Forest Ecology and Management 258 (2009) 712–721

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Thinning and prescribed fire effects on overstory tree and snag structure in dry coniferous forests of the interior Pacific Northwest

Richy J. Harrod [a,*], David W. Peterson [b], Nicholas A. Povak [b], Erich K. Dodson [b]

[a] USDA Forest Service, Okanogan-Wenatchee National Forest, 215 Melody Lane, Wenatchee, WA 98801, United States
[b] USDA Forest Service, Wenatchee Forestry Sciences Laboratory, Wenatchee, WA, United States

ARTICLE INFO

Article history:
Received 10 February 2009
Received in revised form 30 April 2009
Accepted 10 May 2009

Keywords:
Restoration
Fuels reduction
Ponderosa pine
Thinning
Prescribed burning
Forest structure

ABSTRACT

Forest thinning and prescribed fires are practices used by managers to address concerns over ecosystem degradation and severe wildland fire potential in dry forests. There is some debate, however, about treatment effectiveness in meeting management objectives as well as their ecological consequences. The purpose of this study was to assess changes to forest stand structure following thinning and prescribed fire treatments, alone and combined, in the eastern Cascade Mountains of Washington State. Treatments were applied to 12 management units, with each treatment combination replicated three times (including untreated controls). Thinning modified forest structure by reducing overall tree density by >60% and canopy bulk density by 50%, and increased canopy base height by ~4 m, thereby reducing susceptibility to crown fire. The prescribed fire treatment, conversely, did not appreciably reduce tree density or canopy fuel loading, but was effective at increasing the density of standing dead trees, particularly when combined with thinning (37 snags/ha increase). Prescribed fire effects were more pronounced when used in combination with thinning. Thinning was more reliable for altering stand structure, but spring burning was lower in intensity and coverage than desired and may have led to results that downplay the efficacy of fire to meet forest restoration goals.

Published by Elsevier B.V.

1. Introduction

Fire exclusion, livestock grazing, and logging practices have combined to alter forest structure in dry coniferous forests over the past century, generally increasing stand density and basal area, altering forest tree species composition, and increasing densities of smaller trees that can serve as ladder fuels (Hessburg et al., 2005; Peterson et al., 2005). This, in turn, has increased forest susceptibility to severe wildfire (Dahms and Geils, 1997; Peterson et al., 2005), and risks of widespread insect and disease outbreaks (Hessburg et al., 2005). Changes in stand structure, particularly size structure and snag densities, also have important implications for wildlife populations (Lehmkuhl et al., 2006a; Gaines et al., 2007; Lyons et al., 2008), soils (DeLuca and Sala, 2006), and understory vegetation (Covington et al., 1997; Smith and Arno, 1999).

To increase forest resiliency to fire and insect disturbances, forest managers are increasingly using thinning and prescribed fire to modify overstory structure and reduce surface and canopy fuels in dry coniferous forests (Peterson et al., 2005). Recent multisite comparisons have broadly described fuel reduction treatments effects (Schwilk et al., 2009; Stephens et al., 2009), but region specific data is still lacking and our study adds to and expands the range of forest types studied. Such information will be important for managers to make knowledgeable decisions when developing stand and landscape level strategies (Collins et al., 2007). The purpose of this study was to assess the effects of thinning and prescribed fire treatments, applied singly and in combination, on several components of overstory stand structure in dry mixed-conifer forests of north-central Washington State, USA.

Mechanical thinning can alter stand structure, disrupt vertical continuity of canopy fuels, and increase soil resource availability by removing small to moderate sized trees (Agee and Skinner, 2005; Peterson et al., 2005). Mechanical thinning allows managers to control the species composition, size structure, and vigor of residual trees following treatment. Thinning can also provide economic benefits if merchantable trees are removed. However, high treatment costs, concerns about slash increasing surface fuels, aesthetics (e.g., stumps), lack of road access, and soil erosion potential often limit the utility and acceptability of thinning as a fuel reduction or forest restoration treatment (Pollet and Omi, 2002; McIver and Ottmar, 2007). Therefore, reliance on thinning alone to alter forest structure and increase resilience to disturbance is not possible in all forests.

Prescribed fire treatments can reduce surface fuels, kill tree seedlings and saplings, reduce tree density, raise canopy base

* Corresponding author. Tel.: +1 509 664 9331; fax: +1 509 664 9284.
E-mail addresses: rharrod@fs.fed.us (R.J. Harrod), davepeterson@fs.fed.us (D.W. Peterson), npovak@fs.fed.us (N.A. Povak), edodson@fs.fed.us (E.K. Dodson).

0378-1127/$ – see front matter. Published by Elsevier B.V.
doi:10.1016/j.foreco.2009.05.011

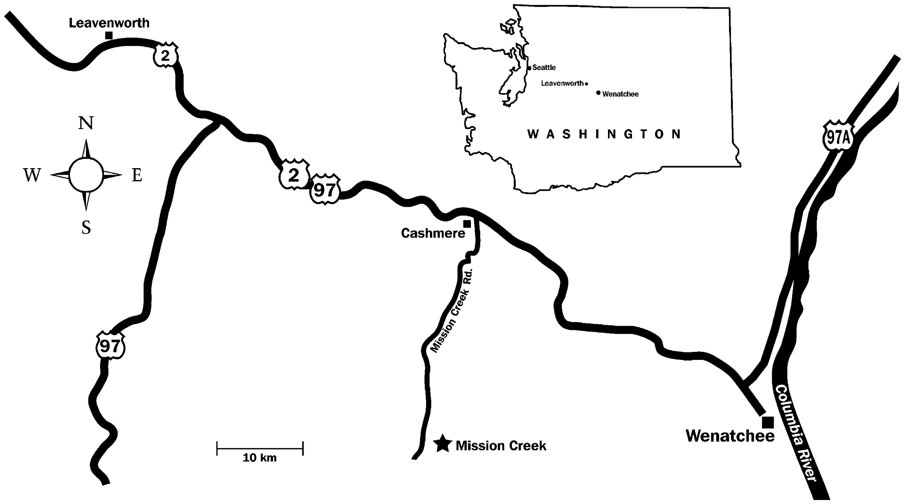

Fig. 1. Study area location.

height, and favor fire-resistant tree species (Agee and Skinner, 2005; Peterson et al., 2005; Stephens and Moghaddas, 2005a). Prescribed fire treatments also help restore fire as an ecological process, which may provide secondary benefits such as seed germination (Harrod and Halpern, 2005a), reduced competition (Harrod and Halpern, 2005b), and increased species richness (Dodson et al., 2008). However, prescribed fire effects can be highly variable, depending on fuels, stand structure, and fire weather. In some cases, early season, prescribed fire alone has been insufficient for modifying overstory structure and achieving restoration objectives (Thomas and Agee, 1986; Knapp et al., 2005; Youngblood et al., 2006). Prescribed fires can also cause undesirable tree mortality, either through direct fire effects or secondary effects, such as increased bark beetle activity (McHugh et al., 2003). For example, Harrington and Sackett (1990) reported that a single prescribed fire caused 35% mortality in old-growth ponderosa pines in northern Arizona.

Finally, thinning and prescribed fire treatments may complement each other when applied in combination by modifying different components of forest structure and fuels. Thinning may be most effective for removing larger trees that are fire resistant but marketable, while prescribed fire may be a more efficient method for killing small trees and creating snags for wildlife habitat. Thinning may also increase the effectiveness of prescribed fire by increasing surface fuels in areas where fuels are lacking. However, the additional costs associated with combining treatments may be hard to justify if management objectives can be met with a single treatment.

The objective of this study was to assess the effects of thinning and prescribed fire treatments, applied alone and in combination, on stand structural characteristics in dry mixed-conifer forests of the interior Pacific Northwest region. Treatments were applied alone and in combination using a factorial experimental design. Structural characteristics of interest included overstory tree density and basal area, snag density and basal area, tree and snag species composition and size structure, canopy base height, and canopy bulk density. For all response variables, we asked whether thinning and prescribed fire treatments alone significantly altered stand structure and whether the effects of thinning and prescribed fire together exceeded that of the individual treatments. This study was part of the U.S. Fire and Fire Surrogates (FFS) research network, which seeks to evaluate prescribed fire and fire-surrogate treatment effects on stand-replacing wildfire risk as well as on a broad array of ecosystem properties and processes in fire-prone forest types throughout the continental United States (Youngblood et al., 2005).

2. Methods

2.1. Study area

The Mission Creek study area is located on the Okanogan-Wenatchee National Forest in the eastern Cascade Mountains of central Washington State (USA; Fig. 1). Forests within the study area are dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*), with some grand fir (*Abies grandis*) and western larch (*Larix occidentalis*) occurring on more mesic sites. Common understory species include *Carex geyeri*, *Calamagrostis rubescens*, *Symphoricarpos albus*, *Spiraea betulifolia* and *Rosa* spp. (*Rosa gymnocarpa*, *R. nutkana* and *R. woodsii*). Soils are stony, sandy loams, and slopes vary from about 15 to 65% (Table 1).

The study area was extensively grazed around the turn of the century to about 1940; extensive logging occurred during the late 1920s and early 1930s (USDA, 1995). Fire suppression became effective in the 1930s and has continued to the present day (USDA, 1995). As a result of these anthropogenic influences, much of the area is now covered by dense stands of 80–140 year old trees (Harrod et al., 1999).

The climate features warm, dry summers (with occasional thunderstorms) and cool, wet winters. Representative weather records for the Mission Creek watershed are from the town of Plain, 32 km north of the study area. Plain receives about 68 cm of precipitation annually, with about 12 cm (18%) falling between April and September, and much of the winter precipitation falling as snow. Monthly mean temperatures range from −3 °C in January to 18 °C in July (Western Regional Climate Center, Plain, www.wrcc.dri.edu).

2.2. Treatments

In 1999, we chose 12 management units (about 10 ha each) for study from the Mission Creek watershed and several smaller adjacent watersheds (selection of sites is described in detail in Dodson et al., 2008). We assigned each unit to one of four combinations of thinning and prescribed fire treatments: (1) no treatment (control), (2) prescribed fire alone (burn-only), (3) thinning alone (thin-only), or (4) thinning followed by prescribed fire (thin-burn). Each treatment combination was replicated on three units. We assigned thinning treatments randomly to 6 of the 12 units. We assigned prescribed fire treatments using a more constrained random process due to concerns about prescribed fire

Table 1
Treatment unit descriptions for the Mission Creek study site.

Treatment	Unit	Topography		Tree density			Tree basal area			Quadratic mean diameter (cm)
		Elevation, mean (m)	Slope, mean (%)	Total (trees/ha)	PSME (%)	PIPO (%)	Total (m^2/ha)	PSME (%)	PIPO (%)	
Control	Crow 3	747	38	450	43	57	32.3	50	50	30.7
Control	Sand 19	780	43	480	89	11	33.8	92	8	31.0
Control	Sand 2	683	58	675	97	3	33.3	92	8	27.1
Burn-only	Pendleton	841	16	323	4	96	22.6	8	92	30.4
Burn-only	Poison	768	40	430	50	49	29.9	69	31	30.6
Burn-only	Spromberg	848	57	546	73	27	42.3	71	29	31.6
Thin-only	Crow 1	738	21	478	3	97	28.9	9	91	28.1
Thin-only	Ruby	975	43	447	72	26	38.2	65	35	33.8
Thin-only	Slawson	838	35	753	90	10	35.0	92	8	24.7
Thin-burn	Camas	1097	43	495	52	19	33.6	68	20	29.7
Thin-burn	Crow 6	718	26	482	4	96	29.0	8	92	27.7
Thin-burn	Tripp	765	67	785	98	2	35.3	91	9	24.3

management in some units, and later switched prescribed fire treatment assignments among two pairs of units after weather and fuel conditions postponed treatment in two units (see below).

We designed the thinning treatment to (1) reduce stand basal area to 10–14 m^2/ha, (2) reduce stand densities to levels that minimize risks from high severity wildfire, (3) retain most large trees, and (4) preserve or promote structural heterogeneity within management units. Leave trees were aggregated spatially, consistent with historical reconstructions of similar forest stands in the area (Harrod et al., 1999). These treatments are similar to those applied to non-study thinning units. Harvest crews yarded merchantable trees by helicopter, leaving tops and branches on-site. Snags were felled due to safety concerns for workers who could be injured by snags falling because of rotor turbulence. Hand crews later cut smaller non-merchantable trees and then lopped and scattered slash from both the commercial thinning and hand-felling. Thinning treatments were started in fall of 2002 and completed by spring 2003.

Prescribed fire treatments were initiated in spring of 2004. Fire crews ignited prescribed fires on four units by hand and from helicopters from late April to early May, which is the same time period and ignition technique used in non-study units. Vegetation was actively growing at that time, so live fuel moisture was relatively high, fire severity was low, and coverage was patchy. Within burned units, fires charred only 23–51% of the soil surface, and fires failed to meet most surface fuel reduction objectives (Agee and Lolley, 2006). Unfavorable fuel and weather conditions also caused two of the six scheduled fires to be postponed until spring of 2006, when they were conducted with the same methods, but under more normal spring prescribed burning conditions. These burns blackened 50–65% of the soil surface and scorched 80% of the overstory trees as compared to 30% in 2004.

Fire behavior varied somewhat among units and among burn years. Fire weather during prescribed fires was moderate with air temperatures of 13–23 °C, relative humidity of 29–42%, and wind speeds of 0–16 km hr^{-1}. Fuel moisture levels for fine woody fuels ($\leq$2.5 cm diameter) were similar (9–13%) for all fires. However, fuel moisture levels for larger woody fuels (2.5–7.5 cm diameter) were lower in 2006 (7%) than in 2004 (10–12%). Live vegetation moisture was also higher in 2004. Flame lengths ranged from 0.2 to 1.0 m in 2004 and 0.1–1.7 m in 2006. Burns were variable in intensity and generally patchy (Agee and Lolley, 2006).

2.3. Sampling procedures

We conducted preliminary reconnaissance of the treatment units to identify areas of continuous forest vegetation and to exclude areas dominated by non-forest vegetation from sampling. Within the forested areas of each unit, we randomly established six permanent sampling plots (20 m × 50 m) in 2000. Following completion of the prescribed fire treatments, however, crews noted that two permanent plots in burned units escaped fire because errors in fire-line construction prevented the plots from being burned. We therefore omitted data from these two plots for subsequent analyses (leaving two units with five plots each).

Within each sample plot, we tagged and surveyed all live and standing dead trees (>7.5 cm diameter at breast height), recording status (tree or snag), species, diameter at breast height (dbh), tree height, and height to crown base (vertical distance from tree base to where the lowest live limb connected at the tree bole). The study design called for sampling vegetation 1 year before and after treatment implementation. In practice, however, we sampled pre-treatment vegetation 1–2 years before thinning treatments began (2000–2001) and sampled post-treatment vegetation about 5 months after prescribed fire treatments were completed in 2004 and 2006. With the delays in applying the prescribed fire treatments, the total time between pre- and post-treatment sampling ranged between 3 and 5 years.

2.4. Data analysis

Our data analysis consisted of three basic steps. First, we tested for statistically significant differences in pre-treatment stand structural attributes among units assigned to different treatment combinations. We also used analysis of variance to examine the amount of variability in stand structural attributes within and between units prior to treatment (described in detail in Harrod et al., 2009). Second, we examined the influence of treatments, stem size and species on tree and snag dynamics, including losses due to thinning, tree mortality, and snag creation and loss. Finally, we assessed the net effects of treatments on stand structural attributes.

For each plot, sampling period, and structural component (trees or snags), we calculated three stand-level structural attributes, including stem density (stems per hectare), basal area (m^2/ha), and stem quadratic mean diameter (cm). We also calculated tree and snag densities by species and size class to better describe pre-treatment conditions and treatment effects on overstory species composition and size structure. We used three species categories: Douglas-fir, ponderosa pine, and grand fir/western larch (minor species combined into one category). We used four diameter size classes for analyzing size structure and size-related treatment responses: saplings (7.5–9.9 cm), small (10.0–24.9 cm), medium (25.0–39.9 cm), and large ($\geq$40 cm) trees and snags.

We also estimated two attributes of overstory canopy structure that can influence wildfire behavior: canopy base height and canopy bulk density. Canopy base height is the vertical distance from the ground to the lowest point in the canopy where bulk density is high enough to propagate fire into the canopy (0.011 kg m^{-3}; Scott and Reinhardt, 2001). Canopy bulk density is the crown mass per unit volume in a forest stand and is an estimate of available fuel in the canopy. We modeled these attributes based on tree data from the pre- and post-treatment tree surveys for each sample plot using the Crown Mass® software in the Fuels Management Analyst 3® (FMA) package (Fire Program Solutions, 2005).

Prior to analysis, we adopted a Type I error rate of 10% ($\alpha = 0.10$) rather than the traditional value of 5% for determining statistical significance of treatment effects due to concerns about low statistical power (high Type II error rates) caused by low levels of replication. We also felt that most managers would be inclined to accept a 10% chance of treatment failure (no true treatment effect) if the estimated treatment effect sizes were sufficiently high. Residuals for all models were evaluated (visually, with scatterplots etc.) to ensure compliance with model assumptions and no data transformations were needed for any analysis.

We analyzed treatment effects on stand structural attributes using a multilevel regression modeling approach (Snijders and Bosker, 1999; Raudenbush and Bryk, 2002) using the SAS statistical software (Littell et al., 2006). We included post-treatment structural attributes (e.g., stand density, canopy base height) estimated for each sample plot ($n = 70$) as model response variables. We included pre-treatment structural attribute values for each plot as a plot-level predictor variable (covariate) after preliminary analysis suggested that thinning effects were often proportional to pre-treatment conditions rather than constant (e.g., 50% of trees removed during thinning). Thinning and burning treatments were included in the analysis as independent categorical predictor variables (treatment applied or not) at the unit level. We included the unit identifier as a random variable to account for possible correlations of responses within units. For each response variable, we fit a full model with thinning and burning treatment effects, the pre-treatment covariate, and all possible interactions (including cross-level interactions). Non-significant ($P > 0.1$) terms were eliminated from the model using a backward elimination process, starting with interaction and random terms. Thinning, burning and their interaction were not removed regardless of their significance level to ensure a statistical test of the treatment effects using the complete experimental design. The multilevel model varied the degrees of freedom used in statistical inference so that the pre-treatment covariate and its interactions were tested for significance at the plot level ($n = 70$), while thinning and burning treatments were tested for significance at the unit level ($n = 12$).

To better understand net treatment effects on overstory tree and snag structural attributes, we used multilevel logistic regression methods to assess treatment effects on individual trees and snags, including probabilities of (1) trees and snags being cut during thinning, (2) trees dying and becoming snags, and (3) snags falling and becoming logs. We assigned each tagged tree and snag a pre-treatment status (tree or snag) and a post-treatment status (tree, snag, cut stump, or downed log). We then used the pre- and post-treatment status categories to create binary response variables (live tree cut or not, snag cut or not, standing live tree became snag or not, and snag fall or not) and to extract data subsets for assessing treatment effects on tree and snag transition probabilities. We modeled transition probabilities (e.g., tree to snag, tree to stump, snag to log) as a function of tree-level predictors, including species and size class, and the unit-level treatment variables, thinned (yes or no) and burned (yes or no), and their interactions. We included the unit identifier as a random variable to account for possible correlations of responses within units. We performed the multilevel logistic regression using the SAS statistical software (PROC GLIMMIX, Littell et al., 2006).

3. Results

3.1. Pre-treatment stand structure

Stand structure and composition varied considerably within and between treatment units prior to treatment application. Pre-treatment tree density (in forested areas of treatment units) averaged 529 stems/ha (stems > 10 cm dbh) overall (Fig. 2A), with unit means ranging from 323 to 785 stems/ha (Table 1) and plot estimates ranging from 190 to 1210 stems/ha. Stand basal area averaged 32.8 m^2/ha overall (Fig. 2B), with unit means ranging from 22.6 to 42.3 m^2/ha (Table 1) and plot estimates ranging from 21.1 to 45.6 m^2/ha. The quadratic mean tree diameter (calculated at the plot level) averaged 29.1 cm overall (Fig. 2C), with unit means ranging from 24.3 to 33.8 cm (Table 1) and plot estimates ranging from 14.3 to 51.3 cm. Douglas-fir and ponderosa pine were the two dominant tree species, together comprising 71–100% of tree stems and 88–100% of stand basal area at the unit level (Table 1). The relative importance of the two dominant species varied substantially among units, with Douglas-fir comprising 3–98% of tree stems and 8–92% of live tree basal area, with similar variability for ponderosa pine (Table 1). Despite (or perhaps because of) considerable variability in pre-treatment stand attributes, analysis of variance tests found no significant pre-treatment differences in tree density, basal area, or mean tree diameter among groups of units assigned to receive different thinning and prescribed burning treatments (Harrod et al., 2009).

Tree canopy attributes related to potential fire behavior also varied among units prior to treatment, but were not significantly different among treatment groups. Canopy base height averaged 4.1 m overall, with unit means ranging from 2.7 to 5.5 m and plot estimates ranging from 1.5 to 8.5 m. Canopy bulk density averaged 0.067 kg/m^3 overall, with unit means ranging from 0.055 to 0.084 kg/m^3 and plot estimates ranging from 0.025 to 0.084 kg/m^3.

Standing dead trees (snags) made up a smaller but important component of overstory stand structure. Pre-treatment snag density averaged 38 stems/ha across all units, with unit means ranging from 8 to 63 stems/ha (Fig. 2D). Snag basal area averaged 1.54 m^2/ha, with unit means ranging from 0.24 to 2.76 m^2/ha. Unlike most structural attributes, snag basal varied significantly among treatment groups, with the highest pre-treatment snag basal area found on units assigned to the thin-only treatment and the lowest found on units assigned to the thin-burn treatment combination. Differences in snag density among treatment groups were not statistically significant, however.

Prior to treatment, tree and snag densities varied among species and size classes. Most trees were in small (10–25 cm dbh) and medium (25–40 cm dbh) size classes, with Douglas-firs being somewhat more abundant than ponderosa pines (Fig. 3A). Large trees (≥40 cm dbh) contributed an average of 57 trees/ha, with Douglas-fir densities about twice that of ponderosa pine overall. Snag densities were highest in the sapling and small tree size classes (<25 cm dbh), with ponderosa pine snags more abundant than Douglas-fir snags. Medium and large snags (≥25 cm dbh) contributed less than 7 snags/ha, with ponderosa pine snags more than twice as abundant as Douglas-fir snags (Fig. 3B).

3.2. Treatment effects on tree and snag dynamics

The thinning treatments removed an overall average of 57% of live trees in thinned units. Thinning intensity varied by size class, with the highest proportions of stems cut in the small and

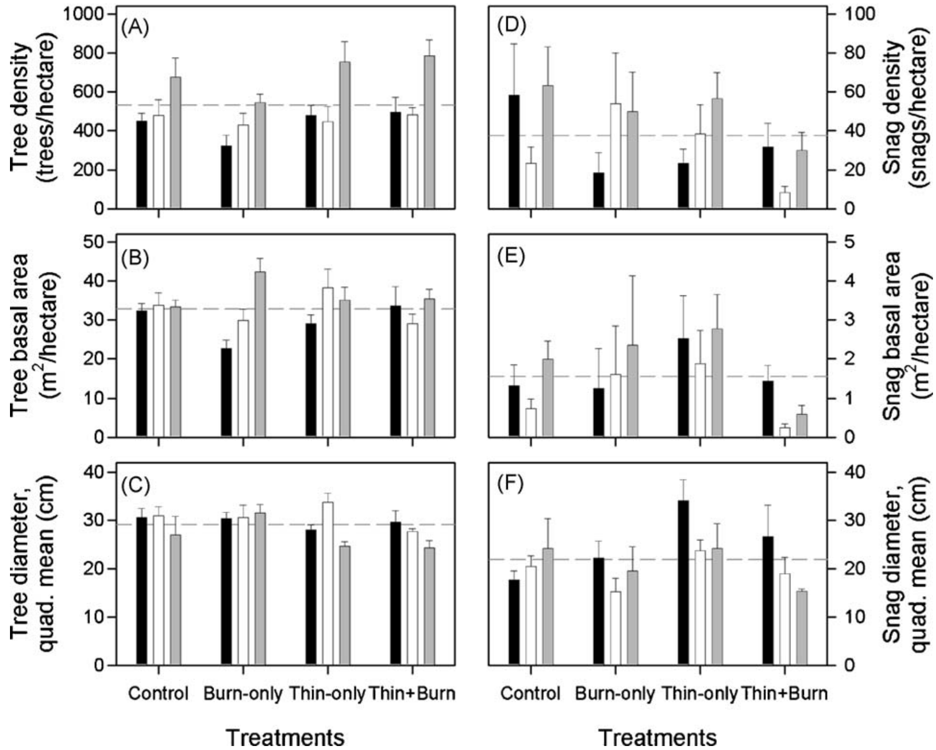

Fig. 2. Pre-treatment stand structural summaries at the unit level for trees (A, B, and C) and snags (D, E, and F), including stem densities (A and D), basal areas (B and E), and quadratic mean stem diameters (C and F). Bars represent unit-level mean values, with units grouped by assigned treatment combination. Error bars indicate standard errors within units. Pre-treatment means for each attribute are indicated by the horizontal dashed reference lines.

medium-size classes (Fig. 4A). Thinning proportions were similar for Douglas-fir, ponderosa pine, and other species, except in the sapling size class where a greater proportion of ponderosa pines were retained (Fig. 4A).

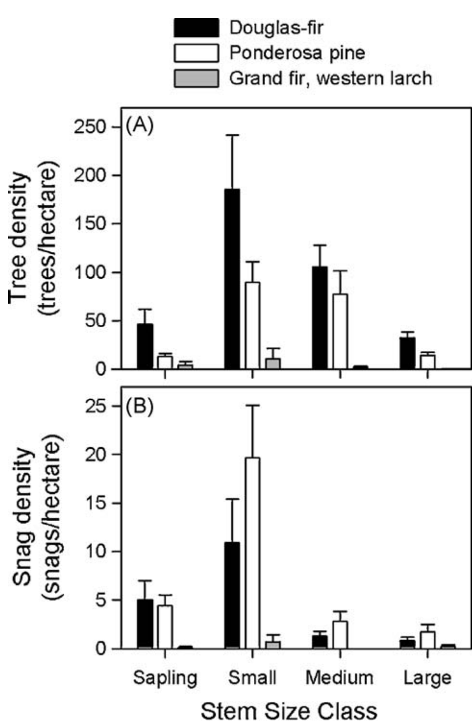

Fig. 3. Size and species distributions of trees (A) and snags (B) across all units prior to treatment. Stem diameter size classes include sapling (7.5–9.9 cm dbh), small (10.0–24.9 cm), medium (25.0–39.9 cm), and large (≥40 cm). Bar shading indicates species classes. Error bars indicate standard errors.

The thinning treatments also removed large numbers of snags. About 70% of snags were cut down during thinning operations. Proportions of snags cut declined with increasing snag diameter, from 78% in the sapling size class to 50% in the large size class

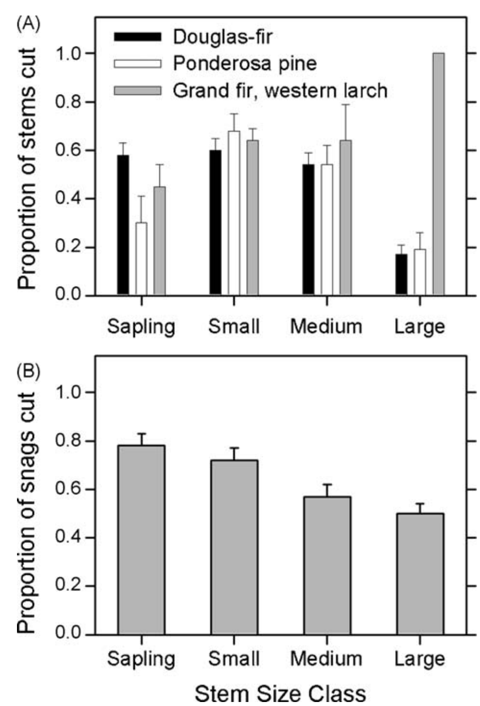

Fig. 4. Proportional reductions in stem densities caused by thinning based on multilevel logistic regression modeling. For trees (A), proportions of stems cut on thinned units are presented by species and size class. Thinning effects on snag densities (B) are presented by size class only. Error bars indicate standard errors of the estimates.

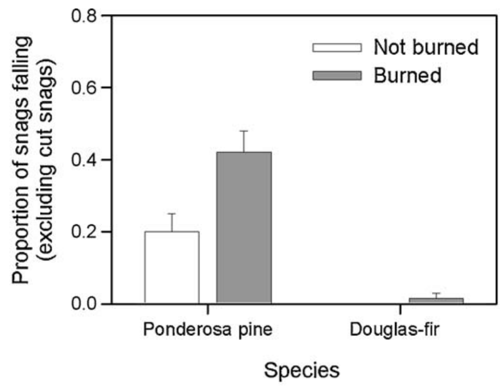

Fig. 5. Snag fall (loss) as a function of species and prescribed fire treatment, based on logistic regression modeling. Error bars indicate standard errors.

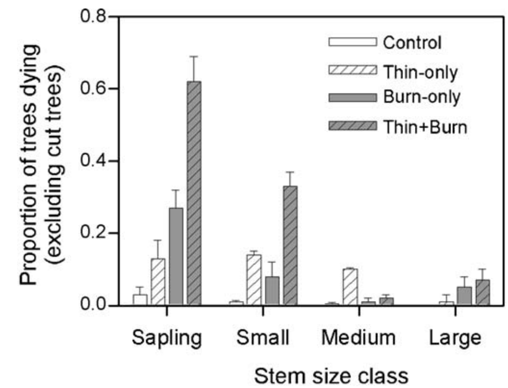

Fig. 6. Tree mortality from logistic regression modeling, by size class and treatment combination, for trees not cut during thinning operations. Error bars indicate standard errors of the estimates.

(Fig. 4B). Proportions of snags cut did not vary significantly among species, but variability among units was high.

Burning treatments further reduced snag populations by increasing the proportion that fell. Logistic regression models of snag fall showed that burning treatments (burn-only and thin-burn) significantly increased the odds of snag fall over those in control and thin-only units (Fig. 5). Ponderosa pine snags were significantly more likely to fall than were Douglas-fir snags (Fig. 5).

Thinning and burning treatments also generated new snags by killing trees. Logistic regression modeling of mortality in trees not cut during thinning operations showed that treatments (thinning and burning), tree size, and species all significantly influenced tree mortality. Tree mortality was very low (<1%) in units assigned to the control treatment. Thinning and burning treatments significantly increased tree mortality (snag creation), with observed overall percentages of 5% for the burn-only treatment, 7% for the thin-only treatment, and 18% for the thin-burn treatment. Mortality was higher overall for ponderosa pine (7%) than for Douglas-fir (3%), and higher for trees in the sapling (17%) and small (8%) size classes than in the medium (2%) and large (1%) size classes. Overall, mortality was highest for small trees in burned units (Fig. 6), but burning (with or without prior thinning) also killed 10–11% of large ponderosa pine trees (Fig. 6 and Table 2), thereby creating new large snags.

3.3. Treatment effects on stand structure

Multilevel modeling showed that thinning and burning treatments significantly influenced different components of overstory stand structure and that treatment effects (especially thinning effects) were often proportional to pre-treatment structural conditions. Thinning treatments significantly altered

Table 2
Mortality percentages during treatment period for ponderosa pine and Douglas-fir trees not cut during thinning operations. Percentages are given as sample sizes (*N*) by treatment combination, tree diameter class, and species. Diameter size classes are large (≥40.0 cm), medium (25.0–39.9 cm), small (10.0–24.9 cm), and sapling (7.5–9.9 cm).

Species	Size class	Control		Burn-only		Thin-only		Thin-burn	
		Mortality (%)	*N*	Mortality (%)	*N*	Mortality (%)	*N*	Mortality (%)	*N*
Ponderosa pine	Large	0.0	24	11.4	35	3.1	32	10.5	19
	Medium	0.0	66	0.6	157	10.2	59	2.2	46
	Small	0.9	107	14.5	173	18.4	49	51.6	31
	Sapling	5.6	18	40.9	44	16.7	6	78.6	14
Douglas-fir	Large	0.0	92	1.4	69	0.0	46	4.4	45
	Medium	0.4	250	1.7	115	9.1	99	2.3	86
	Small	0.7	423	3.0	132	10.5	153	19.7	183
	Sapling	2.2	92	16.7	30	10.6	47	44.7	47

Table 3
Model results for treatment and covariate effects on overstory structural attributes. Significant predictor variables are indicated by bold text. Type column indicates whether the model included different slopes for the pre-treatment condition effect based on thinning treatment alone (T) or the combination of thinning and burning treatments (T × B). Main treatment effects were evaluated at the mean of the pre-treatment condition.

Response variable	Thinning[a]		Burning[a]		Thin × Burn interaction[a]		Pre-treatment[a]		
	F	*P*	*F*	*P*	*F*	*P*	Type	*F*	*P*
Tree density	**137.37**	**<0.0001**	0.77	0.41	0.00	0.96	**T**	**239.87**	**<0.0001**
Tree basal area	**68.58**	**<0.0001**	0.38	0.55	0.13	0.73	**T**	**122.53**	**<0.0001**
Tree quadratic mean diameter	**40.35**	**0.0002**	0.28	0.61	0.09	0.77	**T × B**	**93.39**	**<0.0001**
Tree canopy base height	**42.81**	**0.0002**	3.16	0.11	0.50	0.50	**T**	**11.66**	**<0.0001**
Tree canopy bulk density	**52.35**	**<0.0001**	0.01	0.94	0.30	0.60	**T**	**79.80**	**<0.0001**
Snag density	1.48	0.26	**10.22**	**0.01**	**7.26**	**0.03**	**T × B**	**13.87**	**<0.0001**
Snag basal area	1.55	0.25	**6.66**	**0.03**	**5.11**	**0.05**	**T × B**	**23.60**	**<0.0001**

[a] There were eight denominator degrees of freedom for testing main treatment effects (thin, burn, and thin × burn) and 58 for the pre-treatment covariate. Numerator degrees of freedom were one for all tests.

tree density, tree basal area, mean tree diameter, canopy base height, and canopy bulk density, while burning significantly modified the density and basal area of snags (Table 3). Pre-treatment structure was a significant covariate in all models, with the slope of the covariate varying either with thinning (thinned or not) or across all four treatment combinations (Table 3).

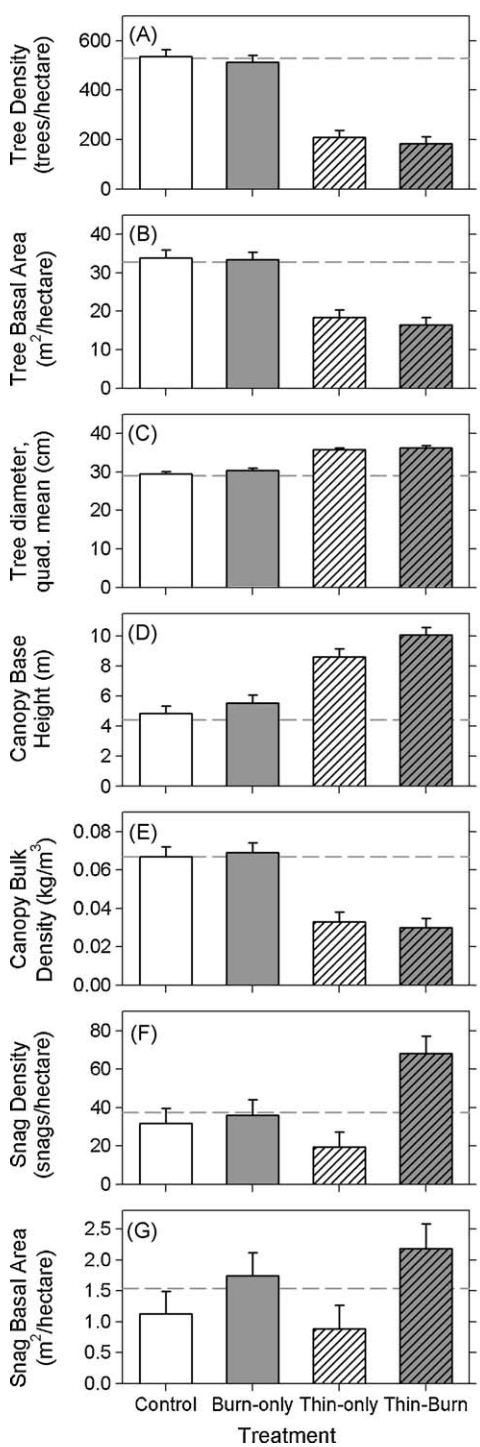

Fig. 7. Thinning and prescribed fire treatment effects on stand-level structural attributes, including tree density (A), tree basal area (B), tree quadratic mean diameter (C), canopy base height (D), canopy bulk density (E), snag density (F), and snag basal area (G). Vertical bars indicate least-square means predictions for post-treatment values for stands with the pre-treatment mean value. Pre-treatment means for each attribute are indicated by the horizontal dashed reference lines.

Thinning significantly reduced tree density and basal area, while increasing mean tree diameter. Thinning reduced tree density by an average of 325 trees/ha (60%), relative to controls, based on a reference pre-treatment mean of 530 trees/ha (Table 3 and Fig. 7A). Thinning also reduced tree basal area by almost 50%, with post-treatment basal area averaging 17.4 m^2/ha on thinned units compared to 33.6 m^2/ha on units not thinned (Fig. 7B). Because thinning preferentially targeted smaller trees, it significantly increased post-treatment mean tree diameters (quadratic mean) from 29.9 cm on units not thinned to 35.8 cm on thinned units (Fig. 7C).

Thinning was also the most effective treatment for increasing canopy base height and reducing canopy bulk density. Thinning increased canopy base height significantly by an average of 3.7 m relative to controls, while burning increased canopy base height by an average of only 0.7 (burn-only) to 1.6 m (thin-burn), which was not statistically significant (Table 3 and Fig. 7D). Thinning reduced canopy bulk density by about 50%, but burning had little effect (Table 3 and Fig. 7E).

Thinning and burning effects on snag density and basal area were more complex, with a significant interactive effect. The thin-burn treatment significantly increased snag density and basal area relative to controls, adding an average of 37 snags/ha and 1.5 m^2/ha respectively (Fig. 7F and G). The burn-only treatment produced smaller increases in snag density and basal area, while the thin-only treatment reduced snag abundance.

4. Discussion

Overstory stand structure influences a wide range of ecosystem properties and processes in dry coniferous forests, including fire behavior (Graham et al., 2004), tree growth and bark beetle activity (Larsson et al., 1983; Fettig et al., 2007), wildlife abundance (Raphael and White, 1984; Lehmkuhl et al., 2006a,b), and understory vegetation biomass and composition (Scholes and Archer, 1997; Moore et al., 2006; Collins et al., 2007). Although these properties and processes may depend on different aspects of overstory stand structure, treatments designed to modify structure to achieve benefits in one area are likely to impact other areas as well. We therefore discuss the outcomes of this study with respect to three potential management objectives to show the tradeoffs involved in selecting fuels and dry forest restoration treatments.

4.1. Restoration of historical stand structure

Decades of fire exclusion have produced significant structural and compositional changes in dry coniferous forests of western North America (Covington and Moore, 1994; Harrod et al., 1998, 1999), and restoration of historical structural conditions (and associated ecosystem processes) is often cited as one objective to be met by the application of thinning and prescribed fire treatments (Covington et al., 1997; Harrod et al., 1999). Not surprisingly, we found that thinning was clearly the most effective treatment for modifying stand structural attributes such as tree density, stand basal area, and mean tree diameter, a result that is consistent with most other sites in the Fire and Fire Surrogates (FFS) network (Schwilk et al., 2009). Thinning, with or without subsequent prescribed fire, reduced mean tree densities by 60% and mean stand basal area by almost 50% at our site. The resulting mean stand basal area (17.4 m^2/ha) was higher than the target value (10–14 m^2/ha), but this was partly because we restricted our sampling to forested areas within the treatment units. Inclusion of non-forest vegetation and rock outcrop areas in our sampling domain would have reduced basal area estimates for most units.

Despite the large thinning effects, post-treatment tree density was still about four times greater than historical stands, due largely to higher numbers of relatively young trees, and mean tree diameters were also much lower than historical stands (Harrod et al., 1999). Although heavier thinning prescriptions could have further reduced tree density and increased mean tree diameter, historical structural conditions cannot be met on these sites without replacing the large, older trees that have been removed through selective logging and other disturbances. This emphasizes the point that thinning and prescribed fire treatments modify stand structure only through selective removal (mortality) of existing trees. Growing large-diameter trees where none currently exist ultimately requires time and protection from high severity disturbances.

In using prescribed fire to modify stand structure, it is difficult to select weather and fuel conditions that will produce significant tree mortality, even among smaller trees, without risking loss of containment. Burning season can also influence prescribed fire effectiveness, primarily because of differences in the moisture content of duff and large fuels. In California, fall prescribed fires have shown the potential to significantly alter overstory structure (Stephens and Moghaddas, 2005a; Kobziar et al., 2006). Knapp and Keeley (2006) found no significant difference in mean bole scorch heights between spring and fall prescribed fires, suggesting that spring fires can produce effects comparable to those of fall fires. However, the higher fuel moisture levels typically found during spring prescribed fires can reduce the amount of surface area burned and increase spatial heterogeneity in fire effects (e.g., Knapp and Keeley, 2006) similar to those found historically (Harrod et al., 1999; Hessburg and Agee, 2003).

In this study, high fuel moistures during the spring prescribed fires on four units in 2004 produced low-severity fires that burned only 23–51% of the forest floor area (Agee and Lolley, 2006) and limited crown scorching to only 30% of trees. The spring fires on two units in 2006, in contrast, were more intense, because green-up had not begun yet, and produced crown scorching on about 80% of trees. On average, burns in 2006 raised ladder fuels about 1 m more than burns in 2004, but other overstory attributes measured were varied similarly across all units, regardless of year. The 2006 burns demonstrate that spring prescribed fires are capable of burning at reasonably high intensities and may be useful for meeting some management objectives, particularly in concert with thinning or as part of a regimen of repeated fires (Allen et al., 2002). The variability in some overstory effects in our study point to the importance of burning under environmental conditions and fuel moisture conditions that will meet management objectives, regardless of season of burn.

4.2. Modification of canopy fuels and potential fire behavior

The primary purpose of the Fire and Fire Surrogates program was to evaluate the effectiveness of prescribed fire and common alternative treatments for modifying surface and canopy fuels to reduce the likelihood of crown fire behavior during wildfires (Youngblood et al., 2005). Crown fire behavior depends on surface fuels, ladder fuels (which allow surface fires to move into the overstory canopy), canopy fuels (which determine the ability of crown fires to spread) (Scott and Reinhardt, 2001; Graham et al., 2004; Peterson et al., 2005), and environmental conditions (Van Wagner, 1977).

Our results suggest that thinning followed by prescribed fire is best for reducing fire hazard because of the complementary effects of thinning and prescribed fire on canopy and surface fuels (see Agee and Lolley, 2006). In this study, thinning raised canopy base height and lowered canopy bulk density, thereby reducing the probability of crown fire to propagate and sustain itself in the canopy. Previous studies have similarly documented the efficacy of thinning treatments for reducing canopy fuels and the severity of modeled (Fulé et al., 2002) and actual wildfires (Pollet and Omi, 2002). Prescribed fire alone was less effective than thinning for modifying canopy fuels, but burning did greatly reduce saplings and small trees (ladder fuels), particularly following thinning. Still, significant modifications to canopy structure required thinning before burning.

A common problem with thinning alone, however, is that it can increase surface fuels and therefore produce higher surface fire intensities, especially when slash is left on site (Stephens, 1998; Raymond and Peterson, 2005; Stephens and Moghaddas, 2005a; Schmidt et al., 2008; Youngblood et al., 2008). At Mission Creek, thinning alone increased surface fuel loadings across all time lag fuel classes (Agee and Lolley, 2006), thereby increasing potential flame lengths and the likelihood of torching. In addition to increasing fire intensity, high surface fuel loading may allow fires to burn or smolder for extended periods and increase tree mortality through basal heating (Swezy and Agee, 1991). Applying prescribed fire following thinning can further reduce fuels and fire hazard by consuming surface fuels (Agee and Lolley, 2006; Stephens and Moghaddas, 2005a), killing tree seedlings and saplings that can become ladder fuels, and further raising canopy base height.

If prescribed fire is not a viable treatment option, the benefits of thinning alone may still outweigh the potential hazards from increased surface fuels, at least during less than severe fire weather conditions. Reduction of post-thinning slash is important for significantly reducing fire hazard within fuel treatment areas (Stephens, 1998), but thinning can be effective at moderating fire behavior as compared to untreated areas (Stephens and Moghaddas, 2005a; Schmidt et al., 2008). Because thinning raised the mean canopy base height to 8.1 m on our Mission Creek sites, models indicate that a flame length of 3.4 m would be required to propagate fire into the canopy (Van Wagner, 1977; Scott, 2003). Such flame lengths would be expected only during high intensity wildfires burning under extreme drought and fire weather conditions. Wildfires burning in slash may hinder fire suppression activities (Stephens and Moghaddas, 2005a) so it is important to consider the tradeoffs, particularly near developed areas, before large amounts of slash are left on site.

4.3. Maintaining or improving wildlife habitat

Although wildlife habitat objectives are rarely the primary reason for modifying overstory stand structure, treatment effects on habitat suitability may still be an important consideration in selecting treatment options, particularly if treatments are to be applied over large areas or in areas containing species of particular interest. Treatment effects on coarse wood are of particular interest as snags and downed woody debris provide habitat for many wildlife species, particularly birds (Gaines et al., 2007; Lyons et al., 2008) and small mammals (Lehmkuhl et al., 2006a,b). While we did not examine treatment effects on downed woody debris, we did find that thinning and prescribed fire affected snag structure and dynamics in different ways.

Thinning resulted in significant losses of pre-treatment snags, with at least 50% of pre-treatment snags being cut across all size classes. Log removal in this study was accomplished by helicopter, so snags were felled primarily due to safety concerns for workers who could be injured by snags falling because of rotor turbulence. Sometimes snags are also removed to improve operational capability or are accidentally knocked over when live trees are felled (Susan Rinke, Timber Sale Administrator, Wenatchee River Ranger District, pers. comm.). Although there was some tree mortality in thin-only units, it was confined mostly to small and medium-size trees and was not sufficient to offset the losses of pre-treatment snags.

Prescribed burning also resulted in the loss of some snags, but those losses were generally offset or exceeded by new snag generation. Stephens and Moghaddas (2005b) also found that prescribed fire alone and combined with thinning resulted in greater numbers of new snags as compared to thinning alone or no treatment. Despite the patchy nature of the spring prescribed fires, the burn-only treatment killed enough trees to offset lost snags, including a small number of large trees. Tree mortality was even higher when prescribed fire followed thinning, presumably due to higher surface fuel loadings and surface fire intensities.

Prescribed fire treatments were generally more favorable for maintaining snag density and basal area, but a couple of caveats are in order. First, many of the trees killed by prescribed fire were small, with diameters less than 20 cm at breast height (1.35 m). Cavity-nesting birds typically select large-diameter trees (>25 cm dbh) for nesting and foraging (Raphael and White, 1984; Ganey and Vojta, 2004; Lyons et al., 2008) so many of the newly created snags will have limited value to birds and other cavity-nesting species (Lyons et al., 2008). Secondly, newly created larger snags may not provide suitable nesting habitat until tree boles decay sufficiently, which can take years to decades (Bull et al., 1997). In replacing large, older snags with large, fresh snags, the treatments may be replacing current habitat with potential future habitat. This could be a problem if large areas of a landscape are treated over a short period of time.

4.4. Management implications

Results from our study suggest that thinning followed by burning is the best treatment for modifying stand structural attributes to restore historical structures and enhance forest resiliency to fire. In contrast, spring prescribed fires alone are not effective for reducing overstory tree density, basal area, and canopy fuel loads, at least not with the low fire intensities realized in this study. However, through their complementary effects, prescribed fire and thinning together can be used to manipulate overstory stand structure and potentially achieve a broader range of management objectives than either treatment alone. Thinning followed by prescribed fire is particularly useful for modifying fuels and fire hazards, due to the combined effects on surface and canopy fuels.

Prescribed fire effectiveness for modifying overstory trees will be limited unless prescriptions can be designed to produce hotter fires while maintaining safety and control. For example, Youngblood et al. (2006) found a similar null effect on large (>25 cm) overstory trees as observed in our study and suggested that more aggressive burning may have achieved more tree mortality. This can be accomplished by burning when fuels are drier and fire weather is more favorable, but such conditions may be difficult to realize, given the availability of such conditions, and planning and staffing limitations. As a result, it may be required to plan subsequent burns to consume additional fuels and cause additional tree mortality. Subsequent burns may also be the least expensive option compared to other treatments.

Finally, thinning and burning treatment applications should be prioritized with consideration of the landscape context. The highest priority areas for applying fuel treatments will likely be areas that historically supported low-severity fire regimes (Brown et al., 2004; Hessburg and Agee, 2003). Within the low-severity fire regime areas, fuel treatments might be strategically placed to limit movement of uncharacteristically severe wildfires across the landscape (Finney, 2001; Finney et al., 2007). In some cases, benefits of modifications in potential fire behavior (e.g., reduced risk of stand-replacing wildfire) will have to be balanced against negative treatment impacts on critical wildlife habitat or other resources (Lehmkuhl et al., 2007). Our findings suggest that thinning and burning treatments can meet a variety of management objectives at the stand scale, but managers will need to develop ecologically sound strategies for effective and efficient placement of treatments on modern landscapes. In addition, managers should consider the need for multiple treatments over time in order to restore historical structure, reduce fire risk, and then maintain structure and fuels within desired limits (Youngblood et al., 2008).

Acknowledgements

This is Contribution Number 164 of the National Fire and Fire Surrogate Project (FFS), funded by the U.S. Joint Fire Science Program. We appreciate the field assistance of numerous workers: Mattias Rudbak, Scott Conlan, Beth Armbrust, Sara O'Neal, Mara McGrath, Dottie Knecht, Todd Jensen, Tim Hatch, Chad Yenney, Megan Whitmore, Kathleen Moran, Darci Carlson, Alisha Toombs, Pete Ohlson, James Dickinson, and many others. Jim Agee, Bob Gray, Andy Youngblood, Rich Fonda, and an anonymous reviewer provided helpful reviews which greatly improved the manuscript. We thank the Wenatchee River Ranger District, Okanogan-Wenatchee NF, for implementing both the prescribed burning and thinning treatments.

References

Agee, J.K., Skinner, C.N., 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211, 83–96.
Agee, J.K., Lolley, M.R., 2006. Thinning and prescribed fire effects on fuels and potential fire behavior in an eastern Cascades forest, Washington, USA. Fire Ecology 2, 142–158.
Allen, C.D., Savage, M., Falk, D.A., Suckling, K.F., Swetnam, T.W., Schulke, T., Stacey, P.B., Morgan, P., Hoffman, M., Klingel, J.T., 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications 12, 1418–1433.
Brown, R.T., Agee, J.K., Franklin, J.F., 2004. Forest restoration and fire: principles in the context of place. Conservation Biology 18, 903–912.
Bull, E.L., Parks, C.G., Torgersen, T.R., 1997. Trees and logs important to wildlife in the interior Columbia River Basin. USDA Forest Service, General Technical Report PNW-GTR-391.
Collins, B.M., Moghaddas, J.J., Stephens, S.L., 2007. Initial changes in forest structure and understory plant communities following fuel reduction activities in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 239, 102–111.
Covington, W.W., Fulé, P.Z., Moore, M.M., Hart, S.C., Kolb, T.E., Mast, J.N., Sackett, S.S., Wagner, M.R., 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. Journal of Forestry 95, 23–29.
Covington, W.W., Moore, M.M., 1994. Southwestern ponderosa forest structure: changes since Euro-American settlement. Journal of Forestry 92, 39–47.
Dahms, C.W., Geils, B.W. (Tech. Eds.), 1997. An assessment of forest ecosystem health in the Southwest. USDA Forest Service, General Technical Report RM-GTR-295.
DeLuca, T.H., Sala, A., 2006. Frequent fire alters nitrogen transformations in ponderosa pine stands of the inland northwest. Ecology 87, 2511–2522.
Dodson, E.K., Peterson, D.W., Harrod, R.J., 2008. Understory vegetation response to thinning and burning restoration treatments in dry conifer forests of the eastern Cascades, USA. Forest Ecology and Management 255, 3130–3140.
Fettig, C.J., Klepzig, K.D., Billings, R.F., Munson, A.S., Nebeker, T.E., Negrón, J.F., Nowak, J.T., 2007. The effectiveness of vegetation management practices for prevention and control of bark beetle infestations in coniferous forests of the western and southern United States. Forest Ecology and Management 238, 24–53.
Finney, M.A., 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. Forest Science 47, 219–228.
Finney, M.A., Selia, R.C., McHugh, C.W., Ager, A.A., Bahro, B., Agee, J.K., 2007. Simulation of long-term landscape-level fuel treatment effects on large wildfires. International Journal of Wildland Fire 16, 712–727.
Fire Program Solutions, 2005. Fuels Management Analyst Suite. Sandy, OR.
Fulé, P.Z., Covington, W.W., Smith, H.B., Springer, J.D., Heinlein, T.A., Huisinga, K.D., Moore, M.M., 2002. Comparing ecological restoration alternatives, Grand Canyon, Arizona. Forest Ecology and Management 170, 19–41.
Gaines, W.L., Haggard, M., Lehmkuhl, J.F., Lyons, A.L., Harrod, R.J., 2007. Short-term response of land birds to ponderosa pine restoration. Restoration Ecology 15, 666–674.
Ganey, J.L., Vojta, S.C., 2004. Characteristics of snags containing excavated cavities in northern Arizona mixed-conifer and ponderosa pine forests. Forest Ecology and Management 199, 323–332.
Graham, R.T., McCaffrey, S., Jain, T.B., 2004. Science basis for changing forest structure to modify wildfire behavior and severity. USDA Forest Service, General Technical Report RMRS-GTR-120.

Harrington, M.G., Sackett, S.S., 1990. Using fire as a management tool in southwestern ponderosa pine. In: Proceedings of Effects of Fire Management on Southwestern Natural Resources, Tucson, AZ, November 1988. USDA Forest Service, General Technical Report RM-GTR-191.
Harrod, R.J., Gaines, W.L., Hartl, W.E., Camp, A.E., 1998. Estimating historic snag density in dry forests east of the Cascade Range. USDA Forest Service, General Technical Report PNW-GTR-428.
Harrod, R.J., McRae, B.H., Hartl, W.E., 1999. Historical stand reconstruction in ponderosa pine forests to guide silvicultural prescriptions. Forest Ecology and Management 114, 433–446.
Harrod, R.J., Halpern, C.B., 2005a. The seed ecology of *Iliamna longisepala* (Torr.) Wiggins, an East Cascade endemic. Natural Areas Journal 25, 246–256.
Harrod, R.J., Halpern, C.B., 2005b. Effects of experimental shading and competitor removal on vegetative and reproductive performance of *Trifolium thompsonii* Morton, a rare endemic herb of eastern Washington. Natural Areas Journal 25, 156–164.
Harrod, R.J., Dodson, E.K., Povak, N.A., Peterson, D.W., 2009. Chapter 2: Pretreatment variation in overstory and understory vegetation. In: Agee, J.K., Lehmkuhl, J.F. (Comps.), Dry forests of Northeastern Cascades Fire and Fire Surrogate Site, Mission Creek, Okanogan-Wenatchee National Forest. USDA Forest Service, Research Paper PNW-RP-577, pp. 13-29.
Hessburg, P.F., Agee, J.K., 2003. An environmental narrative of inland northwest US forests, 1800–2000. Forest Ecology and Management 178, 23–60.
Hessburg, P.F., Agee, J.K., Franklin, J.F., 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management 211, 117–139.
Knapp, E.E., Keeley, J.E., Ballenger, E.A., Brennan, T.J., 2005. Fuel reduction and coarse woody debris dynamics with early season and late season prescribed fire in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 208, 383–397.
Knapp, E.E., Keeley, J.E., 2006. Heterogeneity in fire severity within early and late season prescribed burns in a mixed-conifer forest. International Journal of Wildland Fire 15, 37–45.
Kobziar, L., Moghaddas, J., Stephens, S.L., 2006. Tree mortality patterns following prescribed fires in a mixed conifer forest. Canadian Journal of Forest Research 36, 3222–3238.
Larsson, S., Oren, R., Waring, R.H., Barrett, J.W., 1983. Attacks of mountain pine beetle as related to tree vigor of ponderosa pine. Forest Science 29, 395–402.
Lehmkuhl, J.F., Kistler, K.D., Begley, J.S., 2006a. Bushy-tailed woodrat abundance in dry forests of eastern Washington. Journal of Mammalogy 87, 371–379.
Lehmkuhl, J.F., Kistler, K.D., Begley, J.S., Boulanger, J., 2006b. Demography of northern flying squirrels informs ecosystem management of western interior forests. Ecological Applications 16, 584–600.
Lehmkuhl, J.F., Kennedy, M., Ford, E.D., Singleton, P.H., Gaines, W.L., Lind, R.L., 2007. Seeing the forest for the fuel: integrating ecological values and fuels management. Forest Ecology and Management 246, 73–80.
Littell, R.C., Milliken, G.A., Stroup, W.W., Wolfinger, R.D., Schabenberger, O., 2006. SAS® for Mixed Models, second edition. SAS Institute, Inc., Cary, NC, USA.
Lyons, A.L., Gaines, W.L., Lehmkuhl, J.F., Harrod, R.J., 2008. Short-term effects of fire and fire surrogate treatments on foraging tree selection by cavity-nesting birds in dry forests of central Washington. Forest Ecology and Management 255, 3203–3211.
McHugh, C.W., Kolb, T.E., Wilson, J.L., 2003. Bark beetle attacks on ponderosa pine following fire in northern Arizona. Environmental Entomology 32, 510–522.
McIver, J.D., Ottmar, R., 2007. Fuel mass and stand structure after post-fire logging of a severely burned ponderosa pine forest in northeastern Oregon. Forest Ecology and Management 238, 268–279.
Moore, M.M., Casey, C.A., Bakker, J.D., Springer, J.D., Fulé, P.Z., Covington, W.W., Laughlin, D.C., 2006. Herbaceous vegetation responses (1992–2004) to restoration treatments in a ponderosa pine forest. Rangeland Ecology and Management 59, 135–144.
Peterson, D.L., Johnson, M.C., Agee, J.K., Jain, T.B., McKenzie, D., Reinhardt, E.D., 2005. Forest structure and fire hazard in dry forests of the western United States. USDA Forest Service, General Technical Report PNW-GTR-628.
Pollet, J., Omi, P.N., 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. International Journal of Wildland Fire 11, 1–20.
Raphael, M.G., White, M., 1984. Use of snags by cavity-nesting birds in the Sierra Nevada. Wildlife Monographs 86, 1–66.
Raudenbush, S.W., Bryk, A.S., 2002. Hierarchical Linear Models: Applications and Data Analysis Methods. Sage Publications, Thousand Oaks, CA, USA, 483 pp.
Raymond, C.L., Peterson, D.L., 2005. Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. Canadian Journal of Forest Research 35, 2981–2995.
Scholes, R.J., Archer, S.R., 1997. Tree–grass interactions in savannas. Annual Review of Ecology and Systematics 28, 517–544.
Schmidt, D.A., Taylor, A.H., Skinner, C.N., 2008. The influence of fuels treatment and landscape arrangement on simulated fire behavior, Southern Cascade range, California. Forest Ecology and Management 255, 3170–3184.
Schwilk, D.W., Keeley, J.E., Knapp, E.E., McIver, J., Bailey, J.D., Fettig, C.J., Fiedler, C.E., Harrod, R.J., Moghaddas, J.J., Outcalt, K.W., Skinner, C.N., Stephens, S.L., Waldrop, T.A., Yaussy, D.A., Youngblood, A., 2009. The National Fire and Fire Surrogate Study: effects of alternative fuel reduction methods on forest vegetation structure and fuels. Ecological Applications 19, 285–304.
Scott, J.H., 2003. Canopy fuel treatment standards for the wildland–urban interface. In: Omi, P.N., Joyce, L.A. (Eds.), Fire, Fuel Treatments, and Ecological Restoration. USDA Forest Service, Proceedings RMRS-P-29.
Scott, J.H., Reinhardt, E.D., 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. USDA Forest Service, Research Paper RMRS-RP-29.
Smith, H.Y., Arno, S.F. (Eds.), 1999. Eighty-eight years of change in a managed ponderosa pine forest. USDA Forest Service, General Technical Report RMRS-GTR-23.
Snijders, T., Bosker, R., 1999. Multilevel Analysis: An Introduction to Basic and Advanced Multilevel Modeling. Sage Publications, Thousand Oaks, CA, USA, 266 pp.
Stephens, S.L., 1998. Evaluation of the effects of silvicultural and fuels treatments on potential fire behavior in Sierra Nevada mixed-conifer forests. Forest Ecology and Management 105, 21–35.
Stephens, S.L., Moghaddas, J.J., 2005a. Experimental fuel treatment impacts on forest structure, potential fire, and predicted tree mortality in a California mixed-conifer forest. Forest Ecology and Management 215, 21–36.
Stephens, S.L., Moghaddas, J.J., 2005b. Fuel treatment effects on snags and coarse woody debris in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 214, 53–64.
Stephens, S.L., Moghaddas, J.J., Edminster, C., Fiedler, C.E., Haase, S., Harrington, M., Keeley, J.E., Knapp, E.E., McIver, J.D., Metlen, K., Skinner, C.N., Youngblood, A., 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. Ecological Applications 19, 305–320.
Swezy, D.M., Agee, J.K., 1991. Prescribed fire effects on fine-root and tree mortality in old-growth ponderosa pine. Canadian Journal of Forest Research 21, 626–634.
Thomas, T.L., Agee, J.K., 1986. Prescribed fire effects on mixed-conifer forest structure at Crater Lake, OR. Canadian Journal of Forest Research 16, 1082–1087.
USDA Forest Service, 1995. Mission Creek watershed assessment, Wenatchee National Forest, Leavenworth Ranger District, 61 pp.
Van Wagner, C.E., 1977. Conditions for the start and spread of crown fire. Canadian Journal of Forest Research 7, 23–34.
Youngblood, A., Metlen, K., Knapp, E., Outcalt, K., Stephens, S., Waldrop, T., Yaussy, D., 2005. Implementation of the Fire and Fire Surrogates Study—a national research effort to evaluate the consequences of fuel reduction treatments. In: Peterson, C.E., Maguire, D.A. (Eds.), Balancing Ecosystem Values. USDA Forest Service, General Technical Report PNW-GTR-635, pp. 315–321.
Youngblood, A., Metlen, K.L., Coe, K., 2006. Changes in stand structure and composition after restoration treatments in low elevation dry forests of northeastern Oregon. Forest Ecology and Management 234, 143–163.
Youngblood, A., Wright, C.S., Ottmar, R.D., McIver, J.D., 2008. Changes in fuelbed characteristics and resulting fire potentials after fuel reduction treatments in dry forests of the Blue Mountains, northeastern Oregon. Forest Ecology and Management 255, 3151–3169.