) the difference between the burn treatment and the thin + burn treatment; (5) the difference between the thin treatment and thin + burn treatment. Unadjusted or raw P-values for each contrast were obtained as part of a univariate ANOVA and are reported when the overall F-test was significant. These raw P-values were then adjusted to control family-wise error rate by using the ‘‘step-up’’ procedure of Hochberg (1988) as implemented by Westfall et al. (1999).

Finally, while it is common practice to set $\alpha = 0.05$ for individual comparisons, this value may be too conservative when many tests are performed, increasing the probability of Type II errors (Westfall et al., 1999). As an alternative, we set the significance level for controlling the family-wise Type I error rate to $\alpha = 0.10$. Assumptions of normality and equal variances were tested with the Shapiro–Wilk normality test and normal probability plots (Quinn and Keough, 2002).

2.3.2. Tree structure summarization

Overstory canopy cover, basal area, stem density, diameter class distribution, quadratic mean diameter, stand density index, mean tree height, crown ratio, and crown base height were computed to describe the population structure of standing trees in each treatment unit. Basal area was computed separately by live and dead trees for each species as a summation of the cross-sectional areas of all trees $\geq$1.37 m in height. Absolute density of stems was computed as the sum of all trees $\geq$1.37 m in height and of all seedlings <1.37 m in height. Seedlings were not considered in subsequent summaries of stand density index and mean tree height. Trees were grouped into five diameter size classes to give size-frequency distributions for each unit: (1) saplings were from 0.1 to 9.9 cm dbh, (2) small trees were from 10.0 to 24.9 cm dbh, (3) medium trees were from 25.0 to 39.9 cm dbh, (4) large trees were from 40.0 to 54.9 cm dbh, and (5) very large trees were $\geq$55 cm dbh. Stand density index (SDI) is a relative density measure based on the relationship between mean tree size and number of trees per unit area in a stand (Reineke, 1933), and has proved useful for quantifying relative density across a wide variety of stand conditions because it is independent of site quality and stand age (Long and Daniel, 1990). We used the individual tree summation approach rather than the more easily applied but biased approach based on uniform diameter classes (Woodall et al., 2002). Because the diameter distribution was unknown or was not normal, SDI was calculated as a summation of individual live tree values:

$$\mathrm{SDI} = \sum \left(\frac{1}{25}\mathrm{dbh}_i\right)^a$$

where SDI is stand density index, dbh is diameter in cm at breast height (1.37 m) of the ith tree in the plot, and the exponent a is a species-specific value (Shaw, 2000). Values of the exponent a were 1.77 for ponderosa pine and 1.51 for Douglas-fir and all other species (Cochran et al., 1994). SDI is presented here as a percentage of SDI at full stocking (902 for ponderosa pine and 939 for Douglas-fir) (Cochran et al., 1994). Because these SDI values were percentages, they were altered before analysis with the arcsine square root transformation to meet distribution assumptions. Quadratic mean diameter (QMD) is a measure of average tree diameter (Curtis and Marshall, 2000) and included both standing live and dead trees. Crown base height was computed as the mean of the lower crown heights of live trees in the treatment unit based on the crown ratio and total tree height. Treatment differences among these measures of overstory structure were evaluated by subtracting pre-treatment from post-treatment values and conducting statistical tests on the resulting change variables. The arcsine square root transformation was used to normalize overstory cover values but untransformed values are presented in tabular format to aid in interpretation.

Seedlings selected for aging that were excavated in the field were sectioned at the root collar, the cross-sectional surfaces polished, and annual rings counted under a binocular microscope. Counts were made independently by two people and differences reconciled. Seedling densities for each treatment unit were estimated only for 2004, and ANOVA was used to test for differences among treatments by using these estimated densities. These data were first adjusted with a square root transformation to resolve variance homogeneity. The relationship between seedling age and seedling height was analyzed by species by using an unweighted least squares linear regression (Statistix version 8, Analytical Software, 2003). Because these data came from only thin and control units, we made no attempt to further stratify the seedling structure by treatment. Data for Douglas-fir, grand fir, and the few lodgepole pine were combined.

2.3.3. Coarse woody debris summarization

Log density (number ha^{-1}), large-end diameter, mean log length, cumulative length (m ha^{-1}), log volume (m^3 ha^{-1}), and log cover (m^2 ha^{-1}) were computed to describe the coarse woody debris structure in each treatment unit. Log density was estimated by summing the number of logs having the center of their large ends present within the strip plot, and this value converted to logs per ha. Volume of each log was calculated using Smalian's formula based on the cross-sectional areas of the large and small ends and the length of the log. Log cover was calculated by treating each log as a trapezoid, and determining the percent area of each strip plot covered by the combined areas of trapezoids. We assumed no overlap of pieces. Log density values were transformed by computing the natural log, and a cubed root transformation was used to stabilize variances for log volume.

2.3.4. Understory vegetation summarization

Species richness, species diversity, and species evenness were computed to describe the understory vegetation diversity in each treatment unit. Conifers, including seedlings and saplings, were excluded for this analysis. Species richness (S) was the number of species in a sample plot. Simpson's diversity index may be the most effective index to explain differences among disturbance treatments (Onaindia et al., 2004) and was calculated as the reciprocal of D:

$$D = \sum p_i^2$$

where p is the proportion of the ith species relative to the total number of species. The Shannon–Wiener diversity index (H') (Whittaker, 1972) was used as a measure of heterogeneity in the data, calculated as

$$H' = \sum p_i^2 \log p_i$$

Finally, Pielou's index of evenness J' (Pielou, 1975) was calculated as

$$J' = \frac{H'}{H_{\max}} \quad \text{and} \quad H'_{\max} = \frac{H'}{\log S}$$

where S is the total number of species. All four measures were calculated at the sample plot (0.04 ha) level and then summarized at the treatment level for 2004 post-treatment data.

Multivariate tests for differences in community composition among and between treatments were conducted with 1998 and 2004 data by using multi-response permutation procedures (MRPP), a non-parametric randomization-based alternative to multivariate ANOVA (PC-ORD version 4.33; McCune and

Grace, 2002). A rank-transformed Sørensen distance matrix was used to test the null hypothesis of no difference in average within-group ranked distances, based on abundance of species (percent cover) in all sample plots grouped by treatment. MRPP yields a chance-corrected within-group agreement (effect size, *A*), a measure of the homogeneity within treatments. Values can range from −1 to 1, with values near 0.0 denoting homogeneity as expected by chance and values closer to 1 denoting greater compositional similarities within treatments than among treatments. MRPP also was used to test all pairwise comparisons of treatments.

Species abundance values in 1998 and 2004 were used to test species fidelity to *a priori* groups (treatments) with IndVal, a method for indicator species analysis (Dufrêne and Legendre, 1997). IndVal provides in indicator value for each species based on the product of the relative abundance of a given species in any one group (a measure of exclusiveness), and the relative frequency of the same species in the same group. Frequency (the proportion of times a species occurs in an experimental unit) and average cover (calculated for those sample plots in which it occurs) was obtained for each species in each treatment. Monte Carlo randomization tests with 1000 iterations were used to determine significance of indicator values. All indicator values were accepted if the likelihood of as high an indicator value was ≤ 0.05 and the indicator value was ≥ 10.0.

Non-metric multidimensional scaling ordination (NMS in PC-ORD version 4.33; McCune and Grace, 2002) was used to examine patterns in community composition within and among treatments. This ordination method is well suited to non-normal ecological data because it avoids the assumptions of linear relationships among variables, is robust with respect to large numbers of zero values, and has been used to accurately represent underlying structure in simulated data (Clarke, 1993). We used the global form of NMS with relative Sørensen distance as the measure of compositional dissimilarity between sample plots in 2004. The "slow and thorough" autopilot mode in NMS (random starting coordinates, step length = 0.2, stability criterion = 0.00001, and 400 maximum iterations) was used to generate solutions, from which the lowest stress solution was selected for interpretation. A Monte Carlo test with 400 randomizations was used to determine how likely the observed stress value of the final solution would be by chance alone. The data set included the same array of understory species as used with MRPP.

Because 1998 (pre-treatment) vegetation data were collected at a different level of intensity, they could not directly be used in the same ordination with data from 2004. Instead, a predictive algorithm (NMS Scores procedure, PC-ORD version 4.33; McCune and Grace, 2002) was used to calculate axis scores for the pre-treatment sample plots based on prior ordination with post-treatment data, without altering the positions of the original points. Because of the complexity of the data matrices, each of the three axes was fit one at a time.

Differences between 1998 and 2004 axes scores were used with ANOVA to test the null hypothesis of no difference among treatments; the results of the ANOVA were used to graphically represent treatment means by ordination with standard errors calculated for each axis. Axis scores were correlated to environmental parameters (heat load index, soil depth in cm, percentage soil rockiness, soil ash purity index, and percentage soil clay) using Pearson correlations. The soil parameters came from soil series mapping at each sample plot.[1]

2.3.5. Data quality

Because this study was established as a long-term experiment, with monitoring and evaluation of treatment effects repeated at future intervals and the same treatments reapplied in perhaps 10 years, data quality and documentation of meta-data were emphasized from study inception. Once all field data were converted to digital format and error checking was completed, 5% of the data were independently verified. In addition, 5% of the sample plots were resampled to obtain an estimate of measurement error. Data and the respective meta-data for all measurement variables and the derived response variables were converted to a single database which allowed additional error checking. Finally the data were submitted to an independent database populated with common data from all 13 Fire and Fire Surrogate sites. At this stage, three independent forms of error checking were performed, including conformance to experimental variable definitions, consistency in known or expected relationship between multiple measurement variables, and conformance within pre-established measurement variable limits.

3. Results

3.1. Tree structure

Key structural attributes of trees that tended to differ among treatments included live basal area, live density, QMD, SDI, and seedling density (Table 3). Pre-treatment (1998) mean overstory canopy cover of live trees was 56 ± 3%. By 2004, overstory canopy cover was 51 ± 4% cover. There were no significant differences among treatments in overstory canopy cover between 1998 and 2004 (Table 4).

Mean basal area of live trees in 1998 was 23.5 ± 1.6 $m^2\ ha^{-1}$. Basal area of live ponderosa pine was 15.6 ± 1.2 $m^2\ ha^{-1}$ and represented 66% of the total basal area across all treatments (Fig. 2). In 2004, live tree basal area was 20.8 ± 1.5 $m^2\ ha^{-1}$ across all treatments. The proportion of ponderosa pine remained unchanged. Live basal area was reduced in actively treated units and increased in control units, and thinned units tended to have lower live basal area compared to burned units, yet when *P*-values were adjusted there were no significant differences among treatments (Table 4). Mean basal area of dead trees (snags) in 1998 was 2.8 ± 0.3 $m^2\ ha^{-1}$. Ponderosa pine composed 88% of the total basal area of snags; other species represented were Douglas-fir, western larch (*Larix occidentalis*), and grand fir. Basal area of snags was 2.6 ± 0.5 $m^2\ ha^{-1}$ in 2004. The proportion of basal area that

[1] Data on file, LaGrande Forestry Sciences Laboratory, LaGrande, OR.

Table 3
Summary statistics (mean ± S.E.) for tree and seedling (stem height <1.37 m) structural attributes before and after four restoration treatments in northeastern Oregon

Treatment	Live basal area ($m^2\ ha^{-1}$)	Live density (stems ha^{-1})	QMD[a] (cm)	SDI[b] (% full stocking)	Seedling density (stems ha^{-1})
1998					
Thin	27.3 ± 2.4	698 ± 109	18.9 ± 1.2	48 ± 8	n.a.[c]
Burn	19.2 ± 2.3	286 ± 32	26.1 ± 1.1	32 ± 3	n.a.
Thin + burn	20.1 ± 1.1	398 ± 100	23.3 ± 2.0	35 ± 4	n.a.
Control	27.4 ± 4.7	469 ± 33	21.4 ± 2.0	42 ± 6	n.a.
2004					
Thin	19.6 ± 0.2	452 ± 72	21.5 ± 1.4	36 ± 3	966 ± 252
Burn	19.5 ± 2.4	247 ± 24	27.4 ± 1.1	35 ± 4	27 ± 12
Thin + burn	16.0 ± 1.0	144 ± 15	27.4 ± 1.6	24 ± 2	16 ± 6
Control	28.4 ± 3.0	597 ± 17	20.9 ± 1.5	48 ± 4	1595 ± 612

[a] Quadratic mean diameter.
[b] Stand density index.
[c] Data not available.

Table 4
Results of an ANOVA of the change in tree and seedling structure after four restoration treatments in northeastern Oregon

Response variable	Mean square error	*F*	Contrast	*P*	Adjusted *P*[a]
Overstory canopy cover					
Treatment	0.02	1.81		0.198	
Live basal area					
Treatment	67.20	5.66		0.012	
Control vs. active			−14.40	0.033	0.982
Thin and burn vs. thin + burn			−0.83	0.848	0.982
Thin vs. burn			−8.13	0.006	0.330
Burn vs. thin + burn			−4.48	0.091	0.982
Thin vs. thin + burn			3.65	0.160	0.982
Dead basal area					
Treatment	9.59	1.75	2.93	0.210	
Live tree density					
Treatment	134195.00	9.23		0.002	
Control vs. active			−922.05	0.001	0.051
Thin and burn vs. thin + burn			−222.83	0.157	0.982
Thin vs. burn			−207.47	0.032	0.982
Burn vs. thin + burn			−215.15	0.027	0.982
Thin vs. thin + burn			−7.68	0.930	0.982
Dead tree density					
Treatment	3844.23	3.94		0.036	
Control vs. active			89.88	0.123	0.982
Thin and burn vs. thin + burn			58.33	0.153	0.982
Thin vs. burn			−57.33	0.024	0.982
Burn vs. thin + burn			0.50	0.982	0.982
Thin vs. thin + burn			57.83	0.023	0.982
Live seedling density					
Treatment	1200.36	15.0		0.001	
Control vs. active			65.41	0.007	0.362
Thin and burn vs. thin + burn			27.45	0.028	0.982
Thin vs. burn			−25.69	0.002	0.099
Burn vs. thin + burn			0.88	0.892	0.982
Thin vs. thin + burn			26.57	0.001	0.076
Dead seedling density					
Treatment	3.22	1.47		0.272	
QMD					
Treatment	15.11	11.4		0.001	
Control vs. active			9.50	0.001	0.033
Thin and burn vs. thin + burn			4.18	0.012	0.585
Thin vs. burn			1.34	0.124	0.982
Burn vs. thin + burn			2.76	0.005	0.302
Thin vs. thin + burn			1.42	0.107	0.982

Table 4 (*Continued*)

Response variable	Mean square error	F	Contrast	P	Adjusted P[a]
SDI					
Treatment	145.93	7.46		0.004	
Control vs. active			−24.65	0.007	0.392
Thin and burn vs. thin + burn			−7.20	0.208	0.982
Thin vs. burn			−10.02	0.008	0.395
Burn vs. thin + burn			−8.61	0.017	0.857
Thin vs. thin + burn			1.41	0.661	0.982
Live crown ratio					
Treatment	24.55	2.42		0.117	
Live pine crown height					
Treatment	20.61	7.61		0.004	
Control vs. active			9.77	0.002	0.110
Thin and burn vs. thin + burn			1.79	0.096	0.982
Thin vs. burn			−1.95	0.077	0.982
Burn vs. thin + burn			−0.08	0.563	0.982
Thin vs. thin + burn			1.87	0.026	0.982
Live Douglas-fir crown height					
Treatment	100.18	4.35		0.027	
Control vs. active			9.31	0.273	0.982
Thin and burn vs. thin + burn			22.13	0.002	0.138
Thin vs. burn			−2.44	0.473	0.982
Burn vs. thin + burn			9.84	0.012	0.585
Thin vs. thin + burn			12.29	0.003	0.171
Crown base height					
Treatment	18.05	6.63		0.007	
Control vs. active			5.54	0.076	0.982
Thin and burn vs. thin + burn			4.63	0.041	0.982
Thin vs. burn			−3.85	0.006	0.352
Burn vs. thin + burn			0.39	0.743	0.982
Thin vs. thin + burn			4.29	0.003	0.197

[a] Adjusted to control family-wise error rate by using the "step-up" procedure of Hochberg (1988).

was ponderosa pine snags dropped to 58% in 2004. There was no significant difference in the change in basal area of snags among treatments.

Mean density of live trees in 1998 was 462 ± 52 stems ha^{-1}. Nearly 75% of the live density was composed of ponderosa pine. Douglas-fir was more prevalent in units assigned control and thin treatments (Fig. 3). Mean density of live trees in 2004 was 359 ± 49 stems ha^{-1}. The proportion of live density that was ponderosa pine declined to 69%. Density increased in control units and decreased in actively treated units (Table 4). Live stem density tended to be reduced less in burn units compared to thin + burn units. Mean density of snags was 46 ± 13 stems ha^{-1} in 1998. Nearly 63% of the total snag density was composed of ponderosa pine (Fig. 4). Douglas-fir

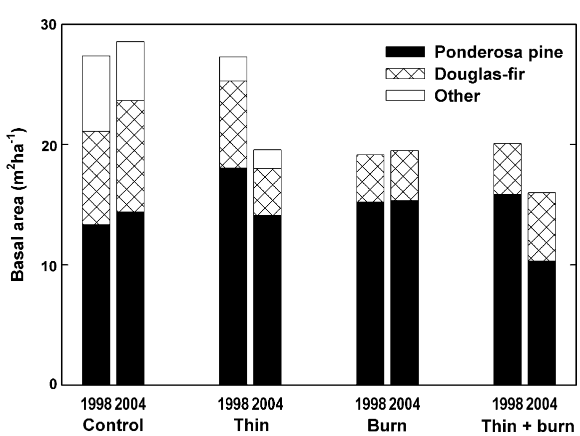

Fig. 2. Pattern of basal area distribution of live trees by species before and after four restoration treatments in northeastern Oregon. Minor amounts of grand fir, lodgepole pine, and western larch are combined as "Other".

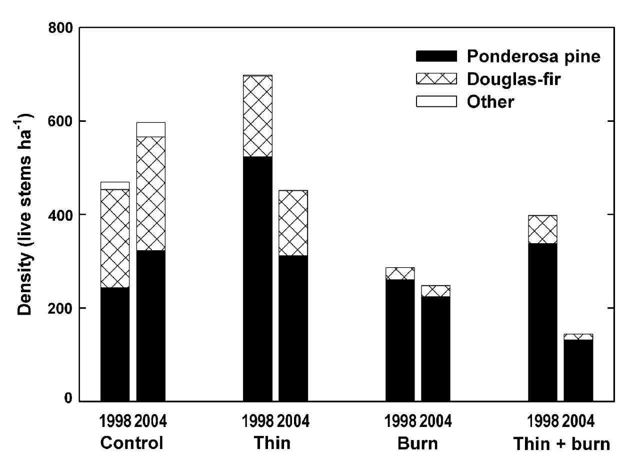

Fig. 3. Pattern of density distribution of live trees by species before and after four restoration treatments in northeastern Oregon. Minor amounts of grand fir, lodgepole pine, and western larch are combined as "Other".

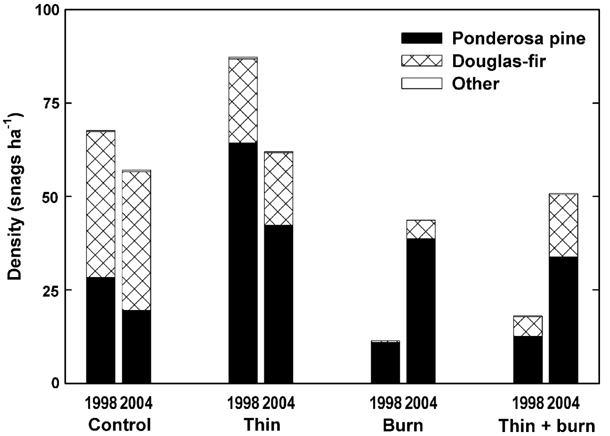

Fig. 4. Pattern of density distribution of dead trees (snags) by species before and after four restoration treatments in northeastern Oregon. Minor amounts of grand fir, lodgepole pine, and western larch are combined as "Other".

snags were more prevalent in units assigned the control and thin treatments. Other species were poorly represented. Mean snag density was 53 ± 11 snags ha^{-1} in 2004. Density of ponderosa pine snags remained unchanged at 63%. Snag density tended to be reduced in thin units and increased in burn and thin + burn units.

Changes in the diameter distribution were determined by QMD, SDI, and graphical representations of frequency by diameter classes. QMD of combined standing live and dead trees before treatments were applied was 22.4 ± 1.0 cm. QMD was 24.3 ± 1.01 cm across all treatment units in 2004. QMD increased in actively treated units and remained constant in control units (Table 4). SDI in 1998 averaged $38 \pm 3\%$ of full stocking. After treatment, the mean SDI was reduced to $36 \pm 3\%$ of full stocking. SDI tended to decline in actively treated units and remained essentially unchanged in control units. Frequency distribution of live and dead trees by diameter classes revealed the differential effect of treatment by tree sizes. Before treatments were applied, the majority of ponderosa pine in thin and thin + burn units was <24.9 cm dbh, resulting in a truncated negative exponential or reverse J-shaped diameter distribution (Fig. 5). Ponderosa pine in control and burn units were present with fewer stems <24.9 cm dbh, presumably because of past management activities. By 2004, density of small-diameter ponderosa pine in control units increased because a large number of seedlings grew into the sapling size

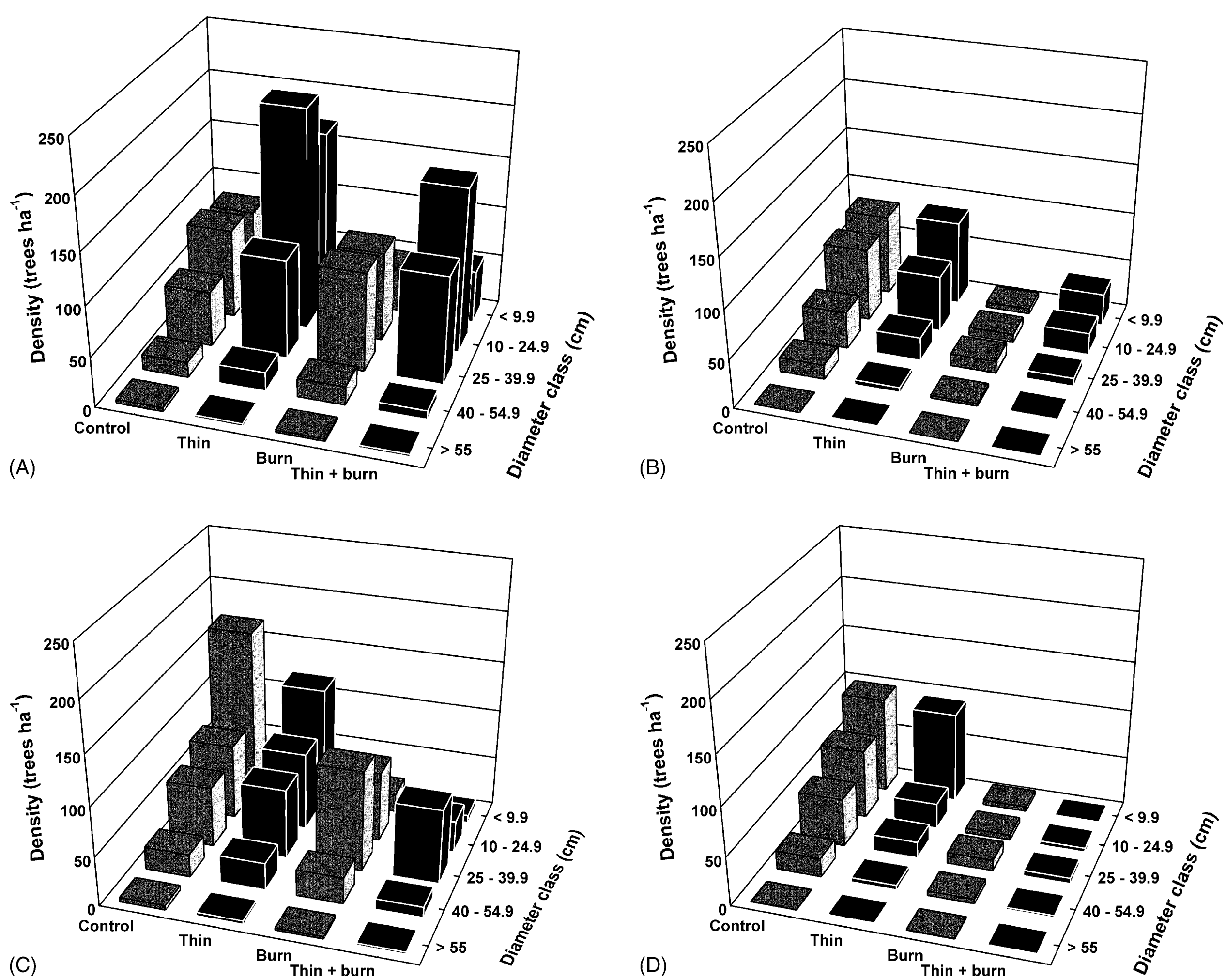

Fig. 5. Pattern of density distribution of live and dead trees by size class after four restoration treatments in northeastern Oregon for (A) ponderosa pine in 1998, (B) Douglas-fir in 1998, (C) ponderosa pine in 2004, and (D) Douglas-fir in 2004.

class. After thinning, density of ponderosa pine stems <24.9 cm was reduced and a reverse J-shaped diameter distribution was established. Little change to the diameter distribution occurred in burn units; the somewhat modal distribution was maintained. In thin + burn units, the number of small-diameter ponderosa pine was reduced, resulting in a modal distribution centered on 25–39.9 cm dbh. The diameter distribution of Douglas-fir in control, thin, and thin + burn units in 1998 followed a negative exponential or reverse J-shaped diameter distribution, characteristic of species encroachment. Douglas-fir in burn units were present in low numbers, but tended to be more modally distributed. When measured again in 2004, the diameter distribution of Douglas-fir in control and thin units followed a reverse J-shaped distribution. The density of small diameter Douglas-fir changed little in thin units; the majority of these stems were probably smaller than thinning specifications. Density of small diameter Douglas-fir was reduced in burn and thin + burn units; these stems were fire sensitive and were readily consumed in the surface fires.

Crown ratio was summarized across all species within a treatment unit. Mean crown ratio was 41 ± 1% in 1998 and 44 ± 1% in 2004. There was no significant difference among treatments. Mean live ponderosa pine tree height was 13.2 ± 0.8 m in 1998 and 14.0 ± 0.8 m in 2004 (Fig. 6). Mean tree height tended to increase in actively treated units and decrease in control units. In addition, height tended to increase more in thin + burn units compared to thin units. Mean live Douglas-fir tree height was 10.1 ± 1.1 m in 1998 and 11.5 ± 1.4 m in 2004. Douglas-fir tree height tended to increase more in thin + burn units than other treatment units. Mean crown base height (lower crown height) was 5.8 ± 0.5 m in 1998 and 5.2 ± 0.8 m in 2004. Trees remaining in thin units tended to have lower crown base height than those in burn and thin + burn units (Fig. 6).

Mean density of live seedlings in 2004 was 649 ± 226 seedlings ha^{-1}. The proportion that was ponderosa pine seedlings was 75%. Live seedling density tended to be lower in actively treated units compared to control units. Seedling density was lower in both burn and thin + burn units compared to thin units (Table 4), and seedling density tended to be lower in thin + burn units compared to burn units. Mean density of dead seedlings in 2004 was 12 ± 3 seedlings ha^{-1} across all treatments. The highest proportion of these was in control units. The proportion of dead seedlings that was ponderosa pine was nearly 80%.

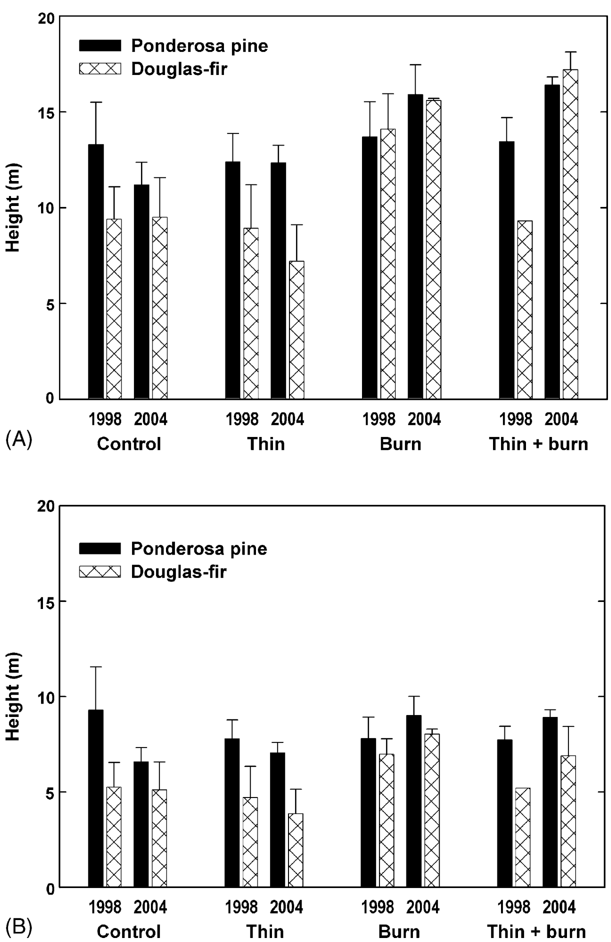

Fig. 6. Pattern of live tree canopy distribution by species before and after four restoration treatments in northeastern Oregon, with (A) mean tree heights with standard error bars, and (B) crown base height with standard error bars.

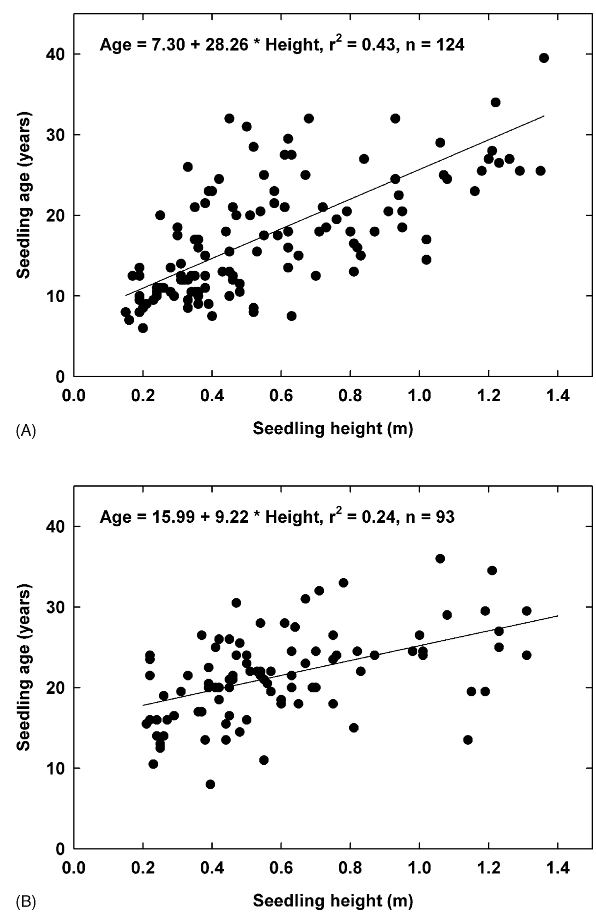

Fig. 7. Pattern of seedling age plotted against height and resulting linear regression for (A) ponderosa pine and (B) Douglas-fir seedlings in both control and thin restoration treatments in northeastern Oregon.

Total age was determined on 217 live seedlings in control and thin units. Of this total, 124 were ponderosa pine, 81 were Douglas-fir, and the remainder was grand fir and lodgepole pine. Ponderosa pine seedlings ranged in height from 15 to 136 cm and averaged 56 ± 3 cm. Douglas-fir seedlings ranged in height from 21 to 131 cm and averaged 58 ± 3 cm. Grand fir and lodgepole seedlings were slightly taller. Linear regression of seedling height and root collar age yielded a y-intercept of ≥7 years for ponderosa pine and 16 years of Douglas-fir, indicating that seedlings across all heights existed as advance regeneration and became established before treatments were implemented (Fig. 7). Seedling ages were normally distributed between 6 and 41 years with no distinct cohort apparent.

3.2. Coarse woody debris

Key structural attributes of coarse woody debris that differed among treatments included log density, cumulative length, volume, and cover (Table 5). One control unit was determined to be an outlier with respect to all log descriptors because it included part of an adjacent lodgepole pine blowdown, and it was excluded from all further analyses of log parameters. Mean log density was 105 ± 20 logs ha^{-1}. Log density in actively treated units was lower than in control units and log density was lower in burn units compared to thin units (Table 6). Log density in thin units tended to be higher than in thin + burn units. Large-end diameter of logs was surprisingly low and averaged 23.7 ± 0.8 cm across all treatment units; there was no difference among treatment units. There was no difference in mean log length (length of individual pieces); the overall mean log length was 3.7 ± 0.2 m. The mean cumulative log length was 116.8 ± 21.4 m. Cumulative log length tended to be lower in actively treated units compared to the control units, and

Table 5
Summary statistics (mean ± S.E.) for coarse woody debris (logs) structural attributes 4 years after four restoration treatments in northeastern Oregon

Treatment	Density (stems ha^{-1})	Cumulative length (m ha^{-1})	Volume (m^3 ha^{-1})	Cover (m^2 ha^{-1})
Thin	156 ± 37	156.2 ± 30.0	6.5 ± 1.0	2.2 ± 0.2
Burn	30 ± 5	43.9 ± 11.4	2.2 ± 0.5	1.4 ± 0.2
Thin + burn	61 ± 8	80.2 ± 26.1	2.3 ± 0.6	1.4 ± 0.2
Control	195 ± 22	210.0 ± 53.0	9.2 ± 1.8	2.6 ± 0.3

Table 6
Results of ANOVA for coarse woody debris structure after four restoration treatments in northeastern Oregon

Response variable	Mean square error	F	Contrast	P	Adjusted P[a]
Log density					
Treatment	3.22	26.20		0.001	
Control vs. active			3.37	0.001	0.011
Thin and burn vs. thin + burn			0.15	0.729	0.914
Thin vs. burn			−1.60	0.001	0.003
Burn vs. thin + burn			−0.73	0.014	0.304
Thin vs. thin + burn			0.88	0.005	0.120
Large-end diameter					
Treatment	5.99	0.68		0.582	
Log length					
Treatment	0.19	0.52		0.678	
Cumulative length					
Treatment	2.27	7.07		0.005	
Control vs. active			349.71	0.010	0.235
Thin and burn vs. thin + burn			39.60	0.590	0.914
Thin vs. burn			−112.28	0.020	0.396
Burn vs. thin + burn			−36.35	0.397	0.914
Thin vs. thin + burn			75.94	0.093	0.914
Log volume					
Treatment	1.53	4.84		0.020	
Control vs. active			1.78	0.002	0.050
Thin and burn vs. thin + burn			0.60	0.054	0.914
Thin vs. burn			−0.56	0.005	0.120
Burn vs. thin + burn			0.12	0.914	0.914
Thin vs. thin + burn			0.58	0.004	0.108
Log cover					
Treatment	0.28	6.26		0.008	
Control vs. active			2.74	0.007	0.178
Thin and burn vs. thin + burn			0.71	0.207	0.914
Thin vs. burn			−0.84	0.019	0.396
Burn vs. thin + burn			−0.06	0.837	0.914
Thin vs. thin + burn			0.78	0.028	0.526

[a] Adjusted to control family-wise error rate by using the "step-up" procedure of Hochberg (1988).

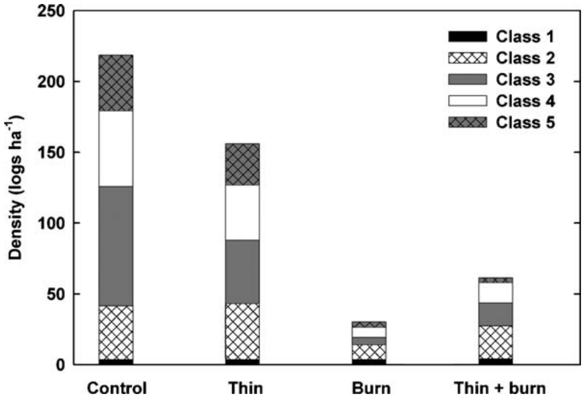

Fig. 8. Number of logs by decomposition class 4 years after four restoration treatments in northeastern Oregon. Log decomposition classes follow Maser et al. (1979).

tended to be lower in burn units compared to thin units. Mean log volume was $9.1 \pm 4.4\ m^3\ ha^{-1}$. Log volume in actively treated units was lower than control units. Log volume tended to be greater in thin units compared to burn or thin + burn units. Mean log cover was $2.1 \pm 0.3\%$. Log cover tended to be greater in control units compared to actively treated units. Log cover also tended to be greater in thin units compared to burn and thin + burn units.

Distribution of logs by decomposition class was highly skewed (Fig. 8) both in absolute density and in the relative proportion by diameter class. The total number of logs present in burn and thin + burn units was <25% of that in control and thin units. Few logs were noted across all units in decomposition class 1, which are newly fallen logs with intact bark, small twigs present, are partially elevated on support points, and the original rounded shape is unchanged. In both control and thin treatments, the proportion of logs in classes 2–5 was relatively uniform. Control and thin units both presumably retained their original component of logs in decomposition class 5, which are logs near the end of their recognizable life span, characterized by the absence of bark and twigs, soft, blocky pieces of wood that have faded to light yellow or gray, the overall log shape is round to oval, and the log rests fully on the ground. About 18% of the logs in control and thin units belonged to decomposition class 5. In contrast, <7% of the logs in burn and thin + burn units were class 5 logs.

3.3. Understory vegetation

Forty-seven species of shrubs, graminoids, and forbs were used to characterize plant communities at each sample plot during the pre-treatment inventory. The list of species included 6 tall shrubs, 11 low shrubs, 18 graminoids, and 12 perennial forbs. These species were present with usually more than a trace of cover. Five of these were considered wide-ranging non-native invasive species, including *Cynoglossum officinale*, *Bromus tectorum*, *Dactylis glomerata*, *Phleum pratense*, and *Poa pratense*.

Understory community composition differed among units in 1998 (Table 7). Three species were associated with thin units (Table 8): *Spiraea betulifolia* and *Symphoricarpos albus* are low shrubs with relatively wide ecological amplitude that are also found associated with Douglas-fir and grand fir on moister sites, and *Arnica cordifolia* is a shade tolerant forb commonly associated with dry forests in the Blue Mountains. These three species were present with limited cover in most units, but occurred with higher frequency and cover in thin units (*S. betulifolia* was in 63% of the sample plots, *S. albus* was in 83% of the sample plots, and *A. cordifolia* was in 77% of the sample plots within the thin treatment units). Six grasses were associated with burn units, including the bunch grasses *Pseudoroegneria spicata* and *Festuca idahoensis*, both commonly found on exposed, non-forest sites. No species were clearly indicative of the thin + burn units before treatment. Five species were indicative of the control units, including the tall shrub *Physocarpus malvaceus*, the rhizomatous graminoid *Carex geyeri*, and three grasses, *F. occidentalis*, *Melica spectabilis*, and *Achnatherum occidentale*.

Table 7
Results of multi-response permutation procedures (MRPP) for differences in understory community composition for four restoration treatments in northeastern Oregon

Comparison	A[a]	P[b,c]
Pre-treatment		
Main effect	0.077	0.001
Thin vs. burn	0.091	0.001
Thin vs. thin + burn	0.045	0.001
Thin vs. control	0.039	0.001
Burn vs. thin + burn	0.012	0.005
Burn vs. control	0.070	0.001
Thin + burn vs. control	0.052	0.001
Post-treatment		
Main effect	0.096	0.001
Thin vs. burn	0.060	0.001
Thin vs. thin + burn	0.068	0.001
Thin vs. control	0.060	0.001
Burn vs. thin + burn	0.031	0.001
Burn vs. control	0.067	0.001
Thin + burn vs. control	0.105	0.001

[a] Chance corrected within-group agreement (effect size).
[b] Probability of randomized groups having as large or larger within-group agreement.
[c] Adjusted to control family-wise error rate by using the ''step-up'' procedure of Hochberg (1988).

The more detailed vegetation survey of understory species within larger plots in 2004 resulted in identifying 191 species. The species list for 2004 included seven tall shrubs such as *Prunus virginiana*, *Salix scouleriana*, and *Holodiscus discolor*, 15 low shrubs such as *Ceanothus velutinus*, *Ericameria nauseosa*, *Linnaea borealis*, *Lonicera utahensis*, and *Ribes cereum*, and 26 graminoids including the rhizomatous *Bromus vulgaris* and the bunch grasses *Elymus elymoides* and *E. glaucus*. A total of 143 forbs were identified, including the non-native invasive annuals *Buglossoides arvensis*, *Logfia arvensis*, and *Sisymbrium altissimum*, non-native invasive perennials *Potentilla recta* and *Rumex acetosella*, the shade tolerant shrub-like forb or sub-shrub *Chimaphila umbellata*, and perennial lilies *Disporum trachycarpum* and *Fritillaria pudica*. Species richness in treatment units was generally low (Table 9).

Table 8
Pre-treatment (1998) indicators, growth form, and frequency and average cover (%) for four restoration treatments in northeastern Oregon

Species	Growth form	INDVAL[a]	P	Frequency (cover %)			
				Thin	Burn	Thin + burn	Control
Spiraea betulifolia	Low shrub	33.0	0.001	**63 (3.9)**	27 (1.7)	44 (2.1)	27 (2.1)
S. albus	Low shrub	45.7	0.001	**83 (6.0)**	61 (1.8)	78 (1.6)	52 (2.3)
Arnica cordifolia	Perennial forb	38.7	0.001	**77 (8.4)**	60 (2.5)	70 (4.2)	58 (2.2)
Bromus carinatus	Graminoid	10.3	0.001	–	**10 (6.5)**	–	–
Elymus glaucus	Graminoid	20.9	0.001	6 (0.3)	**31 (3.3)**	–	21 (2.0)
F. idahoensis	Graminoid	39.9	0.001	36 (5.8)	**80 (15.8)**	74 (12.2)	45 (5.4)
Koeleria macrantha	Graminoid	37.5	0.001	5 (0.3)	**46 (2.2)**	1 (3.0)	21 (0.8)
Pseudoroegneria spicata	Graminoid	15.6	0.014	6 (2.0)	**32 (4.2)**	32 (4.4)	4 (0.9)
Physocarpus malvaceus	Tall shrub	14.7	0.002	3 (0.9)	7 (18.8)	2 (7.7)	**18 (11.5)**
Carex geyeri	Graminoid	25.5	0.003	65 (5.7)	14 (3.3)	30 (5.3)	**60 (4.2)**
F. occidentalis	Graminoid	15.3	0.001	–	1 (0.3)	–	**12 (2.6)**
Melica spectabilis	Graminoid	10.3	0.001	–	–	–	**8 (4.1)**
Achnatherum occidentale	Graminoid	16.9	0.001	19 (2.6)	9 (1.3)	–	**22 (3.2)**

Bold values indicate the treatment for which a species was most strongly associated.
[a] Indicator value = relative abundance × relative frequency (see Dufrêne and Legendre, 1997).

ANOVA of species richness across treatments indicated no difference among treatments ($P = 0.689$); the mean species richness in 2004 was 29 ± 1 species. Simpson's diversity index did not vary among units ($P > 0.811$) and averaged 0.23 ± 0.02, Shannon–Wiener diversity index (H') did not vary among units ($P > 0.831$) and averaged 0.11 ± 0.01, and Pielou's index of evenness (J') was consistent among units ($P > 0.775$) and averaged 0.034 ± 0.002.

Understory community composition differed among fuel reduction treatment units in 2004 (Table 10). Indicator species analysis identified 49 species differentially associated with treatments. Nine species were identified as indicators of thin units. *Spiraea betulifolia* and *S. albus* were the most commonly encountered shrubs in thin units. The only grass indicative of this treatment was the strongly rhizomatous *Calamagrostis rubescens*, although it was frequently encountered in other treatment units as well. Perennial forbs included *A. cordifolia*, *Eurybia conspicua*, *Castilleja* species, and *Hieracium cynoglossoides*. Ten species were indicative of burn units, including *Pseudoroegneria spicata*, the low, open-growing *Danthonia unispicata*, and the bunch grasses *F. idahoensis* and *Koeleria macrantha*. Annual forbs associated with this treatment included *Collomia linearis*, *Galium aparine*, and *Claytonia perfoliata*. Thin + burn units had 15 plant species identified as indicative, including three grasses, seven perennial forbs, and five annual forbs. All three grasses (*Bromus japonicus*, *Bromus tectorum*, and *Ventenata dubia*) are non-native invasives; *B. japonicus* is an annual. A non-native invasive annual indicative of thin + burn units was *Lactuca serriola*. Control units were represented by 15 species. Both the rhizomatous graminoids *Carex geyeri* and *Achnatherum occidentale* apparently had high fidelity in these units, occurring as indicators in 1998 and again in 2004. *Melica spectabilis*, a weakly rhizomatous grass, was also indicative, similar to 1998. Native perennial forbs associated with the control units included *Fragaria vesca*, *Geum macrophyllum*, and *Osmorhiza berteroi*. The only non-native invasive forb was the ordinary *Taraxacum officinale*.

Table 9
Summary statistics (mean ± S.E.) for understory richness and diversity attributes for four restoration treatments in northeastern Oregon

Treatment	Species richness[a]	Simpson's index	Shannon–Wiener H'	Pielou's J'
Thin	26 ± 3	0.25 ± 0.04	0.12 ± 0.01	0.037 ± 0.005
Burn	29 ± 1	0.23 ± 0.05	0.11 ± 0.01	0.033 ± 0.003
Thin + burn	29 ± 2	0.24 ± 0.03	0.11 ± 0.01	0.032 ± 0.003
Control	30 ± 3	0.21 ± 0.01	0.12 ± 0.01	0.034 ± 0.003

[a] Richness based on 400 m^2 plots.

Ordination of plot-level cover data of all species yielded a three-dimensional solution that explained 84% of the variation in the sample-point data (NMS ordination, final stress = 15.26, Fig. 9). Most of the variation (39%) was explained by Axis 1, which tended to separate species along light and moisture availability gradients. Environmental parameters correlated with Axis 1 included percentage soil rock (Pearson's correlation coefficient = 0.449, $P < 0.001$), soil depth (−0.437, $P < 0.001$), soil ash purity (0.283, $P < 0.001$), and heat load index (0.104, $P = 0.040$). Plants with high scores (relatively dry, shade intolerant species) on Axis 1 included *Allium fibrillum*, *A. acuminatum*, *Olsynium douglasii*, *Balsamorhiza incana*, *Lomatium cous*, *Lomatium macrocarpum*, *Eriophyllum lanatum*, the introduced annual *Buglossoides arvensis*, *Navarretia intertexta*, *Eriogonum umbellatum*, *Sanguisorba occidentalis*, *Danthonia unispicata*, and *Ericameria nauseosa*. In contrast, plants with low scores (relatively moist, shade tolerant species) on Axis 1 included *Carex microptra*, *Gentiana calycosa*, *Geranium viscosissimum*, *Spiranthes romanzoffinana*, *C. rubescens*, and the shrubs *Berberis repens*, *Shepherdia canadensis* and *Crataegus douglasii*.

Axis 2 explained an additional 19% of the variation, and tended to distinguish species along a soil texture gradient. Environmental parameters correlated with Axis 2 included percentage soil clay (−0.122, $P = 0.018$) and ash purity index (−0.102, $P = 0.047$). Plants with high scores (species adapted to fine texture soils) on Axis 2 included the relatively deep-rooted

Table 10
Post-treatment (2004) indicators, growth form, and frequency and average cover (%) for four restoration treatments in northeastern Oregon

Species	Growth form	INDVAL[a]	*P*	Frequency (cover %)			
				Thin	Burn	Thin + burn	Control
Spiraea betulifolia	Low shrub	34.7	0.001	**68 (6.8)**	28 (2.6)	43 (5.4)	56 (2.4)
S. albus	Low shrub	38.6	0.001	**89 (10.5)**	87 (3.5)	91 (3.8)	83 (6.9)
Calamagrostis rubescens	Graminoid	33.7	0.001	**91 (29.1)**	81 (23.1)	75 (21.6)	81 (12.4)
A. cordifolia	Perennial forb	29.2	0.018	**86 (9.7)**	74 (10.9)	63 (3.6)	81 (7.3)
Eurybia conspicua	Perennial forb	14.2	0.003	**21 (1.2)**	1 (0.3)	4 (2.0)	12 (0.3)
Castilleja species	Perennial forb	19.5	0.001	**51 (0.4)**	24 (0.4)	48 (0.4)	12 (0.3)
Fragaria virginiana	Perennial forb	26.4	0.029	**70 (2.5)**	63 (1.4)	56 (1.0)	72 (1.9)
Hieracium cynoglossoides	Perennial forb	26.7	0.008	**77 (0.6)**	74 (0.5)	66 (0.5)	55 (0.3)
Sedum stenopetalum	Perennial forb	19.1	0.011	**28 (3.8)**	30 (1.2)	23 (0.3)	31 (0.3)
Pseudoroegneria spicatum	Graminoid	17.6	0.017	25 (2.6)	**47 (4.0)**	39 (5.5)	12 (3.2)
Danthonia unispicata	Graminoid	16.8	0.001	6 (4.7)	**28 (5.5)**	25 (1.3)	5 (8.2)
F. idahoensis	Graminoid	34.1	0.001	73 (5.2)	**93 (9.5)**	81 (9.4)	71 (5.3)
Koeleria macrantha	Graminoid	35.0	0.001	40 (1.7)	**87 (1.5)**	77 (1.2)	41 (0.7)
Achillea millefolium	Perennial forb	32.0	0.031	88 (2.1)	**98 (1.8)**	88 (1.0)	91 (0.9)
Lomatium species	Perennial forb	14.1	0.001	16 (0.3)	**31 (0.4)**	29 (0.3)	8 (0.3)
Phlox viscida	Perennial forb	15.9	0.011	14 (3.7)	**33 (1.9)**	26 (0.5)	14 (0.4)
Claytonia perfoliata	Annual forb	20.2	0.001	10 (0.4)	**33 (0.7)**	8 (0.3)	27 (0.3)
Collomia linearis	Annual forb	24.5	0.001	23 (0.4)	**63 (0.4)**	65 (0.3)	40 (0.3)
Galium aparine	Annual forb	14.1	0.002	7 (0.5)	**31 (0.5)**	2 (0.3)	24 (0.6)
Bromus japonicus[b]	Graminoid	15.1	0.027	11 (0.5)	30 (2.4)	**38 (1.5)**	21 (0.7)
Bromus tectorum[b]	Graminoid	24.6	0.006	28 (3.7)	51 (2.9)	**55 (3.8)**	18 (0.9)
Ventenata dubia[b]	Graminoid	12.8	0.002	1 (0.3)	18 (1.8)	**25 (1.3)**	–
Antennaria stenophylla	Perennial forb	10.8	0.001	–	5 (0.3)	**15 (0.3)**	3 (0.3)
Balsamorhiza sagittata	Perennial forb	24.8	0.003	47 (1.8)	42 (2.2)	**46 (4.7)**	18 (0.7)
Calochortus macrocarpus	Perennial forb	11.1	0.002	12 (0.3)	16 (0.3)	**25 (0.3)**	4 (0.3)
Lithospermum ruderale	Perennial forb	25.3	0.001	44 (0.4)	43 (0.4)	**62 (0.4)**	15 (0.3)
Lupinus caudatus	Perennial forb	33.5	0.001	75 (2.2)	56 (1.0)	**77 (3.1)**	63 (1.5)
Solidago multiradiata	Perennial forb	28.1	0.001	18 (2.2)	35 (1.4)	**64 (1.1)**	6 (0.8)
Zigadenus venenosus	Perennial forb	17.2	0.010	38 (0.3)	43 (0.3)	**51 (0.3)**	21 (0.3)
Clarkia pulchella	Annual forb	23.0	0.001	12 (0.3)	25 (0.4)	**44 (0.3)**	5 (0.3)
Epilobium minutum	Annual forb	33.7	0.001	24 (0.3)	39 (0.4)	**74 (0.4)**	31 (0.6)
Lactuca serriola[b]	Annual forb	12.3	0.001	4 (0.3)	8 (0.3)	**20 (0.3)**	1 (0.3)
Polygonum douglasii	Annual forb	23.7	0.001	24 (0.3)	30 (0.4)	**45 (0.7)**	27 (0.4)
Veronica peregrina	Annual forb	15.7	0.001	2 (0.3)	14 (0.3)	**29 (0.4)**	5 (0.7)
Amelanchier alnifolia	Tall shrub	19.7	0.001	26 (0.4)	21 (0.3)	1 (0.3)	**36 (0.6)**
Rosa species	Low shrub	25.3	0.009	68 (1.2)	54 (1.1)	39 (0.5)	**68 (1.4)**
Achnatherum occidentalis	Graminoid	15.2	0.011	32 (1.0)	13 (1.3)	5 (0.4)	**42 (0.7)**
Carex geyeri	Graminoid	29.9	0.002	66 (11.1)	59 (11.3)	57 (8.3)	**81 (13.6)**
Melica spectabilis	Graminoid	15.4	0.001	–	–	–	**15 (0.8)**
Antennaria neglecta	Perennial forb	11.9	0.001	11 (0.3)	–	5 (0.3)	**14 (1.8)**
Arenaria macrophylla	Perennial forb	25.7	0.001	9 (1.1)	5 (1.1)	4 (0.3)	**36 (1.1)**
Fragaria vesca	Perennial forb	41.0	0.001	34 (3.2)	28 (2.2)	7 (1.3)	**74 (3.0)**
Geum macrophyllum	Perennial forb	16.3	0.001	1 (0.3)	–	1 (0.3)	**17 (0.8)**
Mitella stauropetala	Perennial forb	18.1	0.001	2 (0.3)	2 (0.3)	–	**19 (1.0)**
Osmorhiza berteroi	Perennial forb	26.4	0.001	3 (0.3)	1 (0.3)	3 (0.3)	**32 (0.3)**
Potentilla glandulosa	Perennial forb	19.0	0.048	26 (1.2)	33 (0.5)	27 (0.4)	**55 (0.5)**
Prunella vulgaris	Perennial forb	14.9	0.001	4 (0.3)	–	2 (0.7)	**17 (1.3)**
Taraxacum officinale[b]	Perennial forb	12.4	0.014	21 (0.3)	18 (0.3)	15 (0.3)	**33 (0.3)**
Trifolium longipes	Perennial forb	31.4	0.001	34 (12.3)	55 (11.6)	6 (3.1)	**81 (8.5)**

Bold values indicate the treatment for which a species was most strongly associated.
[a] Indicator value = relative abundance × relative frequency (see Dufrêne and Legendre, 1997).
[b] Non-native invasive species.

perennial shrubs *Physocarpus malvaceus*, *Holodiscus discolor*, *Lonicera utahensis*, and *Berberis repens*, and the perennial forbs *Disporum trachycarpum*, and *Trillium ovatum*. Plants with low scores (species adapted to shallow, coarse texture soils) included a mix of perennials and annuals such as *Bromus tectorum*, *Poa bulbosa*, *Sanguisorba occidentalis*, *Alyssum alyssoides*, *Olsynium douglasii*, *Navarretia intertexta*, and *Pascopyrum smithii*.

Axis 3 explained an additional 28% of the overall variation and also tended to distinguish species along moisture availability gradients. Environmental parameters correlated with Axis 3 included soil depth (0.425, $P < 0.001$), percentage soil rock (−0.346, $P < 0.001$), and ash purity index (−0.146, $P = 0.004$). Plants with high scores (less drought tolerant) included the shrubs *Acer glabrum*, *Linnaea borealis*, *Goodyera*

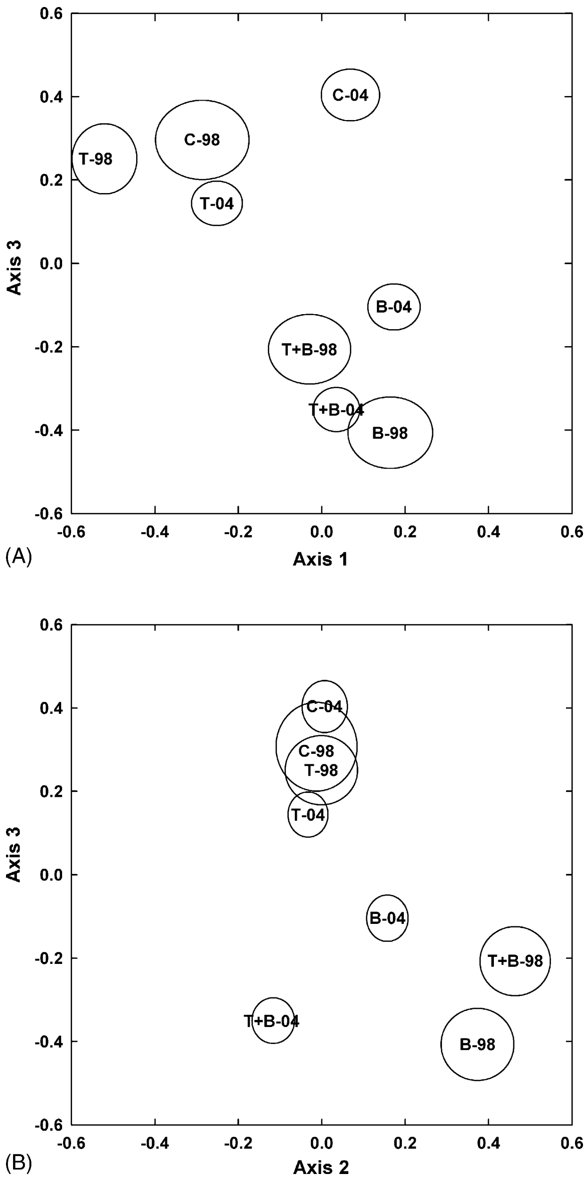

Fig. 9. Non-metric multidimensional scaling (NMS) plot of understory plant communities by restoration treatment in northeastern Oregon. Plotted are mean coordinates with asymmetrical ovals describing the region bounded by standard errors from ANOVA of individual sample plots ($n = 380$) for (A) Axes 1 and 3, and (B) Axes 2 and 3. Regions labeled as follows: T = thin treatment, B = burn treatment, T + B = thin + burn treatment, C = control treatment, 98 = pre-treatment in 1998, 04 = post-treatment in 2004. See text for description of axes.

oblongifolia, *Vaccinium membranaceum*, and *Physocarpus malvaceus*, and the forbs *Disporum trachycarpum*, *Orthilia secunda*, *Anaphalis margaritacea*, *Trillium ovatum*, *Maianthemum stellatum*, *Thalictrum occidentale*, and *Chimaphila umbellata*. Plants with low scores (more drought tolerant) on Axis 3 that may represent scabland communities included the shrub *Ribes cereum*, the forbs *Buglossoides arvensis*, *Blepharipappus scaber*, *Orthocarpus tenuifolius*, *Pyrrocoma carthamoides*, *Grindelia nana*, *Balsamorhiza incana*, *Scutellaria angustifolia*, *Logfia arvensis*, and the grasses *Ventenata dubia*, *Pseudoroegneria spicata*, and *Pascopyrum smithii*.

Two-dimensional representation of understory plant composition by treatment and time in ordination space revealed underlying patterns that differed among treatment units (Fig. 9). Plotted regions for units before and after treatment bounded by asymmetrical confidence intervals for each axis were different in size, reflecting the greater variation in NMS predicted axis scores for pre-treatment units compared to post-treatment NMS axis scores. Treatments were grouped into discrete clusters that indicated similarity among control and thin units and similarity among burn and thin + burn units. Control and thin units were clustered together suggesting abundance in the species arrays of shade tolerant plants occurring on relatively deep soils with few rocks and low ash purity. In contrast, burn and thin + burn units were clustered together, suggesting an abundance in the species arrays of shade intolerant plants occurring on relatively shallow soils with high rock content and a higher heat load index.

Changes in the understory species composition of units between pre- and post-treatment measurements were detected as shifts in the region bounding species space along axes, measured as both direction and magnitude of the vector. Along axis 1, scores of plant communities in burn units remained constant while axis 1 scores of thin units increased ($P = 0.006$). Axis 2 scores for thin + burn units decreased more than burn units ($P = 0.002$) and thin units remained constant ($P < 0.001$). Along axis 3, burn unit scores increased while thin unit scores remained constant ($P < 0.001$), and thin + burn unit scores decreased ($P < 0.001$).

4. Discussion

4.1. Treatment justification

Thinning, burning, and the combination of thinning and burning used in this study each resulted in modified stand structure and composition. Yet the changes noted herein appear minor compared to those reported after stand-replacement wildfires (Crawford et al., 2001; Griffis et al., 2001; Passovoy and Fulé, 2006). Our study was conducted on an operational scale, however, we used unusual care in selecting a suite of potential experimental units that minimized potential site differences by ensuring that environmental parameters such as aspect, slope, and elevation were consistent and units were close to each other. Small scale differences in plant communities were evident across treatment units in response to differences in soil parameters. Consequently, experimental units, especially understory vegetation composition, differed more as a result of pre-existing conditions than as a result of treatment. Failure of our restoration treatments to radically influence the understory vegetation indicates that the majority of understory species have adapted to the same disturbance regimes that our treatments attempted to mimic (Metlen et al., 2004), and that change caused by our treatments was minor compared to the environmental complexity that exists within this portion of northeastern Oregon forests.

Silviculturists throughout the western United States are increasingly being asked to design restoration treatments for low-elevation landscapes that historically supported high frequency, low severity fires. Historical structure in ponderosa pine forests has been described as open and park like, dominated by large, typically uneven-aged trees with little recruitment into the overstory (White, 1985; Arno et al., 1997; Kaufmann et al., 2000; Youngblood et al., 2004; Brown and Cook, 2006). Thinning and burning may reduce stand basal area and density of small trees, remove fire-sensitive trees, reduce accumulations of woody fuels, increase height to live crowns, and result in more fire-resistant forest structure. There is broad agreement that these management practices should move forests more in the direction of historical structures and disturbance regimes (Youngblood et al., 2004).

Thinning is designed to improve growth of the residual trees, to enhance forest health, or to recover potential mortality in immature forests. Thinning may also be used to develop or protect vertical and horizontal forest structure, to develop larger trees, snags, and down wood for terrestrial habitat, and to promote late-successional characteristics (Powell et al., 2001). Broad scale application of mechanical thinning has been suggested as a management strategy for low elevation dry forests in northeastern Oregon that avoids air quality degradation (Mutch et al., 1993). Increasingly, thinning has been suggested for making forests more resistant to uncharacteristically severe fire (Miller and Urban, 2000). In the context of restoration treatments, thinning may be most appropriate when trees are sufficiently large and dense that the structure of the stand is changed when trees are removed, and when treatments with fire would kill too many overstory trees (Brown et al., 2004). We used a low thinning (thinning from below) in which lower canopy trees were removed and larger codominant and dominant trees were retained, aiming to mimic mortality caused by tree-to-tree competition or surface fires under historical conditions (Graham et al., 1999).

Large scale use of prescribed fire has been recommended as a primary means for removing undesirable fuel accumulations, for controlling unwanted dense regeneration that serves as ladder fuels, for increasing forage production, for converting organically bound nutrients to forms more readily availably for plant uptake, and for decreasing the risk of stand-replacement wildfire in low elevation dry forests (Mutch et al., 1993; Harrington, 1996). Burning as a restoration treatment may have the desired attributes of being ‘‘natural’’ within the ecological system and may provide a full range of ecological effects because of varying intensities across burn units (Brown et al., 2004) yet is imprecise within the context of single tree effects. Perhaps the greatest impediment to increased use of burning in low elevation dry forests is legal and social constraints on smoke production. While broad scale fuel treatments likely produce less smoke than that of a large wildfire, the particulate matter associated with treatments is projected to exceed current levels and would persist over longer times (Arno and Ottmar, 1994). Our treatments were purposely applied as late season or fall burns because at this time ground and duff moisture levels are usually lower than during early season or spring burning periods and lower ground and duff moisture levels could enhance the uniform spread of fire. In addition, we assumed that most plant species within our study area are adapted to a fire regime that featured frequent late season or fall wildfires.

Low elevation, dry forests in northeastern Oregon may be particularly suited to applications of thinning coupled with burning for restoring and maintaining forest cover that is resistant to severe insect, disease, and wildfire damage (Mutch et al., 1993; Arno and Ottmar, 1994). In this study, thinning and burning were considered a single treatment, and the effects of initially thinning were not distinguished from the effects of the subsequent burning. Thinning added activity fuel to the forest floor in relatively uniform, parallel strips, with increased fuel moisture relative to fuel outside the strips, greatly affecting the timing of burns.

While thinning and burning are not particularly new silvicultural practices in low elevation forests, better information is needed about both short- and long-term ecological consequences and tradeoffs of treatments. First-year post-treatment understory plant community composition at our study site was described by Metlen et al. (2004). Work reported here extends the initial observations for understory vegetation to community dynamics and links the tree, log, and associated herbaceous strata for a more holistic assessment of structural changes. Little agreement exists about the efficacy across broad regions or among multiple fire-dependent ecosystems in thinning and burning treatments for sustaining healthy forest ecosystems and regulating fuels (Stephens, 1998; van Wagtendonk, 1996; Weatherspoon, 2000). There is as yet no comprehensive comparison of effects across the many fire-dependent forest ecosystems to guide decision-makers. Innovative operational-scale experiments such as those of the Fire and Fire Surrogate study are essential for providing this comparison.

4.2. Thinning effects

In our study, thinning reduced the density of live overstory trees, yet had no effect on overstory canopy cover or basal area. We made no direct measure of crown bulk density, yet we assume the low thinning had minimal effect on reducing crown bulk density because the trees removed were primarily between 10 and 25 cm in diameter and overstory canopy cover was not reduced. Thinning moved the treated units closer to the target basal area of 16 m^2 ha^{-1}. Density (and thus basal area) of snags remained unchanged with thinning because the treatment was applied on an individual tree basis, and these stand features were retained. Thinning increased the QMD because small diameter trees were removed. While thinning tended to reduce SDI, the effect was not dramatic. Consequently, some stands retained leave-tree densities that remain above the threshold where mortality from bark beetles is likely (Cochran et al., 1994). Compared to other treatments, thinning had the greatest effect on the diameter distribution of both ponderosa pine and Douglas-fir by removal of stems between 10 and 25 cm in diameter. This had the effect of maintaining or strengthening the negative exponential or reverse J-shaped diameter distribution. Because our thinning incorporated no additional treatment of the smallest-diameter stems and seedlings, however, the effect was limited and probably fell short of creating fire resilient stands. As noted by Agee and Skinner (2005), low thinning with retention of the smallest diameter stems and seedlings retains much of the ladder fuels and thus the treatment has little effect on increasing crown base height.

Our metrics for large coarse woody debris resources after thinning were well within the range of similarly treated ponderosa pine stands in the Blue Mountains of northeastern Oregon (Torgersen, 2002). These log resources provide important habitat for a host of small mammals, amphibians, and reptiles, primarily as escape cover, shelter, and runways (Bull et al., 1997). While we lack pre-treatment data for log resources, our thinning likely had little effect on log density. Log density after thinning was similar to log density in control units and higher than in burn and thin + burn units. Our use of mechanical thinning provided precise control over the retention of logs. Rarely were new logs added to the thinned units as a result of harvester activity.

Understory species composition was remarkably stable and consistent in thinned units. Indicator species associated with pre-treatment conditions were also indicators of 2004 post-treatment conditions with similar frequency and average cover. Disturbance activities such as our thinning often are cited as potential contributors to the spread and establishment of non-native invasive species because the ground-disturbing activity creates seedbeds for plant establishment and logging equipment serves as vectors in the spread of reproductive plant material (Vitousek et al., 1996). In this study, however, low ground-pressure equipment operated on top of activity fuels to prevent disturbance of the forest floor. By 2004, only three non-native invasive grasses (*Bromus japonicus*, *B. tectorum*, and *Ventenata dubia*) and five annual and biannual forbs (*Rumex acetosella*, *Taraxacum officinale*, *Alyssum alyssoides*, *Dianthus armeria*, and *Lactuca serriola*) were detected. *Bromus tectorum* was the only species of this group to be present in any more than a trace and it showed no evidence of expanding. Thinning also may contribute to the prevention of species invasion in dry forest ecosystems because diversity and productivity of native plant communities are improved (Covington et al., 1997). Our measures of species diversity, however, indicate that the thinning had no effect on diversity. In much the same manner as a similar Fire and Fire Surrogate study site in western Montana (Metlen and Fiedler, 2006), thinning tended to increase the dominance of several rhizomatous species such as *C. rubescens*, *S. albus*, and *Spiraea betulifolia* presumably because of increased resource availability to plants already established under the canopy. Shifts in species space reflected in ordination scores suggested that thinning resulted in fewer shade tolerant, moist-site species.

4.3. Burn effects

In this study, late season burns had little effect on structural attributes of overstory trees. Lack of basal area reduction, density reduction, and decline in SDI was unexpected. There are several possible explanations for the persistence of basal area, density, and SDI. First, because of the densities resulting from 80 years of fire exclusion, burn prescriptions were carefully tailored to each burn unit to consume surface fuels yet prevent movement of the flame into overstory canopies. Thus, direct mortality of large trees likely occurred only when lethal temperatures occurred at cambial layers near the root collar. More aggressive burning prescriptions may have achieved more tree mortality, especially in the smaller diameter stems, with correspondingly greater risk of the burning front reaching tree crowns. Our burns were the first entry in a continuing process of returning frequent fire to these stands; future entries may come closer to desired target conditions. Secondly, our description of structural changes only encompasses the first 4 years post-treatment. Latent mortality of overstory trees either directly from burning or from the interaction of burning, insects, diseases, and environmental conditions may continue for years (Swezy and Agee, 1991; Busse et al., 2000).

Burning had the greatest effect in reducing the number of relatively small diameter Douglas-fir and eliminating Douglas-fir seedlings. Live seedling density was lower in burn units compared to control and thinned units. This had the effect of emphasizing the modal or normal distribution, with few stems except in the 25–39.9 cm diameter class. Changes in the diameter distribution were reflected in a higher QMD. Based on the dominant plant associations within burn units, post-treatment SDI values indicated that these stands remained at considerable risk to serious mortality from bark beetles (Cochran et al., 1994).

As expected, burning reduced log density. Tree mortality, longevity of standing snags, and persistence of logs are components of a dynamic process for converting live trees into forest floor biomass. Just as latent tree mortality is expected with burning, latent log recruitment is expected over time to restore log resources to higher levels. Evidence for this exists in the presence of newly recruited logs in burn units.

Our burns had several notable effects on understory species composition. Fire tolerant perennial species such as *S. albus*, *Spiraea betulifolia*, *C. rubescens*, *Carex geyeri*, and *A. cordifolia* increased in frequency and average cover with burning. Surface burns that cycle nutrients, increase water availability, and create growing space may improve their competitive advantage. In contrast, cover of the bunch grass *F. idahoensis* was reduced with burning. Burning may contribute to species invasion in dry forest ecosystems when exposed sites provide suitable seed beds and seed sources are present near the site. Four non-native invasive species were present in burn units before treatment: *Dactylis glomerata*, *Bromus tectorum*, *Phleum pratense*, and *Poa pratensis*. All four probably represent the results of seeding and grazing practices. By 2004, non-native invasive species included the annual forbs *Dianthus armeria* and *Lactuca serriola* perennial forbs *Rumex acetosella* and *Taraxacum officinale*, and grasses *Bromus japonicus*, *B. tectorum*, and *Ventenata dubia*. Both *B. japonicus* and *B. tectorum* occurred with $<3\%$ average cover; the other species were more limited, suggesting that invasion and expansion of non-native species populations was occurring slowly if at all. Burning also had no effect on species diversity. Shifts in species space reflected in ordination scores suggest that the burns affected species composition by stimulating an increase in abundance of species representing greater shade tolerance and finer-textured soils.

4.4. Thin + burn effects

In our study, the thin + burn treatment resulted in the lowest density of live seedlings. The thin + burn was the only treatment to reduce basal area to the target of 16.0 m^2 ha^{-1}. Perhaps most importantly, the thin + burn treatment resulted in strong modal or normal diameter distributions for both ponderosa pine and Douglas-fir. These results are consistent with those predicted for a low, commercial thin followed by prescribed fire (Agee and Skinner, 2005). Based on the dominant plant associations within thin + burn units, post-treatment SDI values were below the threshold where serious mortality from bark beetles would be expected (Cochran et al., 1994) and will likely remain below this level for perhaps the next decade.

Log resources after our thin + burn treatment were similar to the burn treatment with respect to total log length, volume, and cover. This treatment tended to increase log density compared to burn treatments, presumably because of greater overstory tree mortality and more newly recruited logs.

Interestingly, changes in understory species composition associated with the thin + burn treatment did not follow that for the burn treatment. Shifts in species space reflected in ordination scores suggested that the thin + burn treatment affected species composition by stimulating an increase in abundance of species representing shallow, coarse texture soils. Perennial species such as *Pseudoroengeria spicata*, *Carex geyeri*, *S. albus*, and *Spiraea betulifolia* increased in abundance after treatment. Cover of *Physocarpus malvaceus* and *C. rubescens* were unchanged, while the bunch grass *F. idahoensis* decreased almost by half. Two non-native invasive species were present before treatment: *Bromus tectorum* and *Poa pratensis*. After treatment, the list of non-native invasive species was similar to that after burning alone, with the addition of the annual forbs *Alyssum alyssoides* and *Sisymbrium altissimum*. *Bromus tectorum* was the most abundant non-native invader, with $<4\%$ average cover. Continued monitoring will be required to assess the long-term effects of this treatment on non-native species establishment.

4.5. Implications for management and further research

A broad array of structural changes resulted from our four restoration treatments. These three active treatments are the first step in the process of restoring historical forest structure and ecosystem processes and reducing the risk of uncharacteristically severe wildfire (Fiedler et al., 1998; Harrod et al., 1999; Spies et al., 2006), and managers should use our results as indicative of only the first in a series of planned treatments. Thinning and the combination of thinning and burning reduced the density of live overstory trees, while burning and the combination of thinning and burning reduced the density of small diameter trees, converting the diameter distributions from strongly negative exponential to modal or normal. Thinning, burning, and the combination of thinning and burning increased the dominance of the same bunch grasses, rhizomatous graminoids, and low shrubs as those at a similar Fire and Fire Surrogate site in western Montana (Metlen and Fiedler, 2006), providing strong evidence for a consistent response to these restoration treatments. Active treatments failed to completely meet the treatment objectives of restoring stands to structures representing historical conditions, suggesting that repeated treatments over time are necessary. Yet each active treatment was successful in modifying some structural or compositional components such that the risk of stand-replacement fire was likely reduced. It is unlikely that any single treatment or entry will mitigate the nearly 80 years of fire exclusion and fuel accumulation in low elevation dry forests. A second treatment application in 10–15 years may provide greater efficacy and a greater reduction in the risk of uncharacteristically severe wildfires.

What are the implications for future research of stand dynamics in low elevation, dry forests dominated by ponderosa pine and Douglas-fir and restoration treatments? We report evidence that restoration treatments resulted in differential short-term structural and compositional changes on the overstory, coarse woody debris, and understory plant communities. These findings provide a necessary foundation for future and ongoing assessments of wildlife dynamics, treatment economics, and interactions among treatments, various avian guilds, and insect populations that lead to tree mortality. Further work with potentially more severe treatments, such as greater basal area reduction or greater mortality from burning, would aid in quantifying ecosystem resilience. When extended beyond this first set of treatments, our study provides a foundation for assessment of continued long-term restoration treatment goals of reducing fuels and accelerating the development of late-successional stand structure in low elevation dry ponderosa pine and Douglas-fir forests of northeastern Oregon. As part of the national Fire and Fire Surrogate study, this work identifies the nature and strength of ecological responses that cross ecosystem boundaries, and provides managers a framework for predicting the outcome of restoration treatments.

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