

Developing resilient ponderosa pine forests with mechanical thinning and prescribed fire in central Oregon's pumice region

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Abstract: Thinning and prescribed burning are common management practices for reducing fuel buildup in ponderosa pine forests. However, it is not well understood if their combined use is required to lower wildfire risk and to help restore natural ecological function. We compared 16 treatment combinations of thinning, prescribed fire, and slash retention for two decades across a site quality gradient of second-growth pine stands, measuring changes in forest vegetation growth, structure, and composition. Thinning alone doubled the diameter growth increment of ponderosa pine, moderately stimulated shrub production, and resulted in lower tree mortality compared with unthinned plots. In contrast, repeated fire alone did not substantially alter stand structure or increase tree vigor, herbaceous production, or plant diversity. The combined use of thinning and repeated burning reduced shrub cover, yet produced no changes in herbaceous production, plant diversity, stand structure, or tree vigor compared with thin-only treatments. Additional findings identified (1) inconsequential effects of thinning residues on site productivity, (2) the need for multiple entries of prescribed fire if the abatement of shrubs is required, (3) the ineffectiveness of repeated burning to stimulate plant growth, and (4) that the thinning treatment served as an effective surrogate to fire for managing central Oregon forest vegetation.

Résumé : L'éclaircie et le brûlage dirigé sont des pratiques courantes d'aménagement pour réduire la quantité de combustibles dans les forêts de pin ponderosa. Cependant, on comprend peu la nécessité de combiner l'utilisation de ces deux traitements pour réduire les risques de feu et pour aider à restaurer les fonctions écologiques naturelles. Nous avons comparé 16 combinaisons de traitements d'éclaircie, de brûlage dirigé et de rétention de débris ligneux pendant deux décennies le long d'un gradient de qualité de station établi dans des pinèdes de seconde venue, en mesurant les changements de croissance, de structure et de composition de la végétation forestière. La seule application de l'éclaircie a doublé l'accroissement en diamètre du pin ponderosa, a modérément stimulé la production d'arbustes et a réduit la mortalité des arbres comparativement aux parcelles témoins. À l'opposé, l'application seule de brûlages dirigés répétés n'a pas modifié substantiellement la structure des peuplements ou augmenté la vigueur des arbres, la production des plantes herbacées ou la diversité végétale. L'utilisation combinée de l'éclaircie et du brûlage répété a réduit le couvert arbustif, mais n'a pas modifié la production des plantes herbacées, la diversité végétale, la structure des peuplements ou la vigueur des arbres comparativement aux traitements d'éclaircie seule. D'autres résultats ont montré (1) les effets sans conséquence des résidus d'éclaircie sur la productivité des stations, (2) la nécessité d'entrées multiples du brûlage dirigé si la réduction des arbustes est nécessaire, (3) l'inefficacité du brûlage répété pour stimuler la croissance végétale et (4) le fait que l'éclaircie est un substitut efficace au feu pour aménager la végétation forestière du centre de l'Oregon.

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Introduction

Ponderosa pine (*Pinus ponderosa* C. Lawson) forests are a multiresource treasure, offering aesthetic beauty, wildlife habitat, timber, recreational opportunities, historical sites, and open space for urban expansion. They span western North America, from the Pacific Coast to the Great Plains, and from southern British Columbia to Baja California

(Meyer 1938; Oliver and Ryker 1990). Most are adapted to dry, fire-prone climates and have a pre-Euro-American settlement history of frequent, low-severity surface fires that often maintained a dominance of large-diameter, open-grown trees (Youngblood et al. 2004). Beginning in the late 1800s, a progression of management practices (logging, livestock grazing, fire suppression and exclusion) left many of these forests with uncharacteristically high stand den-

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sities, altered composition of understory plants, and increased risk of large-scale stand-replacement wildfire (Weaver 1943; Hessburg and Agee 2003). The integrity and continuity of many old-growth pine stands was further lost during the twentieth century owing to the growing demand for high-quality wood products (Hessburg and Agee 2003).

Because of the risk of wildfire and insect outbreaks, many land managers and ecologists now emphasize the need for restorative treatments to reestablish the historical structure, composition, and function of ponderosa pine forests (Moore et al. 1999; Allen et al. 2002; Synder 2007). A key tenet of restoration ecology is that the process of reestablishing native ecosystems should be compatible with the evolutionary history of a given system (Moore et al. 1999). For ponderosa pine forests, this suggests the reintroduction of low-severity fire and the reduction of stand densities by thinning, burning, or a combination of the two practices. Basic principles for restoring pine forests and mitigating fire hazard include reducing surface fuels; increasing the height of the live canopy (reducing ladder fuels); decreasing tree density; and retaining large, fire-resistant trees within stands (Hessburg and Agee 2003; Agee and Skinner 2005). As a reality check, Hessburg et al. (2005) acknowledged that attempts to recapture the health of these ecosystems will likely require repeated treatment, adaptive prescriptions, and patience.

Recent examples of wildfire behavior in ponderosa pine forests document the value of restoring forest structure and reducing fuel continuity in fire-prone landscapes. The Cone Fire, which burned through ponderosa pine stands in northeastern California in 2002, is a case in point. Crown-fire spread and severe tree mortality were abruptly stopped when advancing flames reached research areas that were recently thinned and underburned (Ritchie et al. 2007). In areas that received thinning alone, the flames dropped to a surface fire with small pockets of tree scorching and mortality. Other postfire reconnaissance studies also reported that thinned (Pollet and Omi 2002; Strom and Fulé 2007), prescribed burned (Finney et al. 2005), and thinned plus burned (Pollet and Omi 2002) stands had reduced fire severity compared with untreated pine stands. But can thinning alone initiate the restoration of ponderosa pine forests to historical or, at a minimum, fire-resilient conditions? Theory suggests that several ecological processes attributable to frequent, low-severity fire are unlikely to be met by mechanical harvesting. These include exposure of bare mineral soil for seed germination, reduction of surface and ladder fuels, nutrient mineralization from forest floor organics, elimination of fire-intolerant conifers, seed scarification, and shift from fire-sensitive to fire-adapted understory vegetation. Because of the many functions of fire, Agee and Skinner (2005) suggest that thinning alone is not a panacea and that careful consideration be given to the thinning method and management objectives in fire-dependent forests. They recommend (i) thinning from below (low thinning) and removal of small unmerchantable material (precommercial thinning) to increase the canopy base height and (ii) whole-tree harvesting to avoid accumulation of surface fuels. Early results from the Fire and Fire Surrogate study, a nationwide study that compared ecosystem responses to combinations of prescribed fire and thinning (Youngblood et al. 2007), also support the combined use of thinning and burning for restoring

forest structure, herbaceous cover, and soil productivity (Metlen et al. 2004; Gundale et al. 2005; Metlen and Fiedler 2006; Youngblood et al. 2006).

Management efforts to restore pine forests to fire-resilient conditions have been on-going in most areas throughout the West, including central Oregon where the natural fire-return interval prior to the twentieth century was every 4–24 years (Bork 1984). Here, the use of prescribed fire as a surrogate for natural, low-severity fires was recognized as an important tool, beginning with the pioneering work of Harold Weaver (Weaver 1943, 1967) and with the work of Bob Martin (Joslin 2007), and thinning prescriptions to improve stand growth and forest health are well-established from classic, long-term research studies (Cochran and Barrett 1999; Oliver 2005). On the Deschutes National Forest, a concerted effort to reduce stand densities began in the 1980s after a mountain pine beetle (*Dendroctonus ponderosae* Hopkins) outbreak devastated the adjoining lodgepole pine (*Pinus contorta* Dougl. ex Loud.) forests. Evidence that maintaining open stands of vigorously growing trees was a restraint to pine beetle outbreaks (Larsson et al. 1983; Mitchell and Preisler 1991) helped trigger this response. Since then, approximately 50% of the ponderosa pine stands on the Deschutes National Forest have been mechanically thinned, and 20% of the stands have been treated with prescribed fire (J. Booser, Forest Silviculturist, Deschutes National Forest, personal communication, 2008).

Our study was imbedded within the second-growth ponderosa pine forests on the Deschutes National Forest. These were densely stocked stands at the study onset in 1989 that had regenerated following railroad logging in the 1930s, with shrub-dominated understory vegetation and high risk of wildfire and insect damage. Knowledge of the combined effects of thinning and burning at that time, both in central Oregon and elsewhere, was limited to anecdotal observations. Thus, our overall objective was to help fill this knowledge gap by determining the ecological effects of thinning and prescribed fire on vegetation composition and growth, fuel succession, and soil processes. Thinning and burning treatments were applied both separately and in combination to determine their additive effects. Replicate sites were strategically located to capture the site productivity gradient for ponderosa pine across the region's pumice plateau. We report here on vegetation responses to the restorative treatments. Specifically, our objectives were to determine (1) long-term changes in tree mortality, stand growth, and understory production following thinning, burning, or their combination, (2) whether thinning residues affect site productivity, (3) the comparative increase in plant production associated with nutrient flushes from burning versus fertilizer application, and (4) if and when additional treatment is needed to limit fuel accumulation.

Methods

Site description

The study was conducted on the Deschutes National Forest, in the rainshadow of the Cascade Range in central Oregon. The landscape is a gently rolling plateau located between the Cascade Range and the northwestern-reach of the Great Basin to the east. Ponderosa pine forests are com-

Table 1. Site characteristics in 1988, 1 year prior to thinning.

Site characteristic	Swede Ridge	Sugar Cast	East Fort Rock
Location (latitude, longitude)	43°84'N, 121°32'W	43°84'N, 121°34'W	43°85'N, 121°35'W
Site index (m at 100 years; Barrett 1978)	35	31	25
Stand age (years)	40	49	56
Stand density (trees·ha ⁻¹)	804	704	477
Quadratic mean diameter (cm)	23.3	24.2	25.2
Basal area (m ² ·ha ⁻¹)	34	32	24
Elevation (m a.s.l.)	1520	1398	1554
Precipitation (cm)	65	50	38
Mean July temperature (°C)	15	18	18

mon throughout the plateau, with lodgepole pine stands more typical on flat-to-concave terrain. Forest productivity is limited by cold winters and dry summers. The annual precipitation of 50 cm occurs primarily as snow during the winter months. Consequently, the growing season for ponderosa pine typically lasts only from mid-May to mid-August.

Ponderosa pine regenerated naturally on the Deschutes National Forest following railroad logging in the 1930s. By the 1980s, an estimated 53 000 ha of second-growth pine was considered at risk for catastrophic insect or fire disturbance as a result of the long-standing policy of fire suppression on public lands. We selected three sites within these second-growth forests as representative of low, medium, and high site productivity for ponderosa pine. East Fort Rock, the least productive site, is located near the desert fringe, 17 km southeast of Bend, Oregon; Sugar Cast, a moderately productive site, is located 5 km east of Sunriver, Oregon; and Swede Ridge, the most productive site, is located 20 km west of Bend, Oregon. Site productivity is dictated primarily by the precipitation gradient that extends east from the Cascade crest to the desert fringe. The sites have been free of major disturbance since seedling establishment with the exception of a single precommercial thinning operation in the early 1960s. Additional site characteristics and stand conditions are listed in Table 1.

The soil at the three sites is a cryic Vitrand, developed from windblown deposits of pumice and ash following the eruption of Mount Mazama approximately 7700 years BP. Soil fertility is low and horizon development is weak, with a 5–10 cm loamy sand surface horizon (42 g·kg⁻¹ organic matter, 1.4 g·kg⁻¹ total nitrogen (N), pH 6.2) above a 30–40 cm transition horizon (10 g·kg⁻¹ organic matter, 0.2 g·kg⁻¹ total N, pH 6.2) and an undeveloped C horizon. Understory vegetation is dominated by two N-fixing shrubs. Antelope bitterbrush (*Purshia tridentata* (Pursh DC.), referred to herein as bitterbrush) is most common at East Fort Rock and Sugar Cast, and snowbrush (*Ceanothus velutinus* Douglas ex Hook.) is most common at Swede Ridge. Green-leaf manzanita (*Arctostaphylos patula* Greene) is also found at all three sites. Bottlebrush squirreltail (*Elymus elymoides* (Raf.) Swezey), western needlegrass (*Achnatherum occidentale* (Thurb.) Barkworth), Idaho fescue (*Festuca idahoensis* Elmer), Ross' sedge (*Carex rossii* Boott), Virginia strawberry (*Fragaria virginiana* Duchesne), cryptantha (*Cryptantha affinis* (A. Gray) Greene), and silverlead phacelia (*Phacelia hastata* Douglas ex Lehm.) are common herbaceous plants. Understory pine seedlings and saplings were

Table 2. Tree diameter class distribution after thinning.

Treatment	% of trees with tree diameter class of:			
	<10 cm	10–25 cm	25–40 cm	40–55 cm
No thin	2 (1)	52 (2)	43 (1)	3 (2)
Burn (no thin)	4 (4)	58 (5)	37 (3)	1 (1)
Thin	0 (0)	25 (2)	71 (3)	4 (1)
Thin + burn	0 (0)	26 (4)	70 (4)	4 (2)

Note: Values are averages for the three sites. Standard errors are in parentheses.

absent. Plant nomenclature follows the USDA PLANTS database (<http://plants.usda.gov>).

Study design

The experiment was a randomized complete block design with three replications of 16 treatments. The treatments were arranged in a 4 × 2 × 2 factorial design that included four thinning plus slash-removal treatments (thinning plus whole-tree removal (WT), thinning plus bole-only removal (BO), thinning plus no removal (NR), and no thin), two prescribed fire treatments (repeated, none), and two fertilizer treatments (N + P + S application, none). All treatments were installed at each of the three sites (blocks). Treatment plots were 0.4 ha (61 m × 61 m) with 20 m or greater between adjacent plots. Tree growth was measured within 0.2 ha interior subplots, providing an average of 40 measurement trees on thinned subplots and 90 measurement trees on unthinned subplots.

Thinning and slash removal

The thinning treatment followed the preferred prescription used by resource managers on the Deschutes National Forest in the 1980s. Target basal area was 13.7 m²·ha⁻¹, with a tree spacing of approximately 5.5 m × 6.1 m that favored the removal of damaged or smaller trees (thinning from below). Trees marked for thinning were cut by chainsaw between November 1988 and October 1989. Felled trees were removed from WT and BO plots using either a rubber-tire skidder or track grapple skidder (≤48 kPa ground pressure). Post-thinning tree diameter distribution was fairly uniform, with 70% of all trees between 25 and 40 cm at 1.3 m height (DBH) and no trees less than 10 cm DBH (Table 2). All harvest material was removed from WT plots; bolewood only was removed from BO plots, with tree crowns lopped and scattered across the plots; and all harvest material was left on site for the NR treatment, with boles left intact on the ground and tree crowns lopped and scattered across the

plots. Thinning residue mass was calculated using a modified planar-intercept method (Brown 1974; Shea 1993). Downed wood was counted on 12 systematically located transect lines per plot. Measurement length on each transect line was 3 m for 1 h (0–0.6 cm diam.) and 10 h (0.6–2.5 cm diam.) time-lag fuels, as defined by the National Fire-Danger Rating System (Deeming et al. 1972); 10 m for 100 h time-lag fuels (2.5–7.5 cm diam.); and 15 m for 1000 h time-lag fuels and larger (>7.5 cm diam.). Litter and duff mass was estimated by collecting twelve 50 cm × 50 cm samples per plot, located adjacent to each transect line. Dry masses were determined following oven drying at 70 °C for 48 h.

Prescribed burning

Twenty-four burns (four thinning treatments × two fertilizer treatments × three replications) were conducted in early June 1991. The burns did not coincide with the natural season of burning in central Oregon (historically in late summer and early fall) because of the excessive fuel buildup and the potential for high fire severity and loss of containment. A strip-head firing technique was used by US Forest Service fire personnel on all burns, with the strip width and ignition speed varied to maintain near-constant fire spread and flame lengths. Mineral soils were moist at the time of burning, ranging from 32% to 43% moisture by mass. Duff moisture averaged 45%, and fine woody fuel (0–2.5 cm diam.) moisture content averaged 18%. The burns were of low to moderate intensity, with average flame lengths of 0.5–1.2 m. Duff consumption averaged 45% by mass (see Shea (1993) for details of the burn conditions and fire behavior).

Burns were repeated in early June 2002. The objective of these burns was to control the rapid buildup of understory shrubs in the 11 years following the initial burns. Only the 12 nonfertilized plots that had been burned in 1991 were re-burned (four thinning treatments × three replications) because of limited funding. Soil, duff, and fuel moisture contents at the time of burning were similar to the moisture contents recorded during the 1991 burns. The burns were of low intensity, with average flame lengths of 0.3–0.7 m. Duff consumption averaged 39% by mass.

Fertilizing

The fertilizer treatment of 224 kg N·ha⁻¹, 112 kg P·ha⁻¹, and 37 kg S·ha⁻¹ was selected based on findings for pine growth in central Oregon soils (Cochran 1977, 1979). Granular formulations of urea, triple superphosphate, and ammonium sulfate (at necessary combinations to meet the application rates) were applied uniformly across plots with hand spreaders in October 1991 and repeated in October 1996.

Tree measurements

All trees within each subplot (5268 trees total) were measured for DBH in 1988, one year prior to thinning. We also collected two increment cores at breast height from opposite sides of each tree to determine pretreatment periodic growth rate. Tree diameters were estimated for 1968, 1973, 1978, and 1983 by measuring increment growth of each core to the nearest 0.05 cm. Five-year basal area increment was calculated assuming no tree mortality or recruitment occurred between 1968 and 1988. We made this assumption since all understory trees had been precommercially thinned

in 1963, and no dead trees were noted during plot measurements in 1988.

Trees were measured for DBH in 1991, 1996, 2001, and 2006. Mortality was recorded at each measurement date, along with its probable source (fire, bark beetle). Tree damage (forking, mistletoe, insects, bole scar) was noted when present. Total height was measured on all trees in 1991 and 1996 with an optical dendrometer, with a subset of 15 trees per plot (representing a cross-section of tree sizes) measured for volume. Regression equations for each plot were then developed that predicted tree volume as a function of DBH. Coefficients of Determination (r^2) ranged from 0.92 to 0.99. These equations were used to estimate tree volumes in 2001 and 2006 based on field measurements of DBH. Canopy length, height to live crown, and crown scorch (if present) were measured on all trees in the fall of 1991. Periodic annual increment growth (diameter PAI and net volume PAI) was calculated for live trees only within each 5 year growth period.

Canopy cover was measured in 1997 and used to examine the relationship between overstory cover and the presence of understory vegetation. Canopy diameters were measured on the north–south and east–west axes of 15 trees per plot and converted to an area basis assuming an elliptical shape. Canopy area was then predicted as a function of DBH in regression analysis (SAS version 9.1; SAS Institute Inc. 2003). Coefficients of determination for individual plots ranged from 0.34 to 0.90, with a median of 0.72. We then estimated total canopy cover on a plot basis by summing the canopy area of all trees within a plot, as determined using the regression equations and input from field measurements of DBH.

A Pandora moth (*Coloradia pandora* Blake) outbreak began at the onset of the experiment, lasting approximately from 1988 to 1994 and resulting in moderate to severe tree defoliation. Visual estimates of defoliation were recorded for the bottom half and the top half of the canopy of each tree in the interior 0.2 ha plots in 1990, 1992, and 1994, matching the insect's 2 year life cycle. Percent defoliation was calculated as the arithmetic mean of all individual trees within a plot. Defoliation estimates were used to help explain periodic tree growth and to determine possible treatment effects of fire, thinning, fertilizing, and their combinations on Pandora moth activity.

Understory vegetation

Shrub cover was estimated ocularly prior to treatment (1988) by a professional ecologist (W.E. Hopkins) with extensive botanical experience in central Oregon forests, and was then measured quantitatively using a belt-transect method in 1993, 1996, 1999, 2003, and 2006. Three belt transects (5 m × 20 m) were located systematically in each subplot, and each shrub within a belt transect was measured for canopy length and width to the nearest centimetre. Coverage of an individual shrub (live + dead foliage and stems) was calculated assuming a rectangular-shaped canopy. This method was used instead of the more traditional line-intercept method because it also provided a measure of shrub biomass for predicting wildlife browse value (Busse and Riegel 2009). Shrub cover within each belt transect was calculated as the sum of the individual shrubs divided by 100 m²; per-

Table 3. Effect of thinning plus slash-removal treatment on downed woody fuel mass ($\text{Mg}\cdot\text{ha}^{-1}$) before burning (1990) and at the end of the study on unburned (2007 no burn) and burned (2007 burn) plots.

Year	WT	BO	NR	No thin
1990	12.1 (3.1)	25.6 (3.3)	47.8 (12.2)	11.9 (2.5)
2007 no burn	8.0 (2.1)	15.7 (3.9)	39.2 (13.3)	9.7 (1.9)
2007 burn	9.7 (2.2)	9.9 (3.0)	38.2 (15.8)	8.6 (2.8)

Note: Values are means with standard error in parentheses. WT, thinning plus whole-tree removal; BO, thinning plus bole-only removal; NR, thinning plus no removal.

cent cover on a plot basis was estimated as the average shrub cover of the three belt transects times 100.

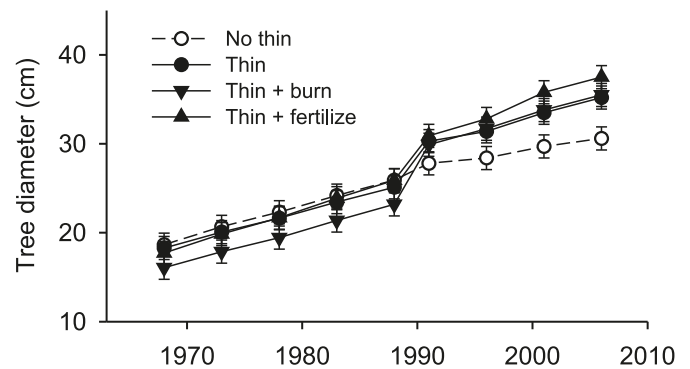
Herbaceous plants were clipped at ground level for biomass determination during peak season (mid-June to mid-July) in 1992, 1993, 1994, 1997, 1998, and 2003. Circular plot frames (1.5 m diam.) were used between 1992 and 1994, and rectangular-shaped frames (0.5 m $\times$ 0.5 m) were used subsequently to meet funding constraints. A minimum of eight systematically located samples were collected and composited from each plot. Samples were collected a minimum of 1 m from previous clipping. All species were identified (for richness and diversity indices), clipped, dried (60 °C for 72 h), and weighed separately for dry matter production for 1992–1994 samples only. Species diversity was estimated by Simpson's Diversity Index (Mangurran 1988), using the dry mass values. After 1994, plants were clipped separately by lifeform (forb, graminoid) prior to drying and weighing. As a consequence, species richness and diversity could not be determined for these years, with the exception that species richness was measured in 2003. Also, results for 1993 were disregarded because of cattle grazing on fertilized plots at Sugar Cast and East Fort Rock. All plots within these two sites were fenced following the 1993 growing season, and livestock grazing was excluded for the remainder of the study.

Statistical analyses

The individual and combined effects of thinning and burning on vegetation dynamics were analyzed by repeated measures analysis (PROC MIXED in SAS version 9.1 (SAS Institute Inc.), with an autoregressive covariance structure that accounted for unequal time periods in the analysis of shrub cover and herbaceous biomass). Data for tree DBH (calculated as quadratic mean diameter; Curtis and Marshall 2000), volume, and periodic annual increments were normally distributed, whereas all understory vegetation data were log transformed to correct for unequal variances. Means and standard errors for these variables were back-transformed for presentation purposes. Multiple linear regression (PROC REG) was used to predict shrub cover as a function of ponderosa pine cover, pretreatment shrub cover, years since treatment, and annual precipitation. Significance for all statistical analyses was set at $\alpha = 0.05$.

Main treatment effects and interactions were analyzed by ANOVA for data collected between 1991 and 2001. However, the experimental design was confounded after 2001, since only 12 out of 24 plots were reburned. Therefore, we used contrast statements to determine the statistical signifi-

Fig. 1. Change in quadratic mean diameter ($\pm$ se) in central Oregon ponderosa pine forests between 1968 and 2006, by treatment. Stands were thinned in 1989.



cance of the treatment comparisons of greatest interest for each measurement year or 5 year growth period. The contrast statements included the following:

1. Thinning — no thin versus the average of WT, BO, and NR.
2. Slash-removal method — WT versus BO versus NR.
3. Prescribed fire in thinned stands — repeated burning versus no burning. Contrasts were run both separately and combined for fertilized and unfertilized plots.
4. Prescribed fire in unthinned stands — repeated burning versus no burning (no-thin plots only). Contrasts were run both separately and combined for fertilized and unfertilized plots.
5. Fertilization — fertilizer versus no fertilizer treatments. Contrasts were run separately for thinned and unthinned treatments.

Results

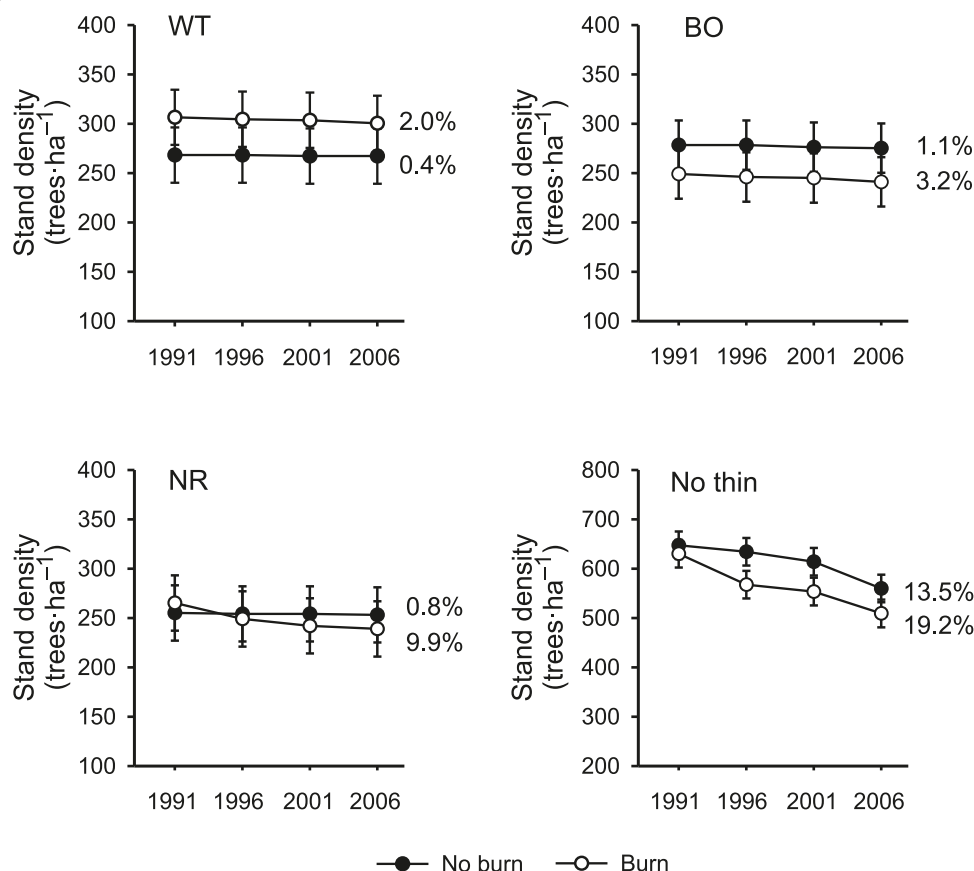
Postharvest residues

Thinning plus slash-removal treatments resulted in a wide range in surface residue mass. As expected, WT had the lowest mass among the slash-removal treatments and was comparable to the no-thin treatment (Table 3). Residue mass was twofold higher for BO and fourfold higher for NR treatments compared with WT. Without fire, forest floor residues were 20%–40% lower by the end of the experiment relative to their 1990 levels, and the proportional differences between the slash-removal treatments were maintained. Repeated burning reduced the residue mass of the BO treatment, yet had little influence on the net residue mass of WT or NR treatments.

Stand density and growth

No differences in stand density, basal area, or DBH were found prior to treatment (1968–1988) among the 48 plots (Fig. 1), with the exception that DBH was slightly smaller on burn plots than on unburned plots (19.9 ± 0.4 cm and 21.0 ± 0.4 cm, respectively; $P = 0.042$). However, no differences in diameter increment ($P = 0.906$) were found between burn and no-burn plots prior to treatment, indicating that tree growth and vigor were consistent between all plots. Basal area and DBH between 1968 and 1988 differed by site ($P < 0.001$) and sampling year ($P < 0.001$) only.

Fig. 2. Stand density changes between 1991 and 2006. Thinning and slash-removal treatments in 1989 were thin plus whole-tree removal (WT), thin plus bole-only removal (BO), and thin plus no removal (NR). Plots were burned in 1991 and 2002. Percentage values are the cumulative mortality between 1991 and 2006. Error bars are standard errors.



Following thinning, stand density $\pm$ SE averaged 266 ± 11 trees·ha⁻¹ for thinned plots and 594 ± 18 trees·ha⁻¹ for unthinned plots between 1991 and 2006. Average basal area was reduced from 29.5 ± 3.0 m²·ha⁻¹ before treatment to 18.0 ± 2.6 m²·ha⁻¹ by the end of the second growing season after thinning. Tree mortality between 1991 and 2006 was minor for thinned trees, averaging 3% or less (Fig. 2), with the exception of the NR treatment, which had higher fire-induced mortality on burned plots. Still, no differences in the 15 year mortality rates resulted because of burning on thinned plots ($P = 0.671$). Mortality averaged 16% on unthinned plots, which was significantly greater than thinned plots ($P < 0.001$). Interestingly, the mortality rate on unthinned plots was similar between burned and unburned treatments ($P = 0.258$), although the pattern of mortality varied between the two treatments. Small openings within the dense stands (10–15 m diameter) were created by fire, whereas a random pattern of mortality from mountain pine beetle attack was evident on unburned plots. Crown scorch from the 1991 burns averaged $19\% \pm 5\%$ for thinned plots and $23\% \pm 3\%$ for unthinned plots. Height to green crown for thinned plots was 7.0 ± 1.0 m following burning and 5.3 ± 0.7 m without burning. For unthinned plots, height to green crown was 6.3 ± 0.6 m with burning and 5.0 ± 0.7 m without burning. Crown scorch was not measured after the 2002 burns, although visual inspection found limited scorch (<15%) across the study plots following the reburns.

Ponderosa pine DBH rose sharply following thinning in 1989, then increased steadily relative to unthinned plots in the succeeding 17 years (Fig. 1). No differences in DBH were detected among the three thinning methods between 1991 and 2006 ($P = 0.646$; Fig. 3), nor was there an effect of repeated burning in thinned ($P = 0.689$) or unthinned ($P = 0.149$) stands. In contrast, fertilizing resulted in significantly greater DBH in thinned stands ($P = 0.016$) but not in unthinned stands ($P = 0.745$) during the experiment. Stand volume was 28% greater for unthinned plots compared with thinned plots between 1991 and 2006 (Fig. 3) because of the higher stand densities. Burning reduced the stand volume in unthinned plots only ($P < 0.001$), whereas fertilizing increased stand volume in thinned stands only ($P = 0.006$).

Periodic annual increment of live trees was differentially affected by thinning, burning, and fertilizing. Diameter increment was twice as great on thinned plots than on unthinned plots in each 5 year measurement period (Fig. 4, Table 4). Repeated burning had little effect on diameter growth in either thinned or unthinned plots, whereas fertilizing significantly increased tree growth between 1991 and 2001. No fertilizer effect was found in the last growth period, which was 5–10 years after the final fertilizer application in 1996. All two-way, three-way, and four-way interactions were nonsignificant between 1991 and 2001 (Table 5), with the exception of the thinning $\times$ year interaction.

Fig. 3. Effect of thinning, slash removal, burning, and fertilizing on quadratic mean diameter (A) and stand volume (B) between 1991 and 2006. Thinning and slash-removal treatments include thin plus whole-tree removal (WT), thin plus bole-only removal (BO), and thin plus no removal (NR). Error bars are standard errors.

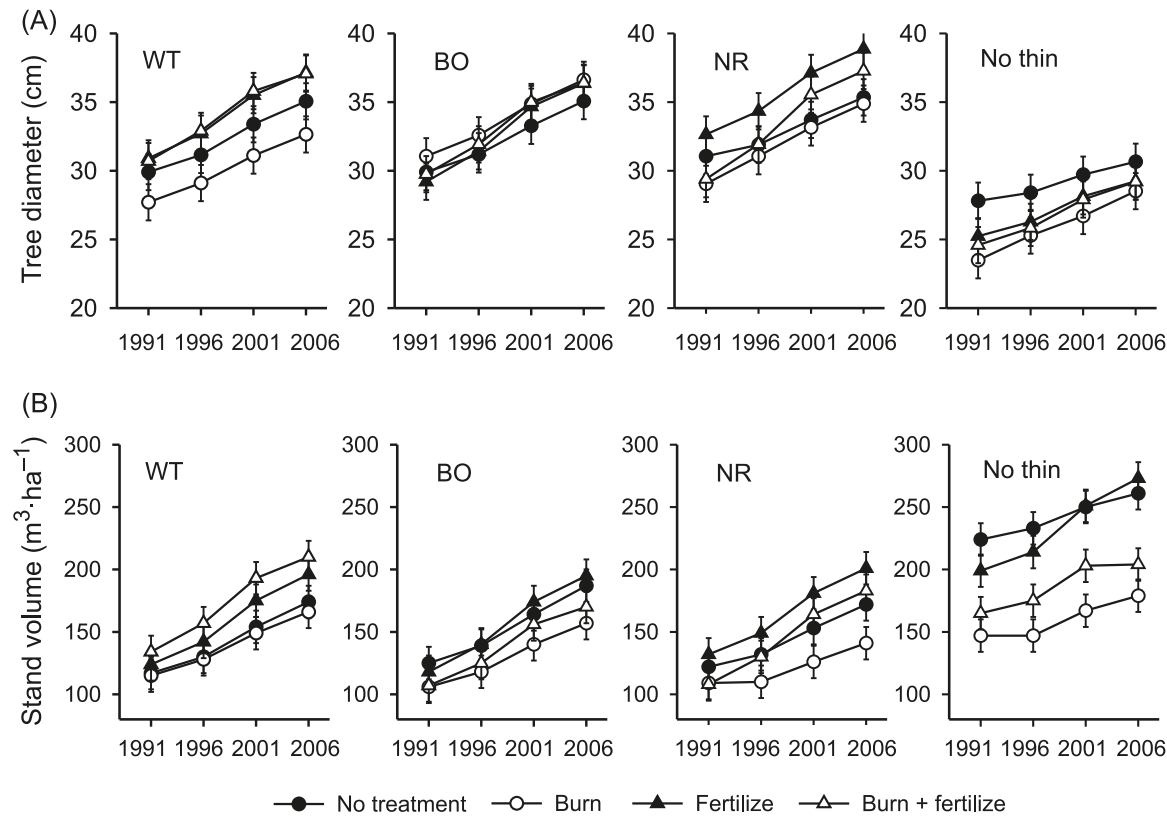


Fig. 4. Ponderosa pine diameter (A) and volume (B) periodic annual increments (PAI) following thinning (1989), burning (1991 and 2002), and fertilizing (1991 and 1996). Values are for survivor trees (net PAI). Error bars are standard errors.

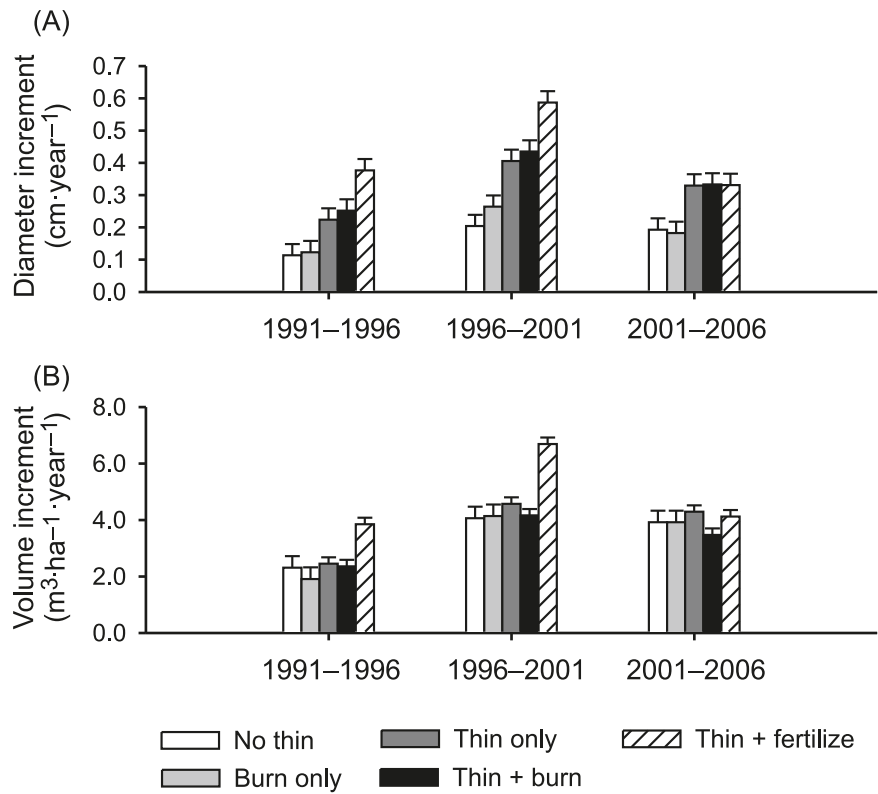


Table 4. Contrast comparisons (*P* values) for the effects of thinning, slash-removal method, burning, and fertilizing on ponderosa pine diameter and net volume annual increment (live trees only) between 1991 and 2006.

Contrast	Growth period	Periodic annual increment	
		Diameter	Net volume
Effect of thinning (no thin versus average of WT, BO, NR)	1991–1996	0.010	0.803
	1996–2001	<0.001	0.220
	2001–2006	0.002	0.509
	1991–2006	<0.001	0.376
Effect of slash-removal method (WT versus BO versus NR)	1991–1996	0.405	0.509
	1996–2001	0.623	0.514
	2001–2006	0.402	0.969
	1991–2006	0.563	0.693
Effect of burning thinned plots (no fertilizer added)	1991–1996	0.397	0.963
	1996–2001	0.531	0.195
	2001–2006	0.954	0.072
	1991–2006	0.771	0.194
Effect of burning unthinned plots (no fertilizer added)	1991–1996	0.849	0.562
	1996–2001	0.245	0.912
	2001–2006	0.839	0.257
	1991–2006	0.659	0.505
Effect of fertilizing thinned plots	1991–1996	<0.001	<0.001
	1996–2001	<0.001	<0.001
	2001–2006	0.642	0.984
	1991–2006	<0.001	<0.001

Note: WT, thinning plus whole-tree removal; BO, thinning plus bole-only removal; NR, thinning plus no removal.

Table 5. Analysis of variance results for diameter and net volume periodic annual increment of live trees (PAI) during consecutive 5 year periods between 1991 and 2001.

Source	Diameter PAI	Net volume PAI
Site	0.234	<0.001
5 year period (Y)	<0.001	<0.001
Thinning (T)	<0.001	0.384
Y × T	0.031	0.775
Burn (B)	0.096	0.422
Y × B	0.520	0.124
T × B	0.161	0.120
Y × T × B	0.184	0.939
Fertilize (F)	<0.001	<0.001
Y × F	0.116	0.003
T × F	0.209	0.722
Y × T × F	0.617	0.689
B × F	0.771	0.797
Y × B × F	0.233	0.674
T × B × F	0.436	0.206
Y × T × B × F	0.999	0.624

Note: Main effect for thinning compares WT (thinning plus whole-tree removal), BO (thinning plus bole-only removal), NR (thinning plus no removal), and no-thin treatments.

Volume periodic annual increment (live trees only) was not significantly affected by thinning, slash-removal treatment, or burning in any of the three growth periods (Fig. 4, Table 4). Only fertilization resulted in greater periodic volume growth. Fertilizing thinned plots resulted in 39% higher volume compared with unfertilized plots in the first growth

period, and 42% higher volume in the second growth period. Similar to the results for diameter PAI, no differences in volume PAI of live trees were found owing to fertilization in the last 5 year period.

Large differences in volume growth were found between the first and second growth periods (Fig. 4). The average volume for all treatments was $2.2 \text{ m}^3 \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ in the first growth period compared with $4.4 \text{ m}^3 \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ for the second growth period. Climatic records indicate that annual precipitation was similar between 1991 and 2001, suggesting that the growth response was not driven by climate. Instead, we believe this growth response was a consequence of the Pandora moth infestation between 1988 and 1994. Average defoliation for the three sites was 55% in 1990, 50% in 1992, and 10% in 1994. In comparison, minimal Pandora moth activity was detected in the second growth period between 1996 and 2001. Interestingly, no differences in defoliation were found among thinning, burning, or fertilizing treatments. Further, the relative tree growth response among treatments was similar between the first growth period and second growth periods (Fig. 4), suggesting that growth recovery following defoliation was independent of treatment.

Shrub cover

Shrub cover declined progressively from 1988 to 2006 on untreated plots, while repeated fire alone nearly eliminated the shrub layer (Fig. 5). In comparison, thinning resulted in a short-term reduction of 28% relative to the pretreatment cover, followed by a steady increase throughout the experiment. Burning in 1991 reduced shrub cover on thinned plots to 6%, which was followed by a rapid recovery during post-fire conditions. By the fifth growing season after burning

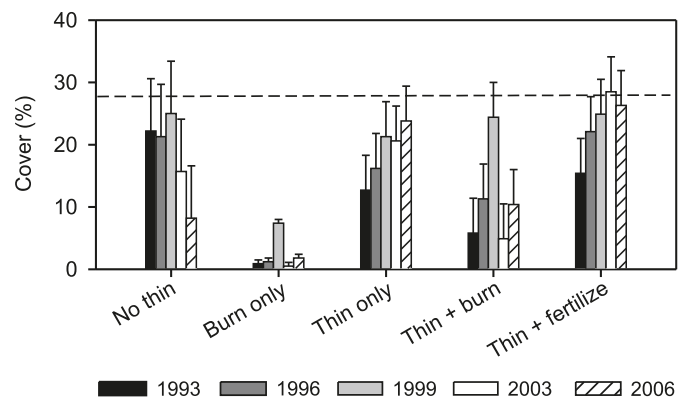
Table 6. Contrast comparisons (*P* values) for the effects of thinning, slash-removal method, burning, and fertilizing on total shrub cover and graminoid and forb biomass between 1991 and 2006.

Contrast	Shrubs		Herbaceous biomass		
	Year	Cover	Year	Graminoid	Forb
Effect of thinning (no thin versus average of WT, BO, NR)	1993	0.398	1992	0.555	0.528
	1996	0.445	1994	0.985	0.952
	1999	0.654	1997	0.070	0.665
	2003	0.854	1998	0.586	0.317
	2006	0.266	2003	0.207	0.777
	1993–06	0.863	1992–03	0.374	0.751
Effect of slash-removal method (WT versus BO versus NR)	1993	0.767	1992	0.689	0.517
	1996	0.937	1994	0.943	0.862
	1999	0.940	1997	0.235	0.641
	2003	0.981	1998	0.346	0.826
	2006	0.963	2003	0.383	0.230
	1993–06	0.962	1992–03	0.619	0.857
Effect of burning thinned plots (no fertilizer added)	1993	0.023	1992	0.599	0.898
	1996	0.532	1994	0.766	0.524
	1999	0.966	1997	0.965	0.111
	2003	0.001	1998	0.970	0.857
	2006	0.057	2003	0.905	0.970
	1993–06	0.069	1992–03	0.820	0.474
Effect of burning unthinned plots (no fertilizer added)	1993	0.009	1992	0.472	0.448
	1996	0.009	1994	0.578	0.539
	1999	0.127	1997	0.551	0.738
	2003	<0.001	1998	0.082	0.883
	2006	0.027	2003	0.846	0.642
	1993–06	0.006	1992–03	0.819	0.684
Effect of fertilizing thinned plots	1993	0.543	1992	<0.001	<0.001
	1996	0.871	1994	0.002	0.006
	1999	0.886	1997	<0.001	<0.001
	2003	0.019	1998	<0.001	<0.001
	2006	0.063	2003	0.036	0.760
	1993–06	0.406	1992–03	<0.001	<0.001

(1996), no differences were found between burned and unburned plots (Table 6). The repeated burn in 2002 reduced shrub cover again to well under 10%. Shrubs, unlike trees, were generally unaffected by fertilizing.

Species composition was dominated by two N-fixing shrubs, bitterbrush at the drier sites (Sugar Cast and East Fort Rock) and snowbrush at the wetter site (Swede Ridge). Other species, occupying less than 5% of the combined cover, included manzanita and rabbitbrush (*Haplopappus* sp.). No major change in shrub composition was found in response to the various treatment regimes. Also, there was no main effect of site on shrub cover ($P = 0.827$) despite site differences in productivity and dominant shrub.

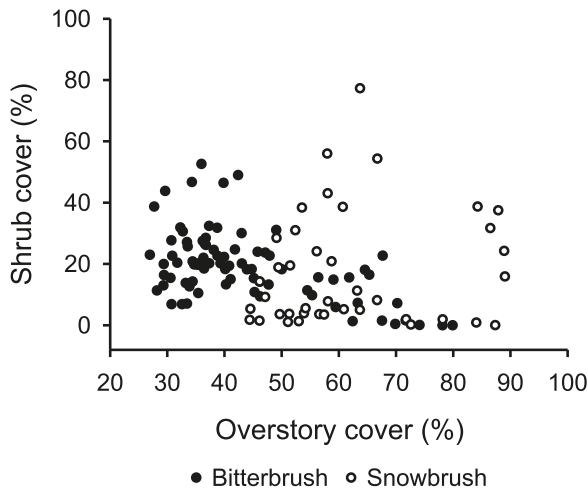
Total shrub cover was significantly related to ponderosa pine cover ($P < 0.001$). However, this relationship was not strong, accounting for only a small part of the variation in the shrub cover data ($r^2 = 0.13$; Fig. 6). Separate regression analyses resulted in a slightly improved prediction for bitterbrush cover ($r^2 = 0.36$) but not for snowbrush cover ($r^2 = 0.08$). Including other site characteristics (time since treatment, pretreatment shrub cover, annual precipitation) along with overstory cover in multiple regression analysis still explained less than 50% of the variation in shrub cover ($r^2 = 0.442$).

Fig. 5. Total shrub cover between 1993 and 2006. Plots were thinned in 1989, burned in 1991 and 2002, and fertilized in 1991 and 1996. Broken line represents pretreatment shrub cover in 1988. Error bars are standard error.

Herbaceous biomass and diversity

Herbaceous biomass was low throughout the study whether plots were thinned, burned, or thinned and burned. Only when fertilizers were added was there a sizable increase in plant production (Fig. 7, Table 6). The fertilizer response was significant throughout the study for graminoids,

Fig. 6. Relationship between shrub cover and ponderosa pine cover on unburned plots.



although peak production was reached in 1997 and then declined 10-fold by 2003. Herbaceous production in thinned stands between 1992 and 2003 averaged $120 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ when fertilized compared with only $16 \text{ kg} \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$ without fertilizing.

Graminoids accounted for 74% of the total herbaceous biomass between 1992 and 2003. No effect of thinning, slash removal, burning, fertilizing, or their interactions on the graminoid:forb ratio ($\sim 3:1$) was detected. Four graminoid species (western needlegrass, bottlebrush squirreltail, Idaho fescue, Ross' sedge) accounted for 93% of the graminoid biomass, with their relative contribution varying between sites. Their dominance was consistent across all treatments, and each of the four species responded positively to fertilization. There was a significant site effect on herbaceous production ($P < 0.001$), as the most productive site (Swede Ridge) had the lowest biomass of both graminoids and forbs presumably because of its higher stand density and overstory cover.

Species richness and diversity were low at the three study sites in both 1992 and 1994. Total species on a plot basis averaged $7.3 (\pm 0.3)$ in 1992 and $6.9 (\pm 0.3)$ in 1994 (Fig. 8). No effect of thinning or slash-removal treatment was found. Burning reduced the species richness of thinned plots in 1994 only ($P = 0.021$), while fertilizer additions significantly increased species richness in both years ($P = 0.003$). No effect of burning on species richness was found for unthinned plots ($P = 0.680$). Species diversity (Simpson's index) was unaffected by any combination of treatments in either measurement year. Finally, no invasive herbaceous species were detected in 1992, 1994, or 2003.

Discussion

We examined the effects of thinning and burning for a sufficient length of time to permit numerous remeasurements, retreatment of understory fuels, and a clear view of the temporal pattern of vegetation development. At question was whether thinning or burning alone could effectively initiate the restoration of ponderosa pine forests in central Oregon. Alternatively, were both practices required to approximate presettlement conditions? By including several slash-removal treatments that produced a gradient of surface

Fig. 7. Effects of thinning, burning, and fertilizing on total graminoid and forb biomass between 1992 and 2003. Plots were thinned in 1989, burned in 1991 and 2002, and fertilized in 1991 and 1996. Error bars are standard errors.

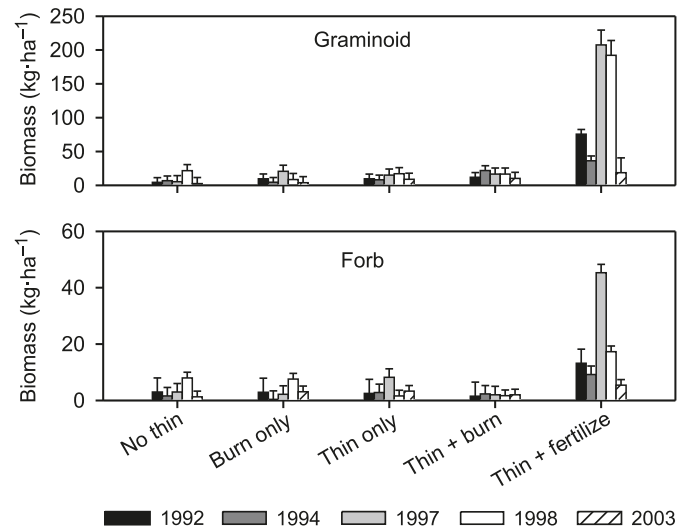
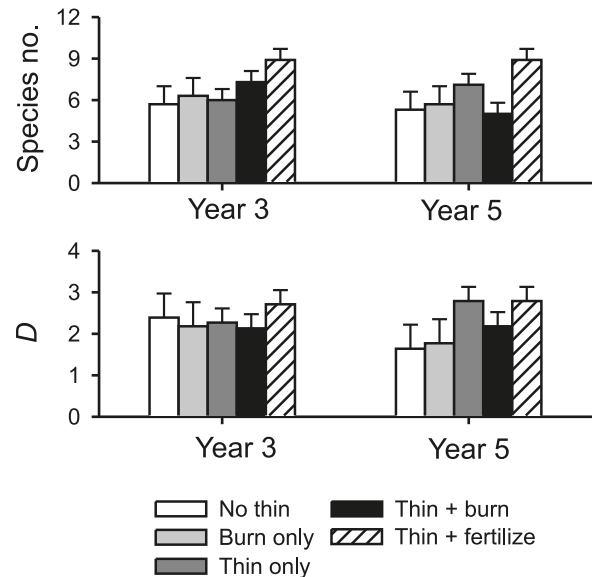


Fig. 8. Species number (richness) and Simpson's diversity index (D) for herbaceous plants in 1992 and 1994, the third and fifth year after thinning.



residues, we were also able to determine the effect of residue management on site productivity.

Thinning alone achieved many of the desired ecological characteristics for restoration of these second-growth ponderosa pine forests. In thinned stands, stand density and basal area were lowered by about 50% to predetermined levels designed to enhance tree growth and reduce crown fire and insect risk; tree vigor was increased as shown by a two-fold improvement in diameter increment compared with unthinned stands; tree mortality from bark beetle attack was not evident (unlike unthinned stands); and nitrogen-fixing shrubs, including an important wildlife browse species (bitterbrush), increased in cover compared with unthinned plots. Many of the basic principles for mitigating fire hazard in

Fig. 9. Typical site conditions in early summer 2006 for burn only (A), thin only (B), and thin plus repeated burn (C). Thin plus fertilize (D) is from 1997, the peak year for understory response to added nutrients.



dry forests (Agee and Skinner 2005) were also met by thinning alone. For example, surface fuel mass was low using whole-tree harvesting; understory trees were removed by thinning from below, resulting in average height to green crown of more than 5 m; and only fire-resistant trees (ponderosa pine) were present. Further, few differences in tree growth dynamics or herbaceous biomass and diversity were detected in thinned stands, whether burned or not. The effects of thinning on soil chemical and biological properties (pH, C, N, C:N, P, microbial biomass, litter decay, wood decay) were also benign at our study sites (Busse and Riegel 2005; Busse et al. 2006).

Despite its overall effectiveness, there were several conditions of fire-adapted systems that thinning failed to meet. These included exposure of bare mineral soil for seedling establishment, reduction of shrub cover, rapid release of available nutrients from forest floor organics, stimulation of fire-adapted herbaceous vegetation, and a more random spatial pattern of overstory trees (clumping and open spaces). However, with the exception of the response of shrubs, these conditions appeared of little significance, since the differences between thin-only and thin plus burn treatments were nominal. For example, bare mineral soil averaged 11% in the first year after thinning compared with 16% immediately after the second cycle of burning (data not shown). Herbaceous biomass was exceedingly low in thinned stands, with or without fire. Instead, herbaceous biomass responded solely to the addition of fertilizers, indicating that the release of available nutrients during repeated burning was in-

consequential and that bare mineral soil was not required for seedbed preparation. Also, most trees were uniformly spaced at the study sites, whether repeatedly burned or not, as was the preferred prescription in the late 1980s. Thinning prescriptions that encourage irregular spacing and random, small openings or clumpiness are an option that many silviculturists may pursue. Even still, the results of our study are likely applicable to the majority of the area in such thinning units.

In effect, shrub cover defined the vegetation difference between thinned plots and thinned plus burned plots (Fig. 9). Shrub cover was twice as great in the absence of fire between 1993 and 2006 (20% versus 11%, respectively). More importantly, the temporal pattern of post-treatment recovery differed between treatments. Without burning, shrub cover was reduced in half by the mechanical disturbance of the thinning operation and then increased steadily to pre-treatment levels by the end of the experiment. With burning, shrub cover was severely reduced and then increased more rapidly to match the coverage on unburned plots within 5–8 years after burning. Thus, if the abatement of shrubs is a priority management objective, then retreatment will be required after a short respite of 5–10 years following initial burning to maintain low shrub coverage. Alternatively, leaving shrubs untreated had no detrimental effect on tree or herbaceous production, and their contribution to crown-fire spread in this stand type may be limited (Ritchie et al. 2007; Busse and Riegel 2009). Our findings suggest that thinned stands may not require follow-up burning in these nutrient-poor and understory-depauperate forests.

Pretreatment stand conditions help explain the finding that forest vegetation was fairly similar between thinned and thinned plus burned treatments. These were densely stocked, even-aged stands that developed with no prescribed fire or wildfire activity following clear-cut harvesting in the 1930s. Tree diameters and tree spacing were fairly uniform, as few saplings or large trees remained following precommercial thinning operations in the early 1960s. The stands remained heavily stocked even after the initial thinning (400–800 trees per hectare), which restricted the recruitment of pine seedlings and saplings as potential ladder fuel. Thinning in 1989 successfully eliminated any remaining small or damaged trees, maintained the uniformity of tree diameters, decreased stand density and basal area, and slightly increased the height to green crown. The resulting simple, even-aged stand structure differs considerably from the ponderosa pine stands in northeastern Oregon studied by Youngblood et al. (2006), who found prescribed burning was required following thinning to adequately reduce the high numbers of ponderosa pine and Douglas-fir (*Pseudotsuga menziesii*) seedlings. Our results do agree with those of Busse et al. (2000), who found only subtle differences in forest vegetation between thinned versus thinned plus burned plots across an extensive area of ponderosa pine forests in southcentral Oregon.

Forest conditions suffered between 1989 and 2006 without silvicultural treatment. Tree mortality from mountain pine beetle averaged 15% on untreated plots, snags and downed wood from killed trees were expected to rise, tree vigor (diameter growth) was 47% lower compared with thinned stands, shrub cover slowly declined, and herbaceous plant production was nominal. These results come as little surprise, as the effect of thinning young pine stands is well established in central Oregon and throughout the west (Mowat 1953; Cochran and Barrett 1999; Oliver 2005). In fact, the thinning prescription of $14 \text{ m}^2 \cdot (\text{ha basal area})^{-1}$ was based on standards established by these earlier studies for reducing bark beetle attack, increasing diameter and volume growth, and extending the stand rotation age.

The gradual reduction in shrub cover found in untreated plots was also anticipated, since the two dominant species, bitterbrush and snowbrush, slowly regress when shaded (Busse et al. 1996). Interestingly, we were not able to identify a specific range of overstory cover that led to the reductions in shrub cover. Less than 50% of the variation in shrub cover was predicted by an assortment of site factors (overstory canopy cover, pretreatment shrub cover, years since thinning, annual precipitation). Thus, the use overstory canopy cover as a predictor of shrub response was unsuccessful at these sites. We attribute this in part to the limited number of unthinned plots in the study. Also, shrub cover was likely the wrong dependent variable for detecting a relationship between overstory and understory vegetation, since it provides no measure of shrub vigor.

In contrast to thinning, fire alone provided few ecological benefits to vegetation structure, growth, or diversity. Although tree density declined 20% with repeated burning, the mortality was isolated in small pockets generally less than 15 m in diameter, leaving the remaining stand intact and densely stocked (Fig. 9A). As a result, tree diameter growth did not differ from untreated plots, shrub cover was essentially eliminated by the initial burns, and herbaceous production remained conspicuously low. These results are

in general agreement with Youngblood et al. 2006, showing that fire alone is a less-than-effective means for ponderosa pine forest restoration. Importantly, our burn prescriptions were held within preferred guidelines for central Oregon for balancing effective fuel reduction with fire safety. Although selecting more aggressive prescriptions to increase tree mortality was possible, fire specialists on the Deschutes National Forest were reluctant to alter their prescriptions in favor of greater fire intensity given the proximity of the study sites to an urban center and the potential for loss of containment. In contrast to our findings, others have used fire alone to successfully eliminate excessive in-growth of tree seedlings and saplings and to rejuvenate understory vegetation in old-growth ponderosa pine stands in central Oregon (Youngblood and Riegel 2000) and in mesic, mixed-conifer forests in western California (Stephenson 1999; North et al. 2007).

In combination, thinning and repeated burning modified the understory by reducing shrub cover (Fig. 9C). Perhaps surprisingly, herbaceous plants did not fill in the niche left by the loss of shrubs, as graminoid and forb production was similar whether thinned plots were burned or not. This finding differs from the traditional concept that fire stimulates plant production (Biswell 1989; Turner et al. 2003) and from recent studies showing slightly greater cover of herbaceous plants when thinned stands are burned (Metlen et al. 2004; Metlen and Fiedler 2006; Youngblood et al. 2006). Evidently, the dry climate, infertile soils, and history of maintaining heavily stocked stands results in a depauperate understory unable to respond to burning and underscores the fact that a positive herbaceous response to burning is not universal (Keeling et al. 2006). In contrast to the herbaceous community, shrubs recovered rapidly after the first entry of fire, which is in agreement with previous fire-effects studies conducted in central Oregon (Martin 1983; Ruha et al. 1996). However, shrub density was substantially reduced at our sites following the second burn (Busse and Riegel 2009), indicating that reburning after 11 years was an effective tool for limiting the presence of shrubs.

Thinning and burning also resulted in few changes in stand structure or tree vigor compared with thin-only plots. This differs from previous central Oregon studies that reported short-term, modest reductions in ponderosa pine growth due to prescribed burning (Landsberg 1993; Busse et al. 2000). Potential mechanisms for reduced growth, such as fine-root mortality, crown scorch, cambial damage, or nutrient loss, were not elucidated in these studies, thus, making it difficult to account for the differences in our results. In our case, tree stress from insect defoliation may have masked any detrimental effects of burning. However, no decline in growth increment was detected following the low-severity repeated burns in 2002, which was well after the Pandora moth infestation had subsided. Therefore, we suggest that low- to moderate-intensity burning in central Oregon does not have a major effect on tree vigor, which differs from findings in other western forests (Fajardo et al. 2007).

Unlike burning, fertilizing with N, P, and S led to sizeable increases in vegetation growth (Fig. 9D). Trees, graminoids, and forbs (but not shrubs) responded strongly between 1991 and 2001. Tree diameter increment was 60% higher when fertilized, and total herbaceous biomass was 3–20 times greater. However, treatment longevity did not extend beyond 5 years after the final fertilizer application in 1996, as differ-

ences between fertilized and unfertilized plots were insignificant in the last measurement period. The unresponsiveness of bitterbrush and snowbrush to fertilizing may have resulted from their rooting architecture, which has proportionally less fine roots than grasses or trees (M.D. Busse, personal observation, 2008), or from the fact that they are N-fixing plants. Busse (2000) found that both shrub species supplied 85% or more of their needed biomass N via symbiotic N fixation, indicating that they are not N limited whether fertilized or not.

Our primary objective in applying fertilizers was to provide a comparative metric for assessing vegetation growth and nutrient release on burned plots. Does the nutrient flush from burning affect plant growth similar to how fertilizing would affect it? Or alternatively, will burning result in either no change in plant production or a decline in production because of a reduction in the total soil N pool? Our results showed that tree growth and herbaceous production were responsive to fertilizer additions, whereas burning provided an inconsequential release of nutrients. In fact, plant-available N in the surface mineral soil was similar between burned and unburned soils in the third growing season after burning or was about fourfold lower than values found on fertilized plots (M.D. Busse, unpublished).

Hart et al. (2005) hypothesized that the nutrient flush often observed following prescribed fire is a modern-day phenomenon and that it was less common in presettlement pine forests. They suggested that frequent burning in presettlement forests of the southwest United States favored the presence of herbaceous plants and led to more rapid nutrient turnover of easily degraded litter than is found currently in pine-litter-dominated forests. Unlike the results of Hart et al. (2005), we found that the reintroduction of repeated fire resulted in neither a flush of nutrients nor a shift in understory composition favoring herbaceous species. Whether this resulted from incomplete duff consumption in our study (<50%) or from possible differences in forest floor nutrient capital between studies is unclear.

Relatedly, we selected the slash-removal treatments (WT, BO, NR) with the objective of testing whether organic residues are important to site productivity. The treatments provided a full range of potential residue levels after thinning, from virtually none following whole-tree removal to maximum retention following thin plus no removal. The intent was to test, indirectly, whether thinning residues offer an advantage to plants by providing a source of slow-release nutrients, carbon as a primer for microbial activity and nutrient turnover, or moderation of microclimate temperature or evaporative water loss. Alternatively, do thinning residues immobilize essential nutrients or serve as a physical barrier that impedes seedling establishment? The results clearly showed that thinning residues did not modify the site potential. No changes in tree growth, insect damage, shrub cover, herbaceous biomass, species richness, or diversity were found. Apparently, there is little need to retain thinning residues at these sites from a 20 year plant productivity perspective. The practice of removing entire trees is also preferred from a fire-risk perspective (Agee and Skinner 2005) and is an important option for those considering biomass harvesting for energy production. We point out, however, that our findings are applicable only to dry ponderosa pine forests in central Oregon, and note that studies of slash retention conducted in the cold-moist climate of Scandinavia (Jacobson et

al. 2000) and at warm-humid sites in the southeast United States (Fleming et al. 2006) have found moderate declines in site productivity following whole-tree removal. In fact, such results have led to policy guidelines of compensatory fertilization after whole-tree harvesting in boreal forests of Sweden (www.svo.se/forlag/meddelande/1545.pdf).

The slash-removal treatments also provided a mechanism for exploring the effects of soil compaction on vegetation growth. Specifically, the NR treatment (thinned, no machinery) provided an “undisturbed” comparison to the WT treatment (thinned, grapple skidded randomly across plots). While concern for any detrimental effects of soil compaction associated with harvesting has been expressed by many (Gomez et al. 2002; Agee and Skinner 2005; Parker 2007), we found no differences in tree, shrub, or herbaceous growth between NR and WT treatments during the study, indicating that soil compaction was a minor issue following thinning and skidding. This observation extends the initial findings of Parker et al. (2007), who found no differences in soil strength or tree growth between the two treatments from 1991 to 1996. They did detect lower growth rates of individual trees in localized pockets of compacted soil, but the declines were insufficient to extrapolate to the plot scale. Moghaddas and Stephens (2008) also found limited effects of thinning on soil strength in mixed-conifer stands in the Sierra Nevada. Although far from definitive, these collective findings indicate that mechanical thinning may not alter long-term soil productivity.

Finally, regarding the Pandora moth infestation, severe defoliation led to a measurable decline in tree growth between 1991 and 1996. Diameter and volume increment growth was about 50% lower during this period compared with the succeeding 5 year period when moth activity was undetected. Perhaps more interesting was the observation that defoliation was comparable across all treatments, which included stand densities from 250 to 800 trees per hectare, fertilized and unfertilized trees, and burning in 1991 when the pupae (resting stage) were near the surface of the mineral soil. A similar growth decline was noted in other central Oregon forests during this infestation (Speer 1997; Cochran 1998), although the long-term implications for stand development and forest health from such outbreaks are not considered acute (Speer 1997). In this respect, we found no indication that tree growth recovery following defoliation varied among the assorted treatments.

Conclusions

Ponderosa pine forests on public lands in central Oregon were typically composed of young, densely stocked, even-aged stands with moderate-to-severe risk of wildfire and insect damage by the latter half of the twentieth century. The results from our study showed that the thinning treatment, when applied across a range of site productivities, was a suitable practice for restoring several ecological characteristics of presettlement pine forests and served as an important first step for restoration of these sites. Following thinning, decisions about reintroducing fire or alternative fuel-reduction treatments can be made on a site-specific basis with knowledge of fuel loads, ladder fuel accumulation, wildlife habitat needs, soil fertility, and public concern. Other findings included the following:

- Low- to moderate-severity prescribed burning following thinning resulted in a short-term reduction in shrub cover. However, repeated burning was required to curb the rapid regrowth of shrubs.
- Prescribed burning did not change herbaceous plants biomass or diversity. In contrast, fertilizing resulted in a large, short-term increase in herbaceous biomass. This result confirmed the nutrient-poor status of central Oregon soils and revealed the inability of the repeated burns to release significant amounts of plant-available nutrients.
- Prescribed burning was not required after whole-tree harvesting from a fire-risk standpoint. Live- and dead-fuel loading was low after this treatment.
- Retaining thinning residues on site was not essential for site productivity. No differences in vegetation growth, structure, or diversity were found among the organic residue treatments.
- Fire alone was an ineffective means for reducing stand density. As a consequence, few changes in tree vigor, herbaceous production, or species composition were found relative to untreated stands.
- Defoliation by a Pandora moth outbreak resulted in reduced tree growth at the onset of the study. The effect was short lived and nondiscriminatory across treatments.

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