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By

Lindsay Aney Chiono

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Committee in charge:

Professor Scott L. Stephens, Chair
Professor Kevin L. O'Hara
Professor David D. Ackerly

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ABSTRACT

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Historic fire regimes in the dry conifer forests of the southern Cascade and northern Sierra Nevada regions of California were characterized by relatively frequent fires of low and mixed severity. Human management practices since the mid-19th century have altered the disturbance role of fire in these dry yellow pine and mixed conifer forest ecosystems. Fire suppression, high-grade timber harvesting, and livestock grazing have reduced the frequency of burning and caused a shift in the structure and species composition of forest vegetation. These changes, including high levels of accumulated fuel and increased structural homogeneity and dominance of shade-tolerant tree species, combined with a warming climate, have rendered many stands susceptible to high-severity fire. In many forests of the western United States, wildfires are increasingly difficult and costly to control, and human communities are regularly threatened during the fire season.

Treating wildland fuels to reduce wildfire hazards has become a primary focus of contemporary forest management, particularly in the wildland-urban interface. The specific objectives of treatment are diverse, but in general, treatments address accumulated surface fuels, the fuel ladders that carry fire into the forest canopy, and surface and canopy fuel continuity. These modifications to forest fuels can alleviate the severity of a future wildfire and support suppression activities through improved access and reduced fire intensity. While fuel reduction treatments are increasingly common in western forests, the long-term structural and ecological effects of treatment remain poorly understood. This dissertation uses a chronosequence of treated stands to examine the temporal influence of treatment on forest structure, the understory plant community, and wildfire hazard.

The first chapter examines the effects of fuels reduction treatment on stand structure, overstory species composition, and ground and surface fuels. The stand structures and reduced surface fuel loads created by fuels modification are temporary, yet few studies have assessed the lifespan of treatment effects. The structural legacies of treatment were still present in the oldest treatment sites. Treatments reduced site occupancy (stand density and basal area) and increased quadratic mean diameter by approximately 50%. The contribution of shade-tolerant true firs to stand

density was also reduced by treatment. Other stand characteristics, particularly timelag fuel loads, seedling density, and shrub cover, exhibited substantial variability, and differences between treatment age classes and between treatment and control groups were not statistically significant.

The second chapter evaluates fuel treatment longevity based on potential wildfire behavior and effects on vegetation. Forest managers must divide scarce resources between fuel treatment maintenance, which is necessary to retain low hazard conditions in treated stands, and the construction of new treatments. Yet the most basic questions concerning the lifespan of treatment effectiveness have rarely been engaged in the literature. In this study, field-gathered fuels and vegetation data were used to aid fuel model selection and to parameterize a fire behavior and effects model, Fuels Management Analyst Plus. In addition, a semi-qualitative, semi-quantitative protocol was applied to assess ladder fuel hazard in field sampling plots. Untreated sites exhibited fire behavior that would challenge wildfire suppression efforts, and projected overstory mortality was considerable. In contrast, estimated fire behavior and severity were low to moderate in even the oldest fuel treatments, those sampled 8-26 years after treatment implementation. Findings indicate that in the forest types characteristic of the northern Sierra Nevada and southern Cascades, treatments for wildfire hazard reduction retain their effectiveness for more than 10-15 years and possibly beyond a quarter century.

Fuel treatment activities disturb the forest floor, increase resource availability, and may introduce non-native plant propagules to forest stands. Non-native plant invasions can have profound consequences for ecosystem structure and function. For these reasons, there is concern that treatment for fire hazard reduction may promote invasion by exotic species. Several short-term studies have shown small increases in non-native abundance as a result of treatment, but the long-term effects have rarely been addressed in the literature. The final chapter examines treatment effects on the understory plant community and on cover of the forest floor, as mineral soil exposure has been linked to invasion. Regression tree analysis provided insights into the influence of treatment and site characteristics on these variables. Treatments increased forb and graminoid cover, but temporal trends in abundance were opposite. An initial increase in forb cover in the most recently treated sites was followed by a gradual decline, while mean graminoid cover was highest in the oldest treatments. Shrubs dominated live plant abundance. Shrub cover showed few temporal trends, but was negatively associated with canopy cover. Mineral soil exposure was increased by treatment and declined slowly over time, remaining elevated in the oldest treatments. Non-native plant species were very rare in the treatment sites sampled in this study. Despite the availability of bare mineral soil and the proximity of transportation corridors, a source of non-native propagules, non-natives were recorded in only 2% of sampling plots. This study suggests that forest disturbance associated with treatment for hazardous fuels reduction may not produce significant invasions in these forest types.

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Spotswood, Eric Waller, Kate Wilkin, and many other friends for sharing their lives and considerable talents.

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CHAPTER 1

Development of Vegetation and Surface Fuels Following Fire Hazard Reduction Treatment

Abstract

In dry western Unites States forests where past resource management has altered the ecological role of fire and stand characteristics alike, mechanical thinning and prescribed burning are commonly applied in wildfire hazard abatement. The reduced surface fuel loads and stand structures resulting from fuels modifications are temporary, yet few studies have assessed the lifespan of treatment effects. I sampled forest fuels and vegetation following fuels reduction in a chronosequence of time since treatment in the northern Sierra Nevada and southern Cascade regions of California. Treatments altered overstory characteristics including stand density, basal area, and species composition. These effects were still present on the oldest treatment sites (8-15 years post-treatment). Other stand characteristics, particularly timelag fuel loads, seedling density, and shrub cover, exhibited substantial variability, and differences between treatment age classes and between treatment and control groups were not statistically significant.

Introduction

The disturbance role of wildfire in many dry, temperate western United States forests has been altered through fire exclusion, timber harvesting, and livestock grazing. These land-use practices have affected forest structure and species composition, increasing surface fuel loads, tree density, dominance of shade-tolerant tree species, and forest homogeneity (Covington and Moore, 1994; Naficy et al., 2010; Scholl and Taylor, 2010). As a consequence, many historically fire-frequent forests are now vulnerable to spatially extensive high-severity wildfire (Skinner and Chang, 1996). A primary focus of contemporary management in these forests is the treatment of fuels and vegetation to address wildfire hazards.

Fuels reduction treatments are intended to reduce the potential for high-intensity, high-severity wildfire by reducing the quantity and continuity of forest fuels. A number of techniques are employed to meet these fuels reduction objectives, and each method has associated effects on forest structure. Mechanical thinning reduces stand density, basal area, and canopy fuels (Fulé et al., 2001; Stephens and Moghaddas, 2005a; Schwilk et al., 2009). To reduce accumulated surface fuel loads and offset the activity fuels produced during harvest operations, prescribed fire is often coupled to forest thinning. Broadcast burning can also be expected to reduce ladder fuels and elevate canopy base height (Raymond and Peterson, 2005; Stephens et al., 2009). Research generally supports the ability of such treatments to alter potential fire behavior and impacts (e.g. Agee et al., 2000; Fulé et al., 2001; Agee and Skinner, 2005; Ritchie et al., 2007; Strom and Fulé, 2007; Stephens et al., 2009; Stephens et al., 2012).

Though the immediate effects of treatment on forest fuels and stand structure are relatively well known, the long-term consequences remain poorly understood. Post-treatment conditions are impermanent: after treatment, the overstory responds to take advantage of newly available growing space, filling the canopy space vacated by thinned trees; the canopy base falls in height as new regeneration joins the overstory; and surface fuels accumulate as the canopy deposits leaves, cones, and branches. Some treatment techniques may actually enhance post-treatment vegetation growth, effectively shortening the lifespan of low fire hazard conditions. Reducing overstory density has long been recognized to promote regeneration (Smith et al., 1997) and increase understory growth (McConnell and Smith, 1970; Bailey and Tappeiner, 1998). Additionally, the exposure of mineral soil by prescribed burning fosters seed germination (Haase, 1986). In order to retain low fire hazard conditions, areas that have been treated must be maintained following their initial establishment. However, few management tools exist to guide the division of resources between establishment of new treatments and maintenance of existing treatments.

In this study, I assessed dead fuel loads, shrub cover, regeneration, and overstory characteristics in a chronosequence since fuel treatment in the northern Sierra Nevada and southern Cascade regions of California. Sample sites were stratified on the basis of forest type: the Sierra mixed conifer and eastside pine forests are present in the study region. My hypothesis anticipated differences in fuel development and vegetation regrowth between the two forest types, with slower accumulation of surface fuels and development of understory and ladder fuels predicted in the xeric eastern slope pine forests. This work should inform resource allocation between fuel treatment implementation and future maintenance.

These management considerations are of particular interest in the study region due to the influence of the Herger-Feinstein Quincy Library Group Pilot Project and prior fuel reduction work enabled by a developing biomass industrial infrastructure. The Project was established in 1998 to promote hazardous fuels reduction in the region, aiming to treat ~16-24,000 ha (40-60,000 acres) per year within a strategic network of fuel breaks (Moghaddas and Craggs, 2007). Support from the local community and relatively abundant economic resources dedicated to fuel treatment implementation have generated many potential sampling sites established over the years since the Project's creation, making this region exceptionally suited for a chronosequence study of fuels reduction. Most of the treatment areas sampled in this study are shaded fuel breaks *sensu* Agee et al. (2000), i.e. areas in which fuels have been modified in order to moderate fire hazard while maintaining some forested cover. Supporting fire suppression activities is an explicit goal of fuel breaks, which are often strategically situated along roads and ridgetops and near communities and other high-value resources. Nonetheless, these findings will be applicable to the broader category of fuel treatments, which include treatments intended to reduce the likelihood of wildfire ignition and/or mitigate potential fire effects and resistance to control (National Wildfire Coordinating Group (NWCG), 2011).

Methods

Study area

This study was conducted in Nevada, Sierra, and Plumas Counties in the northern Sierra Nevada and southern Cascade regions of California (Figure 1.1). Historically, low- to moderate-severity fires here were frequent: a study of fire history in similar forests found a pre-Euro-American settlement mean composite fire return interval of 6-18 years (for fires scarring more than 10% of samples)(Moody et al., 2006). The climate west of the Sierra Nevada crest is Mediterranean with warm, dry summers and cold, wet winters. To the east, the continental climate pattern is prevalent, and is characterized by more extreme daily and seasonal temperature shifts and lower precipitation. Most precipitation falls as snow during the winter months, and annual precipitation ranges from 38 cm on the east side to nearly 230 cm on the west (USDA Forest Service, 1988). The geologic and climatic diversity of this portion of the Sierra Nevada range have produced an equally diverse soil mosaic that includes granitic, volcanic, and serpentine soils. The west side is characterized by relatively deep and productive soils while those of the cool and dry east side are shallow and less productive (USDA Forest Service, 1988). Study site elevations range from 1100-2150 m.

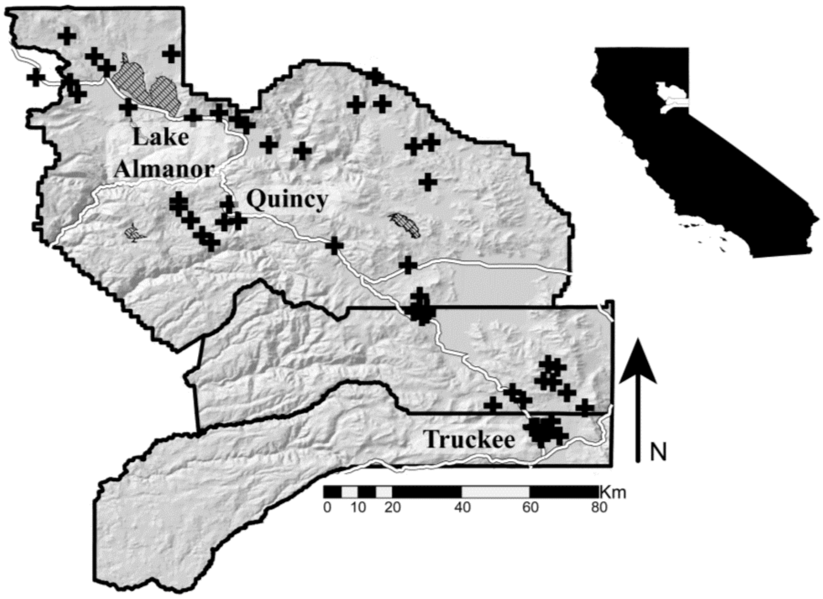

Figure 1.1. Study area in the northern Sierra Nevada and southern Cascades, CA. Black crosses indicate study site locations.

The Sierra mixed conifer forest type is dominated by sugar pine (*Pinus lambertiana* Dougl.), ponderosa pine (*P. ponderosa* Dougl.), white fir (*Abies concolor* (Gord. and Glend.), incense-cedar (*Calocedrus decurrens* [Torr.] Florin.), Douglas-fir (*Pseudotsuga menziesii* var. *menziesii* (Mirb.) Franco), and California black oak (*Quercus kelloggii* Newb.) (Barbour and Minnich, 2000) while the colder and drier lower montane eastside pine type is dominated by Jeffrey pine (*Pinus jeffreyi* Grev. and Balf.) and white fir.

Downed fuels and understory and overstory vegetation were sampled within 51 treatment sites 2-15 years following initial treatment and 13 untreated sites. Local forest managers helped identify treatment units suitable for sampling. Sampling sites had been treated with mechanical thinning alone or in combination with broadcast or pile burning. If applicable, follow-up burning was to occur within three years of the thinning treatment. All treatment projects fitting the study design requirements were sampled. A single treatment *project* often included multiple units treated over a period of several years. In order to avoid possible pseudoreplication arising from adjacent unit locations and identical timber operators, a single unit was randomly selected to represent each individual project.

The mechanical thinning treatments sampled in this study included some prescriptions that were not explicitly designed for hazardous fuels reduction. These included single-tree selection harvests and understory thinning to improve the vigor of residual trees. Incorporating thinning treatments not necessarily intended as fuel treatments in this exhaustive sampling effort permitted a larger sample size. In practice, the stand structures produced by all mechanical thinning types were similar and included reduced ladder fuels and reduced density of small- and mid-diameter trees. To limit variability in post-thinning conditions, hand-thinning and mastication treatments were not included in this study. While most stand treatments (40 of 51) were located on land managed by the US Forest Service, nine sites belonged to the Collins Pine Company, a private forest products company, and two fuel treatments had been implemented by Fire Safe Councils on privately owned land.

Untreated control sites were established in stands adjacent to treatment areas. Control sites were defined as having overstory species composition and slope steepness comparable to those of the adjacent treated unit, without evidence of recent (within ~25 years) wildfire or management. Because mechanical thinning equipment is generally restricted to slopes of less than 30 percent grade, no prospective control site with a slope exceeding 30 percent was sampled. In many cases, potential control sites were deemed unsuitable for sampling because there was evidence of recent thinning or fire, such as intact stumps or char, or because the slope or dominant vegetation differed substantially from that of the adjacent treated area.

Field sampling

Downed woody fuels, understory composition, and overstory characteristics were sampled using a systematic sampling design with a random starting point. Three circular plots, 50 m apart, were established in each treatment unit and placed parallel to the treatment boundary, typically a road. This choice of plot number and spacing represents a compromise between minimizing potential spatial autocorrelation between plots while ensuring that most treatment units could accommodate the sampling design. A fixed number of plots were selected because while treatments varied in areal extent, georeferenced treatment maps were often unavailable. To minimize boundary effects, plots were placed 30 m from the nearest treatment boundary. When gaps in the treatment were encountered during plot placement (e.g. a group selection unit) subsequent plots were placed on the opposite side of the treatment gap with a 30 m buffer from the gap edge.

The elevation, aspect, slope, and slope position for each plot were recorded. Plot centers were permanently marked with wooden stakes and witness trees marked with aluminum tags. Three 17.85 m transects were established within each circular plot (Figure 1.2). The azimuth of the first transect was chosen randomly while the second and third transects were placed with headings of 120˚ and 240˚ greater than the first. Surface and ground fuels were sampled using the planar-intercept method (van Wagner, 1968; Brown, 1974). Beginning at the transect end farthest from plot center, 1-hour (≤0.64 cm diameter) and 10 h time lag fuels (>0.64 cm to ≤2.54 cm diameter) were tallied from 0-2 m and 100 h time lag fuels (>2.54 cm to ≤7.62 cm diameter) were tallied from 0-3 m. The number and diameter of 1000 h time lag (>7.62 cm) and larger fuels were recorded along the full length of the transect, and fuels were categorized according to condition (sound or rotten). Duff and litter depths were measured at 2.85 and 12.85 m. Total surface fuel depth was recorded at three points along each transect. Fuel loads were calculated using Sierra Nevada tree species-specific estimates (van Wagtendonk et al., 1996, 1998), weighted by the contribution of each species to plot-level basal area (Stephens, 2001).

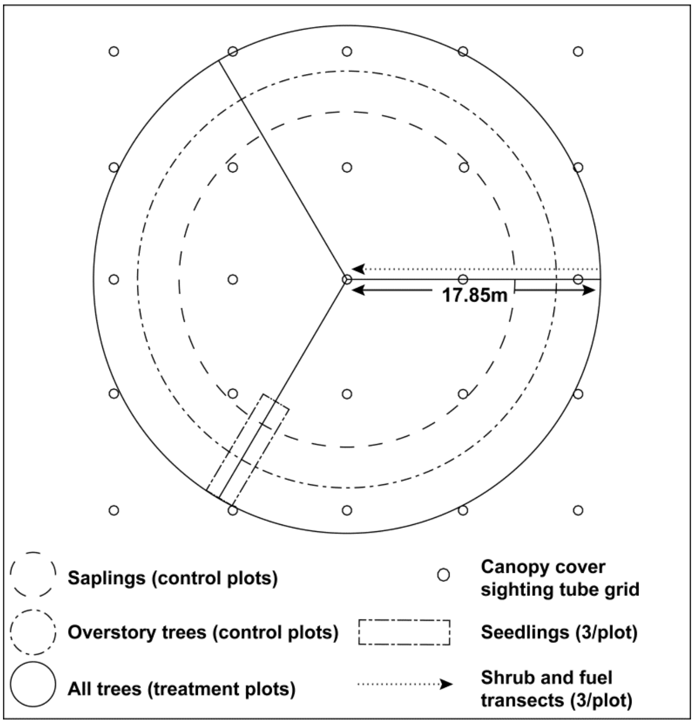

Figure 1.2. Illustration of sampling plot layout. Three sampling plots were placed in each treatment site to investigate patterns of fuel, vegetation, and stand development after treatment. Sampling did not vary between control and treatment sites with the exception of tree data collection.

Shrub measurements including species, average height, and status (live or dead) were taken along each linear transect. Shrub cover was calculated as the transect length occupied by shrub divided by the total transect length. The height, caliper at base, and species of all seedlings (trees <2.5 cm dbh) were recorded in three 2 x 7 m plots, each centered on a transect and positioned at the end farthest from plot center (Figure 1.2). Within each plot, overstory canopy cover was sampled with a densitometer (sighting tube) on a 25-point, 8 x 8 m grid oriented north-south and east-west (Jennings et al., 1999). Percent canopy cover was estimated as the number of canopy "hits" divided by the total number of sampling points (25).

Sampling of treated and untreated plots was identical except with regard to tree sampling. In treatment plots, the total height, height to live crown base, diameter at breast height (dbh), crown class, and species were recorded for all trees ≥ 2.5 cm dbh within 17.85-m radius. Because stand density on the untreated control sites was greater than for treatment sites, overstory trees (≥7.6 cm dbh) and saplings (2.5-7.6 cm dbh) were sampled in nested subplots 0.075 and 0.05 ha in size, respectively.

Treatment history records associated with two of the oldest sites were incomplete. Using standard dendrochronological techniques (Stokes and Smiley, 1977; Swetnam et al., 1985), the year of thinning in these sites was verified using evidence from tree rings contained in stumps of small- to intermediate-diameter trees and visible logging scars on live trees presumed to have resulted from mechanical thinning operations. Stump cross-sections distributed over the sampling area were removed with a chainsaw; increment cores were removed from trees with visible logging scars. Each cross-section or core was sanded to a high polish to allow rings and scars to be viewed clearly under a microscope. The year of thinning was determined by cross-dating tree rings against a master tree-ring chronology.

Canopy fuel calculations and statistical analysis

The Crown Mass program (v. 3.0.49) within the Fuels Management Analyst Plus (FMAPlus) suite (www.fireps.com) was used to estimate canopy bulk density and canopy base height (Carlton, 2005). FMAPlus uses modified allometric equations to estimate average canopy profile characteristics from field-derived inputs including tree species, dbh, tree crown ratio, and canopy class. Canopy base height is defined in FMAPlus as the height above the ground of the first canopy layer with sufficient density of canopy fuels to carry fire vertically. The canopy bulk density is the maximum value of a running mean of vertically oriented one-foot (30.48 cm) canopy layers.

The effects of treatment and differences between time-since-treatment groups were examined by analysis of variance. Data were log-transformed when necessary to meet the assumptions of statistical tests. Where significant differences occurred ($p < 0.05$), comparisons between means were performed using Tukey's HSD multiple comparisons test. Due to limited sample sizes, data representing both methods of active treatment and major slope aspects (north and south) and were combined for these analyses.

An analysis of covariance (ANCOVA) was performed to examine the relationship between stand density and site factors, including stand productivity measures (combined average height of dominant and codominant trees and site index (Dunning, 1942), estimated from average dominant tree height at a specified base age), forest type, slope aspect, and treatment factors, including method of fuel treatment and ownership.

All statistical analyses were performed in the statistical software package R version 2.10.1 (www.R-project.org.)

Results

Fuel characteristics

Mixed Conifer.—With the exception of leaf litter, surface fuel loads in the mixed conifer forest type did not differ significantly between any of the age classes or between the treated and untreated sites (Figure 1.3). Mean 1-, 10-, and 100-hour fuel loads were respectively 78, 98, and 40% greater in the untreated sites relative to the oldest treatment class, but high variation led to insignificant differences between groups. Control litter loads were nearly double those of the treatment age classes, which were very similar. Mean ground (duff) fuel loads for the untreated sites was ~1.5-3.5 times that of the treated sites, though this difference was significant only for the oldest time-since-treatment class.

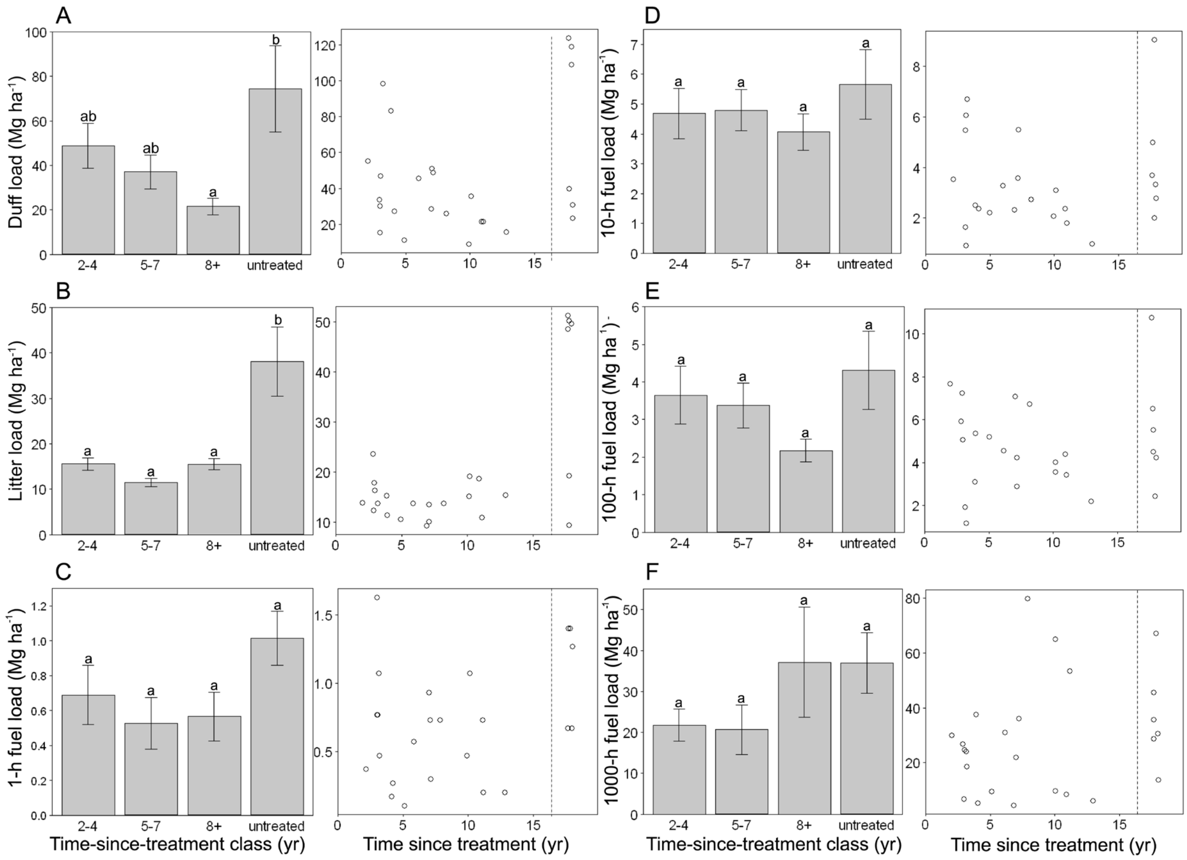

Figure 1.3. Dead surface fuel load by category in the mixed conifer chronosequence (n=19) and control (n=6) sites. Fuel load in the (A) duff layer, (B) litter layer, and the (C) 1-hr, (D) 10-hr, (E) 100-hr, and (F) 1000-hr time-lag categories. Left-hand panels show means ±SE for the three age class groups and the control group. Different letters above each bar indicate significant difference in means between groups (Tukey's HSD multiple comparison test, $p < 0.05$). Right-hand panels show the relationship between time since treatment and each fuel category. Vertical dashed lines separate treated and untreated study sites.

Eastside pine.—For the eastside pine forest type, there was no significant difference in duff or 10- and 100-hr fuel loads over time following treatment and no difference between any time-since-treatment class and the untreated group (Figure 1.4A, D, E). Litter loads were significantly greater in the untreated sites than for the two youngest age classes (i.e. sites treated 2-7 years prior to sampling), but not for the oldest class (8-15 years since treatment). While 10- and 100-h fuel loads exhibited no trend over time since treatment, 1-hour fuel loads were significantly lower in the mid-range treatment age class (5-7 years since treatment) than the oldest class and the untreated group. A similar trend was observed for 1000-hour fuel loads (Figure 1.4F).

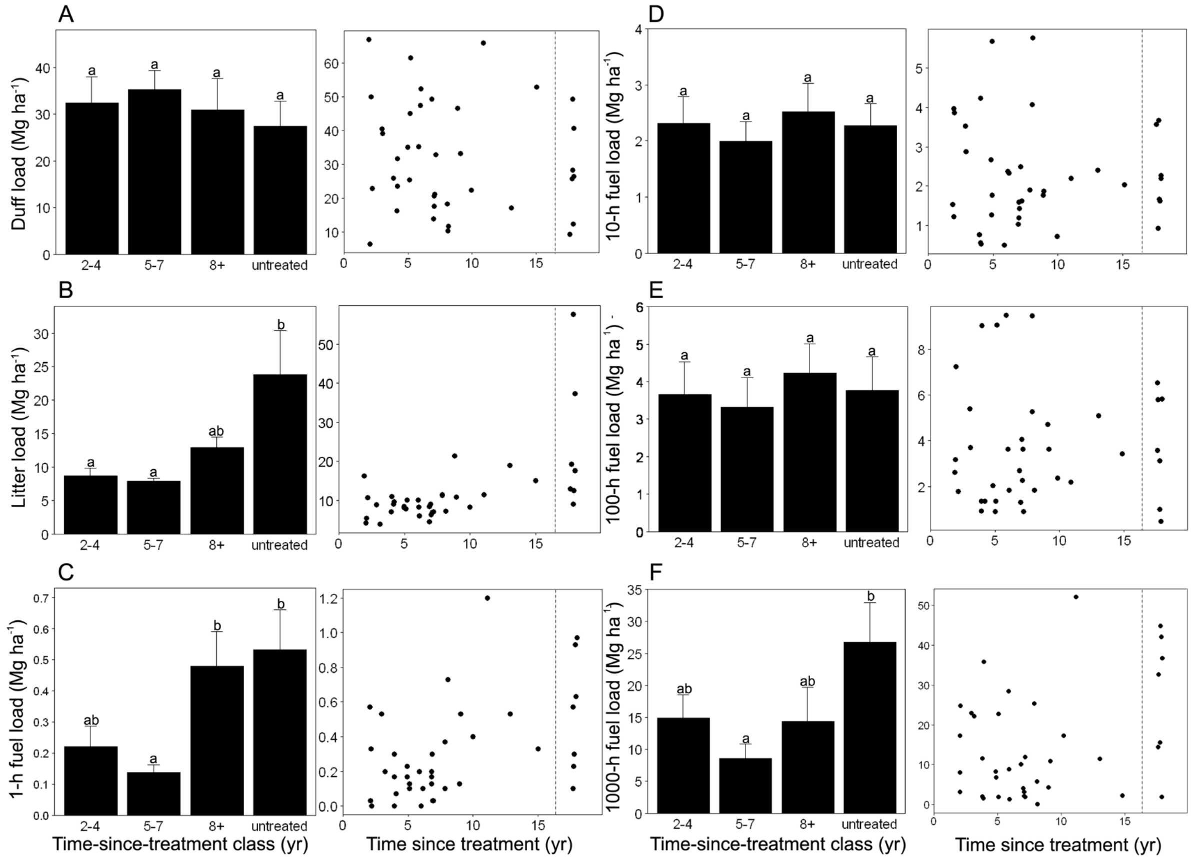

Figure 1.4. Dead surface fuel load by category in the eastside pine chronosequence (n=32) and control (n=7) sites. Fuel load in the (A) duff layer, (B) litter layer, and the (C) 1-hr, (D) 10-hr, (E) 100-hr, and (F) 1000-hr time-lag categories. Left-hand panels show means ±SE for the three age class groups and the control group. Different letters above each bar indicate significant difference in means between groups (Tukey's HSD multiple comparison test, $p < 0.05$). Right-hand panels show the relationship between time since treatment and each fuel category. Vertical dashed lines separate treated and untreated study sites.

Vegetation characteristics

Mixed conifer.— Mean stand basal area (50.3 m^2 ha^{-1}) for untreated stands was double that of treated stands, and this difference was significant across time-since-treatment classes (Figure 1.5A). Treatment also significantly reduced stand density (Table 1.1). The combined contribution of *Abies concolor* and *A. magnifica* to density was reduced (Table 1.1), though not to basal area (Table 1.2), a reflection of the preferential removal of small-diameter trees. Likewise, treatment increased quadratic mean diameter (QMD) by 49.5% (Table 1.2).

Control canopy base height (mean = 1 m) was significantly lower than for any of the treatment groups (overall treatment mean = 4.3 m) (Figure 1.5B). Control canopy bulk density was approximately double that of treated sites, and exhibited no trend over time following treatment (Figure 1.5C). Overstory canopy cover for the two most recent treatment classes (46 and 48%) was intermediate between the oldest time-since-treatment class (41%) and the control (67%). Treatment age classes did not differ significantly from one another with respect to canopy cover, but the 2-4 and 8-15 years-since-treatment groups each had significantly lower percent cover than the untreated control (Figure 1.5D).

Table 1.1. Mixed conifer mean stand density(standard error), quadratic mean diameter (QMD), and mean contribution of each tree species to total density for each time since treatment category. Different letters indicate significant difference in means between groups (Tukey's HSD multiple comparison test, $p < 0.05$). Calculations include all trees with diameter at breast height ≥ 2.54 cm. Tree species codes are *ABSP*: *Abies concolor* and *A. magnifica* (combined), *CADE*: *Calocedrus decurrens*, *PIJE*: *Pinus jeffreyi*, *PILA*: *P. lambertiana*, *PIPO*: *P. ponderosa*, *PSME*: *Pseudotsuga menziesii*, *QUKE*: *Quercus kelloggii*. Species composing less than 1% of total density are not included in % density calculations.

Time Since Treatment (*years*)	**Mean Density (*stems* ha^{-1})**	**QMD (*cm*)**	**% Density by Species**						
			ABSP	*CADE*	*PIJE*	*PILA*	*PIPO*	*PSME*	*QUKE*
2-4 (n=8)	372(47)^{a}	33.7	26	21	12	7	7	14	13
5-7 (n=5)	336(62)^{a}	32.0	32	11	12	11	12	15	8
8+ (n=7)	388(55)^{a}	30.2	13	6	11	4	15	25	26
Untreated (n=6)	1406(119)^{b}	21.4	57	17	1	5	9	9	2

Table 1.2. Mixed conifer mean basal area(standard error) and mean contribution of each tree species to total basal area for each time since treatment category. Different letters indicate significant difference in means between groups (Tukey's HSD multiple comparison test, $p < 0.05$). Calculations include all trees with diameter at breast height ≥ 2.54 cm. Species composing less than 1% of basal area are not included in % basal area calculations. See Table 1.1 for explanation of species codes.

Time Since Treatment (*years*)	**Mean Basal Area (m^2 ha^{-1})**	**% Basal Area by Species**						
		ABSP	*CADE*	*PIJE*	*PILA*	*PIPO*	*PSME*	*QUKE*
2-4	33.7(3.2)^{a}	21	18	15	15	8	18	5
5-7	32.0(6.6)^{a}	34	7	11	22	15	12	0
8+	27.7(4.7)^{a}	28	5	10	6	22	27	3
Untreated	50.3(2.7)^{b}	33	10	2	15	20	17	3

A high degree of variability in understory vegetation characteristics contributed to a lack of significant differences between the treatment age classes and between the treatment and control groups. Tree regeneration was especially variable in the youngest time-since-treatment age class. Though mean seedling density in the youngest class was ~2.5 times greater than that of the next age class, mean densities did not differ at a significance level of 0.05 (Figure 1.5F). This short-lived peak in seedling density following treatment was apparent for sites treated with mechanical thinning alone as well as those in which thinning was followed by burning (not shown). Shrub cover was generally low, and exhibited no trend over time following treatment (Figure 1.5E). Only 4 of 26 mixed conifer sites had >20% shrub cover, and only 2 of these had >30% shrub cover.

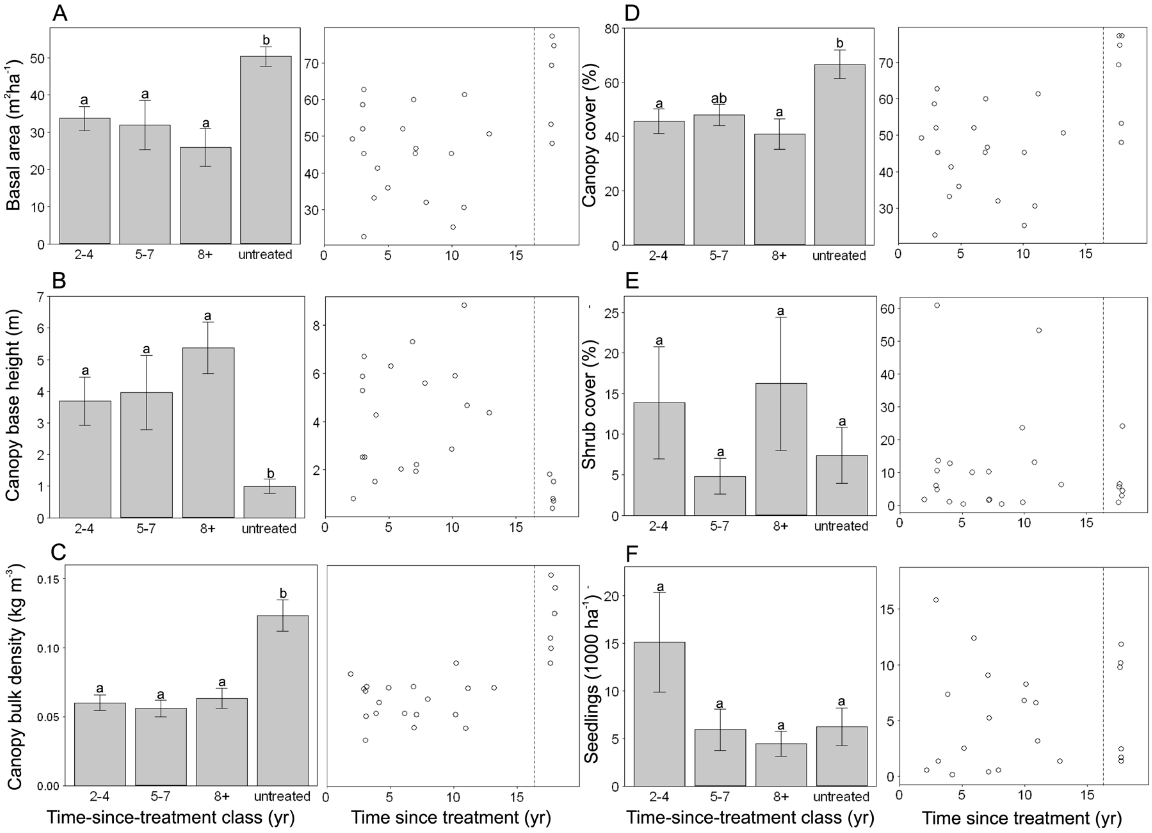

Figure 1.5. Vegetation characteristics in the mixed conifer chronosequence (n=19) and control (n=6) sites. (A) basal area, (B) canopy base height, (C) canopy bulk density, (D) canopy cover , (E) shrub cover, and (F) seedling density (all trees <2.5cm dbh). Left-hand panels show means ±SE for the three age class groups and the control group. Different letters above each bar indicate significant difference in means between groups (Tukey's HSD multiple comparison test, $p < 0.05$). Right-hand panels show the relationship between time since treatment and each stand characteristic. Vertical dashed lines separate treated and untreated study sites.

Eastside pine.— Overstory characteristics generally did not vary between time-since-treatment classes, but were significantly affected by treatment. Control basal area and canopy bulk density were significantly greater than for any treatment age class (Figures 1.6A, C). Basal area of the untreated sites was double that of the oldest time-since-treatment class, while control canopy bulk density was ~4 times that of the oldest treatment class. Likewise, at nearly 50%, mean control canopy cover was 39-82% greater than of the treated sites (Figure 1.6D). Mean canopy base height in the control sites was 1 m compared with 2.5-3.7 m in the treatment units (Figure 1.6C). In terms of stand density, treatment favored Jeffrey pine over true fir (Table 1. 3). As thinning prescriptions generally targeted the smallest-diameter trees, QMD was increased by 55.5% (Table 1.4).

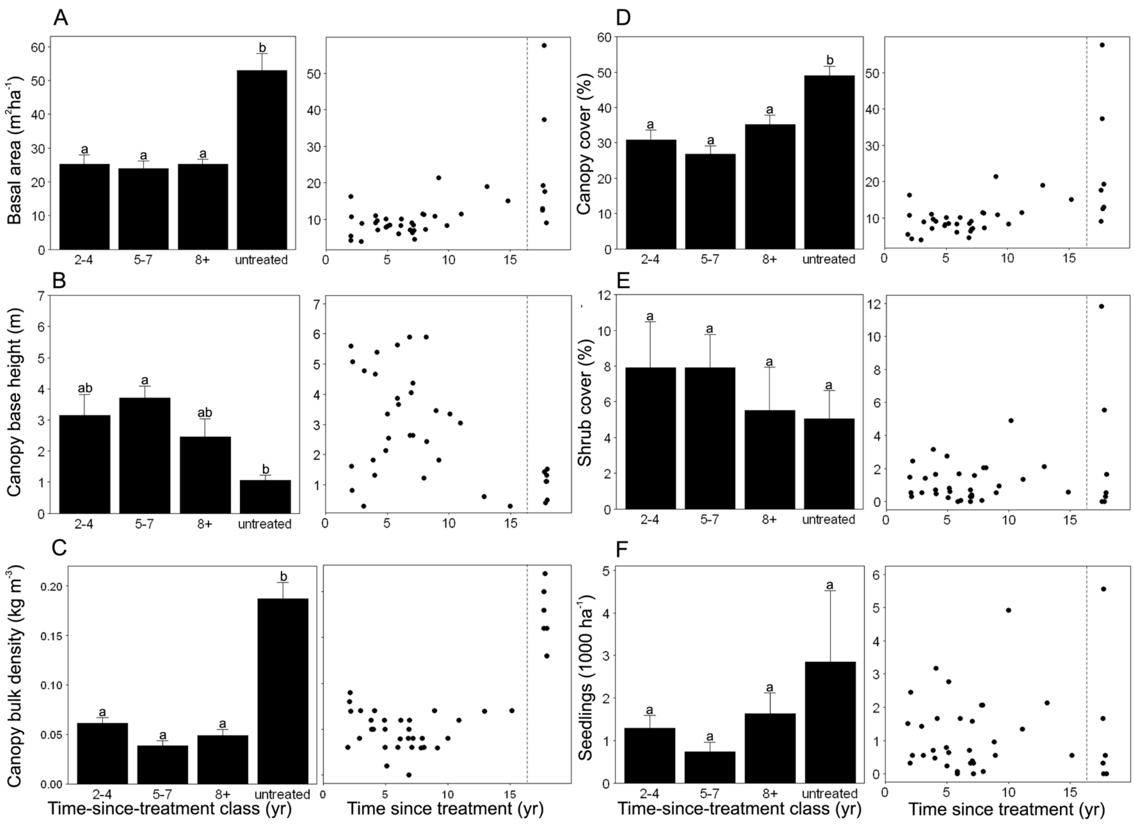

Figure 1.6. Eastside pine vegetation characteristics. (A) basal area, (B) canopy base height, (C) canopy bulk density, (D) canopy cover , (E) shrub cover, and (F) seedling density (all trees <2.5cm dbh) in the eastside pine chronosequence (n=32) and control (n=7) sites. Left-hand panels show means ±SE for the three age class groups and the control group. Different letters above each bar indicate significant difference in means between groups (Tukey's HSD multiple comparison test, $p < 0.05$). Right-hand panels show the relationship between time since treatment and each stand characteristic. Vertical dashed lines separate treated and untreated study sites.

Table 1.3. Eastside pine mean stand density(standard error), quadratic mean diameter (QMD), and mean contribution of each tree species to total density for each time since treatment category. Different letters indicate significant difference in means between groups (Tukey's HSD multiple comparison test, $p < 0.05$). Calculations include all trees with diameter at breast height ≥ 2.54 cm. Tree species codes are *ABSP*: *Abies* concolor and *A. magnifica* (combined), *CADE*: *Calocedrus decurrens*, JUOC: *Juniperus occidentalis*, *PIJE*: *Pinus jeffreyi*, *PIPO*: *P. ponderosa*. Species composing less than 1% of total density are not included in % density calculations.

Time Since Treatment (*years*)	**Mean Density (*stems ha^{-1}*)**	**QMD (*cm*)**	**% Density by Species**				
			ABSP	*CADE*	*JUOC*	*PIJE*	*PIPO*
2-4 (n=10)	258(31.6)^{a}	35.3	34	1	0	64	0
5-7 (n=13)	181.5(12.9)^{a}	41.0	16	0	0	83	0
8+ (n=9)	345.6(90.94)^{a}	30.5	16	0	0	83	0
Untreated (n=7)	1283.5(162.1)^{b}	22.9	42	5	1	52	1

Table 1.4. Eastside pine mean basal area(standard error), and mean contribution of each tree species to total basal area for each time since treatment category. Different letters indicate significant difference in means between groups (Tukey's HSD multiple comparison test, $p < 0.05$). Calculations include all trees with diameter at breast height ≥ 2.54 cm. The "*Abies sp.*" category includes combined contributions of *Abies concolor* and *A. magnifica*. Species composing less than 1% of basal area are not included in % basal area calculations.

Time Since Treatment (*years*)	**Mean Basal Area (m^2 ha^{-1})**	**% Basal Area by Species**					
		ABSP	*CADE*	*PIJE*	*PILA*	*PIPO*	*PSME*
2-4	25.2(2.8)^a	18	1	80	0	0	0
5-7	23.9(2.3)^a	8	0	92	0	0	0
8+	25.3(1.4)^a	38	3	46	1	8	4
Untreated	53.0(4.9)^b	26	11	61	0	2	0

Shrub cover and seedling regeneration did not exhibit the high degree of variability observed for the mixed conifer forest type. Shrub cover was low overall, from 8% in the youngest treatment group to 5% in the untreated group, and did not significantly vary between control and treatment groups. Though mean control seedling density was ~2-4 times treatment group levels, this difference was not significant at $p < 0.05$.

Discussion

Ground and Surface Fuels

Downed coarse wood, often defined as material larger than 7.6 cm (3 in) in diameter (Harmon et al., 1986), is a focus of habitat management, as it is required by many wildlife species for foraging, cover, and substrate (Bunnell et al., 2002). It is also important with regard to ecosystem structure and function (e.g. maintenance of site productivity, protection of soils from compaction and erosion). Yet given the frequent fire regime characteristic of the study area and the high consumption of decomposed coarse wood during burning in low-moisture conditions (Kauffman and Martin, 1989), historic levels of large-diameter surface fuels were likely lower than at present (Brown et al., 2003; Stephens et al., 2007). Reducing pretreatment levels and ameliorating additions produced during fuels manipulation is a management concern, as large quantities of coarse woody fuels can influence potential fire behavior (van Wagtendonk, 1996; Agee et al., 2000; Fulé et al., 2001).

For the eastside pine forest type, loads of large-diameter fuels were significantly higher in the untreated sites than in unburned sites that had been thinned 5-15 years earlier, but they were not significantly different from loads on sites treated 2-4 years prior to sampling (Figure 1.4F). In the xeric environment characteristic of the eastern Sierra Nevada, large-diameter woody fuels may remain on site for many years without follow-up treatment of activity fuels produced during thinning (Laiho and Prescott, 2004). Some sites were treated with prescribed fire after thinning, but while broadcast burning effectively reduces small-diameter fuels and loads of rotten large-diameter fuels (Covington and Sackett, 1984; Stephens and Finney, 2002), it may have less influence on sound logs (Stephens and Moghaddas, 2005b). The initial post-harvest peak in 1000-h loads is particularly clear in Figure 1.7, which shows coarse woody fuel loads in eastside pine sites treated with mechanical thinning alone. The relatively high proportion of solid coarse fuels in the most recently treated sites is likely due to additions produced during harvest. The rapid decline in solid 1000-h fuels is surprising given the expected slow rate of decomposition. Follow-up surface fuel treatment was planned but not yet completed on some sampled sites, and it may be the case that by chance such sites were overrepresented in the first time-since-treatment age class.

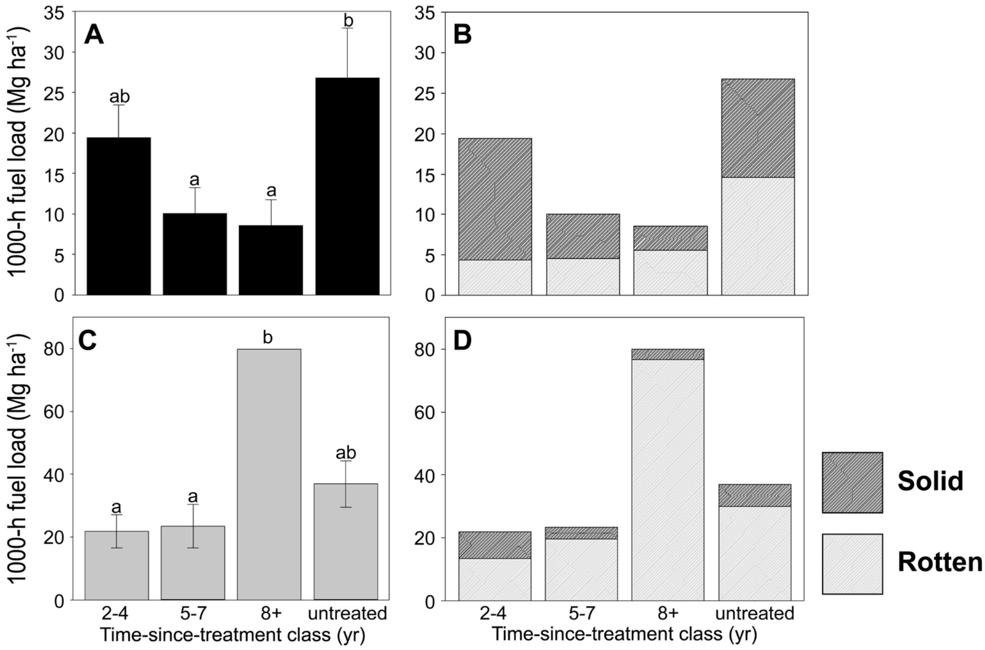

Figure 1.7. Thousand-hour timelag class (diameter > 7.62 cm) fuel load in eastside pine (A, B) and mixed conifer (C, D) sites treated with mechanical thinning alone or untreated. **A, C**: Different letters above each bar indicate significant difference in means between groups (Tukey's HSD multiple comparison test, $p < 0.05$). **B, D**: Total thousand-hour fuels by decomposition class.

The early pulse in coarse fuels was not observed in mixed conifer stands and in general, clear trends among the woody timelag fuel categories over time were not detectable (Figures 1.3 and 1.4). Because of limited availability of treatment sites, sites with varying methods of post-thinning surface fuel treatment were grouped for the analyses shown in Figures 1.3-1.6. While harvest activities transfer woody fuels from the canopy to the surface (Raymond and Peterson, 2005), several prescriptions included in the present study utilized whole-tree yarding, biomass removal, and/or grapple piling, each of which effectively reduce the activity fuels remaining onsite after treatment. Thirty-three percent of sites were treated with either pile or broadcast burning after harvesting in order to address natural and activity fuels. Burning reduces surface fuels but also produces direct and indirect tree mortality (Mutch and Parsons, 1998; Stephens and Moghaddas, 2005b; Youngblood et al., 2009); over time, fire-killed snags enter the surface fuel pool. Indeed, in burned sites, the number of snags in the most recently treated stands was 1.4-6.7 times that of the next age class, indicating that some trees killed during burning that were still standing at 2-4 years after treatment will become part of the surface fuel pool in the coming years. Combining sites treated with burning, which experience additions to the surface fuel pool in years after treatment, with those on which activity fuels were treated mechanically or left untreated would likely have the effect of masking trends in woody fuel loads over time.

Though some studies have assessed long-term dynamics of large-diameter woody fuels (Duvall and Grigal, 1999; Brown et al., 2003), few have looked beyond the initial impacts of treatment with respect to small-diameter woody fuels and the litter layer, which have the greatest influence on surface fire spread. Litter loads in stands belonging to every treatment age class were substantially lower than in untreated stands, and remained at low levels throughout the chronosequence. It appeared that duff loads were not affected by treatment, as they generally did not vary between treated and untreated stands (Figures 1.3A, 1.4A).

Broadcast burning often reduces both ground and surface fuels (Keifer et al., 2006; Kobziar et al., 2009; Vaillant et al., 2009), though these effects may not be long-lived. Keifer et al. (2006) found that surface fuels reached approximately 85 percent of pre-fire levels within 10 years after prescribed burning in ponderosa pine and white fir-mixed conifer forests in Sequoia and Kings Canyon National Parks in the southern Sierra Nevada. In the present study, litter loads in the mixed conifer and eastside pine sites reached only 47 and 54% of untreated levels, respectively, within 8-15 years of treatment. This may indicate that surface fuels in the study area accumulate more slowly than in the southern Sierra, but may also reflect differences in litter deposition between thinned and unthinned sites, as Keifer et al. (2006) did not include mechanical tree-removal in their study. Thinning reduces litter fall (Trofymow et al., 1991) since canopy cover is strongly linked to foliage production and litter accumulation (Hall et al., 2006).

As a general rule, burning reduces surface fuel loads while mechanical treatments tend to increase them (Schwilk et al., 2009), yet it appears that the post-treatment reductions in litter loads observed here are not solely the result of burning. When sites treated with mechanical thinning alone (35 of 51 treatment sites) were analyzed separately, mean litter loads for every forest type/age class combination were at least 40% less than for comparable untreated sites. As a result of the reduced sample size, however, this difference was significant at $p < 0.05$ for only the 2-4 and 5-7 year (mixed conifer) and 5-7 year (eastside pine) time-since-treatment classes.

Others have found reduced duff or litter layer depths after mechanical harvesting. Stephens and Moghaddas (2005a) saw reduced litter depth after removal of chainsaw-thinned trees with rubber-tired and tracked skidders. Fulé et al. (2001) found reduced average duff loads following both mechanical felling and broadcasting of activity fuels, and whole-tree harvesting combined with slash piling. Stephens and Moghaddas (2005c) found that combined duff and litter loads following overstory removal and clear-cutting were reduced relative to either young or old growth reserves, though loads following thin from below and individual tree selection treatments did not differ significantly from the reserves. Harvesting machinery can displace surface and ground fuel layers within a site, though it is not clear by what mechanism this should reduce ground and litter fuel mass. Alternatively, since fuel loads are typically estimated from measurements of fuel depth, compaction of the litter layer by harvesting equipment could produce an apparent reduction in loads. It is unclear whether this influence might be significant, as it has not been addressed in the literature.

Tree Regeneration

The lack of clear trends in tree seedling density over time was not unexpected. Mechanical thinning and thinning with prescribed fire tend to increase tree seedling density, but high variability among sites is common (Schwilk et al., 2009). Species respond to post-treatment conditions independently (Moghaddas et al., 2008; Zald et al., 2008), and regeneration has been linked to stand density, light levels, soil moisture and disturbance, variation in seed production (masting), and site productivity (Bailey and Tappeiner, 1998; Gray et al., 2005; Zald et al., 2008; Schwilk et al., 2009). Interannual climate variation has also been shown to significantly influence recruitment (League and Veblen, 2006; van Mantgem et al., 2006).

The early peak in seedling density in the mixed conifer treatments was not observed in the eastside forest. The relatively productive mixed conifer forest would be expected to promote higher levels of regeneration. In addition, mechanical thinning may have relatively little influence on the light environment in the eastside forest, where canopy cover in untreated stands is relatively low (~50%). Jeffrey pine, a dominant overstory species of the eastside, is associated with indirect radiation, and white and red fir are associated negatively with direct solar radiation and positively with soil moisture (Gray et al., 2005). Irregularity in Jeffrey pine seed crops has also been reported (Hallin, 1959).

Chronosequence studies rely on the assumption that time since treatment is the primary explanatory variable. Variation in conditions at the time of treatment can make this assumption untenable. This shortcoming is particularly relevant with respect to regeneration, which is known to exhibit a high degree of spatial and temporal variation.

Shrubs

Some have suggested that reducing canopy cover during thinning may promote shrub growth (e.g., Vaillant, 2008; Schwilk et al., 2009), thereby shortening the longevity of fuel treatment effectiveness. In the short-term, fuels treatment is expected to reduce shrub cover through mechanical damage (Collins et al., 2007; Wayman and North, 2007; Schwilk et al., 2009) and consumption during burning (Knapp et al., 2006; Wayman and North, 2007; Schwilk et al., 2009). Beyond these initial impacts, thinning could potentially promote shrub growth through reduced overstory competition. Campbell et al. (2009) found live shrub cover increased from 9% in unthinned controls to 32% and 22%, 3 and 16 years, respectively, after thinning-from-below in northern Sierra Nevada ponderosa pine plantations. Many shrubs in the study region are vigorous resprouters, and prescribed fire stimulates seed germination in some species (Knapp et al., 2006). However, potential increases in shrub growth as a result of reduced canopy cover and increased microsite availability may be limited. In the Teakettle Experimental Forest, a southern Sierra Nevada site, North et al. (2005) determined that mixed conifer shrubs were associated with diffuse light and low soil moisture levels. Cover was reduced in both closed canopy and canopy gaps with shrubs preferentially occupying an ecotone between the two cover types.

The lack of clear trends in shrub development in the years following treatment may reflect variability in pre-treatment conditions. Dodson et al. (2008) found that pre-treatment shrub cover was much more influential than treatment with respect to changes in shrub cover over time. Although the high variability in post-treatment shrub cover precluded clear findings with respect to development over time, the hypothesis that thinning and prescribed fire treatments would promote shrub growth, exacerbating potential fire hazards, is not supported by these data, as total shrub cover was nearly always low (<20%) in control and treatment sites alike. Similarly low levels of shrub cover have been observed in other dry western forests following treatment (McConnell and Smith, 1970; Perchemlides et al., 2008).

4.4. Stand Characteristics

Treatments for fuels reduction are often intended to achieve multiple objectives. Apart from fire hazard reduction, the restoration of pre-Euro-American settlement (hereafter "pre-settlement") conditions is a common goal of treatment. Some goals of restoration align with those of hazard reduction: a focus on recreating the conditions associated with pre-settlement can also be expected to reduce fire hazards, as manipulations for restoration typically involve reducing surface fuels and the number of small-diameter trees. While many of the treatments sampled here did not include restoration as an explicit goal, and fuel treatments cannot in general be assumed to achieve restoration, they did move stands toward the structure of pre-settlement forests by some measures.

The abundance of shade-tolerant fir species is a concern from a restoration standpoint as a well as a fire hazard perspective. Shade-tolerant conifers are characterized by vertically continuous crowns that can convey surface fire into the forest canopy. In mixed conifer stands in a Sierra Nevada old growth reserve (Teakettle Experimental Forest), North et al. (2007) compared stand conditions before and after understory thinning with and without prescribed fire to stand reconstructions of 1865. As in their study, treatment in the present study increased QMD and reduced stand density. However, while North et al. found that treatment did not significantly reduce pre-treatment contributions of *Abies* species (red and white fir) to stand density (65.3-71.9%), the contribution of true fir to mixed conifer stand composition in this study was reduced from 57.2% in untreated stands to <26% in treated stands. By comparison, North et al. estimated that true fir composed 36.6% of stand density in their 1865 reconstruction, similar to an estimate of 42% by volume in a 1913 Plumas National Forest survey (McKelvey and Johnston, 1992). Though treatments in the present study reduced the contribution of true fir to stand density, its contribution to stand basal area was not altered, reflecting preferential removal of small-diameter individuals during mechanical thinning. The same pattern was observed in the eastside pine forest type, where treatment favored Jeffrey pine over true fir with respect to density but not basal area.

The structural changes created by fuels management were still evident in the oldest chronosequence class. In both forest types, with respect to stand characteristics, the oldest treated units were statistically indistinguishable from more recently treated stands but were clearly distinct from untreated sites. Treatment effects included reduced vertical and horizontal fuel continuity and a higher proportion of large-diameter, fire-resistant trees. These changes indicate that treated stands are less vulnerable to high-severity fire even 8-15 years after treatment (Agee and Skinner, 2005).

One limitation of the chronosequence approach is that variation at the time of treatment can easily be attributed to variation over time *since* treatment. The changes in stand structure over time may be somewhat confounded with changing mechanical thinning prescriptions over time. Changing forest management over time has frustrated other chronosequence studies (Yanai et al., 2000). Figure 1.8 illustrates the challenge. Stand density appears to exhibit a u-shaped relationship with time following treatment, which likely reflects changes in mechanical treatment prescriptions over time rather than a real trend in stand development. ANCOVA results indicated a significant interaction between treatment age and the period in which thinning occurred. The pre-2002 and post-2002 (inclusive) division was chosen to represent the period before and after the 2001 Sierra Forest Plan Amendment Record of Decision (USDA Forest Service, 2001), which had the effect of reducing harvest levels in fuel treatments through canopy cover targets, diameter limits, and an emphasis on creating stand heterogeneity. While a number of other management directives certainly affected treatment implementation, notably the California Spotted Owl interim guidelines (USDA Forest Service, 1993) and the Herger-Feinstein Quincy Library Group (HFQLG) Forest Recovery Act of 1998 (1998), the 2001 split best described the pattern in stand density based on R^2 and p-values (not reported). Only treatments completed on federal forest land are included in the analysis of density reported in Figure 1.8. The larger chronosequence study includes a significant number of privately managed stands which would not be expected to be influenced by changing federal policies.

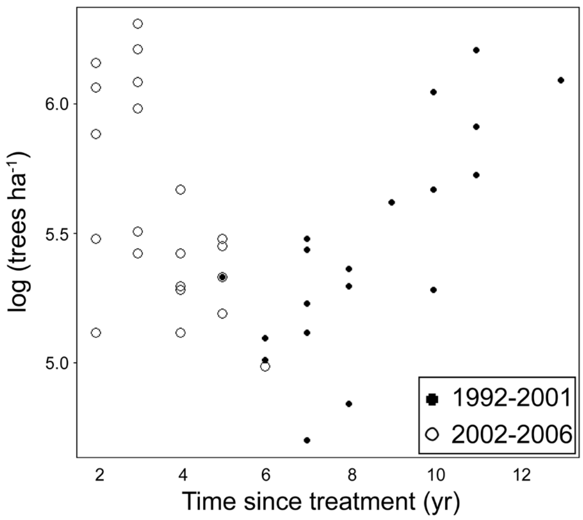

Figure 1.8. Stand density at the time of sampling for treatments sampled in both forest types. Figure includes data from sites located on land managed by the US Forest Service only. Symbol color represents the period in which forest thinning occurred.

Apart from historical changes in mechanical thinning prescriptions over time, I was unable to account for other probable sources of variation, including the seasons of thinning and burning, annual climatic variability, and prescribed burn intensity and fuel consumption. This variation likely contributed to the lack of significant differences between time-since-treatment groups and between treatment and control groups.

Conclusions

Many have noted the need for future maintenance of post-treatment conditions in order to retain low fire hazard (Agee et al., 2000; Peterson et al., 2005; Reinhardt et al., 2008), yet little research exists to guide management planning beyond initial treatment establishment. This chronosequence study indicates that some treatment effects are long-lived in the mixed conifer and eastside pine forests typical of the northern Sierra Nevada and southern Cascade regions of California. Metrics of overstory structure in treated stands were significantly different from those of untreated stands even 8-15 years after treatment implementation. The lack of significant differences between the youngest post-treatment class and the oldest class is further evidence of the longevity of structural changes produced by mechanical thinning alone and in combination with burning. Other effects of treatment, namely on tree seedling regeneration, shrub cover, and most surface fuel categories, were highly variable among sites. Patterns of post-treatment recovery were difficult to discern as a result. As shrub cover across both treated and untreated sites was low (generally <20%), our findings did not validate past concerns that treatment activities could enhance shrub growth, thereby exacerbating wildfire hazards and shortening the lifespan of fuel treatment effectiveness. This work could be used to plan additional fuel treatments and schedule maintenance of existing treatments. A recent analysis of the spatial scale of Sierra Nevada fuel treatments revealed that the current rate of treatment is insufficient to significantly advance restoration goals (North et al., in press), which emphasizes the need for continued and accelerated fire hazard reduction on large forested areas.

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CHAPTER 2

Fuel Treatment Longevity in the Northern Sierra Nevada and Southern Cascades, California

Abstract

Mechanical thinning and prescribed burning treatments are commonly applied to abate wildfire hazards in dry western forests historically characterized by frequent, low-to-moderate intensity fire regimes. Although the stand structures and surface fuel reductions resulting from treatments are temporary, few studies have assessed the lifespan of these effects. I sampled surface fuels and vegetation following treatment for fire hazard reduction in a chronosequence of time since treatment in the northern Sierra Nevada and southern Cascade regions of California. Field data were used to aid fuel model selection and to parameterize Fuels Management Analyst Plus, a fire behavior and effects model. A semi-qualitative, semi-quantitative assessment of ladder fuel hazard was applied to supplement modeled fire behavior metrics. Potential fire behavior and effects were compared among time-since-treatment and untreated control groups. Untreated sites exhibited fire behavior that would challenge wildfire suppression efforts, and projected overstory mortality was considerable. In contrast, fire behavior and severity were low to moderate in even the oldest fuel treatments (8-26 years). Findings indicate that in the forest types characteristic of the northern Sierra Nevada and southern Cascades, treatments for wildfire hazard reduction retain their effectiveness for more than 10-15 years and possibly beyond a quarter century.

Introduction

Historic fire regimes of many North American dry forest types are characterized by frequent, low- to moderate-severity fires. Wildfire exclusion and other forest management activities such as livestock grazing and timber harvest have altered the structure and composition of forest vegetation (Savage and Swetnam, 1990; Fulé et al., 1997) and allowed surface fuels to accumulate (Covington and Moore, 1994; Swetnam et al., 1999). Such changes have adversely impacted forest resistance and resilience to disturbance, because the increased quantity and continuity of forest fuels have increased the proportion of the landscape susceptible to high severity fire (Quigley et al., 1996). At present, wildfires are difficult and costly to control, and human communities are regularly threatened during the fire season.

Wildfire hazard reduction through treatment of wildland fuels has become a primary focus of forest management, particularly in the wildland-urban interface. Modifications to surface, ladder, and canopy fuels can reduce the severity of a future wildfire (Agee, 1996; van Wagtendonk, 1996; Pollet and Omi, 2002; Stephens et al., 2009) and support suppression activities through increased access and reduced fire intensity (Moghaddas and Craggs, 2007). However, low hazard fuel conditions must be maintained, or they will eventually lose their effectiveness. Forest managers must allocate scarce resources between the implementation of new treatments and maintenance of existing treatments, yet there is little research to inform future maintenance of treated sites.

Past evaluations of fuel treatment longevity have often relied upon computerized simulations of forest growth linked with fire behavior models (e.g. Collins et al., 2011), such as the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) (Reinhardt and Crookston, 2003). Such simulations require many assumptions on the part of the modeler. In particular, some models or variants within models do not adequately predict some aspects of forest growth, such as seedling regeneration and understory growth, necessitating user input. Yet regeneration is an important influence on surface fire intensity and crown fire potential. Battaglia et al. (2008) estimated that prescribed burning would be required every 10 years in order to maintain low densities of seedling regeneration in ponderosa pine stands. Beyond the 15-year period, regrowth would achieve reduced susceptibility to burning mortality, and within 20 years, prescribed burning would be expected to produce some overstory mortality as a result of the entry of regeneration into the understory (Battaglia et al., 2008).

Given the probabilistic nature of wildfire occurrence, very few evaluations of fuel treatment longevity have been based upon empirical data. Those that exist are largely anecdotal, and constrained by limited replication and a lack of pre-burn data, which restrict consideration of the relative influence of site to site variability and fire weather conditions. Given these limitations, estimates of the lifespan of treatment effects in mixed conifer and yellow pine forests range from roughly 10 to 20 years (Biswell et al., 1973; van Wagtendonk, 1995; Agee and Skinner, 2005). For example, Biswell et al. (1973) reported from a casual survey that wildfires burning in an Arizona ponderosa pine forest that had burned 15-17 years previously had mixed severity effects and included some crown fire activity, while wildfires burning in stands treated with a controlled burn 1-6 years previously produced very little tree mortality. A recent and relatively large empirical study of fuel treatment longevity, Safford et al. (2012), was based on 12 wildfires burning in 8 national forests in California. While the authors found no effect of treatment age on wildfire behavior or severity, the maximum age of sampled treatments was only 9 years.

The present study takes a chronosequence approach to evaluate the longevity or “temporal persistence” (Fernandes, 2009) of reduced fire hazard derived from fuels management activities. Field data collected in 52 stands of varying time since treatment and 13 untreated stands are used to parameterize a fire behavior and effects model in order to assess post-treatment wildfire hazard development. One focus of fuels reduction activities is increasing crown fire resistance by targeting the fuel “ladders” that carry a surface fire into the forest canopy. Despite the hazard represented by ladder fuels, these fuels are difficult to assess in the field. In addition to computerized simulations of fire behavior, a semi-qualitative, semi-quantitative protocol developed by Menning and Stephens (2007) is applied to assess ladder fuel hazard in treated and untreated stands. Data are stratified by forest type (eastside pine and mixed conifer), method of fuel treatment (mechanical thin or mechanical thin and burn), and major slope aspect (north and south). Fire hazards, including fire behavior and projected tree mortality, are expected to increase with time following treatment, while untreated sites are predicted to exhibit high fire hazards relative to recently treated sites. Alternatively, temporal trends in potential fire hazards may be obscured if variations in stand conditions have been influenced by other factors, such as growing conditions and treatment prescriptions.

Methods

Fuel treatments

The fuels reduction treatments sampled in this study were established to alter fire behavior and effects through reduced surface fuel loads and/or continuity of vegetation. Most treatment sites selected for sampling are located within the Quincy Library Group Forest Recovery Act (Herger and Feinstein, 1998) Pilot Project purview area, which includes the Lassen and Plumas National Forests and the Sierraville District of the Tahoe National Forest in the northern Sierra Nevada and southern Cascades bioregions. Treatments located on the Truckee District of the Tahoe National Forest were also included. Of the 52 treatments sampled in this study, 11 were located on private land, and supplemented the National Forest sites. Nine of these were located on land belonging to the Collins Pine forest products company while the remaining two had been implemented by Fire Safe Councils on privately owned land.

Local forest managers helped to identify treatment units suitable for sampling. Sites were treated with mechanical thinning (hereafter “thin only”) or mechanical thinning in combination with burning (hereafter “thin and burn”). The thin and burn treatments included both broadcast prescribed fire and slash pile burning. If applicable, all follow-up burning was to occur within three years of the thinning treatment. US Forest Service treatment projects are typically implemented in multiple forest stands over a period of several years. In order to avoid possible pseudoreplication arising from adjacent unit locations and identical timber operators, a single unit was randomly selected to represent each treatment project. All treatments fitting the study design specifications were sampled.

Field sampling also included untreated control sites, which were intended to approximate pre-treatment conditions. Untreated sites were defined as those having experienced neither harvesting activity nor burning within the preceding 25 years. All stands located directly adjacent to sampled treatment sites were considered for sampling, and all candidate sites were evaluated in the field. The presence of recent stumps was used as an indicator of past management, while stem charring and ash signified fire, and such sites were excluded. Sites were also excluded from sampling if the species composition of the overstory clearly differed from that of the adjacent treated site. This occurred in only a small number of cases (1-2), and appeared to be the result of a change in soil type or elevation near a treatment boundary. In addition, because mechanical thinning equipment is generally restricted to slopes of less than 30 percent grade, candidate control sites with slopes exceeding 30 percent were excluded from sampling.

Dead fuels and understory and overstory vegetation were sampled within 3 plots representing each treated and untreated site. In total, 52 treatment sites with ages 2-26 years following initial treatment and 13 untreated sites were sampled. Sampling sites were stratified by treatment type (thin only, thin and burn, and untreated), forest type (mixed conifer and eastside pine), and major slope aspect (north- and south-facing). Appendix 1 contains a description of the sites sampled in this study including geographical coordinates.

Field sampling

Downed woody fuels, understory composition, and overstory characteristics were sampled using a systematic sampling design with a random starting point. Three 0.1-ha circular plots were established within each treatment unit. Plots were placed parallel to the treatment boundary (usually a road) and arranged 86 m (50 m plus two times plot radius) apart and 48 meters (30 m plus plot radius) from the boundary. The sampling design and collection of fuels and vegetation data are fully described in chapter 1 of this dissertation. Tables 2.1 and 2.2 contain surface fuel and stand data (respectively) for time-since-treatment and untreated groups.

The following outlines the hazard assessment protocol applied to each sampling plot to characterize ladder fuel hazards; for a thorough description of the method, the reader is referred to Menning and Stephens (2007). The protocol uses a flow chart method to assign a descriptive hazard rating to each quadrant of each sampling plot to quantify the continuity of the best fuel ladder within each quadrant. The protocol was applied to each quadrant within a 12.6 m radius of plot center. Within each quadrant, the height to crown base (defined as the base of either live crown or clumped dead branches) and the size of the gap in the best fuel ladder was estimated to the nearest 1 m. A hazard rating was assigned based on overstory and ladder fuel continuity and concentration of understory fuels. The following describes the ladder fuel hazard categories (categories are illustrated in Figure 2.1):

A. High Hazard: Low aerial fuels, defined as small trees, shrubs, and low-hanging tree branches, are clumped (filling a minimum contiguous area of approximately 4 m^2) and the size of the gap in the best fuel ladder in the quadrant is < 2 m. Overstory fuels are present and continuous.

B. Moderate Hazard: Low aerial fuels are clumped but the largest discontinuity in the best fuel ladder equals or exceeds 2 m

C. Moderate Hazard: Low aerial fuels are not present or not clumped and the largest gap in the best fuel ladder is < 2 m.

D. Low Hazard: Low aerial fuels are not sufficiently clumped and ladder gaps are ≥ 2 m in height.

E. No Canopy/Low Hazard: quadrant contains no trees, or the tree canopies within the quadrant are not linked with other overstory fuels. A potential wildfire would not likely spread into the forest canopy.

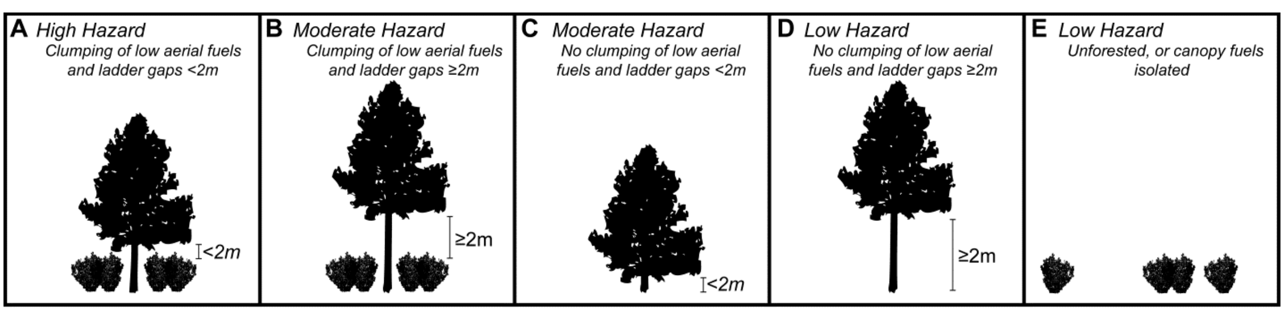

Figure 2.1. Illustration of ladder hazard categories. Redrawn with permission from Menning and Stephens (2007).

Table 2.1. Forest floor and woody fuel loads [mean Mg/ha (standard error)] in treated and untreated sites in eastside pine and mixed conifer sites in the southern Cascades and northern Sierra Nevada, California.

Forest Type	TST*	Duff	Litter	1-h (<0.64 cm diameter)	10-h (0.64–<2.54 cm diameter)	100-h (2.54–<7.62 cm diameter)	1000-h (≥7.62 cm diameter) rotten	1000-h (≥7.62 cm diameter) sound
Eastside Pine	2-4	32.4 (5.6)	8.7 (1.2)	0.2 (0.1)	2.3 (0.5)	3.7 (0.9)	3.1 (1.7)	11.8 (3.6)
	5-7	35.2 (4.2)	7.9 (0.4)	0.1 (0.0)	2.0 (0.4)	3.3 (0.8)	3.7 (1.3)	4.8 (1.3)
	8+	30.9 (6.6)	12.9 (1.6)	0.5 (0.1)	2.5 (0.5)	4.2 (0.8)	8.0 (3.2)	6.3 (3.0)
	Untreated	27.4 (5.4)	23.8 (6.6)	0.5 (0.1)	2.3 (0.4)	3.8 (0.9)	14.6 (4.6)	12.2 (3.9)
Mixed Conifer	2-4	48.8 (10.2)	15.6 (1.4)	0.7 (0.2)	3.6 (0.8)	4.7 (0.8)	13.9 (4.0)	7.9 (2.1)
	5-7	37.1 (7.6)	11.4 (0.9)	0.5 (0.1)	3.4 (0.6)	4.8 (0.7)	15.8 (6.2)	4.8 (1.3)
	8+	21.5 (3.7)	15.5 (1.3)	0.6 (0.1)	2.2 (0.3)	4.1 (0.6)	23.4 (12.4)	13.8 (5.9)
	Untreated	74.4 (19.5)	38.1 (7.6)	1.0 (0.2)	4.3 (1.0)	5.7 (1.2)	30 (7.2)	6.9 (2.4)

Hour categories refer to time lag classes. Different letters in a column (blocked by forest type) indicate significant difference at $\alpha = 0.05$. *TST: Time Since Treatment (yrs)

Table 2.2. Stand characteristics in treated and untreated sites in eastside pine and mixed conifer sites in the southern Cascades and northern Sierra Nevada, California.

Forest Type	TST*	Shrub cover (%)	Tree density (stems ha^{-1})	Basal area (m^2 ha^{-1})	QMD (cm)	Canopy base height (m)	Canopy bulk density (kg m^{-2})	Canopy cover (%)
Eastside Pine	2-4	7.9 (2.6)	258 (32)	25.2 (2.8)	36.0	3.1 (0.7)	0.06 (0.01)	31 (3)
	5-7	7.9 (1.8)	182 (13)	23.9 (2.3)	41.1	3.7 (0.4)	0.04 (0.01)	46 (5)
	8+	5.5 (2.4)	346 (91)	25.3 (1.4)	35.0	2.5 (0.6)	0.05 (0.01)	27 (2)
	Untreated	5.1 (1.6)	1285 (162)	53.0 (4.9)	23.5	1.1 (0.2)	0.19 (0.02)	48 (4)
Mixed Conifer	2-4	13.9 (6.9)	372 (47)	33.7 (3.2)	35.1	3.7 (0.8)	0.06 (0.01)	35 (3)
	5-7	4.8 (2.2)	336 (62)	32.0 (6.6)	37.4	4.0 (1.2)	0.06 (0.01)	45 (6)
	8+	13.9 (7.3)	388 (55)	27.7 (4.7)	31.1	5.3 (0.7)	0.06 (0.01)	49 (3)
	Untreated	7.4 (3.4)	1406 (119)	50.3 (2.7)	21.6	1.0 (0.2)	0.12 (0.01)	67 (5)

*TST: Time Since Treatment (yrs)

Fire Behavior and Effects Modeling

The Crown Mass program (v. 3.0.49) within the Fuels Management Analyst Plus suite (FMA Plus, www.fireps.com) was used to estimate overstory metrics and potential fire behavior and tree mortality at the stand level (Carlton, 2005). Fire behavior output variables include surface rate of spread, flame length, and torching (TI) and crowning indices (CI). TI and CI, respectively, are the wind speeds at a height of 6.1-m that would permit crown fire initiation and active crown fire; low values represent higher susceptibility (Scott and Reinhardt, 2001). FMA Plus incorporates published methodologies to compute canopy fuel metrics (canopy base height and bulk density, stand height), fire behavior indices, and potential tree mortality. Inputs to the model include fire weather conditions, fuel models, topography, and tree lists. Tree lists are derived from field measurements of species, height, crown position, live crown ratio, and diameter. Stephens and Moghaddas (2005b; 2005a) provided a detailed summary of the methodologies used by FMA Plus to calculate canopy characteristics and potential fire behavior and effects. Fire behavior and effects were estimated for each sampling plot.

Fuel models are required to describe fuelbed attributes such as load, moisture of extinction, and heat content for modeling of potential fire behavior and effects. The process of assigning discrete fuel models to represent site conditions is susceptible to modeler subjectivity and is difficult to repeat across studies. To make the process more objective, I employed a classification and regression tree analysis (De'ath and Fabricius, 2000) following the methods of Collins et al. (2011) to bin plots according to their similarity with respect to fuel characteristics. The R package mvpart (Therneau et al., 2012) was used to construct regression trees predicting shrub cover, small surface fuels (combined loads of litter and 1-, 10-, and 100-h fuels), and coarse woody fuels. Forest type, basal area, tree density, canopy cover, dominant and codominant tree height, treatment method, slope aspect, and site index summarized at the plot level served as predictor variables. Statistical fits were generally moderate ($R^2 = 0.2$-0.3). Binned plots were then assigned to standard Scott and Burgan (2005) fuel models using the distribution of fuels among size class categories, plot photographs, and field notes to aid in model selection. The assignments were reviewed by local fire managers and fire science researchers familiar with the study area. Figure 2.2 describes the final selection logic used in binning, which was based on combined results from multiple regression tree analyses. Table 2.3 contains mean fuelbed characteristics of fuel model assignments and the proportion of plots assigned to each model with respect to forest type and method of treatment.

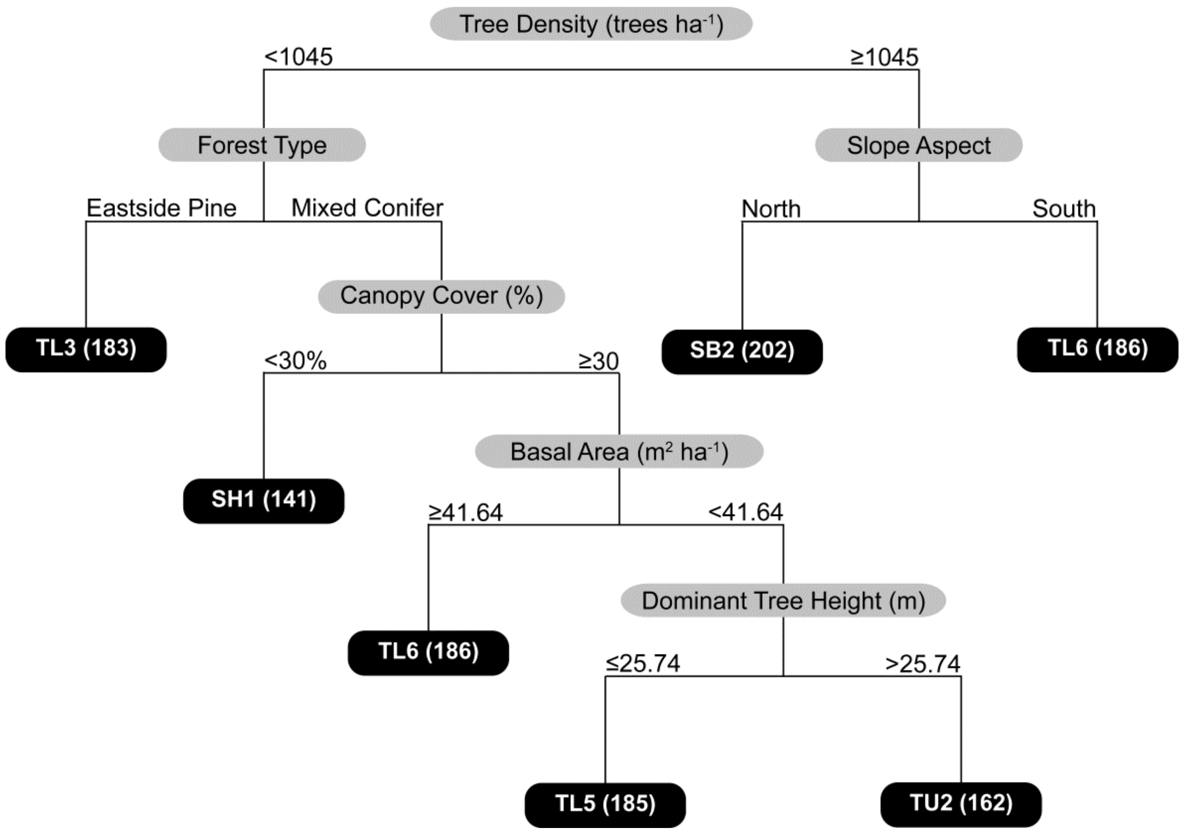

Figure 2.2. Categorical and regression tree describing fuel model selection logic. Terminal nodes are fuel model code (fuel model number)(Scott and Burgan, 2005). See Methods section for description of tree development.

Table 2.3. Mean surface fuelbed characteristics of plots assigned to Scott and Burgan (2005) fuel models for fire behavior modeling, and their relative proportions according to forest type and method of treatment.

Fuel model code (model number)	Plots assigned (count)	Shrub cover (%)	Litter (Mg ha^{-1})	1, 10, 100-h (Mg ha^{-1})	1000-h (Mg ha^{-1})	Proportion of plots (% by forest type)	Proportion of plots (% by treatment method)
SB2(202)	15	8.9	47	11.8	43.3	33% EP, 67% MC	100% U
SH1(141)	12	25.6	12.3	6.8	42.7	100% MC	67% TB, 33% TO
TL3(183)	102	7.2	9.8	6.1	13.0	100% EP	26% TB, 67% TO, 7% U
TL5(185)	31	6.4	15.5	8.2	15.4	100% MC	42% TB, 52% TO, 6% U
TL6(186)	28	2.5	20.3	8	24.7	36% EP, 64% MC	11% TB, 36% TO, 54% U
TU2(162)	8	24.1	17.4	9	32.8	100% MC	50% TB, 50% TO

Forest types: **EP**: eastside pine, **MC**: mixed conifer; Fuel treatment methods: **TB**: mechanical thin and burn, **TO**: mechanical thin only, **U**: untreated

Table 2.4. Weather inputs to FMAPlus representing moderate, high, and extreme conditions (upper 80th, 90th, and 97.5th percentile, respectively).

	Remote Automated Weather Station											
Weather parameter	Chester			Pierce			Quincy Rd			Stampede		
Weather percentile	80	90	97.5	80	90	97.5	80	90	97.5	80	90	97.5
Probable maximum 1-min wind speed (km/h) (Crosby and Chandler, 1966)	21	24	30	19	21	23	32*	35*	41*	19	23	29
Dry-bulb temperature (°C)	31	33	35	30	31	34	36	37	40	29	31	33
Relative humidity (%)	14	12	8.5	13	11	8	12	9	6.5	12	10	6.5
1-h fuel moisture (%)	2.3	2	1.5	2.2	1.9	1.4	2	1.5	1.1	2.1	1.7	1.1
10-h fuel moisture (%)	3.2	2.9	2.3	2.9	2.5	2.0	3.2	2.7	2.1	3.2	2.7	2.1
100-h fuel moisture (%)	6.4	5.8	5.2	5.2	4.6	3.9	7.6	6.5	5.5	6.8	6.1	5.0
Herbaceous fuel moisture (%)	30	30	30	30	30	30	30	30	30	30	30	30
Woody fuel moisture (%)	70	70	70	70	70	70	70	70	70	60	60	60
Foliar fuel moisture (%)	105	100	90	105	100	90	105	100	90	105	100	90

*Calculated from Cashman RAWS weather data

Fire behavior was modeled for upper 80^{th}, 90^{th}, and 97.5^{th} percentile weather conditions, representing moderate, high, and extreme fire weather, respectively. Percentile weather indices were calculated using Fire Family Plus software (Main et al., 1990). Four remote automated weather stations (RAWS) were selected to represent conditions for the 53 treatment and 13 control sites (Figure 2.3) based on recommendations from local USDA Forest Service fire and fuel managers. Twenty years of weather data (1991-2000) for the June 1-September 30 period, the typical fire season in the northern Sierra Nevada and southern Cascades, were included in the analysis. In the case of one RAWS (Quincy Rd), managers noted that wind estimates tended to poorly represent local conditions. As a remedy, wind data from a fifth station located near the Quincy Rd RAWS, Cashman, were substituted for the Quincy wind data in percentile wind calculations. Table 2.4 contains percentile weather indices used in fire behavior and effects modeling.

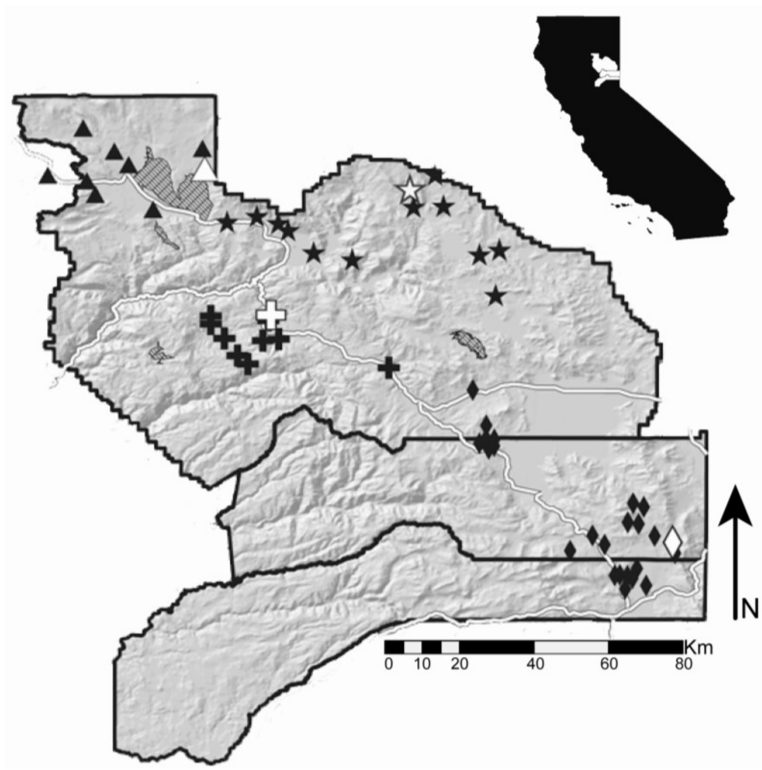

Figure 2.3. Map of study sites and remote automated weather stations (RAWS). Black symbols represent sampled treatment areas (n=53). Control sites (n=13) were located within 200 m of treatment areas, and are not shown. White symbols represent RAWS. Historical weather data from a given RAWS was used to parameterize the fire behavior model for treatment areas with matching symbol types.

Data analysis

To examine chronosequence trends in potential fire behavior and effects, sites were divided into 3 post-treatment age classes, 2-4 years (mean = 3.0, median = 3.0, n = 18), 5-7 years (mean = 6.2, median = 6.5, n = 18), and 8+ years (mean = 11.3, median = 10, n = 16) and 1 untreated class (n = 13). Classes were defined to equalize, to the degree possible, the number of samples within each class. For all continuous variables, values reported in figures and tables are the means calculated at the site level. With the exception of ladder fuel hazard ratings, which are reported at the quadrant level, categorical variables such as fire type were calculated from estimates assessed at the plot level.

Mean and standard errors were calculated for fire behavior indices including rate of spread, flame length, and torching and crowning indices. Analysis of fire behavior across treatment groups was conducted using nonparametric permutation tests because many measures did not meet assumptions of parametric statistical tests even after transformation. Where significant differences between groups were indicated ($p < 0.05$), two-sample tests were performed. Permutation tests were conducted using the functions 'permKS' and 'permTS' of the R package 'perm', package version 2.13.2; (Fay and Shaw, 2010). Because the parameters applied to represent fuel and fire weather conditions in fire behavior modeling were not unique for each sampling plot, fire behavior estimates are less variable than would be expected under real-world circumstances. For this reason, significance levels of the results should be interpreted with caution.

The statistical software R, version 2.13.1 (R Development Core Team, 2011), was used to conduct analyses and develop figures.

Results

Surface fire behavior

Potential surface fire behavior estimated for treated sites was generally of low intensity, while untreated sites exhibited more extreme fire activity. In the eastside pine forest type, predicted flame lengths for untreated sites within a given fire weather scenario were approximately three times those predicted for treated sites (Table 2.5). Surface fire rates of spread were low in treatment units and moderate in untreated sites. For a given weather scenario, mean predicted rate of spread in the untreated units was four times that of any treatment age class. Both measures of surface fire behavior were significantly different between the untreated control and every time-since-treatment class but varied little with time since treatment.

For the mixed conifer forest type, predicted surface fire behavior increased from low to moderate with increasing fire weather severity for treated sites, and moderate to high for untreated sites (Table 2.6). Mean flame lengths and rates of spread were lowest in the 5-7 years-since-treatment class. Mean values for both surface fire behavior metrics were approximately doubled between untreated and treated sites within a given fire weather scenario.

Table 2.5. Modeled fire behavior for eastside pine forest type under moderate (80th percentile), high (90th percentile), and extreme (97.5th percentile) fire weather conditions. Different letters indicate significance (permutation test, $p < 0.05$).

Weather percentile	Time since treatment (yrs)	Surface fire flame length (m)	Surface fire rate of spread (m min^{-1})	Torching index (km h^{-1})	Crowning index (km h-^{1})	Fire type
80th	2-4	0.4 (0.0) ^{a}	0.6 (0.0) ^{a}	337.7 (78.6) ^{a}	50.8 (4.3) ^{a}	90% SF, 10% PCF
	5-7	0.4 (0.0) ^{a}	0.6 (0.0) ^{a}	322.4 (52.1) ^{a}	122.3 (35.6) ^{b}	95% SF, 5% PCF
	8+	0.4 (0.0) ^{a}	0.7 (0.1) ^{a}	255.5 (67.7) ^{a}	65.1 (7.0) ^{ab}	78% SF, 22% PCF
	Untreated	1.1 (0.2) ^{b}	2.5 (0.7) ^{b}	33.0 (11.0) ^{b}	20.3 (1.4) ^{c}	38% SF, 38% PCF, 24% APDCF
90th	2-4	0.4 (0.0) ^{a}	0.8 (0.0) ^{a}	299.9 (69.4) ^{a}	50.3 (4.2) ^{a}	90% SF, 10% PCF
	5-7	0.4 (0.0) ^{a}	0.8 (0.0) ^{a}	285.9 (46.5) ^{a}	119.6 (34.8) ^{b}	95% SF, 5% PCF
	8+	0.4 (0.0) ^{a}	0.8 (0.1) ^{a}	227.8 (60.7) ^{a}	63.8 (6.9) ^{ab}	78% SF, 22% PCF
	Untreated	1.2 (0.3) ^{b}	3.2 (0.9) ^{b}	29.0 (9.7) ^{b}	19.8 (1.4) ^{c}	38% SF, 10% PCF, 52% APDCF
97.5th	2-4	0.5 (0.0) ^{a}	1.1 (0.0) ^{a}	241.2 (55.9) ^{a}	48.7 (4.0) ^{a}	90% SF, 10% PCF
	5-7	0.5 (0.0) ^{a}	1.1 (0.0) ^{a}	232.4 (37.9) ^{a}	116.1 (33.7) ^{b}	95% SF, 5% PCF
	8+	0.5 (0.0) ^{a}	1.1 (0.1) ^{a}	183.6 (48.6) ^{a}	62.0 (6.7) ^{ab}	78% SF, 22% PCF
	Untreated	1.5 (0.3) ^{b}	4.7 (1.3) ^{b}	22.8 (7.8) ^{b}	19.1 (1.4) ^{c}	24% SF, 5% PCF, 71% APDCF

Values are means (standard errors). **APDCF**: active plume-dominated crown fire, **PCF**: passive crown fire, **SF**: surface fire

Table 2.6. Modeled fire behavior for mixed conifer forest type under moderate (80th percentile), high (90th percentile), and extreme (97.5th percentile) fire weather conditions. Different letters indicate significance (permutation test, $p < 0.05$).

Weather percentile	Time since treatment (yrs)	Surface fire flame length (m)	Surface fire rate of spread (m min^{-1})	Torching index (km h^{-1})	Crowning index (km h-^{1})	Fire type
80th	2-4	1.1 (0.1) ^{a}	3.7 (0.5) ^{ab}	80 (18.9) ^{a}	56.9 (7.3) ^{a}	75% SF, 25% PCF
	5-7	0.9 (0.1) ^{a}	2.6 (0.5) ^{a}	92.3 (27.2) ^{a}	57.1 (5.4) ^{a}	93% SF, 7% PCF
	8+	1.1 (0.1) ^{a}	3.7 (0.5) ^{ab}	116.1 (7.0) ^{a}	51.6 (5.1) ^{a}	91% SF, 9% PCF
	Untreated	1.9 (0.3) ^{b}	5.9 (1.2) ^{b}	8.7 (4.3) ^{b}	29.2 (2.4) ^{b}	11% SF, 61% PCF, 28% APDCF
90th	2-4	1.2 (0.1) ^{a}	4.5 (0.6) ^{ab}	71.0 (16.8) ^{a}	55.5 (7.1) ^{a}	67% SF, 33% PCF
	5-7	1.0 (0.1) ^{a}	3.1 (0.7) ^{a}	82.9 (24.4) ^{a}	56.0 (5.4) ^{a}	93% SF, 7% PCF
	8+	1.2 (0.1) ^{a}	4.4 (0.7) ^{ab}	104.0 (6.3) ^{a}	50.5 (5.0) ^{a}	91% SF, 9% PCF
	Untreated	2.1 (0.3) ^{b}	7.2 (1.5) ^{b}	7.5 (3.8) ^{b}	28.5 (2.4) ^{b}	0% SF, 44% PCF, 56% APDCF
97.5th	2-4	1.4 (0.1) ^{a}	6.0 (0.8) ^{ab}	59.4 (14.3) ^{a}	53.9 (7.0) ^{a}	50% SF, 46% PCF
	5-7	1.2 (0.1) ^{a}	4.1 (0.9) ^{a}	69.2 (20.7) ^{a}	54.4 (5.2) ^{a}	87% SF, 13% PCF
	8+	1.3 (0.1) ^{a}	5.6 (0.9) ^{ab}	87.4 (5.4) ^{a}	49.1 (4.9) ^{a}	82% SF, 14% PCF, 4% APDCF
	Untreated	2.5 (0.4) ^{b}	9.8 (1.9) ^{b}	5.9 (3.1) ^{b}	27.6 (2.3) ^{b}	0% SF, 22% PCF, 78% APDCF

Values are means (standard errors). **APDCF**: active plume-dominated crown fire, **PCF**: passive crown fire, **SF**: surface fire

To relate the surface fire behavior parameters estimated here with the challenges such behavior would present to fire suppression operations, mean estimates for the time-since-treatment age classes and the untreated group are displayed on fire characteristics charts in Figure 2.4. The fire characteristics chart, or "hauling" chart, was originally developed by Andrews and Rothermel (1982) to simultaneously display multiple surface fire behavior metrics, and was adapted as a flexible stand-alone computer application by Andrews et al. (2011). Curves represent several flame length ranges that correspond to rules of thumb for fire suppression practitioners. Flame length and fireline intensity are related to the heat felt by a person standing near the flames, and so have been linked to fire suppression activities. For a fire burning with low intensity, direct attack suppression methods at the head of the fire can be employed safely and effectively. At the next range of intensity, represented by the second curve in Figure 2.4, bulldozers and aircraft are likely needed, as intensity levels are too high to allow direct attack at the head of the fire and hand line cannot be relied upon to control the fire. For the third intensity range, with flame lengths between 2.4 and 3.4 m, passive and active crown fire behavior and spotting are possible or likely, and direct attack efforts at the head of the fire may be ineffective. At the highest levels of fire intensity, when flame lengths exceed 3.4 m, active crown fire behavior and spotting are likely and direct attack will not be an effective method of control.

For eastside pine sites, mean fire intensity metrics for all treatment age classes and fire weather conditions fell within the low intensity category, indicating potential wildfires could be effectively controlled through direct attack using hand crews (Figure 2.4). Under moderate fire weather conditions, even the untreated sites did not exhibit surface fire behavior that would render direct attack at the head of the fire with hand crews impracticable. Surface fires burning under extreme weather conditions in the untreated sites may be of moderate intensity which could preclude hand line control but permit control using other firefighting equipment (dozers, aircraft). Mixed conifer sites appear more susceptible to surface fire behavior that could challenge fire suppression activities. Under extreme fire weather conditions, many treated sites fell within the second intensity category. The untreated mixed conifer sites were especially likely to exhibit surface fire behavior that would present a serious challenge to control efforts. Under the most extreme conditions, mean surface fire intensities placed the untreated mixed conifer sites within the third intensity category.

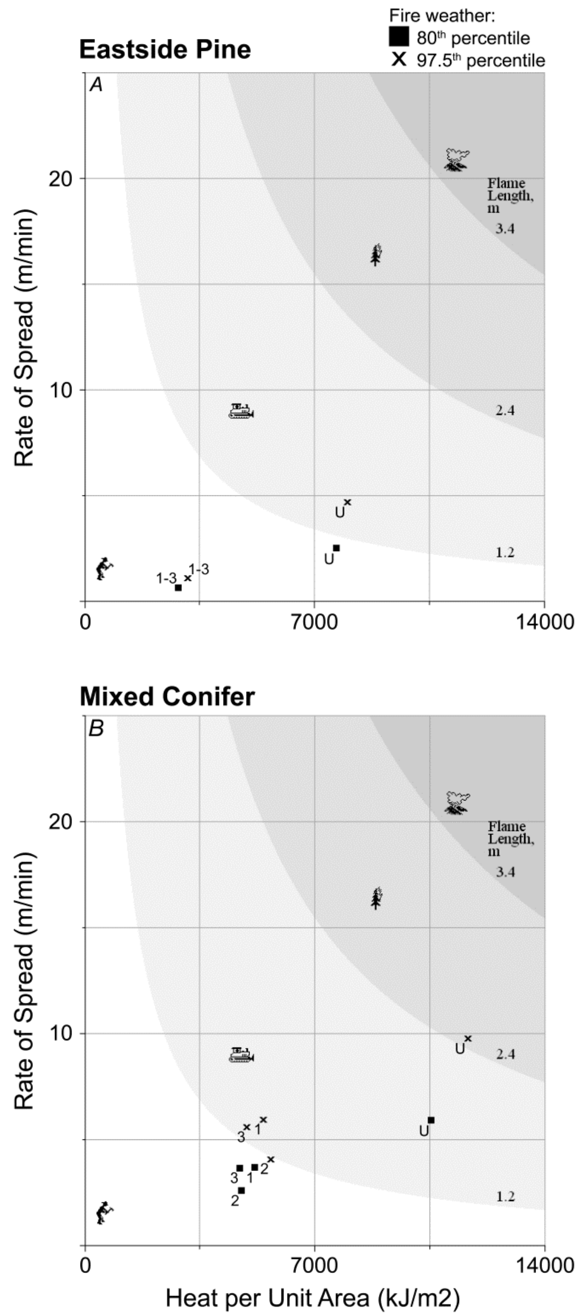

Figure 2.4. Mean surface fire behavior in the eastside pine (A) and mixed conifer (B) forest types under 80th and 97.5th percentile fire weather conditions. Symbol labels indicate time-since-treatment and control classes: “1”, “2”, “3”, and “U” represent 2-4, 5-7, and 8+ years-since-treatment and untreated classes, respectively. Filled curves indicate flame length ranges. Illustrated symbols represent effective control methods for the corresponding surface fire behaviors. See text for further discussion of the charts.

Ladder fuel hazard

In the assessment of ladder fuel hazard, the height to crown base and the size of the gap in the best fuel ladder were lowest for the untreated category in each forest type (Table 2.7). Estimates for the oldest treatment class tended to be intermediate between the five to seven years since treatment category and the untreated group, though mean values for both metrics were still approximately double those of the untreated group. The proportion of quadrants assigned a high ladder fuel hazard rating exhibited no trend with time since treatment (Figure 2.5). Quadrants treated within seven years of sampling were frequently rated low hazard. 60% of eastside pine quadrants belonging to the oldest time-since-treatment group received a low hazard rating compared with only 30% of untreated quadrants. For the mixed conifer forest type, the proportion of quadrants assigned a low hazard rating was similar between the oldest treatment class and the control (42 and 44%, respectively). For both forest types, the proportion of quadrants given a high (A) or moderate (B or C) rating was highest in the untreated category (56 and 70% in the mixed conifer and eastside pine forest types, respectively). Very few untreated quadrants received an E rating, which describes a stand with either few trees or trees with crowns that are poorly connected to the forest canopy. Only 1 and 2% of untreated quadrants received an E rating in the mixed conifer and eastside pine forest types, respectively, compared with 12 and 30% of treated quadrants.

Table 2.7. Mean and standard error values for the height to crown base and largest gap size in the best fuel ladder measured in each quadrant, and proportion of quadrants grouped according to ladder fuel hazard rating.

Forest type	Time since treatment (yrs)	Height to crown base (m)	Gap size (m)	Ladder fuel hazard rating (proportion of quadrants)
Eastside Pine	2-4	6.0 (1.1)	5.9 (1.1)	77% low, 13% mod, 11% high
	5-7	5.3 (0.9)	5.2 (0.9)	86% low, 11% mod, 3% high
	8+	3.2 (0.7)	3.1 (0.8)	60% low, 31% mod, 8% high
	Untreated	1.6 (0.2)	1.5 (0.2)	30% low, 62% mod, 8% high
Mixed Conifer	2-4	5.3 (0.9)	5.1 (0.9)	66% low, 20% mod, 15% high
	5-7	5.6 (0.9)	5.5 (0.9)	78% low, 17% mod, 5% high
	8+	4.5 (0.9)	4.1 (0.9)	42% low, 42% mod, 16% high
	Untreated	2.2 (0.6)	2.2 (0.6)	44% low, 50% mod, 6% high

low = D and E ratings, mod = B and C ratings, high = A rating (see Figure 2.1 for category descriptions)

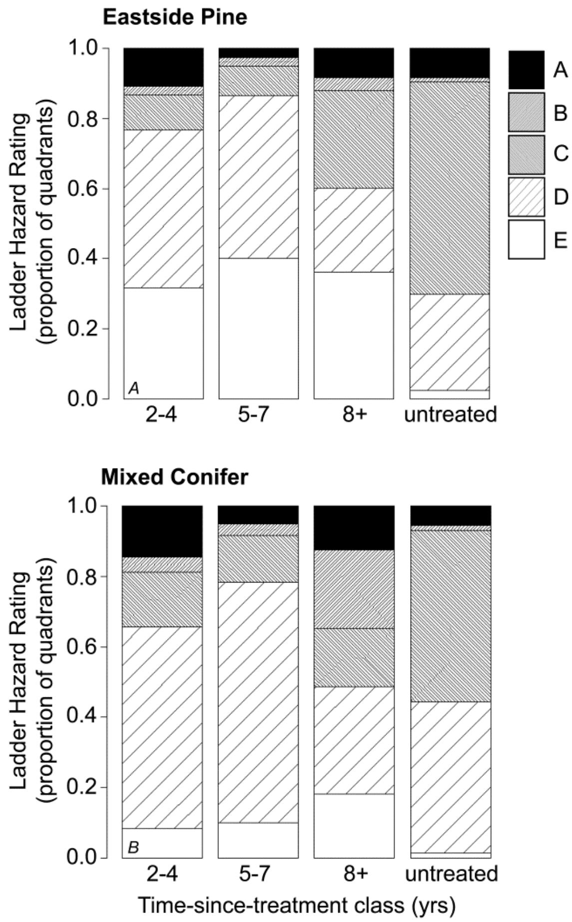

Figure 2.5. Ladder hazard in eastside pine (*A*) and mixed conifer (*B*) sites, calculated as a proportion of quadrants representing each age class/forest type combination. Bar fill patterns represent hazard rating categories: high hazard (A), moderate hazard (B and C), low hazard (D), and nonforest or discontinuous canopy fuels (E).

Potential crown fire behavior

Torching and crowning indexes (TI and CI, respectively) reflect stand susceptibility to crown fire. In the eastside pine forest type, probable maximum one-minute wind speed winds exceeded the predicted TI for very few treated sites, and did not exceed predicted CI for any treated site (Figure 2.6). Very high mean TI's and high CI's were estimated for all treatment age classes. In contrast, under extreme weather conditions, TI and CI values calculated for most untreated sites fell below the estimated mean 97.5th percentile one-minute wind speed (30.9 km h^{-1}). Under moderate conditions, mean predicted torching index for untreated sites exceeded the 80th percentile one-minute wind speed by a margin of 10.1 km h^{-1}. Mean CI for the control fell below 80th percentile one-minute wind speed. The estimation of predicted fire type takes into account both the torching and crowning indexes. Surface fire behavior is predicted when winds do not exceed either the TI or the CI; passive crown fire occurs when winds exceed the TI, but not the CI; and active crown fire behavior is predicted when percentile winds exceed both the predicted TI and CI. Even under extreme weather conditions, active crown fire was predicted only for untreated plots (Table 2.5). While the proportion of treated plots within each fire type category remained constant as fire weather increased in severity, the proportion of untreated plots exhibiting passive and active crown fire activity increased. Active crown fire behavior was predicted in greater than half of untreated plots in the high and extreme fire weather scenarios.

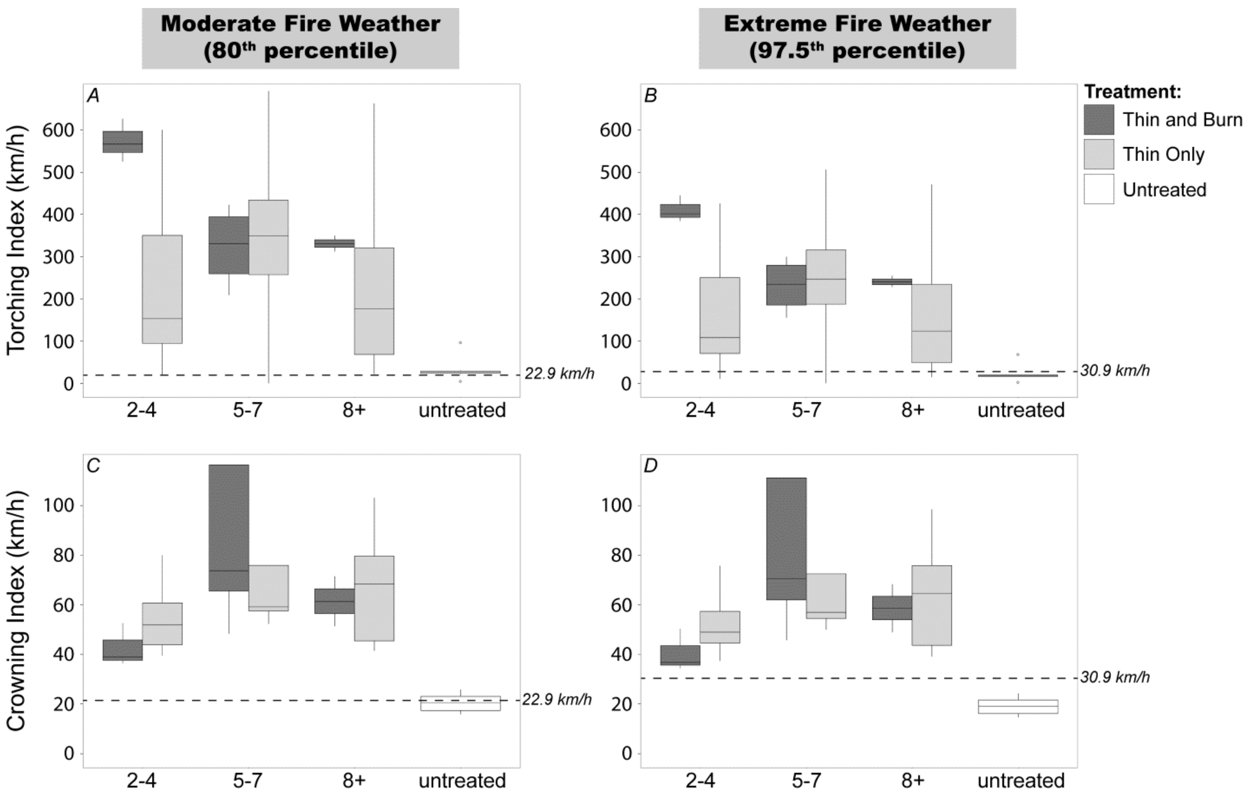

Figure 2.6. Potential crown fire behavior under 80th and 97.5th percentile fire weather conditions, modeled for eastside pine sites. Horizontal lines within each box denote the median, box length reflects the interquartile range, and whiskers extend to the data point furthest from the box hinge that is within 1.5 times the interquartile range. Outliers are represented by unfilled circles. Three high value outliers (between 200 and 500 km/hr) were removed from each crowning index chart to improve readability. All 3 occurred in the 5-7 years since treatment class. For reference, dashed lines indicate probable maximum 1-minute wind speed (Crosby and Chandler, 1966), calculated as the average of percentile values from four remote automated weather stations (see text for station descriptions and Table 2.4 for weather parameters used in fire modeling).

Estimated mean TI values were substantially lower in the mixed conifer sites in comparison to eastside pine sites. However, treatment mean values still greatly exceeded the estimated probable one-minute wind speeds calculated for a given fire weather scenario (Figure 2.7). CI values were similar between forest types, and again, treatment means exceeded one-minute wind speeds. For untreated sites, mean TI values were very low (8.7 and 5.9 in the moderate and extreme weather scenarios, respectively). Mean CI under 80th percentile weather conditions exceeded the probable one-minute wind speed, but under 97.5th percentile conditions, mean CI fell below the one-minute wind speed. The low thresholds for torching and crowning in the untreated sites are reflected in the proportion of plots exhibiting crown fire behavior. Crown fire activity was predicted for all plots in the high and extreme fire weather scenarios. Untreated plots appear particularly vulnerable under the most extreme conditions, as 78% of plots were predicted to experience active crown fire. In contrast, treated plots appear very resistant to crown fire, with only surface fire behavior predicted for 50-90% of plots, depending on treatment age and fire weather. Active crown fire activity was predicted for only one treatment plot.

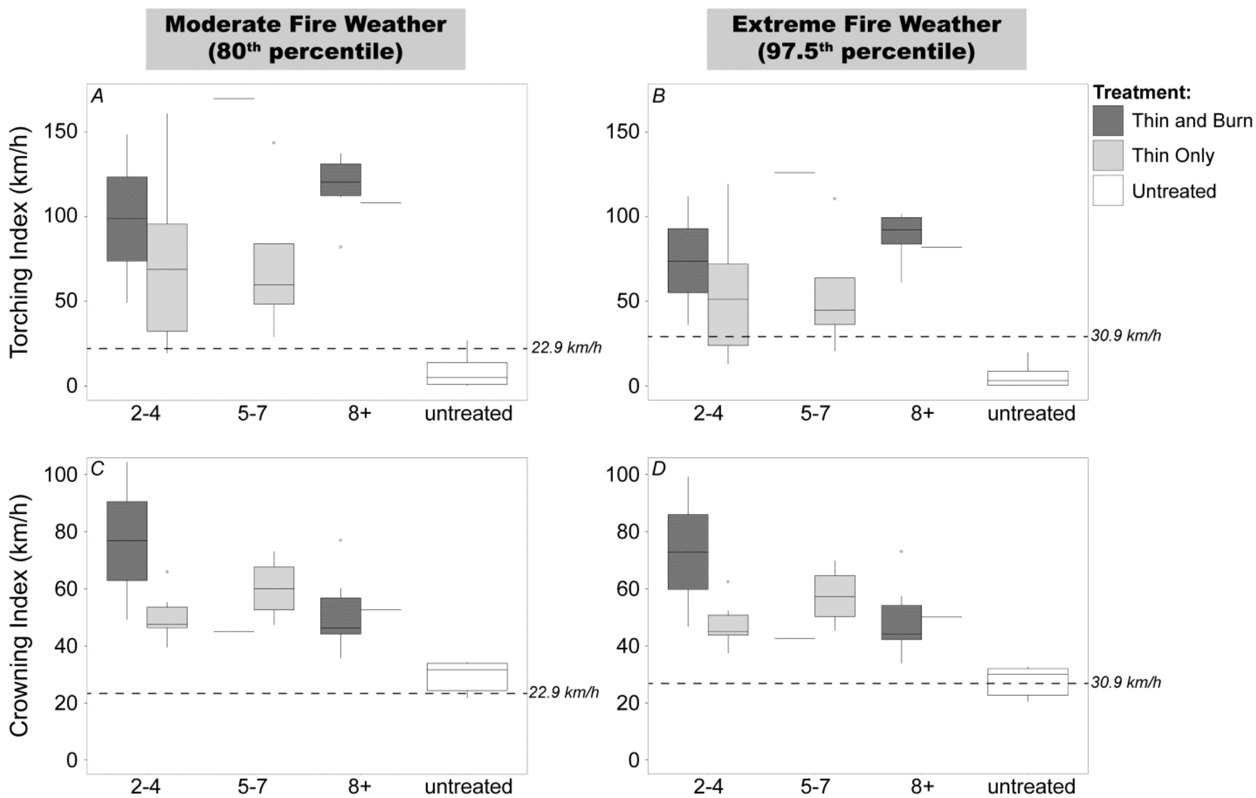

Figure 2.7. Potential crown fire behavior under 80th and 97.5th percentile fire weather conditions, modeled for mixed conifer sites. Horizontal lines within each box denote the median, box length reflects the interquartile range, and whiskers extend to the data point furthest from the box hinge that is within 1.5 times the interquartile range. Outliers are represented by unfilled circles. For reference, dashed lines indicate probable maximum 1-minute wind speed (Crosby and Chandler, 1966), calculated as the average of percentile values from four remote automated weather stations (see text for station descriptions and Table 2.4 for weather parameters used in fire modeling).

Fire effects

Changes in tree density and basal area were calculated with a very high threshold for probable tree mortality. FMA Plus produced an estimated probability of mortality for each tree in the input tree list. An estimated probability of mortality of 90% was selected for density and basal area change calculations. At lower levels of probability of mortality (50-75%), reductions in basal area for untreated sites were near 100% in both forest types.

Treated eastside pine sites appear very resistant to changes in basal area, even under extreme fire weather conditions (Figure 2.8). Predicted changes in tree density and basal area loss were somewhat greater for the oldest treatment class, with mean change in basal area of 17% compared with 7 and 1% in the 2-4 and 5-7 years-since-treatment classes. However, most sites had very low projected basal area loss, with median basal area values for all age classes below 1%. Untreated sites, in contrast, were predicted to experience substantial losses in live tree basal area, even under moderate conditions. Percent change in basal area increased from 50 to 67% for 80th and 97.5th percentile conditions, respectively. On average, the probability of mortality for trees larger than 60 cm dbh in the untreated sites was 48, 62, and 73% under moderate, high, and extreme weather conditions. In comparison, mean probability of mortality for the largest trees did not exceed 4% in any treatment age class group.

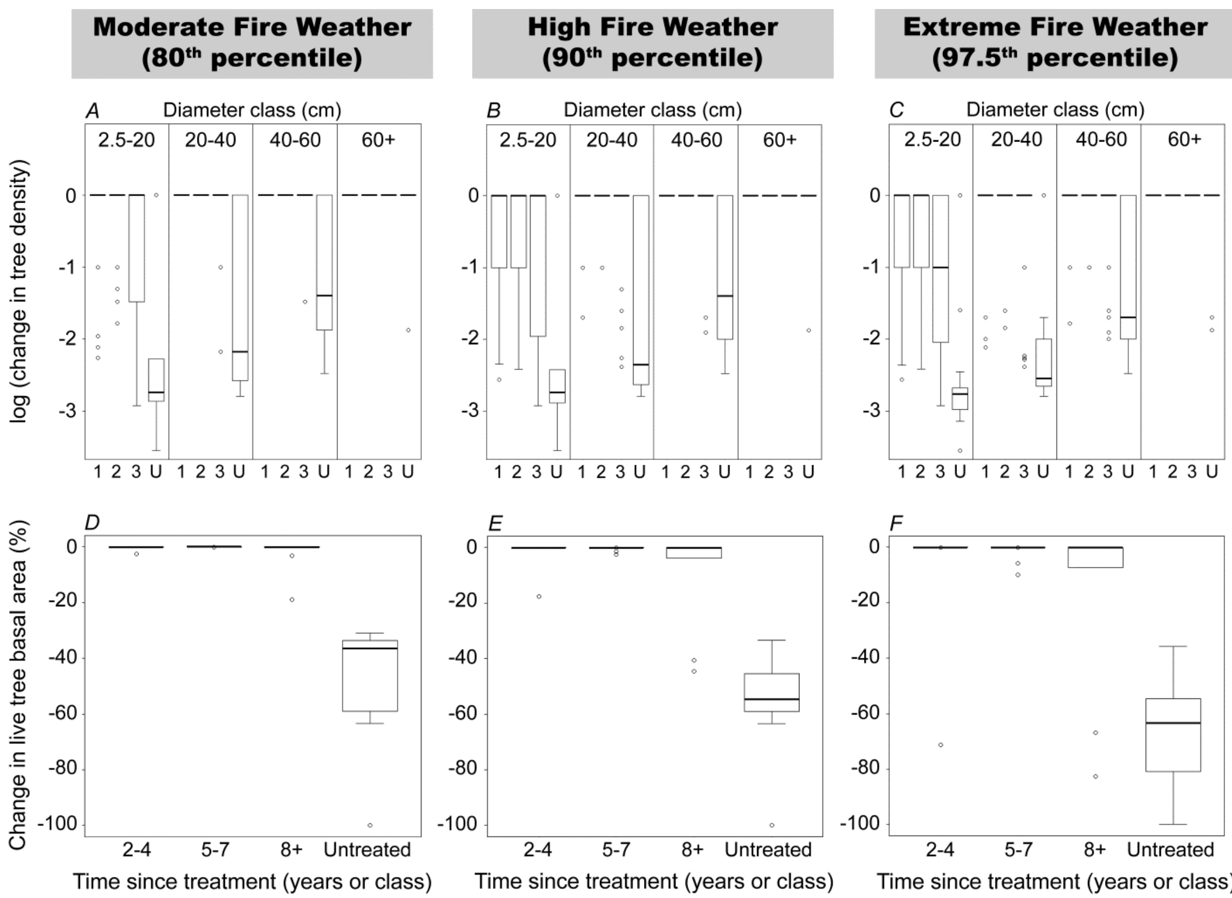

Figure 2.8. Changes in eastside pine stand density and basal area due to modeled wildfire. For calculations of density and basal area reductions, it was assumed that trees with a high (>90 percent) estimated probability of mortality would be killed by fire. Horizontal lines within each box denote the median, box length reflects the interquartile range, and whiskers extend to the data point farthest from the box hinge that is within 1.5 times the interquartile range. Outliers are represented by unfilled circles.

Relative to the eastside pine forest type, mixed conifer sites appear generally more susceptible to changes in tree density and basal area (Figure 2.9). Large changes in tree density and basal area were much more likely for the untreated sites. Mean predicted mortality for pole-size trees (2.5-20 cm dbh) was high in all treatment groups (>87% under moderate fire weather conditions). Predicted mortality under 80th and 97th percentile weather conditions for the largest trees (>60 cm dbh) was 13 and 44%, 4 and 10%, 9 and 21%, and 72 and 89% for the 2-4, 5-7, and 8+ years since treatment groups and the untreated group, respectively. Projected live tree basal area losses in untreated sites greatly exceeded those of treated sites. Absolute change in basal area for these sites increased from 76 to 82% with increasing fire weather severity. Among treatment groups, the most recently treated stands exhibited the highest rates of basal area loss (16 and 38% change for 80th and 97.5th percentile conditions).

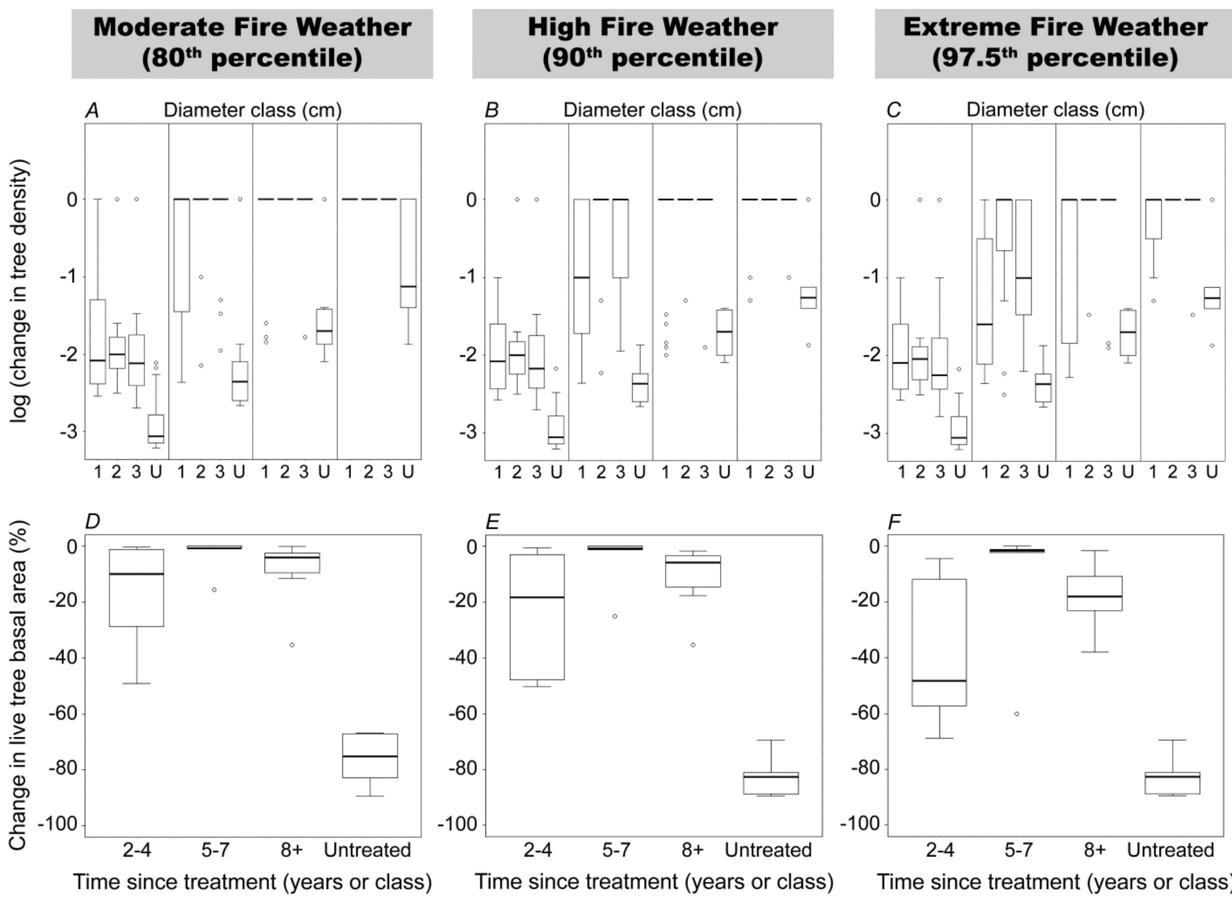

Figure 2.9. Changes in mixed conifer stand density and basal area due to modeled wildfire. For calculations of density and basal area reductions, it was assumed that trees with a high (>90 percent) estimated probability of mortality would be killed by fire. Horizontal lines within each box denote the median, box length reflects the interquartile range, and whiskers extend to the data point farthest from the box hinge that is within 1.5 times the interquartile range. Outliers are represented by unfilled circles.

Discussion

Surface fire potential

Evidence for fuel treatment effectiveness, in the short term, has been well established (Miller and Urban, 2000; Martinson and Omi, 2003; Graham et al., 2004). Yet very few studies have empirically evaluated the lifespan of treatment effects. Those that have, often focused on the accumulation of surface fuels following burning, and in particular, the period of time required for surface fuel loads to reach a threshold beyond which they are sufficient to support subsequent burning. Working in similar forest types, both Thomas and Agee (1986) and van Wagtendonk and Sydoriak (1987) found that surface fuels recovered to ~70% of pre-burn levels within 5-10 years after burning. As a result, surface fuel loads may not be sufficient to permit a subsequent burn for several years. In Yosemite National Park, largely free-burning wildfires burning in upper elevation mixed conifer forests became self-limiting when a previous fire occurred within nine years (Collins et al., 2009). Historic fire regimes may provide additional insight into the length of this fire-resistant period. In the study area, mean fire return intervals were 8-22 years (Moody et al., 2006).

While the potential for burning is linked with surface fuel recovery, it is an inadequate measure of treatment effectiveness. Fuel treatments are not intended to exclude wildfires altogether, but rather to moderate fire intensity and reduce site impacts from burning (i.e. severity). In the wildland-urban interface, where treatments are relied upon to facilitate protection of human communities, treatment longevity should be evaluated from the standpoint of fire control in addition to resource benefits. Reinhardt et al. (2008) argued that managing fuels in order to facilitate fire suppression is counterproductive, as fire exclusion is largely responsible for creating the fuel hazards that treatments are intended to address. Rather, fuels management should increase forest resilience with the focus of allowing the reintroduction of fire as an ecosystem process. Yet where wildlands and human communities intersect, there is a need for fuels management to aid fire protection. Indeed, many of the fuel treatments sampled in this study are defensible fuel profile zones that were established with supporting suppression activities as an explicit goal (Weatherspoon and Skinner, 1996).

Surface fire behavior

Even under extreme fire weather conditions, surface fire intensity and rate of spread were low or moderate in treated sites belonging to every age class. Within a given forest type, estimated surface fire behavior in untreated sites was higher than for any treatment class. Differences across forest types arise due to the influence of site conditions on fuel deposition and vegetation growth. The relatively xeric conditions of the eastside pine forest type manifest in lower surface fuel loads (Table 2.1) and lower predicted surface fire behavior for treated and untreated sites relative to their mixed conifer counterparts. These estimated surface fire hazards have practical implications for fire management. The potential surface fire characteristics displayed in Figure 2.4 indicate that wildfires burning in untreated sites, particularly in the mixed conifer forest type, would challenge wildfire control efforts even without consideration of potential crown fire behavior.

The low to moderate surface fire hazards projected for even the oldest treated sites were not entirely surprising. Fernandes (2009) evaluated the effects of prescribed fire treatments on surface fire behavior through experimental burns in a maritime pine stand in Portugal. He found that flame lengths were reduced for at least 10 years (the duration of the study). Others have similarly predicted long-lived impacts of treatment on surface fire behavior (Vaillant, 2008; Safford et al., 2012).

The apparent lack of temporal trends in surface fire behavior was also expected. Previous analyses of surface fuel loads according to size class categories showed high variability with time since treatment, and method of treatment was not a significant predictor of fuel load (chapter 1 of this dissertation). This lack of significance can likely be attributed, in part, to the diversity of treatment methods contained within each nominal treatment category (mechanical thin only and thin plus prescribed burning). Following forest thinning, surface fuel loads may be elevated for several years (Carlton and Pickford, 1982; Youngblood et al., 2008). Yet the mechanical thinning treatments sampled here spanned several thinning techniques which may have variable effects on the post-thinning fuelbed. For example, cut-to-length harvesting can double pre-treatment fuel loads, while whole-tree removal may have only minimal effects (Walker et al., 2006). The thin and burn treatment comprised both broadcast prescribed fire and slash pile burning; the latter can be expected to reduce total fuel loads but directly impacts a much smaller proportion of the treatment area than broadcast burning. In order to prevent the appearance of relationships that were likely spurious in reality, fire behavior metrics for all treatment categories were merged for presentation in tables and most figures.

Potential crown fire behavior and effects

Ladder fuel hazard

A complete evaluation of fuel treatment longevity requires an assessment of crown fire behavior and fire effects on vegetation in addition to surface fire hazard. Agee and Lolley (2006) highlighted the importance of scale in evaluating the potential for crown fire intiation. They found high average torching indexes when data were aggregated at the unit level, yet when examined at the plot level, torching was predicted for a considerable number of plots (17% under 97th percentile weather conditions). In this study as well, very high mean torching indexes were estimated for treated sites, yet passive crown fire activity was predicted for 11% (eastside pine) and 26% (mixed conifer) of treated plots under extreme fire weather conditions.

The ladder fuels that promote crown fire initiation are highly variable over fine spatial scales. The ladder fuel hazard assessment protocol (Menning and Stephens, 2007) applied in this study is intended to account for spatial variability of fuels by evaluating laddering potential at comparable scales. It has not yet been assessed with respect to actual fire behavior, but allows a comparison of relative hazard. The categorical estimates of ladder fuel hazard are not directly comparable with modeled crown fire metrics; the protocol describes the ability of aerial fuels to convey fire into the forest canopy while the modeled crown fire estimates reported in Tables 2.5 and 2.6 incorporate fire weather, topography, and surface fuels in addition to aerial fuels.

Between the intermediate and oldest treatment age classes, there was a decreasing trend in the mean estimates of both height to crown base and the size of the largest gap in the best fuel ladder. This may be the result of post-treatment recovery, including ingrowth of understory vegetation and tree regeneration. Despite the changes that have occurred since treatment, it appears that the ladder fuel structure in treatments completed 8-26 years before assessment are still distinct from untreated conditions. Based on these measures of ladder fuel hazard, the structure of ladder fuels in the untreated sites should support crown fire more readily than in any treatment age class.

Since mechanical thinning and prescribed burning treatments tend to increase canopy base height, either through burning or direct removal of understory trees and shade-tolerant species, it may seem surprising that the high qualitative rating of ladder fuel hazard was no more common among untreated than treated quadrants. However, the high hazard (A) rating requires concentrated understory vegetation, which was rare for all sites and especially so for untreated sites (Table 2.2). Overall, the proportion of quadrants assigned high or moderate hazard ratings was highest in the untreated category for both forest types. This was largely due to a high frequency of C hazard ratings among untreated quadrants. The C rating describes a fuel complex composed of continuous tree crowns that extend to the forest floor along with the absence of concentrated understory fuels. In contrast, the B categorical ranking was relatively common in the 8+ years-since-treatment class for the mixed conifer forest type. This may indicate that in this forest type, understory vegetation occupies the growing space made available during fuels treatment, thereby reducing the effect of increased canopy base height on laddering potential. However, shrub cover was highly variable across treatment categories and rarely exceeded 20% at the site level. As a result, it is difficult to draw firm conclusions regarding the relationship between shrub response and fuels management and the implications for post-treatment ladder fuel hazard dynamics in the mixed conifer forest type. In the eastside pine type, where shrub cover was consistently low (<10%) in treated and untreated sites alike, the B rating was uncommon.

The E ("no canopy") rating describes a quadrant in which tree crowns, if present, are isolated from the forest canopy. This was an important factor in this study, as reduced tree density and canopy fuel continuity are common goals of forest thinning treatments. Many treated quadrants were assigned an E rating, but the "no canopy" rating was rare among untreated quadrants. Interpretation of the E rating is that the risk of "laddering", or conveying fire into the forest canopy, is low. However, torching of individual trees or small groups of trees may or may not be likely in E hazard quadrants, as connectivity with the forest canopy is the primary factor determining the E rating.

<u>Wildfire simulation</u>
It is important to consider measures of crowning potential within the context of expected conditions in order to allow for site-specific interpretation (Agee and Lolley, 2006). On the basis of expected winds, even under extreme fire weather conditions, the predicted likelihood of torching and crowning fire behavior in the treated sites was very low. In comparison, probable maximum one-minute wind speeds exceeded crowning index thresholds for many untreated eastside pine sites, even under moderate fire weather conditions. This is evidence that treatment prescriptions, which reduce vertical fuel continuity (ladder fuels) and density of canopy fuels, were successful in altering active crown fire potential in the eastside pine forest type. In the mixed conifer type, wind speeds exceeded crowning index thresholds for very few untreated sites

under moderate conditions, though hazards increased with fire weather severity. This may indicate that many untreated mixed conifer stands, at the time of sampling, were resistant to crown fire spread. Yet given the ease of passive crown fire, wildfires in this forest type are likely to both challenge fire suppression efforts and produce high levels of tree mortality.

Indeed, very high levels of live tree basal area change were predicted for untreated sites in both forest types. Levels of expected small tree mortality were similar between treated and untreated sites in the mixed conifer forest type, but it should be noted that relatively few small trees remain after treatment. The larger size classes that comprise the majority of post-treatment stand basal area appear to be protected from fire-induced mortality as a result of treatment. These large trees appear very vulnerable in the untreated sites, where the average probability of mortality under moderate fire weather conditions was 48 and 72% in the eastside pine and mixed conifer types, respectively. This simulation approach did not account for secondary tree mortality, such as bark beetle attack, which can significantly impact cumulative tree mortality (Fettig et al., 2010).

The chronosequence methodology

In a chronosequence approach, multiple sites representing a range of time since disturbance are sampled as a proxy for direct observations of post-disturbance development occurring at a single site. The approach allows for short-term study of process that may require years to elapse, such as ecological succession and the development of soils. The chronosequence approach relies on a number of assumptions. Differences in site conditions arising from factors other than time since treatment, such as topographical influences, can confound inferences regarding temporal trends. A related source of error that has vexed other chronosequence studies (e.g.Yanai et al., 2000) is correlation between time *since* treatment and time *of* treatment. Examples include variation in climatic influences on seedling recruitment and differences in initial treatment effects.

Applied to the study of potential wildfire hazard development following fuels management such as this one, the chronosequence approach assumes homogeneous post-treatment conditions and that the pattern of fuels and vegetation development on recently treated sites follows that which transpired on the oldest sites. In order to isolate the influence of time on potential wildfire behavior and effects, sampling was stratified with respect to site conditions (forest type, slope aspect) and method of treatment. However, investigation of temporal dynamics in surface fuel accumulation, understory growth, and overstory development revealed few trends with time since treatment (Chiono et al., in press). This may be the result of several factors. If change occurs slowly in these forest types, it is possible that the time frame investigated here is too brief to encompass significant change, making trends difficult to detect. Second, there is evidence that over time, forest thinning prescriptions on federal lands have not been uniform with regard to post-treatment stand density (Chiono et al., in press), which indicates that post-treatment conditions have varied. Finally and perhaps most importantly, sample sizes limited the ability to fully account for variability resulting from bottom-up controls such as topography and site productivity. Although this study included an exhaustive sampling effort that incorporated all treatment sites meeting study design specifications over an extensive geographic region (~one million ha), sample sizes were ultimately constrained by the availability of suitable sample sites.

Conclusions

In northern California forest types susceptible to uncharacteristically severe wildfire, forest managers seek to efficiently allocate resources between treating hazardous fuels and maintaining low-hazard conditions within existing treatments. Based largely on forest growth modeling or post-fire observations with limited replication, past authors have estimated that treatment conditions may be retained for 10-20 years (Biswell et al., 1973; van Wagtendonk, 1995; Agee and Skinner, 2005; Collins et al., 2011) to more than 50 years (Vaillant, 2008).

The present study extends the length of the reference period based on replicated, field-sampled data. Ten treatments evaluated here were implemented more than 10 years prior to sampling, and 1 exceeded 15 years. While the fire behavior and effects observed here are based on computerized simulations rather than actual wildfires, this approach does allow control of fire weather inputs, effectively isolating the influence of fuel conditions on treatment effectiveness. In the mixed conifer forest type, the proportion of plots expected to experience only surface fire activity under extreme fire weather conditions was similar between these oldest treatments and the younger treatment age classes, while for the eastside pine type, passive crown fire was more likely in the oldest treatments (42% of plots) than in the younger age classes (5-10% of plots). Even so, crown fire hazards were much more severe where fuels management had not occurred, as active crown fire was predicted for more than 70% of untreated plots in each forest type. By comparison, active crowning was projected in only one treated plot, which was located on the oldest treatment site (26 years).

It is clear that treatments implemented in the northern Sierra Nevada and southern Cascades effectively moderate potential wildfire behavior and increase stand resistance to wildfire disturbance. While variability in both natural and management-induced conditions likely obscured temporal trends in wildfire hazard following treatment, even the oldest treatment evaluated here (26 years since treatment) exhibited low-intensity fire behavior and low-severity effects. These continued low hazards are evidence that fuels reduction treatments in the dry forest types studied here remain effective for at least a decade, though limited sample sizes preclude strong statements regarding treatment longevity beyond 10-15 years.

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CHAPTER 3

Long-term Effects of Fuels Reduction Treatments on Native and Exotic Plant Abundance

Abstract

Forest management in western US forests with altered wildfire regimes prioritizes treatment for hazardous fuels reduction. Fuel treatments, which disturb the forest floor and increase resource availability, have the potential to promote invasion by exotic plant species. Invasives can have profound consequences for ecosystem structure and function. Yet the consequences of these treatments for understory plant communities remain poorly understood, particularly beyond the first few years after treatment. This study investigates the temporal effects of mechanical thin only and mechanical thin and burn treatments. Regression tree analysis was used to explore relationships of plant abundance by lifeform and ground cover with treatment and site characteristics. Ground cover by litter and woody debris was positively associated with tree canopy cover, while higher levels of shrub cover occurred under lower canopy cover. Mineral soil exposure was negatively associated with time after treatment, and recovered slowly. Despite the availability of bare mineral soil and the proximity of treatments to forest roads and sources of plant propagules, non-native plant species were recorded in very few treatment plots (4 of 195). This study suggests that these forest types may be resistant to invasion of non-native understory plants following treatment for hazardous fuels reduction.

Introduction

Treating hazardous fuels has become a management priority in many western forests. Where historical fire regimes were characterized by frequent, low-to-moderate-intensity burning, management activities such as fire exclusion, logging, and grazing have increased fire return intervals, creating conditions that promote large, high severity wildfire (McKelvey and Busse, 1996). The ecological consequences of altered disturbance regimes include densification and dominance of late-seral conifer species, lower spatial diversity in stand structure and species composition, and reduced understory species diversity and production (Mutch et al., 1993; Covington and Moore, 1994; Hessburg and Agee, 2003; Hessburg et al., 2005). Altered disturbance regimes may also promote invasion by non-native plant species (Alpert et al., 2000).

Exotic species invasions are recognized as a threat to natural ecosystems because they can alter ecosystem structure and function (Vitousek, 1990; Mooney and Hobbs, 2000). These changes to ecosystem properties may come into being through their influence on natural disturbance regimes (Mack and D'Antonio, 1998). There are many examples of modifications to wildfire disturbance regimes created by non-natives in the literature; plant invasions have been shown to alter the frequency, severity, extent, and seasonality of burning (D'Antonio and Vitousek, 1992; D'Antonio, 2000; Brooks et al., 2004). Disturbance regime modifications can entrench alien dominance by creating positive feedback loops which favor the invader (Mack and D'Antonio, 1998).

Though managed forest landscapes are not necessarily more prone to invasion than their unmanaged counterparts (Fornwalt et al., 2003), disturbed areas are generally more vulnerable to invasion than undisturbed areas (Elton, 1958; Rejmánek, 1989; Hobbs and Huenneke, 1992; Mack et al., 2000). This has led to a concern that fuel treatments could foster invasion by non-native understory plants within treated areas which, once established, might transition into undisturbed stands. Typical fuel treatment activities such as forest thinning and prescribed burning increase the quantity of unused resources available for plant growth. This is one suggested mechanism by which disturbance could promote invasion (i.e. the resource availability hypothesis (Davis et al., 2000)). Prescribed burning increases mineral soil exposure (Gundale et al., 2005; Moghaddas et al., 2008) and nutrient availability (DeBano, 1990). While soil disturbance, in isolation, may not advance invasion (Hobbs, 1989) treatments often involve tree removal and coincident increases in light availability. Post-treatment non-native cover has been shown to associate positively with bare ground levels (Freeman et al., 2007) and negatively with tree basal area (Fornwalt et al., 2003).

Another cause for concern with respect to the potential for invasion is the proximity of roads to sites treated for fuels reduction. Fuel hazard reduction treatments are often situated along roads because accessibility is a consideration with respect to both treatment implementation and utility – treatments are often intended to aid wildfire suppression activities. In order for invasion to occur, disturbances must coincide with non-native propagules (Davis et al., 2000), and by harboring non-native plant populations, highly disturbed environments such as transportation corridors can promote invasion into adjacent habitats (Milberg and Lamont, 1995). Fuel treatments may therefore represent a perfect storm with respect to non-native plant invasion, with seed sources and disturbance coinciding in space and time.

Though fuel treatments in other plant community types have been shown to dramatically increase the abundance of non-native plant species (Merriam et al., 2006), post-treatment non-native abundance in California forests is generally low (i.e. <10% cover) (Kerns et al., 2006; Merriam et al., 2006; Collins et al., 2007; Kane et al., 2010). There is limited evidence from short-duration studies that non-native abundance may increase over time following treatment initiation (Keeley et al., 2003; Collins et al., 2007; Owen et al., 2009), but the relationship between treatment and invasion over time remains poorly understood.

This chronosequence study takes advantage of a large number of fuel treatments implemented over a period of more than a decade in northern California to examine the long-term effects of treatment on understory plant abundance, with a focus on non-native species. Study objectives were to determine whether treatments for fire hazard reduction promote invasion by exotic species, and if so, how treatment method and site characteristics (e.g. plant community type) influence invasion. As invasion has been associated with exposed mineral soil (Crawford et al., 2001; Freeman et al., 2007) and cover of native plants (Keeley and McGinnis, 2007), this study also assesses the long-term effects of treatment on these characteristics.

Methods

Study site

This study was conducted in Nevada, Sierra, and Plumas Counties in the southern Cascade and northern Sierra Nevada regions of California. It includes forests managed by one forest products company, two private landowners, and three US National Forests. Until fire suppression began in the early 20th century, fire was a common process in the study area. A fire history study conducted in the region found that for the era prior to Euro-American settlement, the mean composite fire return interval was 6-18 years (for fires that scarred more than 10% of samples)(Moody et al., 2006). Study sites range from 1100 to 2150 m in elevation. Forest soils in the study region are well-drained Alfisols and Ultisols, and Haploxerults and Haploxeralfs are common. Common soil series include Deadwood, Kistirn, Holland, and Tahoma. The study area encompasses two forest types: eastside pine and Sierra mixed conifer. The eastside pine forest is dominated by yellow pines (Jeffrey pine (*Pinus jeffreyi* Grev. and Balf.) and ponderosa pine (*P. ponderosa* Dougl.)) and white fir (*Abies concolor* (Gord. and Glend.)) while the Sierra mixed conifer type is characterized by six dominant tree species: California black oak (*Quercus kelloggii* Newb.), Douglas-fir (*Pseudotsuga menziesii* var. *menziesii* (Mirb.) Franco), incense-cedar (*Calocedrus decurrens* [Torr.] Florin.), ponderosa pine, sugar pine (*Pinus lambertiana* Dougl.), and white fir (Barbour and Minnich, 2000). Common shrubs include *Arctostaphylos patula* Greene*, Ceanothus cordulatus* Kellogg, *C. integerrimus* Hook. & Arn., *C. velutinus* Douglas, *Purshia tridentata* (Pursh) DC, and *Symphoricarpos mollis* Nutt. Frequently occurring understory plants include *Wyethia mollis* A. Gray, *Achillea millefolium* L., and *Collinsia parviflora* Lindl.

Treatments

Treatment areas were identified with the assistance of local USDA Forest Service managers, University of California cooperative extension specialists, and the Collins Pine Company. Potential sample sites were restricted to those treated with mechanical thinning with and without burning (prescribed or pile burning). Both mastication and hand-thinning treatments were omitted from this study. For treatments that included burning, the burn treatment was to follow thinning within three years. All treatments that met study design requirements were sampled.

Untreated control sites were located directly adjacent to treatment areas. Control sites were defined as those having experienced neither management nor burning activity within the preceding 25 years. Sites containing recent stumps or char and ash were excluded from control sampling. In an attempt to isolate treatment effects, adjacent treatment and control areas were required to have similar overstory species composition and slope steepness. As timber harvesting equipment is generally restricted to slopes of <30%, control sites were also limited to those with <30% grade. In total, 52 treatment units 2 to 26 years after treatment and 13 control units were sampled.

Field sampling

Overstory characteristics, tree seedling density, and ground cover were measured using three 0.1-ha circular sampling plots in each treatment unit. Elevation, aspect, and percent slope were also recorded on each plot. Because digital treatment maps were often unavailable, individual plots were placed systematically with a random starting point. Running parallel to the treatment edge, plots were located 50 m apart with a 30-m buffer between each plot and the treatment boundary. When treatment gaps (untreated or group selection areas) were encountered during plot placement, the remaining plot or plots were placed beyond the treatment gap, with a 30-m buffer between the next plot and the treatment gap. Within each plot, three 17.85-m transects running outward from plot center were established (Figure 3.1). The azimuth of the first transect was chosen randomly. The remaining two transects were placed at headings 120º and 240º degrees greater than the first.

For all trees ≥2.5 cm diameter at breast height (dbh), height, dbh, and species were collected. Tree heights were measured using a Haglöf Vertex Laser hypsometer. For treated sites, all trees larger than 2.5 cm dbh were sampled for the entire 0.1-ha plot area. Due to higher tree densities on most untreated sites, overstory trees (≥7.6 cm) and saplings (2.5-7.6 cm dbh) were sampled within 0.075 and 0.05 ha nested subplots, respectively. Percent canopy cover was sampled with a densitometer (vertical sighting tube), on a 25-point, 8-by-8 meter grid oriented north-south and east-west and centered on plot center (Jennings et al., 1999).

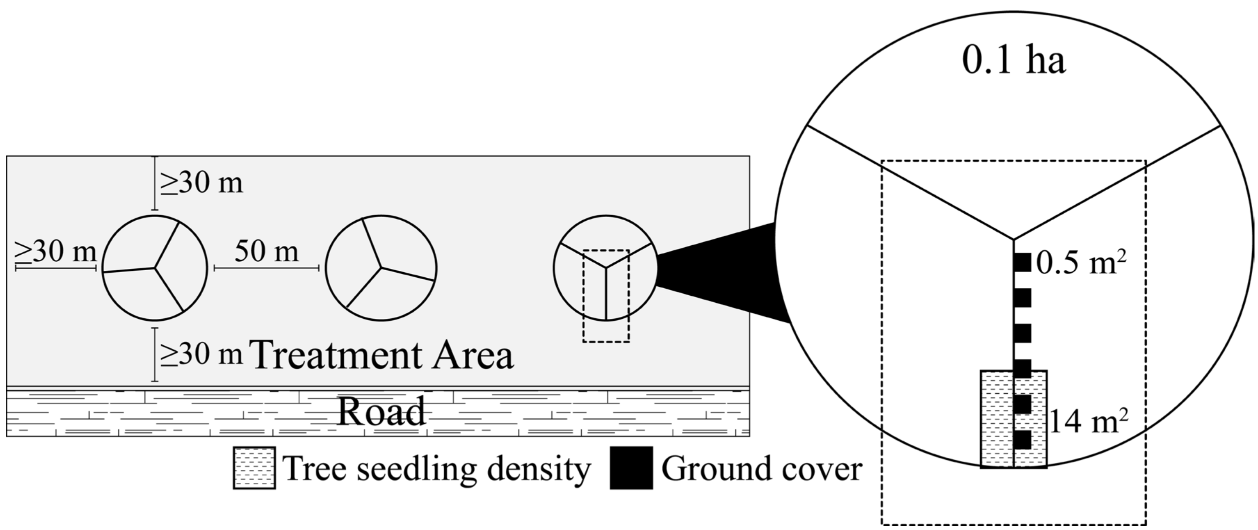

Figure 3.1. Fuel treatment sampling design. Three 0.1-ha sampling plots were placed within each treatment area. Ground cover data was collected within 0.5-m^2 sampling frames placed along each of three transects within each plot. Tree seedlings were tallied within three 7 x 2 m subplots positioned at the outer end of each transect.

Understory sampling included percent ground cover by category and tree seedling density. Ground cover was estimated within 0.5-m^2 sampling frames placed along each transect (6 frames/transect)(Figure 3.1). Total area sampled within each site was 9 m^2. To aid visual estimation, sampling frames were marked to indicate 5, 25, and 50% of quadrat area. Cover categories included exposed mineral soil, litter, rock, woody debris, and vegetation percent cover by growth form (forb, graminoid, shrub, and tree seedling). Plant cover was defined as the vertical projection of foliage and supporting parts onto the ground (Caratti, 2004). Vegetation cover was further divided according to species origin (native and exotic). When exotic species were encountered, total cover by species was recorded. Identification of species and origin were based on Hickman (1993). Sampling of non-plant ground cover (exposed mineral soil, litter, rock, and woody debris) began with the second season of data collection and cover data was collected for 74% of plots (144 of 195). Tree seedlings, defined as trees < 2.54 cm diameter at breast height, were sampled on 42 m^2 within each plot. Seedlings were tallied along the outermost 7 meters of each transect within an area extending 1 meter to either side of the transect (Figure 3.1).

Site index

Site index, a measure of site productivity, was estimated for each treatment site. For each treatment site, samples were collected within the area bounded by the two outermost plots. Sample trees were well-formed dominants without evidence of past suppression or significant crown or bole damage. Tree ages were estimated from tree cores, and total tree age was calculated as the sum of latewood rings plus 10 (an approximation of the number of years required for a tree to grow to breast height)(Fritts, 1976). Site index was then estimated for each sample tree based on the site index curves of Dunning (1942); site index for each treatment area was taken as the most frequently occurring site index estimate.

Statistical Analyses

To examine the effects of treatment (treated vs. control), differences in ground cover by cover category were assessed using unpaired t-tests. $P < 0.05$ indicated statistical significance. The effects of time-since-treatment were assessed by analysis of variance. Sites were grouped into three age classes describing time since treatment: 2-4, 5-7, and 8-26 years. The range of time since treatment contained in each class was selected to approximately equalize the number of observations across classes. An additional class was included to represent the control treatment. Where significant differences occurred ($p < 0.05$), pair-wise comparisons between age class categories were performed with Tukey's HSD multiple comparisons test. Data transformations were applied to meet the assumptions of statistical tests. Due to a high frequency of 0% cover values, treatment effects on tree seedling cover were not assessed. Instead, seedling density was evaluated.

To explore the relationships between ground cover and treatment and site factors, classification and regression tree (CART) analyses were implemented in the "tree" package by Ripley (2011) version 1.0-29 within the R software environmental (http://www.r-project.org/)(R Development Core Team, 2011). The predictor variables evaluated are described in Table 3.1. Regression tree analysis, used for continuous response variables, recursively partitions the response observations into subsets based on the value of a single predictor variable. The output is represented as a dichotomous tree. The CART procedure was chosen because it allows straightforward interpretation of complex relationships between response and predictor variables (Breiman et al., 1984; De'ath and Fabricius, 2000). CART analysis also makes few analytical assumptions and is not sensitive to response or predictor variable distributions (De'ath and Fabricius, 2000), a benefit in analyzing percent cover data, whose distributions are frequently skewed.

Control parameters used in tree fitting include a minimum of 10 observations in a node before attempting a split, and 5 minimum observations within each child (terminal) node. Initial CART models tend to be overly complex, overfitting the response variable data (Breiman et al., 1984). As a remedy, optimal tree size was selected using 10-fold cross-validation with the "cv.tree" function in which 90% of the data were used to fit the tree model while holding out the remaining 10% to evaluate the model. Cross-validation is an iterative process to relate deviance to tree size. Final trees were selected by maximizing the deviance explained while minimizing tree size. Each optimal tree model was then constructed with the "prune.tree" function. Variance explained by each final tree was calculated as: 1 - deviance(model) / deviance(null model) where null model deviance is equivalent to the response variable sum of squares, or deviance at the tree "root".

All statistical analyses were performed in the statistical software R version 2.13.1.

Table 3.1. Descriptive statistics of independent variables used in t-tests, analysis of variance, and regression tree modeling.

Variable	Description	Range	Mean	SE
Ageclass	Time since treatment index: 1: 2-4 years, 2: 5-7 years, 3: 8-26 years, 4: untreated			
Aspect	General site-level slope aspect (north- or south-facing)			
Canopy cover	%	13.3-77.3	40.9	2.0
Elevation	meters	1112-2155	1644	34.6
Forest type	Eastside pine or mixed conifer			
Treatment type	Treatment type (thin only, thin and burn, untreated)			
Site index	Tree height at reference age 300 (meters)	22.9-53.3	38.2	0.7

Results

Non-native plant abundance

Non-native plants were extremely uncommon in the sampled area. Non-native plant species were recorded on only 4 of 195 (2%) plots. Species found were cheatgrass (*Bromus tectorum* L.) and two species of thistle, Canada thistle (*Cirsium arvense* (L.) Scop.) and bull thistle (*C. vulgare* (Savi) Ten.). Non-native cover within these sites was also low (< 2% at the plot level). Table 3.2 displays characteristics of the plots in which non-natives were sampled. The sites in which non-natives occurred were treated for fuels reduction by thinning as well as thinning and burning, and treatment ages spanned 3-11 years. Sites in which non-native species occurred were evenly divided between the eastside pine and mixed conifer forest types and between private and federal ownership.

Table 3.2. Non-native plant species sampled and characteristics of the plots in which they occurred. Since non-native plant species were found in only 4 plots, cover is given for each plot in which the species was found. Abbreviations are Life form: BF: biennial forb, AG: annual grass; Ownership: USFS: United States Forest Service, P: privately owned; Forest type: EP: eastside pine, MC: mixed conifer; Treatment: TB: thin and prescribed burn, TO: thin only; TST: time since treatment.

Scientific name	Life form	Cover (%)	Ownership	Forest type	Treatment	TST (yrs)
Bromus tectorum	AG	0.1	P	EP	TO	8
Bromus tectorum	AG	0.3	USFS	MC	TO	4
Cirsium arvense	BF	0.2	USFS	EP	TB	11
Cirsium vulgare	BF	1.4	P	MC	TO	3

Ground cover and tree seedling density

Treatments had consistent, predictable effects on overstory structure, reducing stand basal area and density and canopy cover. The effects of treatment on ground cover were more varied (Table 3.3). Treatment increased mineral soil exposure by 396% (eastside pine) and 185% (mixed conifer) relative to the controls. Woody debris cover in treated sites was half that of untreated sites in the eastside pine type. Total live plant cover ranged from 2-50% on treated sites and 2-36% on untreated sites. Cover by graminoids, while below 3% on average across forest types, was significantly higher for treated eastside pine sites than for untreated sites ($p = 0.00461$); this difference was not significant for the mixed conifer type ($p = 0.4853$). Mean forb cover was also very low (≤3%). While mean forb cover for treated sites was approximately double that of untreated sites in both forest types, this difference was not significant at $p < 0.05$. Both seedling density and shrub cover were highly variable, and differences between treatment and control means were not statistically significant.

Analysis of variance and Tukey's HSD multiple comparisons test revealed relationships between ground cover and time since treatment. Mineral soil exposure was elevated (>10% cover) in all time since treatment age classes relative to controls (~3% cover)(Figure 3.2A). Mean percent cover by woody debris was lower in every treatment age class (10-13% cover) than the control (19%), but this difference was significant only for the 5-7 years since treatment class ($P = 0.006$) (Figure 3.3). Mean litter cover in the 2-4 years-since-treatment group (73%) was low relative to the control (84%)($p = 0.007$). The 5-7 and 8-26 years-since-treatment classes, with 81 and 79% litter cover, respectively, were not significantly different from either the youngest treatment class or the control.

Table 3.3. Mean (standard error) stand characteristics , ground cover, and live plant cover. Differences between treatment groups within a forest type ($p < 0.05$) are signified by different letters in rows (unpaired t-test). Differences in tree seedling cover between treatment groups were not assessed. Basal area and tree density calculations include trees with diameter at breast height (dbh) ≥ 2.5 cm; tree seedling calculations are for trees with dbh < 2.5 cm.

	Eastside pine		**Mixed conifer**	
	Treated	*Untreated*	*Treated*	*Untreated*
Forest stand characteristics	$n = 32$	$n = 7$	$n = 20$	$n = 6$
Elevation (m)	1749.6 (40.4)	1906.9 (96.2)	1423 (46.9)	1510.7 (84.1)
Basal area (m^2 ha^{-1})	24.7 (1.3) *a*	53 (5) *b*	31.2 (2.6) *a*	53.4 (4.5) *b*
Canopy cover (%)	30.5 (1.6) *a*	49 (2.7) *b*	45.9 (2.9) *a*	68.6 (5.8) *b*
Tree density (stems ha^{-1})	251.9 (30.3) *a*	1287.1 (162.4) *b*	368.7 (29.7) *a*	1389.5 (147.7) *b*
Seeding density (1000 ha^{-1})	1.2 (0.2)	2.8 (1.7)	9.9 (2.5)	6.2 (2)
Ground cover (%)	$n = 27$	$n = 7$	$n = 9$	$n = 6$
Bare mineral soil	12.9 (1.2) *a*	2.6 (0.5) *b*	13.4 (2.5) *a*	4.7 (1.2) *b*
Rock	2.6 (0.4)	2.8 (0.6)	2.0 (0.7)	2.1 (1.5)
Litter	76.3 (1.5)	83.2 (2.7)	80.1 (2.8)	83.0 (3.5)
Woody debris	9.0 (0.8) *a*	18.9 (2.4) *b*	17.3 (1.8)	20.9 (2.3)
Live plant cover (%)	$n = 32$	$n = 7$	$n = 20$	$n = 6$
Graminoid	2.8 (0.5) *a*	0.3 (0.1) *b*	1.0 (0.2)	0.4 (0.1)
Forb	3.0 (0.4)	1.7 (0.6)	2.6 (0.4)	1.1 (0.4)
Shrub	9.4 (1.1)	9.3 (2.8)	7.0 (1.5)	12.4 (4.7)
Seedling	0.4 (0.1)	1.1 (0.4)	2.7 (0.5)	2.6 (0.6)
Total live plant cover	15.7 (1.7)	12.4 (4.3)	13.4 (2.7)	13 (4.1)

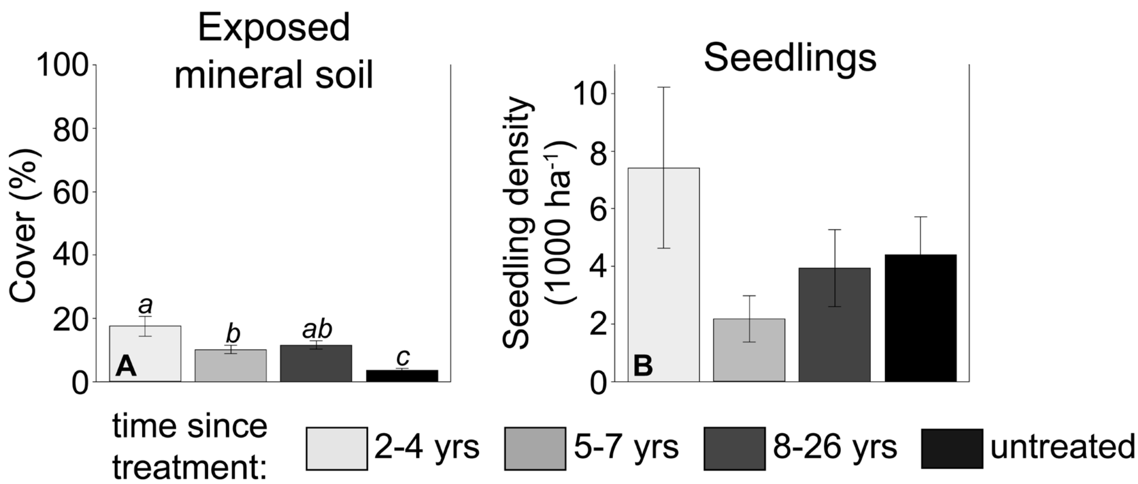

Figure 3.2. Exposed mineral soil (A) and seedling density (B) by time-since-treatment class. Error bars represent +/- 1 standard error of the mean. Where analysis of variance tests were significant, ageclass differences were evaluated using Tukey's HSD multiple comparisons tests. Different letters indicate statistical significance ($p < 0.05$); no letters indicate nonsignificance.

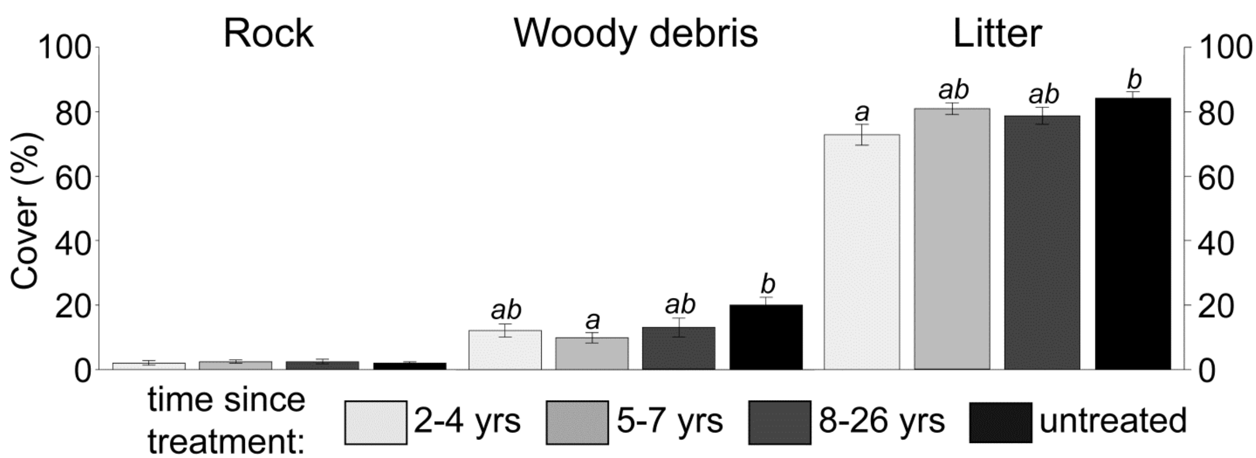

Figure 3.3. Mean ground cover by cover type. Error bars represent +/- 1 standard error of the mean. Where analysis of variance tests were significant, ageclass differences were evaluated using Tukey's HSD multiple comparisons tests. Different letters indicate statistical significance ($p < 0.05$); no letters indicate nonsignificance.

Mean graminoid and forb cover were higher in every time-since-treatment class than in the untreated controls, though differences were not significant among some time-since-treatment classes (Figure 3.4). Seedling density (Figure 3.2B), shrub cover (Figure 3.4), and total live plant cover (not shown) did not differ with respect to time following treatment or between treated and untreated sites.

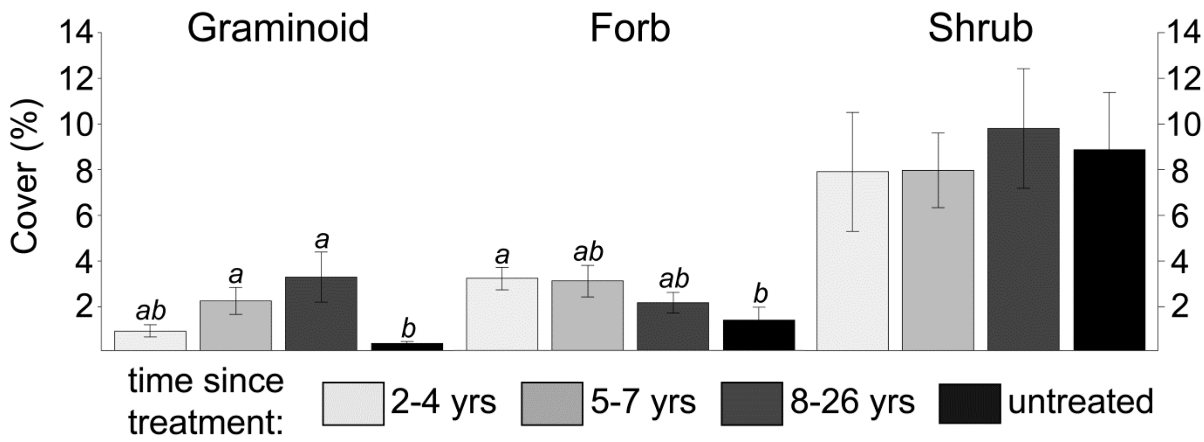

Figure 3.4. Mean plant abundance by growth form. Error bars represent +/- 1 standard error of the mean. If analysis of variance tests were significant, ageclass differences were evaluated using Tukey's HSD multiple comparisons tests. Different letters indicate statistical significance ($p < 0.05$).

Regression tree models explained between 35 and 60% of the variance in seedling density and ground cover by cover type. Canopy cover was the main determinant of ground cover for all response variables examined with the exception of mineral soil exposure, which was primarily associated with treatment and treatment age. Regression trees are shown in Figures 3.5 and 3.6. For both seedling cover and graminoid cover, because fewer than 10% of cover observations were greater than 5%, no regression tree models were fitted. For percent forb cover, no fitted model represented an improvement over the null model.

Time-since-treatment class was the most important explanatory variable for mineral soil exposure. An initial split between the three treated classes (high exposure) and the control (low exposure) was followed by a second split dividing sites treated 2-4 years prior to sampling from those treated 5-26 years before sampling. The highest mineral soil exposure was associated with the most recently treated sites.

A canopy cover division at 44.7% explained a large portion of the variation in litter cover (Figure 3.5). Lower litter cover was predicted below the value of the split. Within the low canopy cover branch, the lowest litter cover was associated with the youngest time-since-treatment class. For sites with high canopy cover, those located on south-facing aspects were predicted to have higher litter cover than those on north-facing aspects.

High canopy cover (≥50.7%) was also associated with high woody debris cover. Within the low canopy cover branch, lower debris cover was predicted for eastside pine sites. Within this forest type, low debris cover was predicted for sites located below 1925 m elevation.

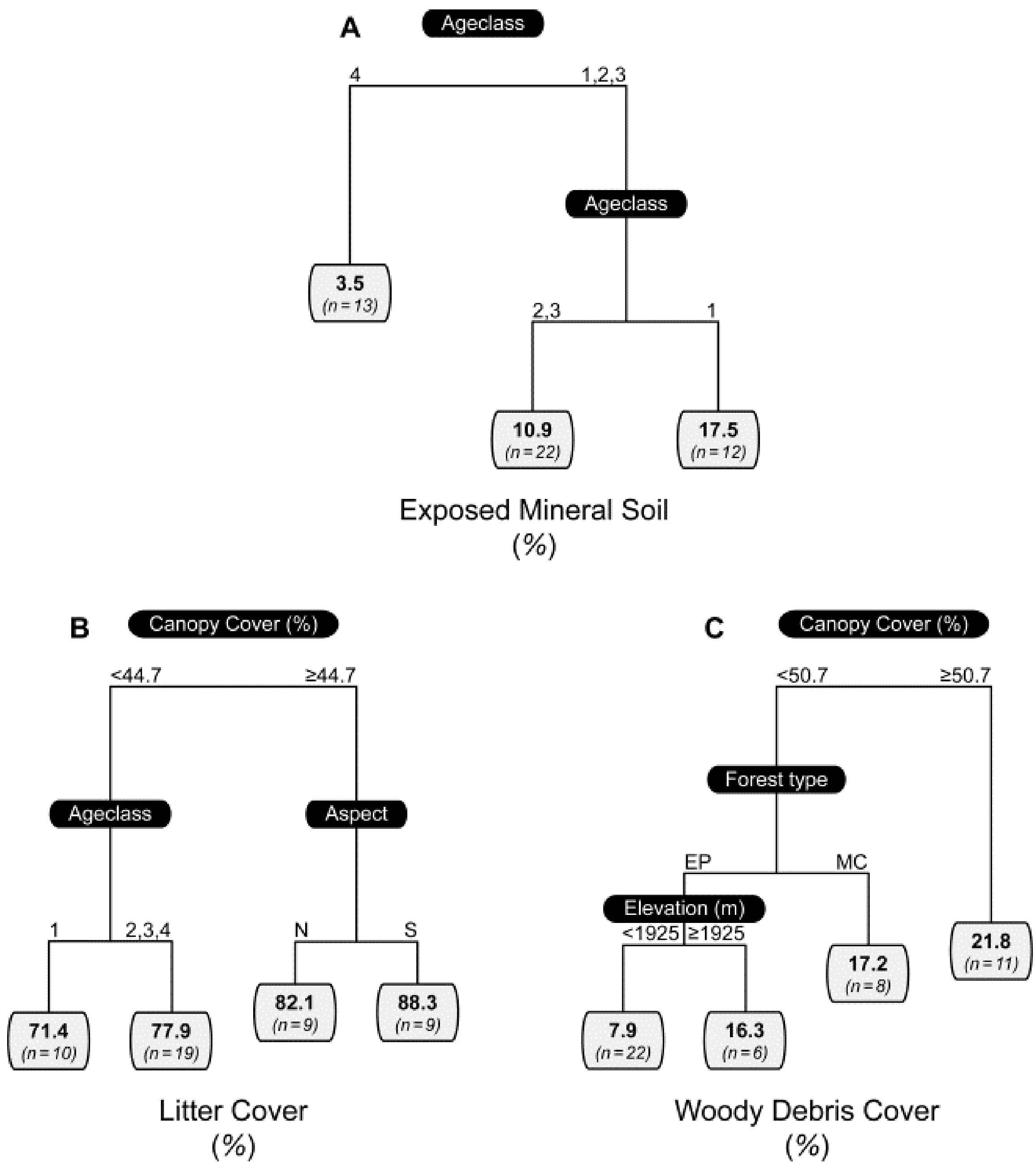

Figure 3.5. Regression tree analyses of ground cover by cover type. Explanatory variables tested include a number of site and treatment factors (Table 3.1); the variables with the most explanatory variable for each cover type were ageclass (exposed mineral soil), canopy cover, ageclass, and slope aspect (litter cover) and canopy cover, forest type, and elevation (woody debris cover). Ageclasses are 1: 2-4 years since treatment (yst), 2: 5-7 yst, 3: 8-26 yst, and 4: untreated. Forest types are EP: eastside pine and MC: mixed conifer. For each tree, each of the 2 splits (nonterminal nodes) is labeled with the variable and the values or levels which determine the split. Each of the 3 leaves (terminal nodes) is labeled with the mean rating and the number of observations in the group (italic, in parentheses). The trees explained 41% (exposed mineral soil), 38% (litter cover), and 49% (woody debris cover) of the total variance in each variable. The vertical depth of each split is proportional to the variation explained.

Live plant cover

The regression tree model for shrub cover (Figure 3.6A) indicated low cover under high canopy cover conditions (≥38.7%). Method of treatment further divided sites with low canopy cover. Under low canopy cover, lower shrub cover was associated with the mechanical thin only treatment. Higher cover was linked to the other treatment age classes and the untreated control. The regression tree model for total live plant cover was very similar to that describing shrub cover, which is not surprising as shrub cover was a large component of total plant cover. The final tree (Figure 3.6C) retained the same structure, splitting variables, and split levels as the shrub cover model. This model explained 23% of the variation in total live plant cover.

Tree seeding density was well described by canopy cover and site elevation. Sites with high seedling density (~20,000 ha^{-1}) had low canopy cover (<51.3), or, if they had high canopy cover, were located below 1504 m elevation. This elevation value is near the mean for mixed conifer sites (1443 m) and below the 25^{th} percentile for eastside pine sites.

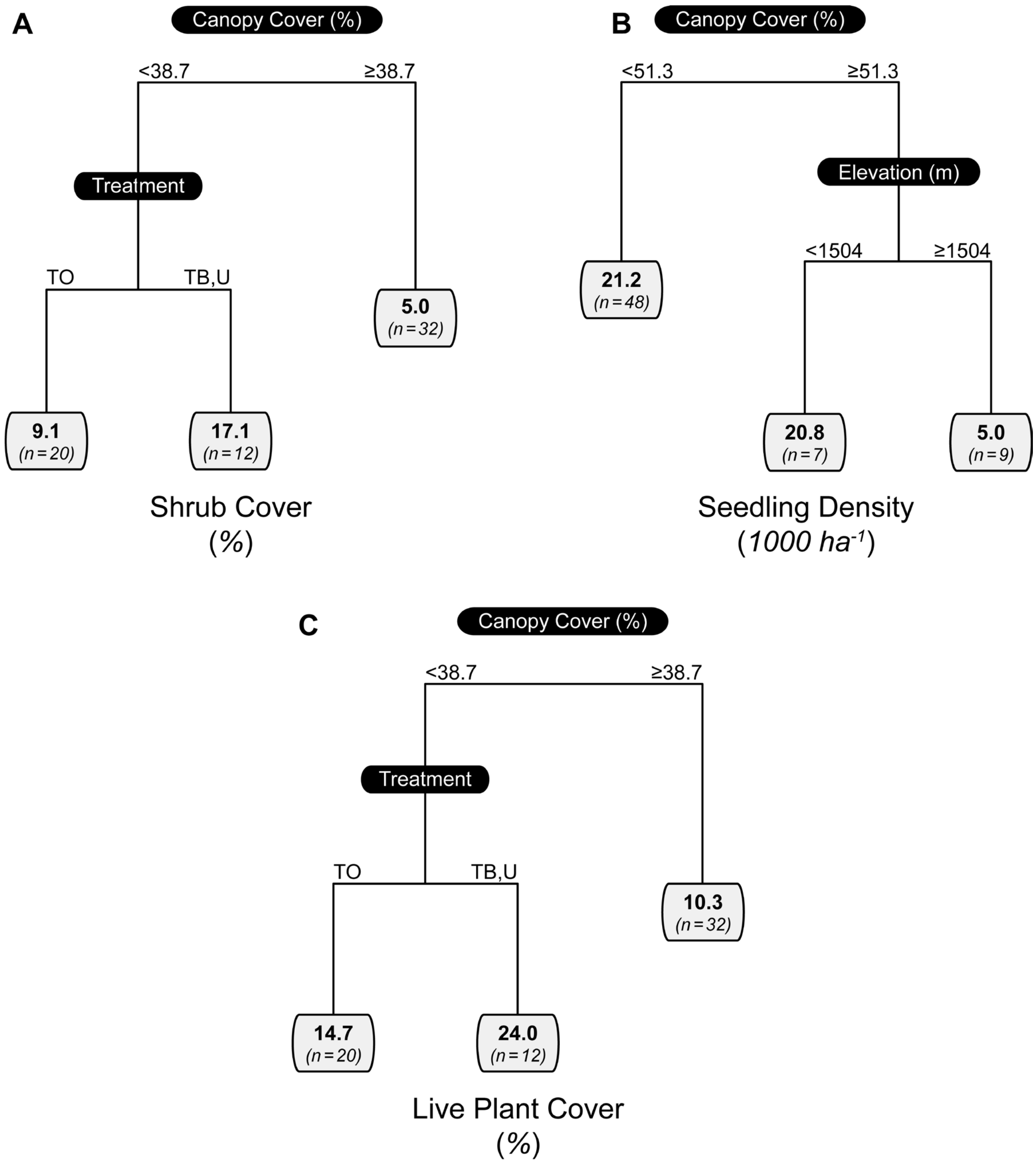

Figure 3.6. Regression tree analyses of shrub cover, tree seedling density, and total live plant cover. Table 3.1 contains a description of the explanatory variables tested; the variables with the most explanatory power were canopy cover and treatment method (shrub cover); canopy cover and elevation (seedling density); and canopy cover and treatment method (live plant cover). For each tree, each of the splits (nonterminal nodes) is labeled with the variable and the values or levels which determine the split. Each of the leaves (terminal nodes) is labeled with the mean rating and the number of observations in the group (italic, in parentheses). The trees explained 23% (shrub cover), 60% (seedling density), and 23% (live plant cover) of the total variation in each variable. The vertical depth of each split is proportional to the variation explained.

Discussion and Conclusions

Understory vegetation in comparable forest communities is known to recover quickly after disturbance (Knapp et al., 2006). Short-term studies, however, have shown variable impacts of thinning and burning on graminoids, forbs, and tree seedling density (Cain et al., 1998; Metlen et al., 2004; Wienk et al., 2004; Metlen and Fiedler, 2006). In this study, both graminoids and forbs demonstrated a relationship with treatment age. Fuel treatments appeared to enhance graminoid cover with an upward trend with time after treatment and the largest mean difference between treatments and controls seen in the oldest treatment age class. Conversely, treatment increased forb cover in the short term (2-4 years after treatment) and mean cover declined with treatment age. Tree seedling density was highly variable with respect to treatment age. Instead, seedling density was positively associated with low (<51.3%) canopy cover, though density was also high for a small number of sites with high canopy cover at low elevations (below 1504 m) (n=7).

Regression tree analysis clarified relationships between treatment and shrub cover, which were determined by direct and indirect treatment effects. Plant cover in the study area was dominated by shrub species, many of which resprout after burning or have persistent soil seed banks that germinate after fire, such as *Ceanothus* and *Arctostaphylos*. While burning can greatly reduce shrub cover in the short term (e.g. Metlen et al., 2004), shrubs tend to recover rapidly after burning (Schwilk et al., 2009). The effects of thin only treatments may be more nuanced, as evidenced by regression tree analysis. Mechanical thinning increases light availability which promotes understory growth, but it also causes mechanical damage to established individuals. Here, higher shrub cover was associated with lower canopy cover. Yet for sites with low canopy cover, the thin only treatment was associated with relatively low shrub cover levels. Elsewhere, thin-only treatments have been shown to reduce shrub cover while thin and burn treatments did not (Collins et al., 2007).

As in other studies, treatments increased bare mineral soil coverage (Boerner et al., 2007; Moghaddas et al., 2008). Soil exposure is often shown to decline quickly after treatment (Boerner et al., 2007; Boerner et al., 2009). In this study, mean percentage of exposed mineral soil declined over time following treatment from 18% in the youngest treatments sampled to 12% in the oldest (mean 10 years), but bare ground levels were still elevated in the oldest treatments relative to untreated controls (4% exposed). The xeric eastside pine type was overrepresented in measurements of soil exposure on treated sites (69% of observations), though not for controls. However, mean soil exposure by treatment age class differed little between forest types, so this discrepancy does not explain the difference between percent soil exposure in the active treatment and control sites. The long-lasting effects of treatment on mineral soil exposure may be linked to the effects of forest thinning on canopy cover, which was positively associated with both litter and woody debris cover.

Despite the link between bare ground levels and non-native species (Freeman et al., 2007) and the long-lasting increases in mineral soil exposure resulting from treatment, the fire hazard reduction treatments studies here did not enhance invasion by non-native plant species. Others too have found that low-intensity disturbances (e.g. burn only treatments, single-tree selection harvesting) did not increase non-native abundance (Battles et al., 2001; Collins et al., 2007). Yet in similar forests, relatively extreme alterations of the growing environment such as clearcutting

(Battles et al., 2001) and severe wildfire (Crawford et al., 2001; Hunter et al., 2006) have been shown to promote non-native species, indicating the potential for invasion.

While many past studies have found small or nonsignificant effect sizes (Griffis et al., 2001; Wienk et al., 2004; Hunter et al., 2006; Freeman et al., 2007), treatment intensity has been linked to invasion. Mechanical thinning and burning tend to increase exotic species more than thinning or burning alone (Schwilk et al., 2009) and higher severity prescribed fire is associated with increased non-native cover (Dodson and Fiedler, 2006; Kerns et al., 2006). Dramatic reductions in overstory cover and disturbance of the soil surface are particularly to be avoided. In a study of fuel treatments in many plant communities across California, Merriam et al. (2006) found that fuel treatments increased non-native cover by 200%, and non-native species were most strongly associated with the method of treatment. Non-natives were found on half of plots in treatments constructed by bulldozers, which were also associated with the highest non-native cover. In contrast non-natives were found on only 4% of mechanically thinned plots. Bulldozers are typically used to construct fuel breaks in grassland and shrubland ecosystems, and are not used in the forested areas in the study region. Here, sampling was confined to mechanical thinning and mechanical thin and burn treatments. The very low abundance of non-native plant species seen here precluded statistical analysis of associations between non-native cover and specific treatment methods and intensity, treatment age, and site factors.

Intact forested ecosystems generally appear to be less vulnerable to invasion than other plant communities, such as grasslands and oak savanna (Keeley et al., 2003; Huston, 2004; Von Holle and Motzkin, 2007). This may apply even to forested areas, such as fuel treatments, which are located near roads and therefore in contact with a readily available source of invasive plant propagules. A study in Banff National Park found that the while transportation corridors (highways and railways) promoted the spread of non-natives in both grassland and forest ecosystems, increased non-native frequency extended 150 m into the grasslands from the corridor edge but only 10 m into the forested habitat (Hansen and Clevenger, 2005). In the present study, non-native plant species were commonly observed at treatment edges but were nearly absent within treatment interiors, indicating that these treated areas may be resistant to invasion. The proximity of transportation corridors may have increased non-native abundance within the treatment boundary, but because sampling plots were located 30 m from boundaries, this edge effect was not assessed. Nevertheless, the long period of time elapsed since treatment indicates that treatments have not led to rapid spread of invasives.

One factor likely to influence treatment impacts on the understory plant community is the preventive actions taken by those conducting forest management activities. In the study region the USDA Forest Service, which constructed a majority of the fuel treatments sampled, has employed a number of strategies to avoid spreading non-native plants in areas treated for hazardous fuels reduction. Standards and guidelines include equipment washing, avoiding pre-existing infestations during treatment construction, and timing burning to target established invasives. Beyond initial treatment implementation, sites are monitored and non-native plants are removed (R. Bauer, personal communication). The influence of these preventive management techniques were not evaluated in the current study.

Several authors have called for long-term monitoring of stands treated for hazardous fuels reduction (e.g. Keeley et al., 2003), yet very few studies have assessed treatment effects beyond

1-3 years. In a large-scale, long-term study of treatment effects in ponderosa pine forests of eastern Washington, Nelson et al. (2008) found that thinning and burning led to a small but significant increase in non-native species richness and cover. The most intense treatments (combined thin and burn) led to an average of 2% non-native cover, and non-native abundance remained relatively constant over time. While single-entry fuels treatments may not substantially increase non-native abundance, treatment areas must be maintained over time in order to retain their effectiveness with respect to wildfire hazard. These repeated disturbances could enhance the potential for invasion by non-native plant species (Hobbs and Huenneke, 1992; D'Antonio, 2000).

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APPENDIX 1: Description of Sampling Plots

Plot	*Forest Type*	*Treatment*	*Elevation*	*Aspect*	*Completion year*	*Age*	*Latitude*	*Longitude*
1	EP	TB	1562	140	2006	2	39.705933	-120.475756
2	EP	TB	1555	149	2006	2	39.705989	-120.476687
3	EP	TB	1543	343	2006	2	39.705917	-120.477530
4	EP	TO	1957	61	2006	2	39.477106	-120.219047
5	EP	TO	1963	57	2006	2	39.477763	-120.219800
6	EP	TO	1954	337	2006	2	39.478236	-120.221850
7	EP	TB	1585	140	2006	2	39.733360	-120.477638
8	EP	TB	1595	117	2006	2	39.733957	-120.477779
9	EP	TB	1600	170	2006	2	39.735198	-120.477267
10	MC	TO	1705	232	2006	2	39.698030	-120.491427
11	MC	TO	1684	276	2006	2	39.697432	-120.492078
12	MC	TO	1687	262	2006	2	39.696721	-120.492115
13	MC	TO	1347	310	2006	3	39.861252	-120.686516
14	MC	TO	1355	278	2006	3	39.860541	-120.686540
15	MC	TO	1352	2	2006	3	39.859851	-120.687651
16	EP	TO	2125	336	2005	2	39.465960	-120.294643
17	EP	TO	2128	326	2005	2	39.465399	-120.295315
18	EP	TO	2106	32	2005	2	39.464967	-120.296134
19	EP	TO	2158	349	2005	3	39.525298	-120.170200
20	EP	TO	2158	50	2005	3	39.525799	-120.171168
21	EP	TO	2149	78	2005	3	39.526246	-120.171417
22	EP	TO	1565	265	2005	3	39.703758	-120.460383
23	EP	TO	1573	299	2005	3	39.704794	-120.459971
24	EP	TO	1563	305	2005	3	39.705161	-120.459841
25	EP	TO	1502	357	2005	4	39.811838	-120.506099
26	EP	TO	1509	7	2005	4	39.811832	-120.507092
27	EP	TO	1512	29	2005	4	39.812051	-120.508042
28	MC	TB	1615	341	2004	3	39.868551	-120.993050
29	MC	TB	1591	9	2004	3	39.868159	-120.992243
30	MC	TB	1599	111	2004	3	39.867666	-120.991871
31	MC	TB	1317	130	2004	3	39.968171	-121.069229
32	MC	TB	1347	209	2004	3	39.970511	-121.071786
33	MC	TB	1368	137	2004	3	39.971243	-121.071918
34	MC	TO	1186	306	2004	3	39.924461	-121.042504
35	MC	TO	1176	349	2004	3	39.923635	-121.042153
36	MC	TO	1177	304	2004	3	39.922960	-121.042676
37	MC	TO	1490	215	2004	3	39.887833	-121.016216
38	MC	TO	1488	226	2004	3	39.886605	-121.014965

39	MC	TO	1479	263	2004	3	39.885962	-121.014235
40	EP	TO	1894	54	2003	4	39.524209	-120.148957
41	EP	TO	1910	43	2004	4	39.523482	-120.148696
42	EP	TO	1900	42	2004	4	39.522779	-120.148318
43	MC	TO	1672	10	2004	4	39.684175	-120.469892
44	MC	TO	1643	30	2004	4	39.684376	-120.469628
45	MC	TO	1653	37	2004	4	39.684816	-120.471606
46	MC	TO	1608	336	2004	4	39.692443	-120.459935
47	MC	TO	1592	362	2004	4	39.691989	-120.460196
48	MC	TO	1617	345	2004	4	39.690847	-120.460273
49	EP	TO	1686	90	2004	4	40.102483	-120.493166
50	EP	TO	1699	88	2004	4	40.101899	-120.493222
51	EP	TO	1705	86	2004	4	40.104672	-120.491912
52	EP	TB	1421	252	2004	4	40.203343	-121.197321
53	EP	TB	1435	285	2004	4	40.202426	-121.196875
55	EP	TB	1441	285	2004	4	40.201762	-121.196458
56	EP	TO	1986	326	2003	5	39.557906	-120.138220
57	EP	TO	2014	324	2003	5	39.558781	-120.137183
58	EP	TO	2000	335	2003	5	39.559536	-120.137117
59	EP	TO	1733	40	2003	5	40.114329	-120.447160
60	EP	TO	1737	17	2003	5	40.114638	-120.448088
61	EP	TO	1738	17	2003	5	40.114873	-120.448947
62	EP	TO	1804	100	2003	5	39.384022	-120.180016
63	EP	TO	1793	12	2003	5	39.383638	-120.179033
64	EP	TO	1787	90	2003	5	39.382891	-120.177206
65	MC	TB	1146	169	2002	5	39.953683	-121.071264
66	MC	TB	1146	203	2002	5	39.954138	-121.070467
67	MC	TB	1159	253	2002	5	39.954305	-121.071843
68	EP	TB	1890	170	2002	5	40.208642	-120.568470
69	EP	TB	1885	206	2002	5	40.209072	-120.569206
70	EP	TB	1884	250	2002	5	40.209726	-120.569465
71	EP	TB	1843	16.4	2002	6	39.411984	-120.171468
72	EP	TB	1846	198	2002	6	39.411255	-120.171892
73	EP	TB	1846	27	2002	6	39.410559	-120.172164
74	EP	TB	1830	144	2002	6	39.407053	-120.161414
75	EP	TB	1820	318	2002	6	39.406314	-120.161002
76	EP	TB	1820	154	2002	6	39.405457	-120.161711
77	MC	TO	1562	74	2002	6	40.234103	-121.321396
78	MC	TO	1555	360	2002	6	40.234900	-121.322294
79	MC	TO	1563	335	2002	6	40.235103	-121.323252
80	EP	TO	1830	240	2002	6	39.495867	-120.115444

81	EP	TO	1841	238	2002	6	39.496272	-120.113985
82	EP	TO	1862	226	2002	6	39.496904	-120.113668
83	EP	TO	1912	93	2001	7	39.568507	-120.158642
84	EP	TO	1912	92	2001	7	39.568000	-120.159256
85	EP	TO	1919	96	2001	7	39.567388	-120.160038
86	EP	TO	1714	30	2001	7	40.017757	-120.458369
87	EP	TO	1703	40	2001	7	40.017526	-120.457698
88	EP	TO	1720	23	2001	7	40.016859	-120.456891
89	MC	TO	1346	140	2001	7	40.176222	-121.038316
90	MC	TO	1377	172	2001	7	40.176084	-121.037087
91	MC	TO	1369	164	2001	7	40.175246	-121.036536
92	EP	TO	1385	110	2001	7	40.297048	-121.251035
93	EP	TO	1374	200	2001	7	40.297648	-121.250220
94	EP	TO	1393	230	2001	7	40.298186	-121.249465
95	MC	TO	1225	250	2001	7	40.172503	-120.927929
96	MC	TO	1191	176	2001	7	40.172338	-120.927300
97	MC	TO	1190	210	2001	7	40.172455	-120.926298
98	MC	TO	1555	252	2001	7	40.262791	-121.339115
99	MC	TO	1553	200	2001	7	40.263532	-121.339285
100	MC	TO	1541	221	2001	7	40.263165	-121.340070
101	EP	TB	1836	170	2001	7	39.413009	-120.187697
102	EP	TB	1820	150	2001	7	39.413448	-120.186820
103	EP	TB	1822	119	2001	7	39.412889	-120.188666
104	EP	TO	1867	135	2001	7	39.412906	-120.199780
105	EP	TO	1875	138	2001	7	39.412906	-120.199780
106	EP	TO	1847	144	2001	7	39.411730	-120.199943
107	EP	TO	1821	44	2001	7	39.426709	-120.151834
108	EP	TO	1820	60	2001	7	39.425942	-120.151064
109	EP	TO	1829	60	2001	7	39.426114	-120.149965
110	EP	TO	1938	230	2000	8	39.496515	-120.247133
111	EP	TO	1928	215	2000	8	39.497264	-120.247604
112	EP	TO	1942	224	2000	8	39.497845	-120.247767
113	EP	TO	1810	282	2000	8	39.388621	-120.131744
114	EP	TO	1803	279	2000	8	39.387941	-120.132283
115	EP	TO	1818	284	2000	8	39.390185	-120.131959
116	EP	TO	1522	205	2000	8	40.233336	-121.091513
117	EP	TO	1529	141	2000	8	40.333025	-121.089378
118	EP	TO	1530	180	2000	8	40.333336	-121.090193
119	MC	TO	1723	260	2000	8	40.376365	-121.347436
120	MC	TO	1737	248	2000	8	40.376424	-121.347776
121	MC	TO	1745	255	2000	8	40.377290	-121.347849

122	EP	TO	2110	264	1999	9	39.457730	-120.066504
123	EP	TO	2075	242	1999	9	39.458880	-120.069221
124	EP	TO	2075	238	1999	9	39.459552	-120.069449
125	EP	TO	1418	316	1998	9	40.274486	-121.426123
126	EP	TO	1418	311	1998	9	40.274422	-121.425407
127	EP	TO	1420	319	1998	9	40.275344	-121.424362
128	MC	TB	1579	261	1997	10	39.919366	-120.959824
129	MC	TB	1570	258	1997	10	39.918759	-120.959632
130	MC	TB	1597	260	1997	10	39.918171	-120.959485
131	MC	TB	1256	252	1997	10	40.155900	-120.905950
132	MC	TB	1253	295	1997	10	40.156672	-120.905798
133	MC	TB	1260	319	1997	10	40.157108	-120.905009
134	EP	TB	1600	30	1997	10	40.206341	-120.635908
135	EP	TB	1590	150	1997	10	40.206991	-120.634229
136	EP	TB	1581	50	1997	10	40.207765	-120.634202
137	MC	TB	1119	290	1996	11	40.109980	-120.848392
138	MC	TB	1104	317	1996	11	40.109434	-120.849078
139	MC	TB	1115	293	1996	11	40.109032	-120.849747
140	MC	TB	1269	216	1996	11	39.922553	-120.921200
141	MC	TB	1284	186	1996	11	39.921654	-120.924316
142	MC	TB	1298	166	1996	11	39.921473	-120.925292
143	EP	TB	1951	300	1996	11	40.278207	-120.588877
144	EP	TB	1956	283	1996	11	40.277572	-120.588653
145	EP	TB	1956	235	1996	11	40.277056	-120.587660
146	EP	TO	1263	140	1995	13	40.187753	-120.970467
147	EP	TO	1252	192	1995	13	40.187497	-120.971320
148	EP	TO	1254	208	1995	13	40.187821	-120.972321
149	MC	TB	1620	95	1994	13	40.094165	-120.769257
150	MC	TB	1636	182	1994	13	40.093801	-120.767673
151	MC	TB	1655	194	1994	13	40.093854	-120.766698
152	EP	TO	1466	50	1993	15	40.327256	-121.278164
153	EP	TO	1467	70	1993	15	40.327425	-121.279195
154	EP	TO	1475	90	1993	15	40.327135	-121.280309
155	MC	TB	1122	349	1981	26	39.960965	-120.949386
156	MC	TB	1116	337	1981	26	39.960797	-120.950070
157	MC	TB	1126	314	1981	26	39.961311	-120.948591
158	MC	TB	1082	326	1981	26	39.961576	-120.947799
159	EP	U	1899	198	NA	NA	39.467177	-120.217513
160	EP	U	1903	226	NA	NA	39.467466	-120.218641
161	EP	U	1905	240	NA	NA	39.467785	-120.219918
162	MC	U	1681	13	NA	NA	39.681673	-120.473318

163	MC	U	1681	348	NA	NA	39.681950	-120.471955
164	MC	U	1683	17	NA	NA	39.682206	-120.471305
165	EP	U	1546	201	NA	NA	39.701094	-120.461297
166	EP	U	1544	236	NA	NA	39.700698	-120.460518
167	EP	U	1549	243	NA	NA	39.700504	-120.459849
168	MC	U	1210	55	NA	NA	39.921314	-121.039119
169	MC	U	1234	48	NA	NA	39.920697	-121.040494
170	MC	U	1234	61	NA	NA	39.920098	-121.040710
171	MC	U	1522	340	NA	NA	39.887463	-121.009865
172	MC	U	1516	320	NA	NA	39.886898	-121.008957
173	MC	U	1530	301	NA	NA	39.886945	-121.008523
174	EP	U	2155	4	NA	NA	39.461562	-120.291501
175	EP	U	2149	0	NA	NA	39.462245	-120.290661
176	EP	U	2122	66	NA	NA	39.464195	-120.292387
177	EP	U	1909	120	NA	NA	39.527840	-120.150437
178	EP	U	1904	112	NA	NA	39.527817	-120.151334
179	EP	U	1900	110	NA	NA	39.528309	-120.151581
180	MC	U	1732	134	NA	NA	39.699075	-120.487657
181	MC	U	1728	161	NA	NA	39.698519	-120.488202
182	MC	U	1713	156	NA	NA	39.697867	-120.488482
183	EP	U	2157	240	NA	NA	39.562552	-120.127589
184	EP	U	2121	215	NA	NA	39.561787	-120.125794
185	EP	U	2118	256	NA	NA	39.561877	-120.126907
186	MC	U	1295	115	NA	NA	39.966242	-121.067527
187	MC	U	1286	146	NA	NA	39.965660	-121.067216
188	MC	U	1292	141	NA	NA	39.964695	-121.067126
189	EP	U	2118	12	NA	NA	39.526946	-120.169446
190	EP	U	2131	351	NA	NA	39.526664	-120.168585
191	EP	U	2148	351	NA	NA	39.526256	-120.167764
192	MC	U	1623	329	NA	NA	39.690357	-120.459311
193	MC	U	1609	352	NA	NA	39.689873	-120.460658
194	MC	U	1609	56	NA	NA	39.690670	-120.461644
195	EP	U	1594	41	NA	NA	39.702118	-120.478461
196	EP	U	1580	12	NA	NA	39.702691	-120.479525
195	EP	U	1592	63	NA	NA	39.703736	-120.480362

Table shading indicates plots within the same treatment area (3 plots treatment^{-1}). *Forest types* are: EP: Eastside Pine, MC: Mixed Conifer; *Treatments* are TB: Thin and Burn, TO: Thin Only, U: Untreated Control. *Elevation* is given in meters and *Aspect* is in degrees. *Completion year* is the year of thinning (TO treatments) or the year of burning (TB treatments). *Age* is years since treatment at the time of sampling.