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Title: Fuel Treatment Longevity in the Blue Mountains of Oregon

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Wildland fires are an increasingly extensive, expensive, and frequent occurrence in dry forests of the western United States. Fuel reduction treatments are designed to reduce extreme fire behavior, promote resilient forest structure, and facilitate fire control efforts. Although there is widespread recognition that repeated treatments are needed to maintain desired stand structure and fuel loading, few empirical studies have evaluated the length of time that treatments meet objectives. Fuel treatments tend to open the forest canopy, which increases light and stimulates understory vegetation growth. The length of time fire hazard is decreased within treated stands therefore varies with different forest types and treatment approaches.

Dry ponderosa pine and mixed conifer forests are commonly targeted for fuel reduction in the Pacific Northwest. This study re-measured the Blue Mountains Fire and Fire Surrogate (FFS) study site in the Blue Mountains of northeastern Oregon. In 1998, sixteen units were delineated and assigned to four treatment groups: mechanical thin, prescribed burn, both thin and burn, and no treatment control. My primary research question was: How do fuel loading, tree regeneration, and understory vegetation vary among fuel treatments, measured repeatedly over a 15-17 year period post-treatment, in the Blue Mountains of Oregon?

I examined treatment longevity by comparing pre- and post-treatment fuel loading, tree regeneration, and understory vegetation. The principal findings are: 1) total woody fuel loading15-17 years post-treatment was similar to pre-treatment values; 2) all active treatments result in similar cover by graminoids and shrubs 15-17 years post-treatment; 3) thinning increased tree regeneration over time, and 4) none of the treatments noticeably increased the cover of two invasive grasses of concern.

The intensity of fuel reduction treatments may play a role in the longevity of fire hazard reduction. Low-intensity prescribed fire and thinning from below resulted in few long-term modifications to woody fuel loading and understory vegetation. Quantifying persistent changes in forest conditions can aid in the planning and analysis of future fuels treatments, along with scheduling maintenance of existing treated areas.

Fuel Treatment Longevity in the Blue Mountains of Oregon

by

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Kat Morici, Author

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Fuel Treatment Longevity in the Blue Mountains of Oregon

Kat Morici

CHAPTER 1: INTRODUCTION

Dry forests of the western United States (US) co-evolved with fire. Summer thunderstorms are a common occurrence, resulting in fires that alter plant communities through successional processes. In return, vegetation impacts fire spread and behavior. The interaction of fire with fuels, weather, and topography creates a range of fire intensities, giving rise to a heterogeneous landscape. Plant and animal communities depend on fire as an essential disturbance for the structural diversity it generates (Lindenmayer et al. 2006). A forest that supports a mix of successional stages is predicted to be more resilient to future disturbance (Stevens et al. 2014).

Recent management practices have directly and indirectly homogenized the structure and composition of many dry forests in the western US, prompting concerns about wildfire and sparking debate over future management (Covington and Moore 1994; Hessburg and Agee 2003; Collins et al. 2011). Fuel reduction treatments, such as mechanical thinning and prescribed fire, are widely proposed to address the current state of dry forests, but site-specific examinations of their long-term effects and efficacy are lacking across many geographic areas. This study delves into the longevity of fuel reduction treatments in the Blue Mountains of northeastern Oregon.

The legacy of fire in the Blue Mountains

Historical fire regimes in the Blue Mountains were reconstructed using fire scars and tree ring evidence dating back to the 1400s (Heyerdahl et al. 2001; Johnston 2016). Ponderosa pine (*Pinus ponderosa*) forests east of the Cascade range generally experienced small fires with regularity, and large fires less frequently (Bork 1984). An inverse relationship between fire size and frequency indicates that areas with a longer fire return interval experience large fires more frequently than areas with a shorter fire return interval (Bork 1984; Soeriaatmadja 1965; Heyerdahl et al. 2001).

Topography, elevation, and climate interact to influence characteristics of the fire regime, such as seasonality, frequency, and intensity (Heyerdahl et al. 2001). Most fires occur between midsummer and fall, when fuel moisture is low and thunderstorms are most common (Agee 1996; Holle et al. 2016). The northern Blue Mountains are more topographically complex than the southern Blue Mountains and have a climate influenced by the Columbia River Gorge, which brings higher precipitation and lower summer temperatures (Heyerdahl et al. 2001). This combination of factors allows snow cover to persist longer in the northern area, which maintains higher fuel moisture into the summer, resulting in a shorter fire season. In general, higher elevations experience lower temperatures and higher precipitation, resulting in less frequent fire than low elevations (Johnson and Clausnitzer 1992; Beaty and Taylor 2001; Fulé et al. 2007). However, the importance of elevation is complicated by slope and aspect, as relatively steep north and east aspects receive significantly less solar radiation than south and west facing slopes. Thus, in the more topographically complex terrain at the northern end of the Blue Mountains, moderate to high severity fire was most prevalent on north and east aspects, while low severity fire was most common on south and west aspects (Heyerdahl et al. 2001).

Physical patterns in the environment and temporal variation in weather produced an array of fire effects across the landscape, which are evident in early photographs of the eastern Cascades (Hessburg et al. 2007). Areas of high, moderate, and low fire severity are visible across the spectrum of dry and moist mixed conifer forests, creating a patchy landscape that included all seral stages. Hessburg et al. (2007) concludes that dry mixed conifer forests incorporated patches of moderate and high severity fire as a portion of the mostly low severity fire regime, although the study excludes stands with ponderosa pine potential vegetation types.

The pre-1900 fire regime varied between the northern and southern Blue Mountains, but a reconstruction show fire size and frequency within individual watersheds was relatively consistent from 1687-1900 (Heyerdahl 1997). Johnston (2016) estimated a mean fire return interval of 11-18 years in ponderosa-dominated sites in the southern Blue Mountains. Though Heyerdahl (1997) does not provide an estimate of the mean fire return interval, my own calculations based on data from the paper's fire regime reconstruction suggests the northeastern Blue Mountains had an approximate fire return interval of 10-25 years. An area on the Warm Springs reservation with similar forest type, elevation, and topography had an average minimum interval between fires of 6 years, and average maximum interval of 29 years (Soeriaatmadja 1965). While researchers were able to reconstruct fire frequency, size, and severity from tree ring records, it is impossible to separate human ignitions from lightning ignitions.

Humans and land use in the Blue Mountains

The first human inhabitants in the Blue Mountains are estimated to have arrived 11,000-13,000 years ago (Gilbert et al. 2008). Nomadic people from the Nez Percé, Cayuse, Pauite, Umatilla, and Shoshone tribes seasonally occupied the Blue Mountain territory and used fire as a tool to modify the landscape for hunting and gathering, in support of their way of life (Schwantes 1989). In the mid to late 1700s, the first Europeans travelled into the Blue Mountains to trap animals for the fur trade (Schwantes 1989). The trappers introduced diseases such as smallpox that devastated existing populations of native people, reducing their numbers by 80% over the following century (Boyd 1990). In the mid-1800s, Euro-American settlement began to increase, bringing an estimated eight- to tenfold increase in the number of animals grazing in the area, mostly cattle, sheep, and horses (Irwin 1994). The number of livestock continued to increase until the Taylor Grazing Act in 1934 (Hessburg and Agee 2003; Irwin 1994). Widespread overgrazing limited the amount of fuel available for fires to carry across the landscape.

Many Euro-American settlers also brought the preconceived notion that fire was destructive to the forest's timber resources, although some human ignitions may have occurred to improve livestock range. In the late 1800s, the completion of the transcontinental railroad initiated a long period of selective timber harvest, in which large ponderosa pine trees were removed to feed the growing need for lumber (Hessburg and Agee 2003). Forests of large trees with little understory and litter were noted by Langville (1903) as being "almost immune from fire." These trees had thick bark and no lower branches, adaptations that allowed them to survive the frequent low-intensity fires that characterized the area. The large, knot-free boles of ponderosa pine were preferred by mills. To protect timber resources, an official federal policy of fire suppression was adopted in the early 1900s, and a system was in place to effectively suppress fire on a widespread scale by the 1940s (Pyne 1982).

Changes in human habitation of the Blue Mountains correspond to changes in the fire regime, with the most drastic signal appearing as the near cessation of regular fires after about 1900 (Heyerdahl et al. 2001; Soeriaatmadja 1965; Bork 1984). Around the turn of the 20th century, livestock grazing reduced herbaceous fuels, which carry low-intensity fire, and timber harvesting practices removed the vast majority of old fire-resistant ponderosa pine (Hessburg and Agee 2003). The conditions under which the forest had once developed were no longer present; the removal of regular fire from the ecosystem fundamentally changed the forest from a collection of open ponderosa-dominated stands to favor dense regeneration of shade-tolerant species such as Douglas-fir (*Pseudotsuga menziesii*) and grand fir (*Abies grandis*). While young trees of these species are susceptible to mortality from fire, the long fire-free period in the 1900s allowed for the development of a denser forest with a higher proportion of fire intolerant species than was historically present.

Current state of fire management

Despite widely acknowledged ecological benefits, wildfires are considered a threat in many present-day dry western US forests, as we have evidence that past management has left these forests ill equipped to cope with the current set of challenges. Fire suppression alongside timber harvesting and grazing practices disrupted historically frequent fires, resulting in dense stand structure and high fuel accumulation in ponderosa pine-dominated forests. High fuel loading and periods of drought have been linked to an increase in extremely large and severe wildfires in western US forests over the last several decades (Miller et al. 2008; Dennison et al. 2014).

In addition, human induced climate change has been linked to longer fire seasons and increased frequency of large fires (Westerling et al. 2006; Westerling 2016). Climate models for the Pacific Northwest predict an increase in temperatures of 0.1°C to 0.6°C per decade (Mote and Salathé 2010). The change in precipitation is less certain, but may take the form of a slight increase in precipitation over winter and spring with a decrease in summertime precipitation (Mote and Salathé 2010). Warm springs followed by warm and dry summers have been linked to widespread fire activity in the Blue Mountains (Heyerdahl et al. 2008). Historically, large fires were more common during dry years and El Niño years, while small fires burned regardless of climatic variation (Heyerdahl et al. 2002).

The cost of fighting wildland fire is rising quickly, and is likely to continue with increasing human encroachment into the Wildland Urban Interface (WUI), rising mid-summer temperatures in the western US, and lengthening fire seasons (Westerling et al. 2006). Using suppression as the only method to deal with wildfires is a reactive approach that is not always successful, and it ironically compounds the hazard posed by fire over time.

Fuel reduction treatments

Dry forests that have been effectively treated for fuel reduction are more resilient to wildfire (Agee and Skinner 2005; Stevens et al. 2014). Treating stands with mechanical thinning, prescribed fire, or a combination of the two reduces fuel loading, a measure of biomass weight per unit of area. A reduction in available fuel produces lower-intensity fires, with decreased mortality of mature trees (Prichard et al. 2010, Stephens et al. 2009). However, forest type, initial stand conditions, specific treatment implemented, and time since treatment impact the direction and magnitude of this effect.

Forest fuels are categorized as canopy, ladder, surface, and ground fuels, respectively ranging from the tops of overstory trees, vegetation between the canopy and the surface, understory vegetation, and subsurface organic material (Stephens et al. 2012). Ladder fuels, such as tall shrubs and understory trees, create vertical continuity capable of carrying fire from the forest surface into the canopy, which is considered severe fire behavior. Most fuel treatments aim to reduce surface fuels, create gaps in the canopy, and break up ladder fuels (Graham et al. 2004; Agee and Skinner 2005). These modifications decrease the likelihood of fire spreading into the crowns of trees.

Fuel reduction treatments alter the amount and arrangement of fuels, but require careful implementation to produce the desired reduction in fire hazard. Mechanical thinning without slash removal may result in more woody material on the forest floor, potentially increasing fire intensity (Raymond and Peterson 2005). Prescribed fire applied with the goal of reducing tree density yields snags, which fall over time and contribute to surface fuel loading. Overall, fire hazard in western dry forests is most impacted by the combination of thinning to reduce tree density and prescribed burning to consume the additional slash (McIver et al. 2013). However, Chiono et al. (2012) suggest that treatments opening the forest canopy also encourage vegetation growth in the understory, resulting in a faster return to hazardous fire conditions.

Invasive plant species are another vital consideration when planning fuel treatments, as some aggressive invaders capitalize on disturbed areas. Dodson and Fiedler (2006) correlated higher treatment intensity with increased invasive species. The disturbance associated with fuel treatments must be weighed against wildfires and post-fire seeding, which have a significant effect on the establishment and spread of invasive species (Hunter et al. 2006). Some invasive annual grasses, such as cheatgrass (*Bromus tectorum*), have a positive feedback loop with fire; they cure early in the fire season, provide continuity in areas that are otherwise devoid of fuel, and are capable of rapid establishment following disturbances like fire (Brooks et al. 2004). These characteristics lend a competitive edge in frequently disturbed areas, with a detrimental effect to native vegetation cover.

The effectiveness of fuel reduction treatments also depends on weather conditions and time elapsed since treatment. Extreme fire weather conditions include high temperature, low relative humidity, and strong winds. Wildfires that occur under such conditions are essentially unstoppable, and treated areas may not slow fire spread or lower fire intensity enough to impact suppression efforts. However, case studies provide evidence that treated areas locally modify fire behavior, resulting in increased tree survival even under extreme weather conditions (Graham et al. 2004; Agee and Skinner 2005). Fire behavior becomes less severe within fuel treated areas due to changes in fuel loading and forest structure, the effects of which diminish with increasing time since treatment. The effective lifespan of fuel reduction treatments is estimated to be 10-20 years, although longevity is also assumed to depend on productivity (Omi and Martinson 2010; Chiono et al. 2012). Few long-term studies of fuel treatment longevity exist (Kalies and Yocom Kent 2016). Expanding our understanding of this topic is essential to plan for the maintenance of treated areas in different geographic regions.

Research topic

I conducted a re-measurement of an existing Fire and Fire Surrogate study site (McIver and Erickson 2007) in the Blue Mountains of northeastern Oregon. In 1998, sixteen units were delineated and assigned to four treatment groups: mechanical thin, prescribed burn, both thin and burn, and no treatment control. The minimum fuel treatment objective was to have at least 80% of the basal area of dominant and co-dominant trees survive a head fire under 80th percentile fire weather conditions. Trees, understory vegetation, and woody fuels were measured at a series of research plots within each unit. Pre-treatment and repeated post-treatment measurements took place, with the final measurement 15 years post-burn and 17 years post-thin.

In the first study, I evaluated temporal trends in woody fuel loading. The change from pre-treatment fuel loading was examined 4 years post-burn and 6 years post-thin, as well as 15 years post-burn and 17 years post-thin. These comparisons were made for fine fuels, rotten and sound coarse fuels, as well as total woody fuel loading.

In the second study, my objectives were to characterize the short- and long-term effects of different fuel treatments on understory life forms, tree regeneration, and invasive species.

Goals

The overarching goal of this research is to guide management of these fire-prone landscapes in a way that sustainably incorporates fire with the conservation of a host of forest values. For forests that have not experienced fire since the onset of suppression in the early twentieth century, active forest management can promote a more varied, heterogeneous stand structure (Lindenmayer et al. 2006, Lydersen and North 2012).

Due to the range of treatment types, intensities, and timing, a wide variety of treatments may be implemented to emulate natural disturbance processes and sustain forest heterogeneity. Understanding the effects and lifespan of different fuel treatments is valuable for planning and management. Re-evaluating the fuel loading and understory composition after fifteen to seventeen years will help inform land managers of long-term treatment effectiveness. Tracking the change in common invasive species over time also gauges ecosystem resilience to disturbance. With widespread agreement that continuing on the current management trajectory for fire in dry western forests is not economically or ecologically acceptable, monitoring and adaptive management fill knowledge gaps to illuminate alternative paths for the future.

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CHAPTER 2: LONG-TERM EFFECTS OF FUEL REDUCTION TREATMENTS ON FUEL LOADING IN THE BLUE MOUNTAINS

Introduction

Dry ponderosa pine and mixed conifer forests in the western US are commonly targeted for fuel reduction treatments, though few studies examine the length of time treated areas remain effective. Fuel reduction treatments are designed to reduce extreme fire behavior, promote resilient forest structure, and facilitate fire control efforts. Treating stands with mechanical thinning, prescribed fire, or a combination of the two reduces stand density and alters fuel loading (Agee and Skinner 2005). As a result, a wildfire that occurs in recently treated stands generally has lower potential fire severity, which increases tree survival (Prichard, et al. 2010; Stephens et al. 2009).

The effectiveness of fuel reduction treatments varies based on forest type, stand conditions, specific treatment implemented, and time elapsed since treatment. Dry forests that were characterized by frequent, predominantly low-severity fire regime prior to Euro-American settlement currently support unusually high stand density and fuel accumulation generated by a century of fire suppression along with grazing and timber harvesting practices (Covington and Moore 1994). These forests are good candidates for fuel reduction treatments, while forests with a predominately high-severity fire regime may not realize ecological benefits (Dennison et al. 2014). Treatment implementation is a vital consideration, as the desired reduction in wildfire severity can be negated by residual slash left on site after mechanical thinning (Raymond and Peterson 2005). In addition, applying prescribed fire to reduce tree density can result in fire-killed trees that eventually fall and increase surface fuel loading over time (Schwilk et al. 2009). In western dry forests, using mechanical thinning to reduce stand density followed by prescribed fire to consume surface fuels provides the greatest reduction of fire hazard and movement towards historical stand structure (McIver et al. 2013; North et al. 2007).

Treatment longevity is expected to vary by forest type and productivity, therefore long-term empirical studies in a range of geographic areas are necessary (Chiono et al. 2012; Collins et al. 2011). Such studies can help identify timeframes for recurrent treatments to maintain fuel structure and loading associated with low to moderate intensity fires. Two main components of treatment longevity are stand structure and fuel loading. Treatments designed to reduce stand density create vertical and horizontal openings, which fill in with trees and understory vegetation over time. Additionally, woody debris re-accumulates on the forest floor post-burn or slash decomposes post-thin. Changes in surface fuel loading may have a substantial impact on wildfire behavior, and are the focus of this paper.

Surface fuel loading is a measure of the weight of nonliving woody biomass on or near the forest floor per unit of area (Mg/ha), and is categorized by size and decay status. While fire intensity, or energy released per second per unit of fire edge, is largely affected by the total amount of woody material that is available to burn, rate of spread, or speed of forward fire movement, is primarily driven by fine fuel loading (NWCG, 2016, Rothermel and Forest 1972). Fine fuels are defined in this paper as dead woody material on the forest floor with a diameter of less than 7.6 cm. Coarse woody fuels have a diameter greater than 7.6 cm, and are categorized as sound or rotten. Total woody loading is the combination of fine and coarse fuel loading.

The effect of single-entry fuel reduction treatments is expected to diminish with increasing time since treatment, though empirical research on treatment longevity has been identified as a knowledge gap (Kalies and Yocom Kent 2016). Several studies from California provide insight into mid and long-term changes in fuel loading following fuel reduction treatments. Keifer et al. (2006) found the majority of fuel re-accumulation in central Sierra Nevada ponderosa pine forests took place within the first decade following prescribed fire, mostly in the form of fire-killed trees falling to the forest surface. In the mixed conifer forests of the northern Sierra Nevada and southern Cascades, a chronosequence of mechanical fuel reduction treatments revealed highly variable fuel loading across the time series, likely due in part to different methods of post-thinning slash treatment (Chiono et al. 2012). Stephens et al. (2012) observed that mechanical treatments without fire did not initially reduce fire hazard due to the residual fine fuels produced during thinning. After 7 years, fine fuels had decayed, resulting in similar fire hazard among thin, burn, and combination treatments. Vaillant et al. (2015) found total surface fuel load to be near pre-treatment values 8 years post-treatment, which was within the range of the mean fire return interval for the area. Quantifying persistent changes in fuel loading within different forest types and geographic regions can aid in the planning and analysis of future fuel treatments, along with prioritizing and scheduling maintenance of existing treated areas.

The Blue Mountains Fire and Fire Surrogate (FFS) study site in northeastern Oregon was initiated to examine the ecological effects of common fuel treatments, as well as the applicability of using mechanical thinning as a surrogate for fire. It is a prime location to evaluate the lasting effects of mechanical thinning and prescribed fire, singly and in combination, as the treatments near the end of their suspected lifespan.

My primary research questions were: (1) Is there a significant difference between pre-treatment and post-treatment woody fuel loading, measured 4 years post-burn and 6 years post-thin, within each treatment? (2) Is there a significant difference between pre-treatment and post-treatment woody fuel loading, measured 15 years post-burn and 17 years post-thin, for both individual treatments and a combination of thinning and burning? These questions were investigated for fine fuel, rotten and sound coarse fuel, as well as total woody fuel loading.

I hypothesized that total woody fuel loading would be reduced 4-6 years post-treatment for all three active fuel reduction treatments. However, 15-17 years post-treatment, no detectable change from pre-treatment values is expected. Fine fuels (diameter <7.6 cm) were hypothesized to recover within the initial 4-6 years post-treatment. The no-treatment control is included in this analysis to represent fuel dynamics over time and space in an unmanipulated stand.

Methods

Study site

The Blue Mountains FFS study area is located within the Wallowa-Whitman National Forest in northeastern Oregon. Study units were spread across 50 km^2, but limited to mid-elevation sites, ranging from 1100 m to 1400 m elevation. Sites are dominated by ponderosa pine (*Pinus ponderosa*), with Douglas-fir (*Pseudotsuga menziesii*) secondary and intermittent grand fir (*Abies grandis*), lodgepole pine (*Pinus contorta*), and western larch (*Larix occidentalis*). Fire regime reconstructions from similar ponderosa pine-dominated forest in the neighboring Malheur National Forest estimate a pre-1900 fire return interval of 11-18 years (Johnston 2016). Timber harvesting, grazing, and fire suppression have altered the fire regime for over a century, contributing to a denser forest with a higher proportion of shade-tolerant species than found in the era prior to Euro-American settlement (Agee 1996).

Treatment implementation

Blue Mountains FFS researchers delineated sixteen treatment units ranging in size from 8 to 20 hectares (Youngblood et al. 2006). Units were then randomly assigned to one of four treatments, for a total of four replications of each treatment. The treatments included mechanical thinning, prescribed burning, a combination of thinning and burning, and a no-treatment control. Thinning treatments took place after the initial measurement in 1998, consisting of a thin from below, with a preference for retaining large trees or snags and fire-tolerant species, specifically ponderosa pine. Residue from thinning remained on site. Units were broadcast burned in the fall of 2000; weather and fuel conditions during ignition are described in Youngblood et al. (2008). Overall, fire effects were low to moderate, with flame lengths averaging <0.3 m, although thin and burn units tended to support more intense fire than burn-only units due to residual slash from thinning. The post-treatment basal area (BA) target was 16 m^2/ha, and desired fine fuel loading was ≤ 4.5 Mg/ha. Five units met the target BA pre-treatment, and all treatments except the control met fine fuel target pre-treatment (Figure A-2: Tree basal area (BA, m^2/ha) by status and species for each treatment and measurement year at the Blue Mountains FFS site.).

Field Sampling

Depending on the size and shape of the unit, approximately 25 grid points with 50 m spacing between points were located as sampling plots, for a total of 380 plots within 16 units. At each plot, researchers measured trees, understory vegetation, and woody fuels. Data were collected in all units pre-treatment in 1998 (1999 for three of the burn-only units) and post-treatment in 2004 and 2015. The 2015 measurement undertaken by this study consisted of eight of the original plots within each unit, selected at random from all of the plots in the unit, for a total of 128 plots. This number of plots was feasible within time and budgetary constraints.

Three 20 m Brown's transects (Brown 1974) were measured at each plot. During the initial data collection, the first transect followed a random azimuth from plot center, and the second and third were placed at 120° offsets. The original azimuths were obtained and used for re-measurements whenever possible. On a 3 m section of each transect, 1-hr (>0.0-0.64 cm) and 10-hr (0.64-2.5 cm) fuels were tallied. Along the entire transect, 100-hr fuels (2.5-7.6 cm) and 1000-hr fuels (>7.6 cm) were counted. For 1000-hr fuels, diameter and decay class were recorded. The transect lengths were 3.048 m (10 feet) for 10-hr fuels in 1998, and 21.68 m (66 feet) for 10-hr fuels in 2004, as well as 100 and 1000-hr fuels. In the 1998-2004 measurements, field sampling methods did not include tallying 1-hr fuels. The 2015 sampling effort showed 1-hr fuels to be 1.5 times the 10-hr fuel count, so this estimation is used as a surrogate for previous years' 1-hr fuel counts.

Fuel loading was calculated for each treatment unit as the mean loading of all plots measured in that unit for a given measurement year. As suggested in Brown's Handbook (1974), inputs to the fine fuel calculations were adjusted by the presence of slash and proportion of dominant tree species. Trees species, diameter at breast height, height, and height to live crown was recorded at each plot during each sampling year in 400 m^2 circular plots.

Data analysis

The starting condition of each unit varied due to site differences and past management; consequently this study compared the change in fine and coarse woody fuel loadings from the initial measurement to the post-treatment measurements for each treatment. Each treatment was applied to four units, allowing for analysis of changes in fuel loading within the treatments over time.

As mentioned in Hungerford et al. (1991), site productivity influences fuel loading. To obtain a surrogate for fine scale differences in productivity, the 30-year monthly averages of the daily maximum vapor pressure deficit (VPD) was gathered for June, July, and August (PRISM Climate Group 2016). Each plot was assigned a summer average VPD value based on its spatial location, and plot values were averaged to the unit level.

Linear mixed effects models (R Core Team 2016; Pinheiro et al. 2015) were created to investigate the long-term effect of mechanical thinning, prescribed burning, and a combination treatment on fuel loading while allowing for different starting conditions and variances among each treatment and year. Fixed effects included VPD, treatment, year, and the interaction of treatment and year. Unit was included as a random effect, with four replicates per treatment. We accounted for temporal autocorrelation by designing models with various correlation structures, and choosing among the models for each response variable using the lowest BIC (Table A-1). Plots of normalized residuals were investigated for normality and constant variance among treatments. If BIC values differed by less than 2, the model with more appropriate residual plots was selected. To examine differences in size and decay classes, separate models were built for 1-hr, 10-hr, 100-hr, all fine fuel, all coarse fuel, sound 1000-hr, rotten 1000-hr, and total woody fuel loading.

To determine how woody fuel loading changed over time, differences in average fuel loading from the 1998 pre-treatment measurement to the 2004 and 2015 measurements were calculated for each treatment. A 95% Bonferroni correction was used to adjust for the eight comparisons of interest (adjusted confidence intervals 99.38%). Due to inherent site differences, such as initial basal area, species composition, and soil type, the treatments were not directly compared to the no action control.

Results

Overall F-tests for model fixed effects revealed statistically significant evidence that the unit-level influence of fuel reduction treatment depended on the study year for fine fuel loading, but not for 1000-hr or total fuel loading, after accounting for site productivity through VPD (Table 2-1). The fuel treatment type and year of measurement were also independently significant influences on woody fuel loading dynamics across most size classes, with the exception of sound 1000-hr fuels. Fuels were highly variable across all the treatments, resulting in few comparisons reaching a statistically significant difference between pre- and post-treatment values in 2004 or 2015. Estimates of differences in fuel loading between years and estimates of mean fuel loading for a given year are derived from the selected mixed model.

Table 2-1: Results of overall F-tests for fuel loading model fixed effects. The response variable is fuel category, which is specified at the top of the corresponding set of results. Bold indicates significance (p-value <.05).

All fine			
Predictor	DF	F-value	p-value
treatment	**3,11**	**9.14**	**0.003**
year	2,24	3.33	0.053
VPD	1,11	0.37	0.558
treat:year	**6,24**	**3.53**	**0.012**
1-hr			
treatment	**3,11**	**4.26**	**0.032**
year	**2,24**	**4.12**	**0.029**
VPD	1,11	0.05	0.820
treat:year	6,24	2.06	0.096
10-hr			
treatment	**3,11**	**5.35**	**0.016**
year	2,24	3.08	0.065
VPD	1,11	1.31	0.277
treat:year	**6,24**	**4.34**	**0.004**
100-hr			
treatment	**3,11**	**10.79**	**0.001**
year	**2,24**	**3.42**	**0.049**
VPD	1,11	0.04	0.851
treat:year	6,24	2.44	0.055

All 1000-hr			
Predictor	DF	F-value	p-value
treatment	**3,11**	**4.02**	**0.037**
year	**2,24**	**8.24**	**0.002**
VPD	1,11	0.59	0.459
treat:year	6,24	0.76	0.611
Sound 1000-hr			
treatment	3,11	2.19	0.146
year	2,24	1.72	0.200
VPD	1,11	2.09	0.176
treat:year	6,24	1.71	0.161
Rotten 1000-hr			
treatment	**3,11**	**9.21**	**0.003**
year	**2,24**	**11.12**	**<.001**
VPD	1,11	0.05	0.835
treat:year	6,24	2.39	0.059
All woody			
treatment	**3,11**	**5.49**	**0.015**
year	**2,24**	**6.51**	**0.006**
VPD	1,11	0.54	0.478
treat:year	6,24	1.17	0.355

Fine fuels

Fine fuel loading was examined as a combination of 1, 10, and 100-hr fuel loading. The no-treatment control unexpectedly resulted in an estimated 2.8 Mg/ha reduction in mean fine fuel loading between 1998 and 2004 (Figure 2-1). Not surprising was a similar pattern in the burn-only treatment, with an estimated decrease of 0.99 Mg/ha. The initial mean fine fuel loading in the control was 6.8 Mg/ha, noticeably higher than the initial 2.6 Mg/ha in the burn-only (Table B-3). After 15 years, a reduction in fine fuel loading is only evident in the burn-only treatment, mainly due to the drop in

100-hr fuels, which were estimated to be 0.7 Mg/ha less than the 1998 measurement. Mechanical thinning with and without burning did not result in statistically detectable shifts in fine fuel loading at any point over the course of the study.

Coarse fuels

Sound 1000-hr fuel loading did not display a statistically significant change in any of the treatments either 4-6 years or 15-17 years post-treatment (Figure 2-1). However, a reduction of mean sound 1000-hr loading was estimated to be 6.2 Mg/ha in the no-treatment control between 1998 and 2015 (Table B-3). The corresponding increase, estimated to be 9.1 Mg/ha, in rotten 1000-hr loading indicates that a major portion of the sound large fuels are decaying. After adjusting for multiple comparisons, evidence for these changes are not statistically significant, but presents a reasonable interpretation for the movement in both rotten and sound large fuel loading. While all active treatments showed decreases in mean rotten 1000-hr fuel loading 4-6 years post-treatment, only the thin and burn combination reached a statistically significant decrease. The estimated reduction in rotten 1000-hr fuel loading of 5.3 Mg/ha did not persist 15-17 years post-treatment.

All fuels

Total woody fuel loading was significantly reduced four years post-burn in the burn-only treatment, with an estimated reduction of 6 Mg/ha (Figure 2-1). Thinning followed by burning appears to have a similar effect, although it was not detectable statistically. The comparisons between pre-treatment and 15-17 years post-treatment yielded no evidence that thinning, burning, or a combination of the two had a statistically significant effect on mean total woody fuel loading. The no-treatment control also did not display a statistically significant difference in mean woody fuel loading over the course of the study. However, fluctuations in mean fuel loading in the control rivaled the changes which took place in the active treatments (Figure 2-2, Table 2-2)

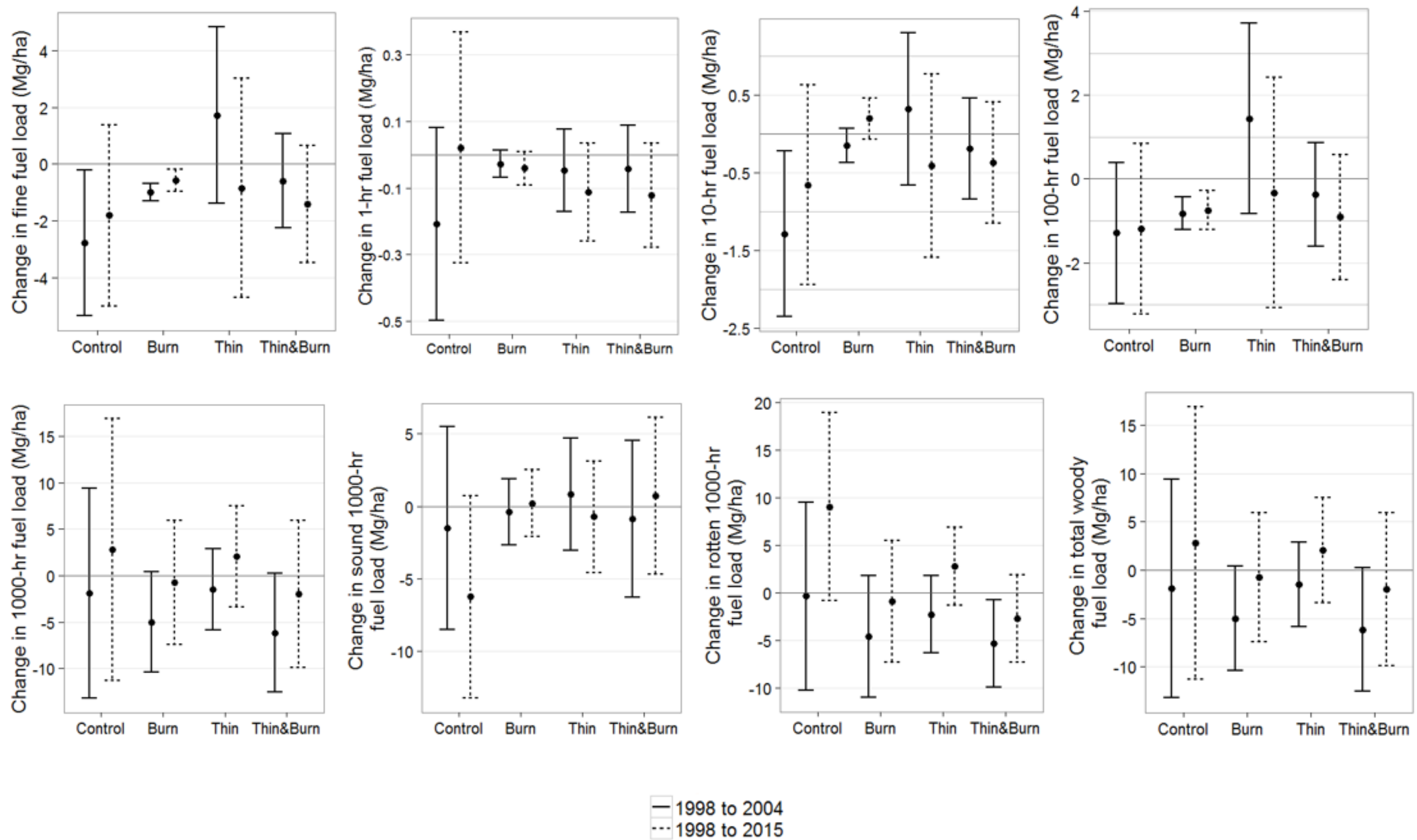

Figure 2-1: Plots displaying the differences in mean average fuel loading between the initial and post-treatment measurements for each treatment at the Blue Mountains FFS site. Error bars represent the 99.38% confidence intervals for the differences. The line at 0 represents no difference.

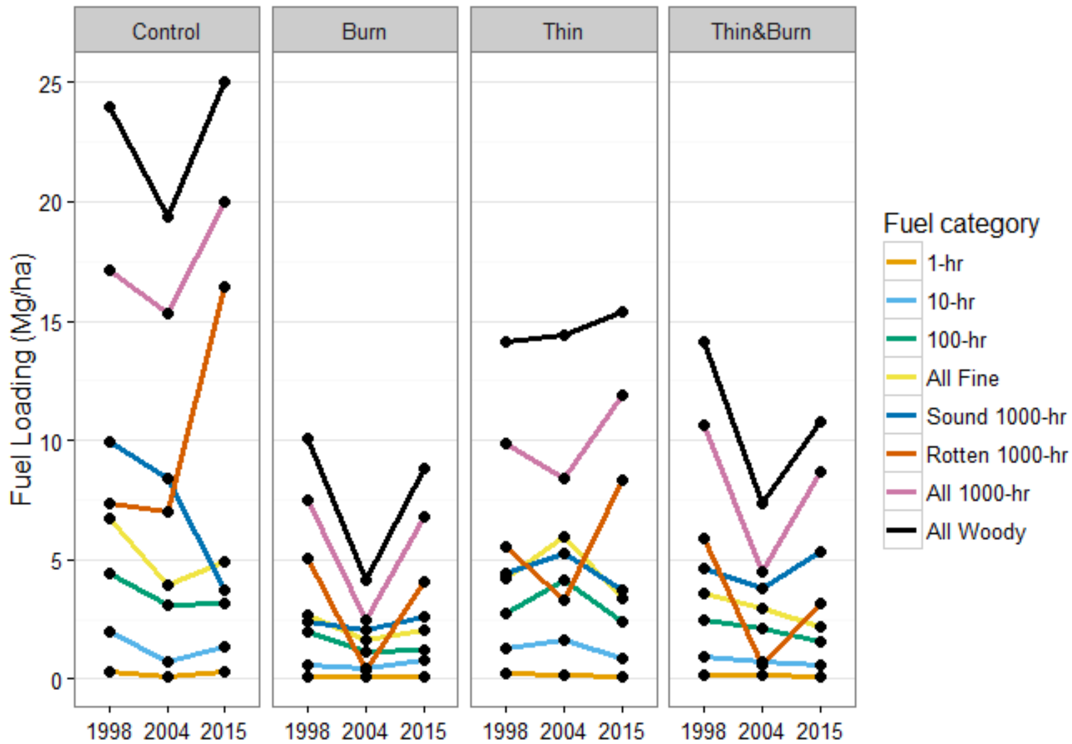

Figure 2-2: Estimated mean fuel loading by treatment and year for the Blue Mountains FFS site. Line color indicates fuel loading category.

Table 2-2: Estimated mean total woody fuel loading and percent change from initial loading for each treatment and year.

Treatment & year	Estimated mean loading (Mg/ha)	% of 1998 loading
Control 1998	24.0	-
Control 2004	19.3	80.6
Control 2015	25.0	104.0
Burn 1998	10.1	-
Burn 2004	4.1	40.8
Burn 2015	8.8	87.2
Thin 1998	14.2	-
Thin 2004	14.4	102.0
Thin 2015	15.4	108.4
ThinBurn 1998	14.1	-
ThinBurn 2004	7.4	52.2
ThinBurn 2015	10.8	76.2

Discussion

The impact of fuel reduction treatment on fine fuel loading depended on time since treatment, which indicates that different treatment approaches set fine fuels on unique trajectories over time. However, this finding did not hold for coarse woody loading. Rotten 1000-hr fuel loading varied by treatment and year, which suggests that treatments differentially impacted course rotten fuels, and decay over time had a noticeable effect on rotten fuel dynamics. Neither treatment nor year greatly affected sound 1000-hr fuels, demonstrating their relative stability as an element of dry forest systems in the face of infrequent low-intensity management actions.

<u>Fine fuels</u>

The impact of fuel reduction treatments on fine fuel loading in the Blue Mountains was nearly imperceptible after 15-17 years. Mid-term change (4-6 years) in fine fuel loading was most evident as a significant reduction in the control, which was unexpected. Perhaps the control units supported a microclimate better suited to rapid decomposition over those years. Several of the control units have been identified as moister sites, with deeper soils and higher soil water holding capacity than are present in most of the actively treated units. Ponderosa pine and Douglas-fir roots have been found to decompose faster with higher moisture levels (Chen et al. 2000), and the same trend may hold for fine fuels on the forest surface. Generally, the rate of decay in fine fuels has not been well studied. It is unclear if all units, left untreated, would have experienced a similarly high rate of decay noticed in the control. Sampling and measurement error are also possible causes of the abrupt decline found in these data. Regardless, the fine fuel loading decline in control units was not maintained in the 2015 measurement.

Temporary decreases in fine fuels are expected after prescribed fire, since fine fuels are the primary carriers of fire, and are consumed in the process (Raymond and Peterson 2005). However, the measured reduction was slight 4 years post-fire and imperceptible 15 years post-fire. Thinning is expected to increase fine fuel loading due to residual slash, but this effect is not evident 6 years post-treatment. Mixed-conifer forests in the north-central Sierra Nevada experienced a significant increase in fine fuel loading 1 year post-thin, and decreased to pre-treatment levels by 7 years post-thin

(Stephens et al. 2012). A chronosequence study of mechanical fuel treatments in the eastern Sierras showed 1-hr fuel loading reaching a low 5-7 years post-treatment, and returned to untreated levels after 8 or more years (Chiono et al. 2012). The study grouped sites that were only mechanically treated with sites that were also broadcast or pile burned, thus it is not possible to tease out the effects of individual treatments.

Coarse fuels

The implementation of thinning and prescribed burning treatments designed to restore resilient forest conditions also maintained coarse woody fuels, which are important habitat for a variety of wildlife species (Bull et al. 1997). Sound 1000-hr fuels did not show obvious changes due to treatment in this study, but were noticeably reduced in the control, where they appeared to transition into coarse rotten fuels over the 17 year study. The evident reduction in rotten 1000-hr fuels post-burn was expected, as found in other studies (Raymond and Peterson 2005, Stephens and Moghaddas 2005a). Rotten large fuels were likely cured when prescribed burning took place in the fall, which, along with their reduced density and higher surface area to volume ratio, increased their flammability and consumption. Sound fuels were not as affected, which is consistent with observations made in other studies (Hyde et al. 2011). The possibility that sound logs were consumed in the prescribed fire and replaced with fire-killed trees is unlikely in this study, given that total basal area increased over the same time period. Many studies of fuel reduction treatment consider all coarse fuels as a single response variable (e.g. Keifer et al. 2006; Vaillant et al. 2015). However, the differing responses of sound and rotten coarse woody fuels in this study demonstrate a need to examine them separately.

All fuels

Total woody fuel loading did not exhibit a long-term detectable difference for any fuel reduction treatment. Keifer et al. (2006) found mean total fuel load for ponderosa pine in the southern Sierra Nevada to be reduced by 99% immediately post-burn, reaching 84% of pre-treatment load 10 years post-burn, mostly due to fire-killed trees falling. As coarse wood makes up the bulk of fuel loading at the Blue Mountains site, it is reasonable to assume that prescribed fire in this study did not have a rate of consumption similar to the aforementioned study. An investigation of prescribed fire and mechanical thinning treatments across California's coniferous forests found total fuel load 8 years post-treatment to be 67-79% and 55-103% of the initial loading for burn-only and thin-only treatments, respectively (Vaillant et al. 2015). The reduction in total woody loading found in California for the burn-only treatment falls between the 4-year and 15-year post-burn measurements our Blue Mountains study (estimated to be 41% and 87% in the burn-only and 52% and 76% in the thin and burn). The thin-only estimates are also similar, estimated to be 102% of initial loading 6 years post-thin, and 108% of initial loading 17 years post-thin.

One possible explanation for the lack of a middle-term effect of treatment on total woody loading in our Blue Mountains study is that treatments were designed to have a light-touch. A portion of the sixteen units met basal area and fine fuel loading targets pre-treatment, reducing the need for a more intense prescribed fire. Another reason could be the length of time between measurements. A full re-measurement was not completed until 4 years post-burn and 6 years post-thin. Partial measurements took place one year post-thin and immediately post-burn, but were not considered in this paper due to lack of corresponding tree data. Perhaps some of the effects, especially on fine fuels, were too transitory to be captured in the mid-term time frame. Finally, the variability of fuels across the landscape may have provided enough noise to obscure trends in surface fuel loading with the number of plots measured in this study. However, the similar range of variance between the 2015 measurement of 128 plots and previous years' 380 plots does not provide evidence for this point.

Downed woody fuel dynamics have not been well studied in these systems, and current fire models are not precise enough to hone in on a definite change in fuel loading that would substantially impact fire behavior (Keane 2013). Published literature reports changes in post-treatment fuel loading, but do not interpret the numerical differences relative to established thresholds for ecological significance given landscape-level variability in fuel loading (Stephens and Moghaddas 2005b, Chiono et al. 2012, McIver et al. 2013). Following conversations with subject matter experts, a 4.5 Mg/ha difference in total woody fuel loading between the initial and post-treatment loading was suggested to be a meaningful difference. In the future, this estimate can be refined by more accurate fire behavior models and additional empirical data. Fuel loading is highly variable across landscapes, so detecting a statistical difference may also be a meaningful result for the size and power of this study.

Conclusions

It is critically important for humans to coexist with wildfire, one of the primary disturbances in dry forested landscapes (Moritz et al. 2014). The current trajectory of increasingly frequent severe fire and rising spending is not sustainable (Dennison et al. 2014). Fuel treatments, such as mechanical thinning and prescribed fire, are tools used to create fire-resilient landscapes through the alteration of live and dead fuels. However, this study did not find light-touch fuel reduction treatments to have a lasting impact on most categories of dead woody fuel loading. It is important to note that applying heavier thinning or more intense prescribed fire may lead to different outcomes. To re-create the heterogeneity produced by a patchwork of fires across the historical dry forest landscape, treatments could be applied every 5-25 years, with more intense treatments following longer treatment-free intervals. Continued monitoring of treated areas will supply a picture of fuel loading dynamics across a range of treatments and ecosystems.

Fuel loading is extremely variable across the Blue Mountain landscape, and detecting small differences may not be practical or biologically important to fire hazard and land management practices. The high variability of fuel loading across landscapes makes the use of controls somewhat speculative in all but the most homogenous study areas. When examining the results of fuel reduction treatments applied to distinct stands, the most accurate method of comparison may be to look at the change in each stand over time, using the control as representative of loading dynamics in an unmanipulated stand, instead of a direct comparison between the treatment and control stands.

Refined estimates of the effect of loading by fuel size class on fire behavior may lead to a more informative estimate of biological significant change. Further research into the environmental drivers and rate of fine fuel decay is necessary. Fine fuel quantity and arrangement is typically the determining factor of fire spread (Rothermel and Forest 1972). A baseline reference for a rate of decay in different forest types would help inform the variability that could be reasonably expected over time. Regular monitoring of fuel loading at intermediate times would provide a more complete picture of how fuels change in each treatment over time. Understanding the trajectory of fuel re-accumulation is advantageous when planning for future management activities.

Design limitations

There are several limitations regarding the applicability of these results to broader implications of fuel reduction treatments. First, the no-treatment control units in this study were not facsimiles of the other treatment units. Factors such as basal area, trees per hectare, species composition, soils, and aspect show the control to skew towards more productive sites. The burn-only and combined thin and burn units tend to inhabit less productive areas. As such, we chose not to compare actively treated units directly to the control units. In the future, care should be exercised in selecting analogous sites or using a blocked design. I attempted to capture the influence of productivity on fuel loading with the inclusion of VPD in the mixed models, but this technique has not been evaluated for accuracy.

Another consideration in applying the results of this study is the limited geographic area and relatively uniform site factors. Thus, the scope of inference is limited to mid-elevation ponderosa pine and dry mixed conifer forests in the locality of the study area. However, patterns can inform treatment longevity estimates in areas with similar forest type, structure, and environmental conditions. Long-term monitoring of fuel loading in treated stands can help inform land managers of the longevity of different fuel treatments. Since site-specific conditions are expected to affect longevity, it is vital to expand the area of inference by examining post-treatment fuel dynamics in an array of geographic areas and forest types.

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CHAPTER 3: FOREST UNDERSTORY RESPONSE TO FUEL REDUCTION TREATMENTS IN THE BLUE MOUNTAINS

Introduction

Fire is a keystone disturbance in mixed conifer forests of the western US. For the past century, forest management practices in the US involved actively suppressing most wildland fires. Fire suppression along with other land management practices in dry ponderosa pine and mixed conifer forests have led to unusually high fuel loading and continuity among fuels, which have contributed to a recent increase in stand-replacing wildfire, especially during periods of drought (Miller et al. 2008; Dennison et al. 2014). Also, the absence of periodic fire has resulted in the establishment of shade-tolerant species and changes in the cover of herbaceous plants in the understory as the canopy closes (Riegel et al. 1995, Naumburg and DeWald 1999, Carr and Krueger 2011). Understory vegetation affects the spread of fire over the forest surface and influences the likelihood of a crown fire, which is considered severe fire behavior.

Fuel reduction treatments, such as mechanical thinning and prescribed fire, are used to moderate potential fire behavior and enhance heterogeneous forest structure in dry ponderosa pine and mixed conifer forests. Most fuel treatments aim to reduce continuity of the canopy fuels (tree crown bulk density, arrangement) and break up ladder fuels (canopy base height, saplings, tall shrubs). The amount of heat produced per unit area is closely related to flammable large woody material; however, fine fuel availability governs the rate of spread and flame height (Rothermel and Forest 1972). Removing fine fuels, such as twigs, graminoids and forbs, temporarily diminishes the ability of the system to carry low-intensity surface fire. However, it has been suggested that treatments opening the forest canopy also stimulate vegetation growth in the understory, resulting in a faster return to hazardous fire conditions (Chiono et al. 2012). The impact of fuel treatments on understory dynamics can be examined by tracking changes in understory structure over time. Understory structure is characterized by percent cover by life forms, such as shrubs and graminoids, as well as seedling and sapling density.

From both a fire management and ecosystem health perspective, invasive plants are an added consideration when determining which treatment to implement, as they tend to colonize disturbed areas (Hobbs and Huenneke 1992; Stapanian et al. 1998). Thinning followed by burning has led to increased cover by invasive species in ponderosa pine-dominated mixed conifer forests (Dodson and Fiedler 2006). Compared to the thin-only and burn-only treatments, the combined treatment had a more intense effect on the overstory, which was linked to higher cover by invasive species that are known to alter ecosystem properties (Dodson and Fiedler 2006). The possible promotion of invasive plants by fuel treatments must be balanced against the risk of severe wildfires and post-fire seeding in certain areas, which have a significant effect on the establishment and spread of invasive species (Hunter et al. 2006). Post-fire grazing has also been cited as a potentially influential factor in the expansion of invasive species (Keeley 2006). Large invasions by some species, such as *Bromus tectorum* and *Ventenata dubia*, allow an area to burn more frequently than the historic fire return interval, with a detrimental effect to native vegetation cover (Brooks et al. 2004, Scheinost et al. 2008).

Examining long-term trends in understory structure and composition of treated stands informs land managers of the ecological effect of different fuel treatments, as well as the length of time fire hazard is reduced. In addition, understanding the trajectory of vegetation re-growth is essential to plan for maintenance of treated stands, whether through repeated treatments or wildland fire use. Regrowth varies as a function of species present, site productivity, topography, the treatment implemented, and time since treatment (Chiono et al. 2012). Since site-specific conditions and annual weather variability are expected to affect longevity, it is vital to expand the area of inference by examining post-treatment regeneration across a range of geographic areas and forest types.

The Blue Mountains Fire and Fire Surrogate (FFS) study site, treated 15-17 years ago, is a prime location to explore the lasting effect of three fuel treatments in northeastern Oregon. One of the original goals of the study was to describe the suitability of using mechanical fuel treatments as a surrogate for prescribed fire in fire-adapted forests (McIver et al. 2012). The length of time fire hazard is reduced within treated stands depends in part on the structure of the understory vegetation. In this study, the objectives were to characterize the effect of different fuel treatments on 1) understory percent cover by life form, 2) seedling and sapling density, and 3) percent cover by invasive species.

Methods

Study Site

The Blue Mountains study area is comprised of dry ponderosa pine-dominated mixed conifer forest within the Wallowa-Whitman National Forest in northeastern Oregon. The study units are spread across 50 km^2, but are limited to mid-elevation sites, ranging from 1100 m to 1400 m. Ponderosa pine (*Pinus ponderosa*) is the dominant tree species in the area, with Douglas-fir (*Pseudotsuga menziesii*) secondary and scattered grand fir (*Abies grandis*), lodgepole pine (*Pinus contorta*), and western larch (*Larix occidentalis*). Prior to 1900, the fire regime was characterized by frequent low-intensity fires on south and west aspects, and more intense, less frequent fire on north and east aspects (Heyerdahl et al. 2001). Timber harvesting, livestock grazing, and fire suppression began in the early 1900s. Grazing and suppression have largely continued to the present day.

Field Sampling

As part of the FFS project, sixteen units in the Blue Mountains were delineated, measured, and randomly assigned to four treatment groups: mechanical thin, prescribed burn, both thin and burn, and no treatment control. A 50 m grid was overlaid across each unit, with circular plots located at grid points. Grid points were designated as permanent research plots. The units ranged from 10-20 ha, with a total of 380 plots established across all sixteen units. The study design and 1998-2004 data collection effort were completed by FFS researchers; we undertook the 2015 data collection effort.

Within each plot, tree species, DBH (diameter at breast height), status (live or dead), height, percent canopy cover, and understory vegetation were recorded. Seedlings over 10 cm and less than 1.37 m were tallied by species. Saplings were defined as trees with a DBH < 7.62 cm. Canopy cover was measured with a moosehorn densiometer at plot center and at a 2m offset in each cardinal direction. In the 1998 measurement, plot size was 200 m^2; post-treatment measurements expanded plot size to 400 m^2. Understory vegetation was characterized by an ocular estimate of percent cover by a subset of 49 common species in 1998. Post-treatment in 2001 and 2004, an estimate of percent cover by all species in each plot was completed. In 2015, the survey was modified to an ocular estimate of the percent cover by life form (graminoids, forbs, shrub), percent cover of uninhabited area (litter/soil or bare/rock), as well as percent cover of a set of 10 invasive species of concern to local managers (Table 3-1). Two of the ten invasive species were not found at any plots, and two common invasive species were also collected in 1998.

While pre-treatment measurement did not encompass all species present at the site, the species collected in 1998 comprised the vast majority of graminoid and shrub cover in 2001. Calculating the percent cover by life form in 2001 using only species collected in 1998 showed a 2.5% difference in the total cover by graminoids and 3.7% difference in the total cover by shrubs in 2001 compared to a calculation using all species collected in 2001. Thus, comparisons to pre-treatment graminoid and shrub data were determined to be reasonable. However, the 2001 forb cover revealed a 145.5% difference when calculated using species collected in 1998 compared to all species collected in 2001. As a result, total forb cover was determined to be incomparable pre- and post-treatment.

Table 3-1: Invasive species collected in 2015. *not found in any plots in 2015. **collected in 1998

Grass		**Forb**	
Scientific Name	**Common Name**	**Scientific Name**	**Common Name**
*Bromus tectorum***	cheatgrass	*Cynoglossum officinale*	hounds tongue
Ventenata dubia	ventenata	*Cirsium vulgare*	bull thistle
*Dactylis glomerata***	orchardgrass	*Hypericum perforatum*	St. John's wort
*Taeniatherum canput-medusae**	medusahead rye	*Centaurea diffusa**	diffuse knapweed
Bromus japonicus	Japanese brome	*Potentilla Recta*	sulphur cinquefoil

All plots were measured pre-treatment in 1998, post-treatment in 2001, 2004, and a random subset of 8 plots per unit were measured in 2015. Data collection took place between June and August. To mitigate the effect of seasonal trends on estimates of vegetation cover, at least one unit from each treatment was measured during every site visit. For detailed description of the original study design, see the FFS Network Study Plan (McIver and Erickson 2007).

Treatment implementation

Mechanical thinning took place in 1998, and prescribed burning was accomplished in the fall of 2000. The mechanical thin consisted of a thin from below, preferentially retaining large ponderosa pine. The prescribed fire was characterized by low to moderate fire effects, with higher-intensity fire in the combined thin and burn treatment compared to the burn-only due to the addition of slash from thinning (Youngblood et al. 2008). See appendix for treatment effects on basal area and trees per hectare by species and status (Figure A-2, Figure A-3).

Data analysis

The vegetation life form data contained the unit average percent cover by graminiods and shrubs at 16 units over 4 measurement years. The tree regeneration data included a unit average density per hectare of seedlings and saplings by species. Life form and tree regeneration data were analyzed using PC-ORD version 7 (McCune and Mefford 2015). The environmental matrix contained measurements to describe habitat variables at each plot, in addition to variables relating to the experimental design. Slope and aspect were combined into a heat load index to account for their combined effect on vegetation (McCune and Keon 2002). The total soil depth of each soil type was gathered from the USDA OSD website (NRCS Soils 2017). Inherent site productivity differences were estimated by including the maximum vapor pressure deficit (VPD) expressed as a 30-year average over June, July, and August (PRISM Climate Group 2016). Increases in VPD often result in decreased available soil water, which is generally a limiting factor for vegetation growth in western US dry forests during the summer.

Non-metric scaling (NMS) was used to examine variation among unit with respect to their understory vegetation composition (Mather 1976; Kruskal 1964). In NMS ordinations, units which are more similar in composition are located closer together. The distance between points widens with increasing dissimilarity in their overall vegetation compositions. This procedure has the advantage of not assuming linear relationships between variables. For the vegetation life form matrix and the tree regeneration matrix, Sørensen distance was used. NMS was initiated under the "medium" setting on autopilot using a random starting configuration. For each version of the main matrix, fifty runs were used for both the real data and the randomized Monte Carlo test. To confirm that a stable solution was reached, plots of stress for all iterations were checked, along with the reported final instability. The real data run with the lowest stress was used to interpret results (Bruce McCune and Grace 2002).

Pairwise comparisons between treatment-year combinations were run with multi-response permutation processes (MRPP) to test for differences in vegetation composition and regeneration density between treatments over time (Zimmerman et al. 1985). The combination of fuel reduction treatment and measurement year was used as the grouping variable. As with the NMS procedures, Sørensen distance was used for the life form matrix and the regeneration matrix. Treatments were compared to each other within distinct measurement years, as well as comparisons within each treatment over the four measurement years to track change over time. Bonferroni adjustments were used to account for multiple comparisons within each set of contrasts.

Individual invasive species were analyzed with R version 3.3.1 using linear mixed models to examine changes in cover over time (R Core Team 2016). The only invasive species of concern present in both the initial data collection and the 2015 measurement was cheatgrass (*Bromus tectorum*). However, a relatively recent invasive grass, *Ventenata dubia*, was recorded starting in 2001, so the post-treatment percent cover trends were analyzed for this species. Separate linear mixed models were built for each of the three invasive grass species, using treatment, year, VPD, and the combination of treatment and year as fixed effects. The sampling unit was included as a random effect. Various correlation structures were examined and the model with the lowest BIC and acceptable residual plots was selected (Table B-4).

Results

<u>Understory vegetation by life form</u>

In the vegetation life form NMS ordination, Axis 1 is positively correlated to percent cover by graminoids, and Axis 2 is negatively correlated to percent cover by shrubs (). Of the environmental variables, canopy cover displays a negative correlation to Axis 2 (r=0.47), indicating higher shrub cover in areas with closed canopies. VPD is positively correlated to both axes ($r_{Axis\ 1}$ =0.41, $r_{Axis\ 2}$ =0.46), showing drier conditions positively relate to graminoid cover and negatively relate to shrub cover.

The ordination reveals similarity in the end state of graminoids and shrubs after mechanical thinning, prescribed burning, and a combination of thinning and burning measured 15 years post-burn and 17 years post-thin. Despite differing initial conditions in 1998, the centroids of the units are nearly co-located in 2015 (). The pairwise MRPP results confirm that active fuel reduction treatments do not differ from one another in 2015 with respect to percent cover of graminoids and shrubs (Table 3-2). All treatments show some departure from the control in the final measurement year, although the differences are not statistically significant after adjusting for multiple comparisons. In 1998, the burn-only is suggestively dissimilar from the thin-only and control treatments, again the differences are not significant after the Bonferroni adjustment. The no action control fluctuates throughout the study across a wide range with relation to Axis 1, and the end state is distinctly apart from the active treatments. Despite the wide fluctuation, pairwise MRPP comparisons of each treatment over time reveal only one statistically significant difference, after adjusting for the number of comparisons. The thin-only increased in both graminoids and shrubs between 1998 and 2004. During this time period, variance between the four thin-only units dropped dramatically.

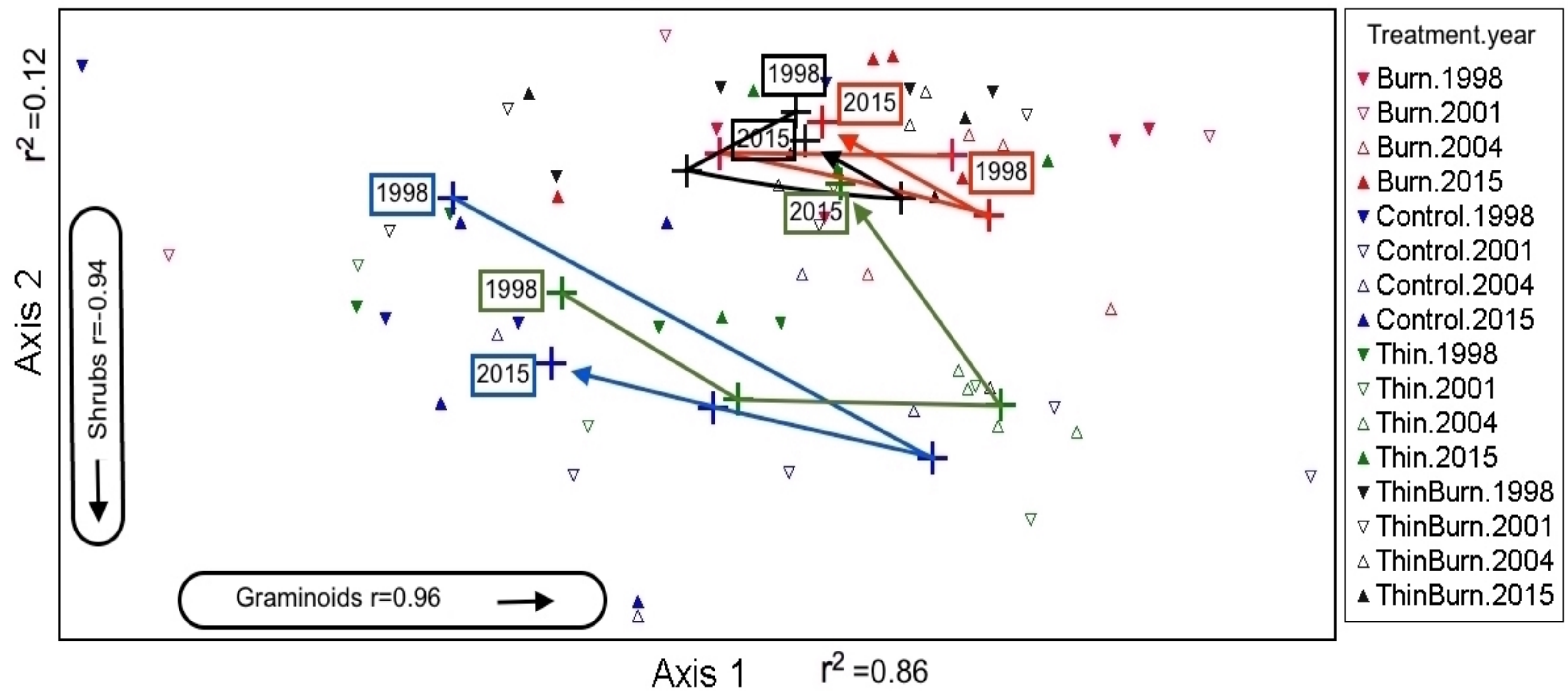


Figure 3-1: NMS ordination of sampling units in life form space for the Blue Mountains FFS site. Each triangle represents one unit, the orientation and fill of triangles indicate year of measurement. The centroid of each treatment-year combination is represented by a cross, and the 1998 and 2015 centroids are labeled. Lines are drawn chronologically between the centroids of each treatment-year (1998, 2001, 2004, and 2015). Blue = control, Red = burn-only, Green = thin-only, Black = thin and burn.

Table 3-2: MRPP pairwise comparisons between treatments for each year, and within each treatment between years for the Blue Mountains FFS site. Bold indicates significance (p-values ≤ 0.008). T is the test statistic, A is chance-correlated within-group agreement.

Life Form

Between treatment comparison	T	A	p-value
ThinBurn.1998 vs. Burn.1998	0.095	-0.011	0.429
ThinBurn.1998 vs. Control.1998	-1.029	0.106	0.132
ThinBurn.1998 vs. Thin.1998	-1.549	0.155	0.078
Burn.1998 vs. Control.1998	-1.950	0.196	0.049
Control.1998 vs. Thin.1998	0.859	-0.075	0.829
Burn.1998 vs. Thin.1998	-2.291	0.244	0.024
ThinBurn.2001 vs. Burn.2001	1.092	-0.103	0.929
ThinBurn.2001 vs. Control.2001	-1.170	0.095	0.120
ThinBurn.2001 vs. Thin.2001	0.170	-0.019	0.442
Burn.2001 vs. Control.2001	-0.574	0.040	0.234
Burn.2001 vs. Thin.2001	0.169	-0.015	0.477
Control.2001 vs. Thin.2001	0.690	-0.061	0.734
ThinBurn.2004 vs. Burn.2004	0.519	-0.037	0.652
ThinBurn.2004 vs. Control.2004	-1.673	0.140	0.062
ThinBurn.2004 vs. Thin.2004	-2.312	0.284	0.036
Burn.2004 vs. Control.2004	-2.662	0.217	0.015
Burn.2004 vs. Thin.2004	-2.767	0.241	0.012
Control.2004 vs. Thin.2004	-2.755	0.237	0.014
ThinBurn.2015 vs. Burn.2015	1.009	-0.113	0.848
ThinBurn.2015 vs. Control.2015	-2.205	0.203	0.038
ThinBurn.2015 vs. Thin.2015	0.949	-0.073	0.895
Burn.2015 vs. Control.2015	-2.297	0.234	0.036
Burn.2015 vs. Thin.2015	0.892	-0.068	0.814
Control.2015 vs. Thin.2015	-2.894	0.219	0.016

Within treatment comparison	T	A	p-value
ThinBurn.1998 vs. ThinBurn.2001	0.540	-0.065	0.632
ThinBurn.1998 vs. ThinBurn.2004	0.194	-0.015	0.484
ThinBurn.1998 vs. ThinBurn.2015	0.985	-0.122	0.893
ThinBurn.2001 vs. ThinBurn.2004	-0.375	0.035	0.318
ThinBurn.2001 vs. ThinBurn.2015	0.621	-0.078	0.685
ThinBurn.2004 vs. ThinBurn.2015	0.849	-0.052	0.811
Burn.1998 vs. Burn.2001	0.478	-0.043	0.600
Burn.1998 vs. Burn.2004	0.480	-0.047	0.613
Burn.1998 vs. Burn.2015	0.320	-0.031	0.570
Burn.2001 vs. Burn.2004	-0.514	0.033	0.274
Burn.2001 vs. Burn.2015	0.752	-0.055	0.758
Burn.2004 vs. Burn.2015	-1.215	0.071	0.117
Control.1998 vs. Control.2001	-2.127	0.155	0.040
Control.1998 vs. Control.2004	-0.681	0.054	0.203
Control.1998 vs. Control.2015	0.760	-0.040	0.762
Control.2001 vs. Control.2004	0.296	-0.023	0.544
Control.2001 vs. Control.2015	-1.636	0.128	0.067
Control.2004 vs. Control.2015	0.225	-0.021	0.531
Thin.1998 vs. Thin.2001	0.058	-0.007	0.422
Thin.1998 vs. Thin.2004	-3.719	0.472	**0.007**
Thin.1998 vs. Thin.2015	-1.808	0.152	0.050
Thin.2001 vs. Thin.2004	-1.000	0.098	0.158
Thin.2001 vs. Thin.2015	-0.963	0.075	0.156
Thin.2004 vs. Thin.2015	-3.470	0.355	0.009

Regeneration

Between treatment comparison	T	A	p-value
ThinBurn.1998 vs. Burn.1998	0.002	0.000	0.401
ThinBurn.1998 vs. Control.1998	-1.664	0.066	0.057
ThinBurn.1998 vs. Thin.1998	-0.598	0.022	0.258
Burn.1998 vs. Control.1998	-3.613	0.221	**0.007**
Control.1998 vs. Thin.1998	0.597	-0.031	0.679
Burn.1998 vs. Thin.1998	-2.303	0.122	0.032
ThinBurn.2001 vs. Burn.2001	-0.201	0.011	0.383
ThinBurn.2001 vs. Control.2001	-3.308	0.192	0.009
ThinBurn.2001 vs. Thin.2001	-3.459	0.183	**0.008**
Burn.2001 vs. Control.2001	-3.256	0.190	**0.008**
Burn.2001 vs. Thin.2001	-3.218	0.191	0.010
Control.2001 vs. Thin.2001	0.457	-0.035	0.597
ThinBurn.2004 vs. Burn.2004	1.289	-0.068	0.912
ThinBurn.2004 vs. Control.2004	-3.736	0.258	**0.008**
ThinBurn.2004 vs. Thin.2004	-3.763	0.251	**0.008**
Burn.2004 vs. Control.2004	-3.532	0.189	**0.008**
Burn.2004 vs. Thin.2004	-3.519	0.177	**0.008**
Control.2004 vs. Thin.2004	0.192	-0.013	0.471
ThinBurn.2015 vs. Burn.2015	-2.011	0.114	**0.039**
ThinBurn.2015 vs. Control.2015	-1.003	0.047	0.152
ThinBurn.2015 vs. Thin.2015	-1.211	0.066	0.118
Burn.2015 vs. Control.2015	-2.013	0.103	0.048
Burn.2015 vs. Thin.2015	-4.105	0.280	**0.006**
Control.2015 vs. Thin.2015	-0.911	0.035	0.179

Within treatment comparison	T	A	p-value
ThinBurn.1998 vs. ThinBurn.2001	0.822	-0.048	0.787
ThinBurn.1998 vs. ThinBurn.2004	0.085	-0.005	0.473
ThinBurn.1998 vs. ThinBurn.2015	-1.438	0.071	0.087
ThinBurn.2001 vs. ThinBurn.2004	-1.122	0.073	0.133
ThinBurn.2001 vs. ThinBurn.2015	-1.629	0.101	0.065
ThinBurn.2004 vs. ThinBurn.2015	-3.127	0.196	0.013
Burn.1998 vs. Burn.2001	0.927	-0.070	0.822
Burn.1998 vs. Burn.2004	1.479	-0.100	0.946
Burn.1998 vs. Burn.2015	1.589	-0.082	0.967
Burn.2001 vs. Burn.2004	1.231	-0.096	0.896
Burn.2001 vs. Burn.2015	0.520	-0.026	0.664
Burn.2004 vs. Burn.2015	1.413	-0.081	0.940
Control.1998 vs. Control.2001	-0.773	0.052	0.194
Control.1998 vs. Control.2004	-0.780	0.049	0.185
Control.1998 vs. Control.2015	-2.193	0.104	0.032
Control.2001 vs. Control.2004	1.193	-0.111	0.913
Control.2001 vs. Control.2015	0.791	-0.037	0.776
Control.2004 vs. Control.2015	0.548	-0.027	0.679
Thin.1998 vs. Thin.2001	0.778	-0.042	0.771
Thin.1998 vs. Thin.2004	0.786	-0.042	0.773
Thin.1998 vs. Thin.2015	-2.009	0.125	0.041
Thin.2001 vs. Thin.2004	1.359	-0.137	0.920
Thin.2001 vs. Thin.2015	-0.064	0.005	0.376
Thin.2004 vs. Thin.2015	0.172	-0.016	0.462

Tree regeneration

In the NMS ordination of tree regeneration, Axis 1 is negatively correlated to ponderosa pine and Douglas-fir seedlings and saplings (**Error! Reference source not found.**). Axis 2 is positively correlated to Douglas-fir saplings, and has modest negative correlation with ponderosa pine seedlings. Other tree species expressed very weak negative correlations with Axis 1. An examination of environmental variables revealed total soil depth had a strong negative correlation with Axis 1 (r=-0.48).

The ordination reveals an increase in tree regeneration 17 years after thinning, with and without burning. The combined thin and burn treatment is initially similar to the burn-only treatment, and by the final measurement year is located in the vicinity of the thin-only and the control (**Error! Reference source not found.**). An examination of these data reveals this dramatic change is driven by an increase in seedlings in the thin and burn treatments, but an increase in both seedlings and saplings in the thin-only (Figure A-4, Figure A-5). MRPP does not explicitly confirm this result, although pairwise comparisons between treatments for each year show a significant difference between the thin and burn and the thin-only in 2001, as well as a significant difference between the thin and burn and the control in 2004, but no difference between any of these treatments in 2015 (Table 3-2). The only statistically significant difference in 2015 is between the thin-only and the burn-only treatments. Although this difference is suggested by the pre-treatment comparison, it does not reach the threshold of statistical significance after adjusting for multiple comparisons. The initial state of the thin-only and control was noticeably lower on Axis 1 than the initial state of the burn-only and thin and burn. Since soil depth was strongly negatively correlated to Axis 1, the thin-only and control appear to inhabit sites with deeper soils. This observation is supported with an investigation of soil depth for each treatment (Figure 3-3).

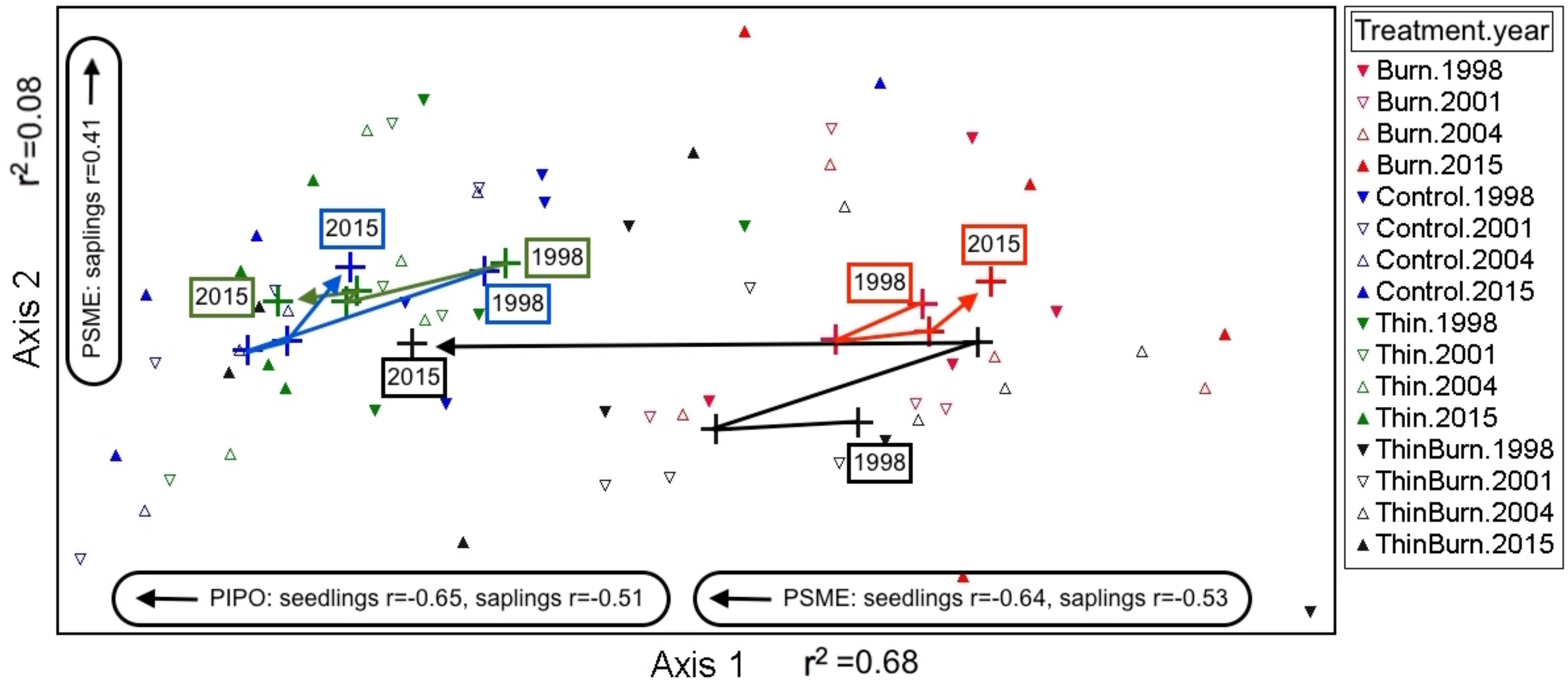
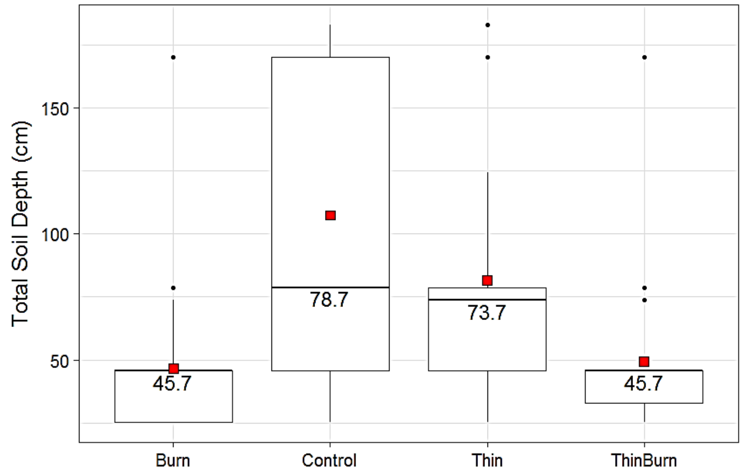


Figure 3-2: NMS ordination of sampling units in tree regeneration space for the Blue Mountains FFS site. Each triangle represents one unit, the orientation and fill of triangles indicate year of measurement. The centroid of each treatment-year combination is represented by a cross. Lines are drawn chronologically between centroids of each treatment-year (1998, 2001, 2004, and 2015). Blue = control, Red = burn-only, Green = thin-only, Black = thin and burn.

Figure 3-3: Total soil depth for each treatment at the Blue Mountains FFS site. The mean is represented by a red square. The median value is given.



Individual invasive species

The influence of fuel reduction treatment on cheatgrass and ventenata was negligible. All treatments, including the control, experienced a temporary increase in mean cover by cheatgrass from 1998 to 2004, the effect of which was not evident for any treatment in 2015. The control was the only treatment to have a statistically detectable change, between the 1998 and 2004 measurements (Figure 3-4). However, the change was slight, increasing an estimated 0.9 percent in mean cover by cheatgrass. The other treatments showed suggestive, though not statistically significant, increases in mean percent cover during this time period. The range of variability in the thin-only and combined thin and burn indicates that cheatgrass cover may be high in certain plots, though it is not a dominant component of the understory vegetation when averaged across the larger area.

Ventenata was not included in the pre-treatment sampling, so all comparisons are between post-treatment measurements at 2001, 2004, and 2015. An examination of post-treatment ventenata dynamics indicate a possible increase, though not a statistically significant one, in all active fuel reduction treatments between 2001 and 2015. In all treatments, estimated mean cover of ventenata increased by less than one percent. Ventenata was found in only two control plots during the study. High variability in the control may be a function of rarity. The range of variability in the combined thin and burn treatment may indicate that ventenata cover is concentrated within particular plots, not evenly distributed across the landscape.

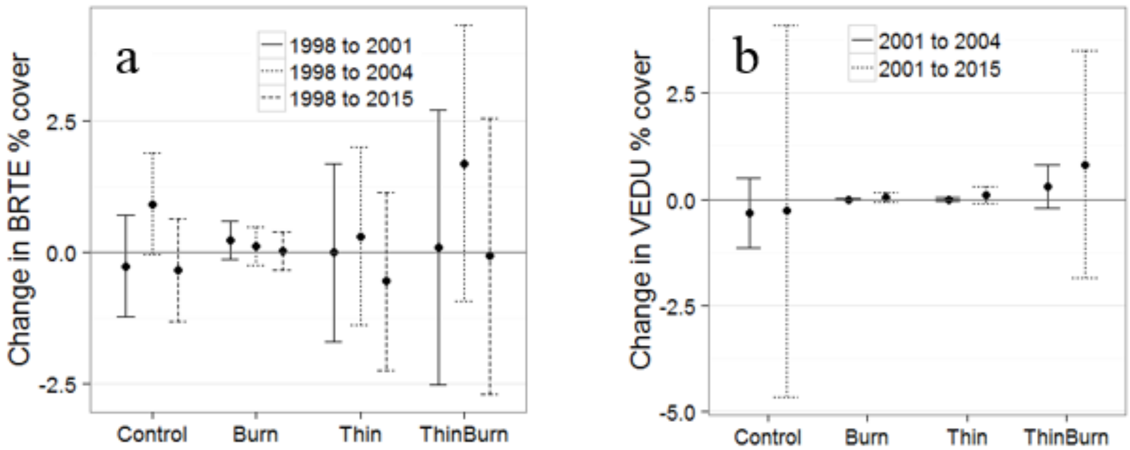


Figure 3-4: Plots displaying the change in mean percent cover of a) cheatgrass (BRTE) and b) ventenata (VEDU) between measurement years for each treatment at the Blue Mountains FFS site. Error bars represent the confidence intervals for the differences (BRTE = 99.58%, VEDU = 99.38%). The 0 line represents no difference.

Table 3-3: Estimated differences in mean percent cover of cheatgrass (BRTE) and ventenata (VEDU) between measurement years for each treatment at the Blue Mountains FFS site. Confidence intervals for BRTE = 99.58% and VEDU = 99.38%

Species	Treatment	Comparison	Estimated difference	t-vaue	DF	p-value	Confidence Interval
BRTE	Control	2001 v 1998	-0.27	-0.86	35	0.394	(-1.19,0.65)
		2004 v 1998	0.91	2.88	35	0.007	(-0.01,1.83)
		2015 v 1998	-0.35	-1.10	35	0.280	(-1.26,0.57)
	Burn	2001 v 1998	0.23	1.90	35	0.066	(-0.12,0.57)
		2004 v 1998	0.12	0.98	35	0.333	(-0.23,0.46)
		2015 v 1998	0.03	0.23	35	0.823	(-0.32,0.37)
	Thin	2001 v 1998	-0.01	-0.02	35	0.982	(-1.62,1.6)
		2004 v 1998	0.30	0.54	35	0.591	(-1.31,1.91)
		2015 v 1998	-0.55	-1.00	35	0.323	(-2.16,1.05)
	ThinBurn	2001 v 1998	0.09	0.10	35	0.920	(-2.4,2.57)
		2004 v 1998	1.68	1.97	35	0.056	(-0.8,4.17)
		2015 v 1998	-0.08	-0.09	35	0.927	(-2.56,2.4)
VEDU	Control	2004 v 2001	-0.32	-1.15	23	0.261	(-1.14,0.51)
		2015 v 2001	-0.28	-0.19	23	0.849	(-4.65,4.09)
	Burn	2004 v 2001	0.00	0.00	23	1.000	(-0.02,0.02)
		2015 v 2001	0.05	1.39	23	0.179	(-0.06,0.16)
	Thin	2004 v 2001	0.00	-0.22	23	0.829	(-0.04,0.04)
		2015 v 2001	0.10	1.48	23	0.153	(-0.11,0.31)
	ThinBurn	2004 v 2001	0.29	1.72	23	0.099	(-0.22,0.8)
		2015 v 2001	0.82	0.92	23	0.365	(-1.85,3.5)

Discussion

Understory vegetation by life form

Thinning, burning, and thinning followed by burning all resulted in a similar composition of graminoid and shrub percent cover in the understory after 15-17 years. Burning with and without thinning did not appear to dramatically alter cover of graminoids and shrubs over the course of the study. Dodson et al. (2008) found little change in understory community composition after similar fuel reduction treatments, measured 2 years post-burn. Low prescribed fire intensity may not have been sufficient to produce a noticeable understory response (Busse et al. 2000, Knapp et al. 2007). Another possible explanation involves the shallow soil depth in most of the burn-only and thin and burn units compared to the thin-only and control units. A study of fuel reduction treatments in the Sierra National Forest revealed soil moisture was associated with understory communities pre-treatment, but post-treatment communities were also associated with soil depth and litter depth (Wayman and North 2007). Shallow soils could dampen vegetative response to treatment.

Thinning alone may temporarily increase cover by woody shrubs, as it opens the canopy and does not cause fire-induced mortality. A woody shrub response likely results in more ladder fuel, capable of bringing fire from the forest surface into tree crowns. While impermanent, the increase in shrub cover in the thin-only treatment may or may not be significant to fire behavior, but was noted at other Fire and Fire Surrogate sites (Schwilk et al. 2009). The no-treatment control experienced a decrease in graminoid cover, which may impede surface fire spread, but wide fluctuations throughout the study do not suggest a strong trend over 17 years. However, the 2002 change in grazing regime from sheep to cattle (USFS, personal communication, 2016) that occurred after the initial measurement in six of the sixteen units complicates the interpretation of changes in understory cover. The affected pasture includes three of the four control units, two burn-only units, and one thin-only unit. Sheep commonly browse all available palatable vegetation, while graminoids make up the majority of the diet of cattle in the Blue Mountains (Hutchings and Stewart 1953; Gade and Provenza 1986; Doran 1943; Holechek et al. 1982). In addition, shrubs tend to be variably distributed across the landscape, with extensive patches present in some units. Forbs were not examined because pre-treatment cover data were not collected for most forb species. Ocular estimates of vegetation percent cover over large plots have a wide margin of error, so small differences are interpreted with caution.

Tree regeneration

In this study, thinning was associated with an increase in tree regeneration, which diminished treatment effectiveness at mitigating crown fire risk at longer time scales. Seedlings increased 17 years after thinning, in both the shallower soils of the thin and burn treatment and the deeper soils of the thin-only treatment. Thinning alone resulted in an increase in saplings as well. In the initial measurement, sites with deeper soils, mostly the control and thin-only, had more seedlings and saplings, as well as higher overall TPH. Both ponderosa pine and Douglas-fir follow the same trend, with a surge of seedlings following thinning with and without burning, across a range of soil depths, evident in the 2015 measurement. This indicates that thinning increases tree regeneration in the Blue Mountains, perhaps by opening the canopy and reducing competition for water and nutrients (Stone and Wolfe 1996). The prescribed fire did not appear to affect much change in tree regeneration over either the short or the long term, perhaps due to the low-intensity fire produced by burning under weather conditions when fire is likely to stay within designated perimeters (Ryan et al. 2013). A moderate-intensity fire would kill many existing seedlings and small saplings, but also has a greater likelihood of entering the crowns of trees in areas with high fuel accumulations and ladder fuels, making the fire more difficult to control.

Mature ponderosa pine are classically adapted to survive low- to moderate-intensity surface fires. The preservation of an intact overstory means that light, water, nutrient availability, and therefore tree regeneration, may be unlikely to increase substantially after low-intensity prescribed fire. However, other studies found that prescribed fire resulted in an increase in pine recruitment, likely due to reduced competition after removal of competing understory vegetation (Bailey and Covington 2002; Zald et al. 2008). Seedling response to fuel reduction treatment was notably variable between Fire and Fire Surrogate sites, potentially due to cone production year and other site-specific conditions (Schwilk et al. 2009).

Individual invasive species

Fuel reduction treatments can be implemented in ponderosa pine and Douglas-fir forests of the northern Blue Mountains area without a large risk of invasion by cheatgrass and ventenata. None of the various fuel reduction treatments experienced a significant invasion of these species over the course of this study. This finding was surprising because disturbance is linked to the spread of many invasive annual grasses (Griffis et al. 2001; D'Antonio and Vitousek 1992). Annual grasses respond strongly to changes in temperature and precipitation (Pitt and Heady 1978), thus a single measurement year may not capture the full extent of a cheatgrass or ventenata invasion. Overall, cover of the two invasive grasses was very low. However, high variability of the results indicates that invasive cover may be concentrated in a small number of plots that could serve as a seed sources for invasion during favorable conditions. Monitoring for potential invasion is beneficial, as proximity to roads, recreation facilities, grazing, and changes in climate may alter these results.

Conclusions

In the northern Blue Mountains of Oregon, light thinning from below, low-intensity prescribed burning, and a combination of the two resulted in few significant changes in understory composition and structure 15 years post-burn and 17 years post-thin. All active treatments resulted in similar cover of graminoids and shrubs. Thinning with and without burning increased seedling density of the dominant tree species, ponderosa pine and Douglas-fir. The increase in tree regeneration was notable in both shallow and moderately deep soils. One important finding is that active fuel reduction treatments did not permanently promote the cover of two invasive grasses of interest, cheatgrass and ventenata.

Pre-existing differences in site environmental characteristics, such as soil depth, and management, such as grazing history limit direct comparisons between treatments. Additionally, the plot size and number of species collected increased after the initial 1998 measurement. The 2015 measurement is a subsample of 128 of 380 plots, potentially influencing dynamics of patchy variables, like shrubs. Due to the variability in sites and sampling, inference into understory vegetation dynamics is limited at this study site. Regardless, monitoring areas treated for fuel reduction will continue to provide insight into understory development and fire hazard over time. Sustained monitoring of treated areas is needed across a range of geographic areas and ecosystems.

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CHAPTER 4: CONCLUSIONS

The Blue Mountains Fire and Fire Surrogate study was originally designed to examine the economic and ecological effects of fuel reduction treatments, testing the applicability of implementing mechanical treatments, with and without prescribed fire, in the place of historically high-frequency, low-severity fire. As a pioneering study, it provided insight into post-treatment effects on a multitude of variables, from the overstory canopy to the ectomycorrhizal fungi in the soil (Schwilk et al. 2009; Smith et al. 2005). I chose to focus this thesis on fuel loading and understory characteristics, as these physical variables are likely to determine whether fire stays on the ground or climbs into the crowns of trees, which is regarded as severe fire behavior in these types of dry forests. As with all pioneering efforts, the study has limitations that can guide better, future research, both in terms of the experimental design and methods. The Blue Mountains project was the first of twelve FFS sites to be set up, and valuable insights were learned from its shortcomings.

The first limitation is the completely randomized study design, which did not consider the range of edaphic conditions within the study area. Despite site selection that accounted for elevation, topographic position, and plant association, control and thin-only units systematically inhabit areas with deeper soils than the burn-only and combined thin and burn units. The inherent differences in productivity complicated direct comparisons between treatments. Since soil is a key driver of aboveground biomass potential, blocking on soil type or available soil water is suggested for future studies. I attempted to account for the impact of the productivity gradient by including Vapor Pressure Deficit (VPD) in statistical models. While this method is far from ideal, VPD has a relationship to forest type and climate.

Other limiting factors are the change in grazing use and data collection methods over time. Six of the sixteen units are located within a grazing pasture that was leased for use by sheep for the first several years of the study, then allowed to rest for two years before being leased for cattle grazing. The remainder of the units were consistently leased for cattle grazing throughout the study. Differences in grazing preferences between sheep and cattle complicate the interpretation of treatment effects on changes in understory vegetation over the course of this study. In addition, pre-treatment data collection included smaller plots and fewer understory species than later measurements. Thus, it is not possible to track all species of concern over the course of the study. This is a common failure of monitoring programs—changes in methodology without a cross-walk step. However imperfect, the overall study and this particular site provides a robust dataset to investigate questions of fuel treatment longevity on fuel loading and understory composition.

Principle findings

Woody fuels

Fuel treatments had little effect on dead and down woody fuel loading 15-17 years any fuel reduction treatment. In the absence of extreme weather, the severity and pattern of historical fires in mixed conifer forests was limited in part by lack of fuel from previous fires (Collins et al. 2009). Heyerdahl et al. (2001) found evidence of frequent fire prior to 1900 in the Blue Mountains area. In a frequent-fire fuel-limited system, we expected that fuel loading would change immediately and then re-build to pre-fire levels within the timeframe of this experiment. The magnitude of the initial changes was nearly imperceptible, given the light nature of both the mechanical thinning and prescribed fire.

Separating fuel loading into size and decay classes, other trends came to light. Prescribed fire reduced fine fuel loading, a trend which held 15 years post-treatment. Burning also appeared to lower rotten coarse woody loading, although this was only evident 4 years post-treatment. In the control, sound coarse fuels decreased as rotten coarse fuels increased, suggesting an overall trend towards higher decayed fuel loading. While fire consumption will vary based on fuel moisture and weather, the control may be ripe for an intense fire given the increase in rotten, flammable fuel loading.

Forest understory

Mechanical thinning shifted the forest understory over 17 years more noticeably than prescribed fire over 15 years, which may be driven by an increase in light and resources resulting from removal of many mid-story trees. Seedling density increased markedly in thinned units, with and without burning, over the course of the study. The strength of this trend is notable across a range of soil depths and applied to both dominant tree species, ponderosa pine and Douglas-fir. Sapling density also increased in the thin-only treatment. By the final measurement year, mean cover by graminoids and shrubs was similar for all active treatments.

Neither cheatgrass nor ventenata increased significantly directly following treatment or over the course of the study. Overall, invasive cover was low, averaging less than 1% across all treatments in 2015. Although the study area was located in proximity to roads and the entire area has been leased for grazing, spatial isolation from large population centers may provide some protection from invasion (McKinney 2002). Another possibility is high endemic site resistance, through intact native vegetation and environmental conditions that are not favorable to widespread invasive establishment. However, continued monitoring is beneficial, as annual grasses respond to yearly changes in temperature and precipitation (Pitt and Heady 1978).

Longevity of fuel reduction treatments

Light thinning from below and low intensity prescribed fire in the Blue Mountains did not have a lasting impact on total fuel loading and understory vegetation composition, nor did they promote two invasive grasses of concern. The increase in tree regeneration evident 17 years post-thinning may ultimately contribute to crown fire risk depending on conditions during any subsequent fire. Future measurement and modeling efforts may examine the change in fire hazard quantitatively.

Fire records in in dry ponderosa pine and Douglas-fir forests of the Blue Mountains indicate historical fires were primarily low- to moderate-intensity and high frequency (11-18 years; Johnston 2016). If treatments are designed to restore a resilient forest structure that has a greater probability of surviving wildland fire under high fire danger weather conditions, then these treatments had a slight and fleeting impact. If designed to mimic a past low-severity disturbance regime, the treatments implemented for this study were an appropriate option, bearing in mind the need for repeated re-treatment at frequent intervals. However, single or double-entry light treatments applied to forests in a departed condition does not restore resilient structure and maintain low fire hazard over succeeding decades. Heavier treatments or repeated treatments are necessary for these objectives.

Additional findings

This study did not conduct an in-depth analysis of changes in tree composition, structure, or growth resulting from treatment, yet certain trends are evident from the data collected. Thinning reduced both tree basal area (BA) and trees per hectare (TPH), while prescribed fire alone did not have a clear effect on either variable (Figure A-2, Figure A-3). Burning with and without thinning appeared to slightly increase the number of dead trees in 2004, although the lack of change in basal area indicates that fire-killed trees are generally smaller. Interestingly, this effect was not brought to light by the analysis of seedlings and saplings, perhaps because the variability between units was too great to detect it. In the final measurement, 17 years post-thin and 15 years post-burn, BA climbed to approximately pre-treatment values in the thinned units, both with and without burning. TPH in the combination thin and burn treatment units remained low, while noticeably increasing in the thin-only treatment in 2015. An increase in BA without an increase in TPH indicates that the combination of thinning and burning may promote size growth in existing trees. The no-treatment control experienced a massive increase in TPH between 2004 and 2015, mainly in ponderosa and lodgepole pine. The magnitude of the increase in tree density, a near doubling of previous TPH over 12 years, calls into question the number of plots measured in 2015. Several of the control units contained plots dominated by extremely dense clumps of understory trees, which may have undue influence on the treatment average.

To look for a treatment-induced change in tree growth, tree core data were collected from half of the plots measured in 2015. We attempted to core a dominant or co-dominant ponderosa pine and Douglas-fir at each selected plot. Cored trees were located outside the measurement plot at a randomly selected azimuth, but well within the treatment unit boundary. Not every plot contained both tree species in the vicinity, so sample sizes are not balanced. We measured cores in the field with a hand lens and calipers, recording the length in mm of the last 15 years of growth, last 17 years of growth, and the segment of growth 20-35 years prior to 2015 were recorded. There was a standard thinning response, with the greatest growth increase for Douglas-fir. Trees in the no-treatment control showed the least growth compared to all other treatments.

Recommendations for future management

Fuel reduction treatments have a finite lifespan. The treatments implemented in this study had a light touch and therefore a negligible or short-lived effect on woody fuels and understory vegetation; it is likely that heavier thinning or hotter prescribed fire may have produced larger and more enduring change. However, more aggressive treatments require careful implementation and monitoring to avoid an unwanted response by invasive plant species. If past disturbance frequency and severity are used as a guide for fuel reduction treatments, then the dry forests of the Blue Mountains will require regular action to maintain desired stand characteristics. A more restorative treatment will be needed prior to wildland fire use under all but the most modest fire weather conditions. To maintain forest heterogeneity and resilience, we recommend treatments of varying intensities applied every 5-25 years. Factors that influence historic fire regime, such as elevation, aspect, and topography, should be used to guide treatment application.

In summary, an understanding of the changes brought about by common fuel reduction treatments is imperative to craft forest management strategies that meets ecosystem and societal objectives. Monitoring treated areas, and subsequent adaptive management, is the only method of tracking these changes and refining management actions in response to actual effects, in addition to climatic and societal changes. The current shortcomings in monitoring programs' funding and staffing, along with data analysis and feedback are a primary factor in our inability to manage forests with respect to maximizing environmental and economic benefits now and in the future.

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APPENDIX A: FIGURES

Figure A-1: Photographs depicting the range of conditions found within each treatment at the Blue Mountains FFS site. Photographs taken by Kat Morici, 2015.

Figure A-2: Tree basal area (BA, m^2/ha) by status and species for each treatment and measurement year at the Blue Mountains FFS site.

Figure A-3: Trees per hectare (TPH) by status and species for each treatment and measurement year at the Blue Mountains FFS site.

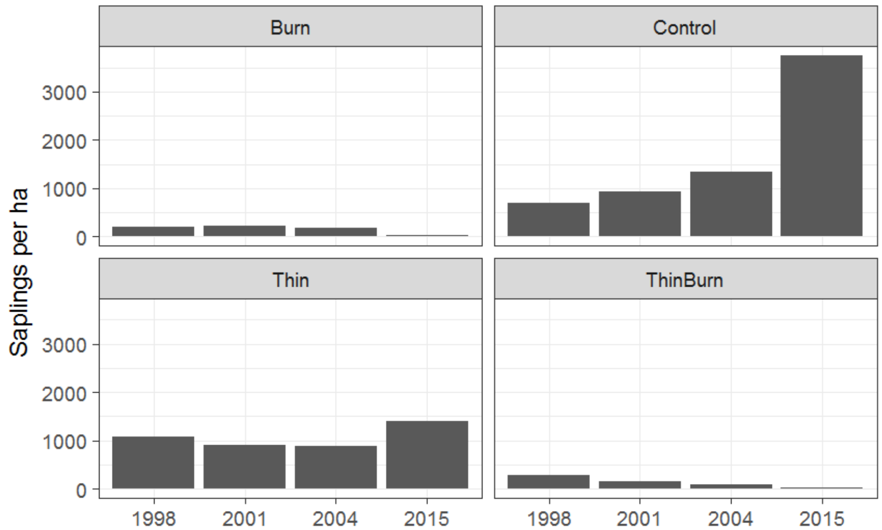

Figure A-4: Saplings (DBH <7.6 cm) per hectare for each treatment and year at the Blue Mountains FFS site.

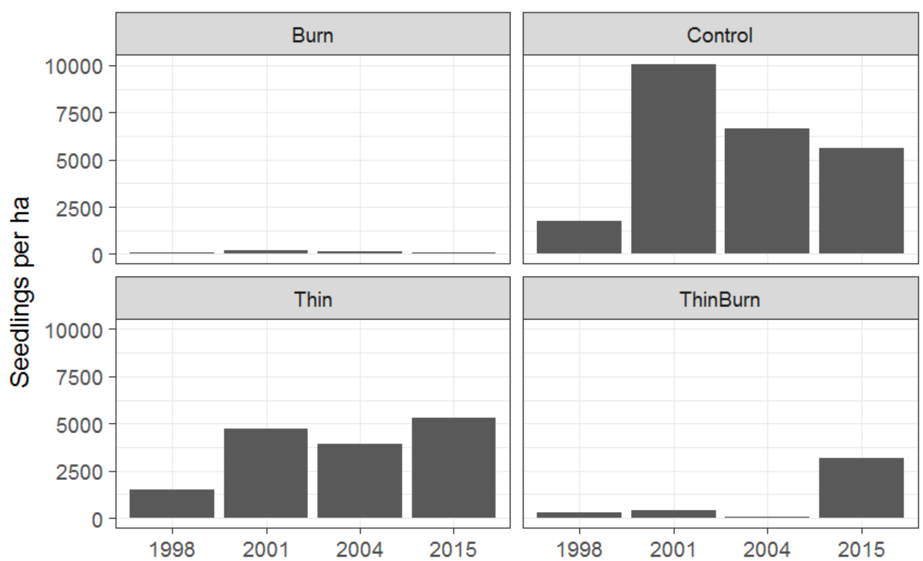

Figure A-5: Seedlings (height <1.37 m) per hectare for each treatment and year at the Blue Mountains FFS site.

APPENDIX B: TABLES

Table A-1: Results of model selection for each fuel variable. Bold names indicate low BIC value and acceptable residual plots. Cshet uses compound symmetry correlation structure, genhet uses general correlation structure, arhet uses autoregressive lag 1 correlation. All models allow for variance heterogeneity among years.

All fine		
Model	DF	BIC
cshet	19	184.01
genhet	21	185.24
arhet	19	**180.51**
1-hr		
cshet	19	3.40
genhet	21	6.83
arhet	19	**0.96**
10-hr		
cshet	19	119.65
genhet	21	124.70
arhet	19	**118.01**
100-hr		
cshet	19	163.95
genhet	21	169.34
arhet	19	**162.51**

All 1000-hr		
Model	DF	BIC
cshet	19	286.18
genhet	21	291.08
arhet	19	**284.07**
Sound 1000-hr		
cshet	19	**254.92**
genhet	21	260.64
arhet	19	257.27
Rotten 1000-hr		
cshet	19	**259.90**
genhet	21	266.29
arhet	19	259.62
All woody		
cshet	19	300.01
genhet	21	305.00
arhet	19	**297.95**

Table B-2: Results of overall F-tests for model fixed effects for each fuel variable, bold indicates significance (p-value ≤ 0.05).

All fine			
Predictor	DF	F-value	p-value
treatment	**3,11**	**9.14**	**0.003**
year	2,24	3.33	0.053
VPD	1,11	0.37	0.558
treat:year	**6,24**	**3.53**	**0.012**
1-hr			
treatment	**3,11**	**4.26**	**0.032**
year	**2,24**	**4.12**	**0.029**
VPD	1,11	0.05	0.820
treat:year	6,24	2.06	0.096
10-hr			
treatment	**3,11**	**5.35**	**0.016**
year	2,24	3.08	0.065
VPD	1,11	1.31	0.277
treat:year	**6,24**	**4.34**	**0.004**
100-hr			
treatment	**3,11**	**10.79**	**0.001**
year	**2,24**	**3.42**	**0.049**
VPD	1,11	0.04	0.851
treat:year	6,24	2.44	0.055

All 1000-hr			
Predictor	DF	F-value	p-value
treatment	**3,11**	**4.02**	**0.037**
year	**2,24**	**8.24**	**0.002**
VPD	1,11	0.59	0.459
treat:year	6,24	0.76	0.611
Sound 1000-hr			
treatment	3,11	2.19	0.146
year	2,24	1.72	0.200
VPD	1,11	2.09	0.176
treat:year	6,24	1.71	0.161
Rotten 1000-hr			
treatment	**3,11**	**9.21**	**0.003**
year	**2,24**	**11.12**	**<.001**
VPD	1,11	0.05	0.835
treat:year	6,24	2.39	0.059
All woody			
treatment	**3,11**	**5.49**	**0.015**
year	**2,24**	**6.51**	**0.006**
VPD	1,11	0.54	0.478
treat:year	6,24	1.17	0.355

Table B-3: Estimated differences in mean average fuel loading between initial and post-treatment measurements for each treatment. Bold indicates statistical significance (p-values ≤ 0.006).

Treatment	**Comparison**	**Estimate**	**t_{24} value**	**p-value**	**99.4% Confidence Interval**
All fine					
Control	2004 v 1998	-2.79	-3.24	0.003	(-5.36, -0.21)
	2015 v 1998	-1.82	-1.70	0.101	(-5.02, 1.38)
Burn	2004 v 1998	-0.99	-9.50	**<0.001**	(-1.3, -0.68)
	2015 v 1998	-0.58	-4.50	**<0.001**	(-0.97, -0.19)
Thin	2004 v 1998	1.72	1.66	0.110	(-1.39, 4.83)
	2015 v 1998	-0.85	-0.66	0.516	(-4.72, 3.02)
ThinBurn	2004 v 1998	-0.60	-1.08	0.292	(-2.26, 1.06)
	2015 v 1998	-1.40	-2.03	0.053	(-3.47, 0.66)
1-hr					
Control	2004 v 1998	-0.21	-2.16	0.041	(-0.5, 0.08)
	2015 v 1998	0.02	0.18	0.857	(-0.32, 0.37)
Burn	2004 v 1998	-0.03	-2.01	0.056	(-0.07, 0.01)
	2015 v 1998	-0.04	-2.47	0.021	(-0.09, 0.01)
Thin	2004 v 1998	-0.05	-1.16	0.258	(-0.17, 0.08)
	2015 v 1998	-0.11	-2.29	0.031	(-0.26, 0.03)
ThinBurn	2004 v 1998	-0.04	-1.00	0.326	(-0.17, 0.09)
	2015 v 1998	-0.12	-2.33	0.028	(-0.28, 0.03)
10-hr					
Control	2004 v 1998	-1.29	-3.61	**0.001**	(-2.35, -0.22)
	2015 v 1998	-0.65	-1.52	0.141	(-1.94, 0.63)
Burn	2004 v 1998	-0.14	-1.94	0.064	(-0.36, 0.08)
	2015 v 1998	0.20	2.29	0.031	(-0.06, 0.47)
Thin	2004 v 1998	0.32	0.99	0.331	(-0.65, 1.3)
	2015 v 1998	-0.41	-1.04	0.311	(-1.59, 0.77)
ThinBurn	2004 v 1998	-0.19	-0.86	0.400	(-0.84, 0.46)
	2015 v 1998	-0.37	-1.41	0.172	(-1.15, 0.41)
100-hr					
Control	2004 v 1998	-1.29	-2.30	0.030	(-2.97, 0.39)
	2015 v 1998	-1.19	-1.74	0.094	(-3.23, 0.85)
Burn	2004 v 1998	-0.82	-6.34	**<0.001**	(-1.21, -0.43)
	2015 v 1998	-0.74	-4.75	**<0.001**	(-1.21, -0.28)
Thin	2004 v 1998	1.44	1.91	0.068	(-0.82, 3.71)
	2015 v 1998	-0.33	-0.36	0.721	(-3.08, 2.42)
ThinBurn	2004 v 1998	-0.37	-0.89	0.380	(-1.6, 0.86)
	2015 v 1998	-0.91	-1.83	0.080	(-2.41, 0.58)

Table B-3 (Continued)

Treatment	Comparison	Estimate	t_{24} value	p-value	99.4% Confidence Interval
1000-hr					
Control	2004 v 1998	-1.88	-0.50	0.623	(-13.2, 9.44)
	2015 v 1998	2.79	0.59	0.558	(-11.3, 16.87)
Burn	2004 v 1998	-4.99	-2.78	0.010	(-10.36, 0.39)
	2015 v 1998	-0.71	-0.32	0.754	(-7.4, 5.98)
Thin	2004 v 1998	-1.44	-0.99	0.333	(-5.81, 2.93)
	2015 v 1998	2.05	1.13	0.271	(-3.39, 7.48)
ThinBurn	2004 v 1998	-6.16	-2.89	0.008	(-12.54, 0.22)
	2015 v 1998	-1.96	-0.74	0.466	(-9.9, 5.98)
Sound 1000-hr					
Control	2004 v 1998	-1.51	-0.65	0.522	(-8.5, 5.47)
	2015 v 1998	-6.23	-2.67	0.013	(-13.22, 0.75)
Burn	2004 v 1998	-0.38	-0.50	0.624	(-2.66, 1.9)
	2015 v 1998	0.22	0.29	0.772	(-2.06, 2.51)
Thin	2004 v 1998	0.83	0.65	0.523	(-3.02, 4.68)
	2015 v 1998	-0.71	-0.56	0.583	(-4.57, 3.14)
ThinBurn	2004 v 1998	-0.85	-0.48	0.639	(-6.25, 4.54)
	2015 v 1998	0.72	0.40	0.691	(-4.67, 6.11)
Rotten 1000-hr					
Control	2004 v 1998	-0.37	-0.11	0.912	(-10.23, 9.5)
	2015 v 1998	9.02	2.74	0.011	(-0.84, 18.89)
Burn	2004 v 1998	-4.61	-2.16	0.041	(-11.02, 1.8)
	2015 v 1998	-0.93	-0.44	0.667	(-7.34, 5.47)
Thin	2004 v 1998	-2.27	-1.68	0.107	(-6.34, 1.79)
	2015 v 1998	2.76	2.03	0.053	(-1.31, 6.83)
ThinBurn	2004 v 1998	-5.31	-3.46	0.002	(-9.9, -0.71)
	2015 v 1998	-2.69	-1.75	0.092	(-7.28, 1.91)
All woody					
Control	2004 v 1998	-4.67	-1.13	0.270	(-17.05, 7.72)
	2015 v 1998	1.00	0.19	0.849	(-14.53, 16.52)
Burn	2004 v 1998	-5.98	-3.26	0.003	(-11.48, -0.48)
	2015 v 1998	-1.29	-0.56	0.580	(-8.18, 5.6)
Thin	2004 v 1998	0.28	0.12	0.902	(-6.41, 6.96)
	2015 v 1998	1.20	0.43	0.671	(-7.18, 9.58)
ThinBurn	2004 v 1998	-6.76	-2.60	0.016	(-14.56, 1.04)
	2015 v 1998	-3.36	-1.03	0.313	(-13.14, 6.42)

Table B-4: Results of linear mixed model selection for cheatgrass (BRTE) and ventenata (VEDU). The selected model is bolded. The correlation structure is described by the model name. All models allow for variance heterogeneity among years.

Species	Model	DF	BIC
BRTE	**compound symmetry**	23	200.14
	general	28	215.84
	autoregressive lag 1	23	200.40
VEDU	compound symmetry	19	87.91
	general	21	87.18
	autoregressive lag 1	19	85.90

Table B-5: Results of overall F-tests for cheatgrass (BRTE) and ventenata (VEDU) model fixed effects, bold indicates significance (p-value ≤ 0.05).

Species	Predictor	DF	F-value	p-value
BRTE	treatment	3,12	2.69	0.093
	year	3,35	5.15	**0.005**
	VPD	1,35	0.21	0.649
	treat:year	9,35	2.65	**0.019**
VEDU	treatment	3,12	0.84	0.496
	year	2,23	0.10	0.908
	VPD	1,23	0.03	0.863
	treat:year	6,23	0.86	0.536

Table B-6: Cheatgrass (BRTE) and ventenata (VEDU) mean percent cover and standard deviation for each treatment-year combination at the Blue Mountains FFS site.

Species	Treatment	Year	Mean	SD
BRTE	Control	1998	0.01	0.02
		2001	0.23	0.36
		2004	0.13	0.15
		2015	0.04	0.04
	Burn	1998	0.51	0.54
		2001	0.24	0.11
		2004	1.42	0.71
		2015	0.16	0.09
	Thin	1998	0.68	1.34
		2001	0.66	1.17
		2004	0.98	1.43
		2015	0.12	0.15
	ThinBurn	1998	0.24	0.42
		2001	0.33	0.13
		2004	1.93	2.69
		2015	0.17	0.13
VEDU	Control	2001	0	0
		2004	0	0
		2015	0.05	0.1
	Burn	2001	0.68	1.31
		2004	0.36	0.56
		2015	0.4	0.59
	Thin	2001	0.01	0.01
		2004	0	0.01
		2015	0.11	0.19
	ThinBurn	2001	0.03	0.05
		2004	0.32	0.47
		2015	0.85	0.95