THESIS

IMPACTS OF TREATMENTS ON FOREST STRUCTURE AND FIRE BEHAVIOR IN DRY WESTERN FORESTS

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ABSTRACT

IMPACTS OF TREATMENTS ON FOREST STRUCTURE AND FIRE BEHAVIOR IN DRY FORESTS

Forest managers are increasingly using mechanical treatments in dry forests of the western US in order to produce stands with spatially complex structure while also reducing crown fire potential. However, there has been a lack of evaluation of these treatments on spatial patterns in dry forest types of the western US. In addition, the implications of heterogeneous fuels complexes on fire behavior are not well understood due to a lack of experimental data and the use of semi empirical models which cannot account for the structural complexity of fuel beds. The lack of well quantified studies on changes in spatial heterogeneity and limitations on quantifying the associated fire behavior suggest there are gaps in our knowledge regarding the implications of mechanical fuels treatments.

The primary emphasis of this thesis is in Chapter 1. I comprehensively stem-mapped seven 4 ha plots, after treatment in dry, coniferous treated stands across the Southern Rockies and Colorado Plateau. Then, I estimated pre-treatment structure by constructing linear allometric regressions of tree characteristics and applying these to mapped stumps thus producing stem-maps before treatment. To investigate how these treatments altered structural complexity, I used spatial statistical analyses to assess spatial relationships of trees, before and after treatment, occurring at stand and within-stand scales as well as horizontal and vertical dimensions. Then, I assessed the cumulative effects of the reduction and spatial alterations of structure on potential fire behavior, measured by rate of spread, fireline intensity and percent

of canopy consumed, across a range of open wind speeds using the Wildland urban-interface Dynamics Simulator (WFDS). WFDS is a physics-based model capable of representing the 3-D complexities of the fuels complex and captures fuel-atmosphere-fire dynamics through space and time. Results from this chapter suggest (1) treatments impact facets of structural complexity in varying ways, though avoided large-scale homogenization of forest structure, (2) within canopy wind speeds increase following treatments and, (3) fire behavior can be altered in two distinct manners following treatments. In most cases, the alterations in the fuels complex coupled with greater within canopy wind speeds resulted in an overall decrease in potential fire behavior and crown fire activity, especially at high open wind velocities. However, in two cases I examined there were increases in fire behavior following mechanical treatments. In these cases the increases were primarily associated with increased surface fire behavior. The results from this chapter suggest that these mechanical treatments may not always enhance, but can promote, a degree of structural complexity, and that mechanical treatments are effective if implemented strategically.

Chapter 2 reports on litter bulk density values for use by managers to improve fuel loading assessments. Litter bulk density as a factor, in conjunction with litter depth, is used to estimate litter load necessary for fuel hazard assessments as litter is a primary carrier of fire and its load impacts potential rate of spread, fireline intensity and smoke production. In addition litter load estimation is needed for estimating carbon and wildlife habitat availability. However, available litter bulk density factors are limited by region, forest type, and site history. This chapter uses data collected on litter bulk density in both ponderosa (*Pinus ponderosa* Lawson) dominated and dry mixed conifer stands throughout the southern Rockies across sites

that have been recently mechanically treated and in recently undisturbed sites. Results show litter bulk density was much lower in ponderosa pine forest than mixed conifer, and the impact of treatment was relatively negligible. These results provide managers in the southern Rockies with a regionalized value that may improve accuracy for estimation of litter load.

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1. A SPATIALLY DRIVEN EVALUATION OF STRUCTURE AND FIRE BEHAVIOR FOLLOWING HETEROGENEOUS TREATMENTS IN DRY FORESTS

1.1 INTRODUCTION

Land management practices following Euro-American settlement such as livestock grazing, road building, logging, fire suppression, and timber-oriented management have contributed to an altered forest structure particularly in dry western forests that historically had frequent low severity fire regimes (Veblen et al., 2000; Fulé et al., 2009; Naficy et al., 2010 Franklin et al., 2013). As a result, modern forests are often regarded as exhibiting a build-up of fuels, leading to greater potential for extensive crown fires and tree mortality and lower long term forest resiliency (Savage and Mast, 2005). This concern has prompted an increased emphasis on the use of mechanical treatments to mitigate crown fire potential (Agee and Skinner, 2005).

Mechanical treatments also seek to reduce fuels in a manner that results in greater structural heterogeneity (North et al., 2009). It is believed that treatments which promote a heterogeneous arrangement of structure are more likely to produce ecological functions characteristic of dry forests than homogeneous forests given that stand dynamics tended towards heterogeneity before settlement (Larson and Churchill, 2012). For example, mechanical treatments are utilized to create greater diversity of vertical canopy structure for avian species (McElhinney et al., 2005), such restoring northern goshawk (*Accipiter gentilis*) habitat (Youtz et al., 2008).

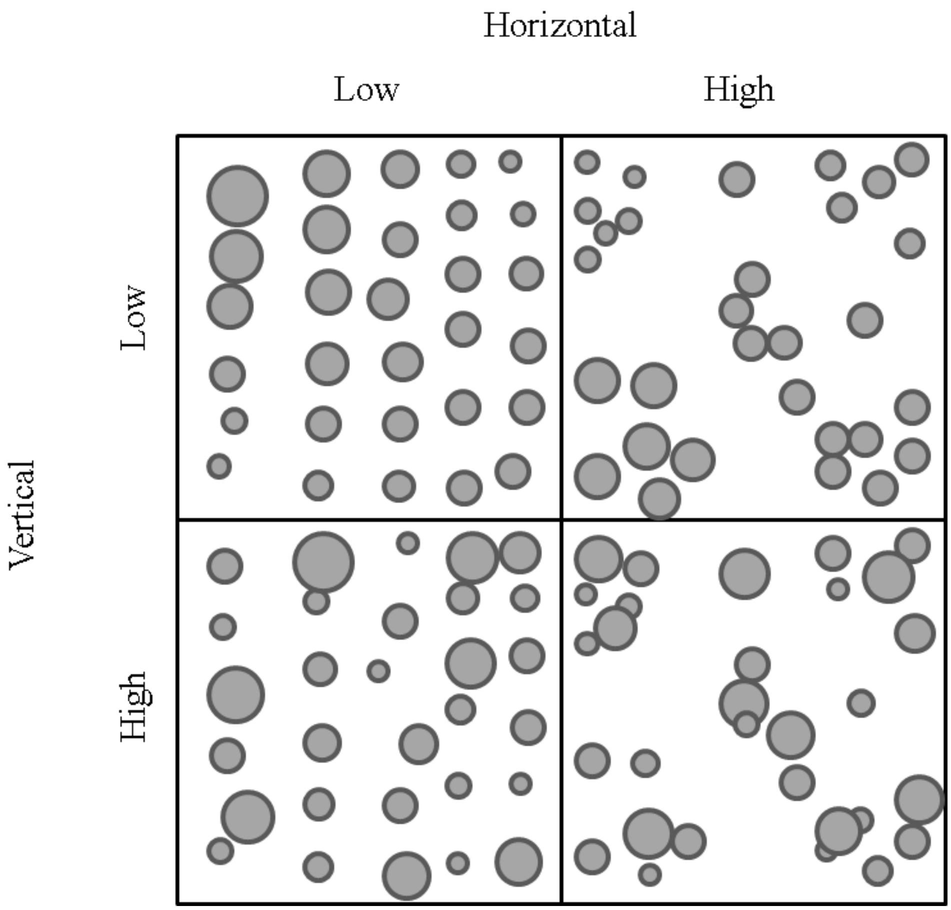

Figure 1.1. An overhead view of 30 trees rearranged in various levels of vertical and horizontal structural complexity. Complexity can be further described at the stand scale (e.g. describing tree patterns) and patch scale (e.g. enumerating patches, openings, and individual trees). Circle areas represent relative tree heights.

The description of heterogeneity for a given stand can be thought of as a spatial problem that consists of multiple dimensions and scales (Figure 1.1). At stand scales, structure can be described horizontally, such as classifying the general spatial pattern exhibited by trees (e.g. Harrod et al., 1999) and vertically, by measuring the degree to which differently sized trees spatially intermingle (e.g. Franklin and Van Pelt, 2004). At finer spatial scales, dry forest stands are also spatially decomposed into a horizontal mosaic of tree patches, individual trees, and openings (Larson and Churchill, 2012). The vertical dimension of complexity at this scale describes the variability in tree heights within patches (Cooper, 1960; White, 1985; Mast and Veblen, 1999).

Despite the increased emphasis on altering structural complexity using mechanical treatments, there are relatively few studies that document changes following treatments (Larson and Churchill, 2012). While it has been shown that treatments which utilize spatially-explicit reference conditions to guide silvicultural implementation can meet structural complexity objectives (Harrod et al., 1999; Churchill et al., 2013a), a paucity of available reference conditions coupled with biophysical disparities between a reference stand and the stand in question limit the potential use of reference conditions in many areas. This often leaves managers to rely on a non-spatial view of forest structure reinforced by current fuels management planning tools to guide implementation (Larson and Churchill, 2012; Churchill, 2013). It is unknown if a non-spatial approach can promote a heterogeneous forest structure with high complexity (Larson and Churchill, 2012).

The creation of structural complexity challenges examination of the primary goal of these mechanical treatments, reduction of undesired, potentially hazardous fire behavior. Past work has suggested that mechanical treatments could reduce fire hazard following a combination of decreasing surface fuel load, canopy bulk density and increasing canopy base height (Agee and Skinner, 2005). For example, several studies have assessed the impact of mechanical treatments on potential fire behavior linking empirical and semi empirical point-functional modelling systems to evaluate fuels reduction and restoration treatment effects (Reinhardt et al., 2008; Fulé et al., 2012). Specifically, these studies have utilized linkages between the surface and crown fire spread models developed by Rothermel (1972, 1991) and the crown fire initiation and spread models developed by Van Wagner (1977). Modeling approaches based upon these linkages are not able to account for the spatial distributions of fuels, as well as fire behavior dynamics arising from fuel-fire-atmosphere interactions; this assumption of homogeneity has been attributed to poor model performance in heterogeneous fuels complexes (Van Wilgen et al., 1985).

Recently developed physics-based fire models, such as Wildland-urban interface Fire Dynamics Simulator (WFDS) (Mell et al., 2007, Mell et al., 2009), and HIGRAD/FIRETEC (Linn, 1997), use a three dimensional numerical grid to represent the fuels complex and simulate the major physical processes that influence fire behavior. Physics-based models have recently been used to explore how wind and fire behavior behave in heterogeneous fuels arrangements with varying degrees of structural complexity. Hoffman et al. (2012) used WFDS to investigate the interaction between several levels of bark beetle-caused tree mortality and the spatial pattern of overstory trees on fire behavior. They found that the modeled fire behavior was dependent the stand scale spatial pattern, with highly aggregated patterns exhibiting greater fire behavior compared to random or homogeneous spatial patterns, when the canopy fuel load and vertical structural complexity was constant. Pimont et al. (2011) used FIRETEC to investigate the effect of both canopy fuel load and tree aggregation fire behavior. Their study showed that the level of canopy fuel load had the greatest effect on potential fire behavior, but that the degree of tree aggregation influenced how variable winds were within the canopy. The findings of these studies suggest that alterations in the fuel loading and the spatial pattern of the fuels complex influence the potential fire behavior. Unfortunately these studies depend on hypothetical changes which may not be representative of actual mechanical treatment operations.

The overall goal of this study was to use a combination of field inventories and physics based fire behavior modeling to examine changes in forest structure and to investigate the associated changes in fire behavior arising from mechanical treatments in dry forest types of the eastern Colorado Plateau and southern Rocky Mountains. To meet this overall goal, I investigated two questions. First, how are the sampled treatments altering structural complexity? Second, given the changes in structural complexity what impacts do these treatments have on potential fire behavior? To answer the first question, I examined structural complexity in both the horizontal and vertical dimensions determined at the stand and patch scale. To address the second objective I utilized WFDS to simulate the alterations in within canopy wind flow and the associated changes in fire behavior. Owing to a current paucity in spatially-explicit forest data and analysis (Larson and Churchill, 2012), this study's results regarding effects on structural complexity provide for discussion on how spatial objectives could be integrated into treatment planning and evaluation. In addition, WFDS simulations suggest possible ramifications of structurally complex mechanical treatments on fire behavior.

1.2 METHODS

1.2.1 Study areas

Seven sites were sampled that had been recently treated with a mechanical thinning 1-3 years prior, across the Colorado Plateau and southern Rocky Mountains (Table 1.1). Sites were selected in consultation with local forest managers based on the criteria that they must have relatively little (< 20%) slope and represent typical dry forest structure within the locale. The majority of sites were dominated by ponderosa pine (*Pinus ponderosa* Lawson) except one site (PC) which was mixed, primarily with ponderosa pine, Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) and spruce (Picea spp.). No site had been recently managed or disturbed, with exception of a low-thinning, fuels reduction treatment ten years prior to the recent treatment.

Table 1.1. Non-spatial stand structure characteristics of sampled sites pre- and post-treatment.

Managing agency	Site name	Thin status	Trees ha^{-1}	QMD (cm)	BA (m^2 ha^{-1})	HT (m)	Species comp.
Boulder County Open Spaces	HB	Pre	418	24.1	19.1	9.8	98% PIPO, 2%JUSC2
		Post	295	25.0	14.5	10.1	98% PIPO, 2%JUSC2
Kaibab NF	LC	Pre	487	25.9	25.6	20.9	100% PIPO
		Post	203	35.2	19.7	25.3	100% PIPO
Pike NF	MG	Pre	685	20.5	22.7	14.9	95% PIPO, 5%PSME
		Post	269	22.4	10.6	16.6	97% PIPO, 3%PSME
Pike NF	PC	Pre	934	18.4	24.8	15.8	20% PIPO, 54% PSME, 7%POTR5, %19 PICEA
		Post	352	18.5	9.5	16.6	32% PIPO, 41% PSME, 17%POTR5, %10 PICEA
Roosevelt NF	DL	Pre	516	18.7	14.1	14.2	95% PIPO, 4%PSME, 1% POTR5
		Post	181	19.0	9	15	97% PIPO, 3%PSME
Uncompahgre NF	UM	Pre	504	27.5	30	22.1	87% PIPO, 6% POTR5, 7%PICEA, 1% QUGA
		Post	314	26.7	17.6	25.5	87% PIPO, 7% POTR5, 6%PICEA, 1% QUGA
Cibola NF	BW	Pre	327	27.4	19.3	14.3	81% PIPO, 8% JUSC2, 11% PIED
		Post	63	38.9	7.5	18.2	88% PIPO, 3% JUSC2, 9% PIED

QMD, quadratic mean diameter at breast height; BA, basal area; HT, 90th%ile tree height; species codes follow NRCS PLANTS Database (USDA NRCS, 2014)

Although exact objectives in silvicultural prescriptions varied slightly among sites, all sampled areas explicitly stated objectives that sought to reduce potential fire behavior and qualitatively described the desired structure in such a way that might increase structural complexity (e.g. creation of a mosaic of tree patches and openings, increase tree aggregation, promote vertical diversity within patches). All sites were mechanically thinned with targets of 7 to 14 m2 ha-1 of residual basal area. Removal or retention of trees was based on utilizing exiting forest structure rather than strictly guided by locations of remnant material (i.e. pre-settlement trees, stumps, and logs), and no reference conditions were used to guide implementation.

1.2.2 Field sampling

In the summers of 2012 and 2013, a single 200 m x 200 m plot was established within each site following mechanical treatments. Plots were aligned with cardinal directions and randomly located within the thinned stand such that significant edaphic features imposing additional heterogeneity (e.g. roads, streams, ridges, unit edges) were avoided.

All trees at least 1.4 m tall in the post-thinned stand were mapped to a Cartesian coordinate system and had the following information recorded: species, tree height (m), diameter at stump height (DSH; cm), diameter at breast height (DBH; cm), crown radius (m), and crown base height (m). Crown radii were measured as the average of two distances, in a random direction from tree bole to crown edge. Crown base height was measured by visually compacting the crown (e.g. mentally “lifting” the lowest branches until a complete whorl accumulated) (USDA Forest Service 2005).

In addition to measuring all the residual trees that were at least 1.4 m tall, all of the stumps resulting from thinning were mapped and recorded their species and DSH. To estimate tree properties before each stump was cut, this study used a mixed-effects, linear regression using measured residual trees as samples. Site, species, and DSH were fixed, independent factors used. An interactive term, site x DSH, accounted for potential site differences in the relationships of DSH to tree measurements. Preliminary results suggested linear regressions fit well and resulted in normal and homoscedastic distributions of residuals. Regressions were then applied to stumps to reconstruct the pre-treatment forest structure.

Surface fuels were sampled within 64 sub-plots spaced regularly every 25 m in each treated site as well as in an adjacent untreated site. Untreated sites were chosen in consultation with local forest managers with requirements that such sites had similar biophysical conditions as treated sites and species composition and structure resembling that of treated sites prior to their own treatment. Stemming from plot centers, two 15 m planar transects were laid out in random directions for woody fuel estimation following Brown (1974); 1-hr time lag (>0.6 cm diameter) and 10-hr time lag (0.6-2.5 cm diameter) fuels were tallied for 2 m, and 100-hr and 1000-hr time lag fuels, 2.5cm to 7.6cm and over 7.6 cm diameter classes respectively, were measured for 5 m and 15 m of each transect, respectively. In each sub-plot, litter load and depth were measured in a randomly placed 0.10 m^2 quadrat following methods by Ottmar and Andreu (2007). Herbaceous and shrubby samples were clipped in randomly places 1 m2 quadrats within each subplot and weighed ex situ for 3 days at 80°C for at least 48 hours to achieve equilibrium dry weight.

1.2.3 Analysis of structural complexity

A specific method was used to examine the effects of mechanical treatments on structural complexity on each combination of dimension, horizontal and vertical, and scale, stand or patch (Table 1.2). Each method is explained in more detail below.

Treatment effects on horizontal complexity at the stand scale were identified using univariate and bivariate forms of the pair correlation function, g(*r*). The mechanics of the function and numerical implementation are provided by Wiegand and Moloney (2004). Conceptually, from each tree location, this function sums the number of trees within an annulus of a fixed width that expands with each step of *r*, the distance from the i^{th} tree. Averaging these counts across all trees before and after treatment, the empirical statistics $g_{pre}(r)$ and $g_{post}(r)$, respectively, are produced. The univariate form, which produces $g_{pre}(r)$ and $g_{post}(r)$, was used to separately determine spatial patterns of trees both before and after treatment. To accomplish this, the locations of trees were independently and identically redistributed using a homogeneous point process multiple times to determine the null range of g(*r*) statistics representing complete spatial randomness (CSR). By testing each observed g(*r*) statistics against the null, at each step of *r*, the pattern was determined as random (not statistically different from CSR), uniform (significantly below the null range), or aggregated (significantly above the null range). In an aggregated pattern, the distribution of trees is such that a greater proportion of trees are within the annulus than under CSR; conversely, fewer trees than CSR at *r* represent uniformity.

Table 1.2. Methodological framework for assessing alterations in structural complexity across dimensions and scales following mechanical thinning.

Scale	Method	What was examined
Horizontal dimension		
Stand	Univariate and bivariate point correlation functions	Thinning impacts on the spatial pattern of trees and whether the pattern is more or less aggregated following thinning
Patch	Spatial patch detection	Changes in aerial cover of individual trees relative to patches and changes in patch size distributions, following thinning
Vertical dimension		
Stand	Height differentiation index	Thinning impacts on spatial mingling of differently sized trees
Patch	Patch scale coefficient of variation of tree heights	Changes in the variability of tree heights among patches following thinning

The bivariate form, $g_{post}(r)$ - $g_{pre}(r)$, was used to compare the difference in degree of aggregation from before treatment to after treatment. For this function, the chosen null was of no difference, $g_{post}(r) = g_{pre}(r)$. For each site, the pre-treatment locations of trees are fixed, while the labels of trees that were either removed or retained are permuted several times, simulating a random treatment with no difference in aggregation (Wiegand and Moloney, 2004). Whether the observed $g_{post}(r)$ - $g_{pre}(r)$ was similar to the null range or departed significantly was tested. A significant departure below, or above, the expected range, would signify the post-treatment pattern was less, or more, aggregated, respectively.

The bivariate form of the pair correlation function compliments the univariate form. For example, one might find that before and after treatment, the observed spatial pattern was uniform using the univariate form. Yet, results using the bivariate form suggest the post-treatment pattern was significantly more aggregated than before treatment. This would indicate that, though treatment significantly altered the degree of aggregation of trees, the magnitude was too low to alter the overall pattern.

Formal inference to detect departure from the null was accomplished using a goodness-of-fit test (Diggle, 2003). For each test, the range of null statistics was produced from 999 simulations. For these functions, functions were implemented across a range of *r*, from 1 m to 10 m. The depth of each annulus was chosen as 1 m. These analyses were performed in Programita (Wiegand and Moloney, 2004), incorporating Ripley's (1988) method of edge correction. Inference was gained using $\alpha = 0.05$ for the goodness-of-fit test, as well as all subsequent formal tests in this study.

Mechanical treatment impacts on arrangements of patch scale patterns was evaluated using the Plotkin et al. (2002) cluster analysis in a manner similar to Sanchez-Meador et al. (2009). This analysis was used to identify whether treatments were enhancing a mosaic of patches and openings, in lieu of predominance of continuous cover or individual trees. First, each tree's crown radius was projected to crown area assuming a circular geometry. Then, each tree was joined into a unique patch if its crown area overlapped with at least one other tree's crown area but need not have overlapped with every other crown in a unique patch. Identification of unique patches was used to examine how treatments were altering the proportion of area covered each by patches, individual trees, and openings. Patches were also classed by size, as determined by number of trees per patch, similar to Churchill et al. (2013a). Patch size classes were divided into small (2-4 trees), medium (5-10 trees), large patches (11-20 trees), and patches of continuous cover (21+ trees). The largest size class was distinguished separately as it is assumed forest dynamics in dry forests uncommonly produce patches of this size (Sanchez-Meador et al., 2009; Churchill et al., 2013a). Following Plotkin et al. (2002), no edge correction was applied. Details on implementation of the cluster analysis are available by Churchill et al. (2013b).

Changes in vertical complexity at both stand and patch scales were also examined. At the stand scale, mechanical treatment impacts were evaluated with the height differentiation index (TH) (Kint et al., 2000). This index accounts for the juxtaposition of differently sized trees by comparing heights of each tree's height (HT_i) to its nearest three neighbors (HT_j), and is calculated as:

$$TH_i = \frac{1}{3}\sum_{j=1}^{3}\left[1 - \frac{min(HT_i, HT_j}{max(HT_i, HT_j}\right]$$

Potential values range from no height differentiation, 0, to extreme differentiation, 1. At the patch scale, unique patches were identified from the cluster analysis and I calculated the coefficient of variation of tree heights per each patch, $CV_{HT[i]}$. The impact of treatment on vertical complexity was then evaluated using distributions of calculated TH_i and $CV_{HT[i]}$. As values were distribution non-normally, a Wilcoxon rank sum test was used to evaluate differences in median values following treatment per site using R v2.15.2.

1.2.4 WFDS background

WFDS was developed by the National Institute of Standards and Technology (NIST) and US Forest Service, Pacific Northwest Research Station, and is an extension of NIST's Fire Dynamics Simulator (FDS). WFDS uses computational fluid dynamics techniques to describe the spatial and temporal evolution of fire through a three dimensional numerical grid. WFDS uses a numerical form of the Navier-Stokes equations suitable for low-speed, thermally driven flow suitable for wildland fires, a large-eddy simulation approach to account for turbulence at a sub grid scale and the use of a finite volume method to account for thermal radiation (McGrattan et al., 2010). The fuels complex can be represented in WFDS in two ways: The fuel element model and the boundary fuel model. The fuel element model represents fuel within the same numerical grid as wind flow as a porous medium described by its bulk or mean quantities such as surface area to volume ratio, moisture content, and density within each computational cell. The boundary fuel model was developed for situations in which the surface fuel bed must be resolved at spatial scales that are smaller than the computational grid used for wind flow and simulates the fuel bed properties in its own computational grid. In the boundary fuel model, the surface fuels represent a source of drag and heat and mass flux occurs at the interface between the two computational grids. The combination of modeling approaches used in WFDS for the fuels complex facilitates the user to simulate fires in areas with complex fuel beds, including both vegetative only and a mix of vegetative and urban fuels, and accounts for the radiative and convective heat transfer between the gas phase and the vegetation and the drag of the vegetation on the airflow. Thermal degradation of solid fuels can be simulated using one of two approaches a linear function or an Arrhenius type equation. The linear model for thermal degradation was used in all simulations described here. A detailed description of the physical and mathematical formulations in WFDS can be found in McGrattan et al. (2010), and Mell et al. (2007, 2009). Both FDS and WFDS have undergone various forms of model validation and verification throughout their development including over 1,200 separate FDS simulations covering a wide set of physical aspects including fluid dynamics, heat transfer, flame heights and temperatures and room fire behavior. More detail about the verification and validation of FDS can be found in Mcdermott et al. (2013) and McGrattan et al. (2013). Specific evaluation studies of WFDS for applications in vegetative fuels is limited to a handful of studies including fire spread through Australian grassland fuels (Mell et al., 2007), surface fire spread for laboratory scale fires (Bova et al., manuscript in preparation), fire spread through individual tree crown burning experiments (Mell et al., 2009), and wind flow through forested canopies (Mueller, 2012). Further validation studies of WFDS in vegetative fuels are ongoing but are limited by a lack of well quantified field and lab experiments that cover the broad range of scenarios and quantities of interest across the field of wildland fire science. As discussed by Linn et al. (2013), one of the advantages of modeling systems based on a general representation of physical properties governing fire behavior, is that they can be extended beyond the conditions for which experimental evaluation has occurred. However, it is important to remember that given the uncertainties within the model, the results should be interpreted with caution, as with any model. Simulation tools such as WFDS should however, provide insights into the interactions between the fire, fuels and atmosphere that can drive the development of new hypotheses and experiments, as well as provide insight into data collection methods and analysis of field experiments aimed at verifying model-driven hypotheses.

1.2.5 WFDS simulation details

Simulation domains were set to 1000 m x 400 m x 100 m with a voxel resolution of 1m by 1 m in width and length (Figure 1.2). Vertical voxel resolution changed depending on height in domain spanning from 0.5 m at the bottom of to 2 m at the top of the domain. In each simulation fuels from stem-mapped sites were populated, using the fuel element model, between x= [700,900] and y= [100,300], oriented randomly, with respect to the domain. The remainder of the domain was populated with randomly oriented, replicated stem-mapped fuels. Each tree was defined by its measured x- and y- coordinates, dbh, height, cbh, and crown radius. Individual tree crowns were represented as right circular cones defined according to the latter three metrics.

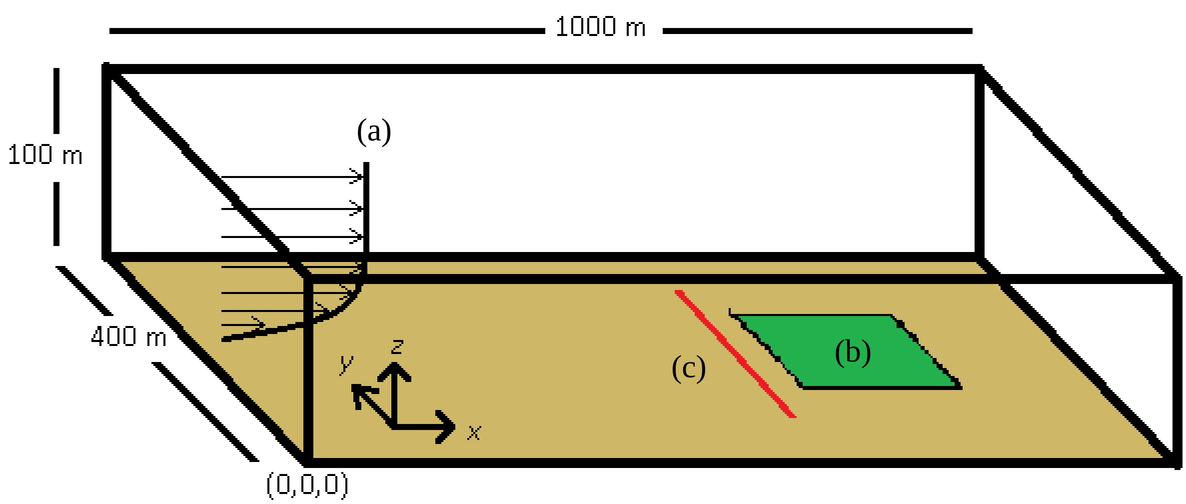

Figure 1.2. Schematic of WFDS simulation domains. Note the (a) inflow wind profiles originating at x=0 m, (b) the site located 700 to 900 m downwind of the inflow, and (c) the fireline origin at x=650 m.

Individual tree crowns and surface fuels were defined by a series of bulk properties. As crowns fuels represent a combination of fine diameter woody twigs and needles, 4000 m^{-1} was used to define surface area/volume of each crown (Brown, 1970; Scott and Burgan, 2005; Mell et al., 2009). The drag coefficient was set to 0.125 (Mueller, 2012), and the crown bulk density was set at 1.2 kg m^{-3}. The crown bulk density is lower than that used by Mell et al. (2009) or Contreras et al. (2012), however, given the total volume occupied by crowns, the canopy biomass would have been much higher than expected in ponderosa pine forest (Cruz et al., 2003). Surface fuels were parameterized with a surface area/volume ratio of 5710 m^{-1}, within the range commonly observed for herbaceous and litter fuels (Brown, 1970). All fuels were also defined with a high heat content of 17,700 (Mell et al., 2009); crown and surface fuel moistures were set at 100% and 5%, respectively, representing typical assigned fuel moistures in a wildland fire scenario (Scott and Reinhardt, 2001)

Surface fuels were populated using the boundary fuel model in WFDS, defined by load and depth for each site, before and after treatment. Load was determined as the sum of the site-averaged load of litter, 1-hr woody fuels, herbaceous and shrub material. Surface fuelbed depth was determined by adding the average litter depth to the average herbaceous and shrub material height. Larger diameter woody fuels do not contribute greatly to fire propagation relative to fine fuels (Anderson, 1969; Mell et al., 2009). Because the focus of this study was the examine effects of altered canopy fuels, surface fuels were distributed homogeneously across each site.

Parameterized wind profiles entered the domain at the inflow boundary (x = 0). Inflow wind speed profiles followed the atmospheric power law function,

$$u/u_r = (z/z_r)^{0.143}$$

, where u is the streamwise wind velocity at height z and u_r is the reference streamwise velocity at reference height z_r (Plate, 1971). In these simulations, u_r, was set at 2.2 m s^{-1}, 4.0 m s^{-1}, 9.0 m s^{-1}, and 13.4 m s^{-1} at, z_r of 20 m; 20 m corresponds to a meteorological standard for open wind. Hereafter, these are referred to as very low, low, moderate, and high wind scenarios. Simulations were run for 2500 simulated seconds. At t = 1300, a fireline 300 m in length was ignited 50 m upwind of site location and allowed to burn downwind through the site where fire behavior sampling occurred.

The outflow boundary was set to an open boundary condition, in effect allowing air and energy to freely exchange across the boundary. Lateral boundaries and the top of the domain were set to a mirror boundary condition, in effect a free-slip, no-flux wall which assumes the conditions outside the domain are identical to conditions on the boundary surface. A typical WFDS input file is provided in section 3.1

1.2.6 WFDS output and analysis

Though WFDS simulations were parameterized with the same inflow winds, the unique structure of each site before and after treatment produces different regimes of wind flow. The functional relationship of streamwise wind velocity for a given height, *U(z)*, is strongly influenced by the canopy bulk density at *z* (Poggi et al., 2004). Since each treatment would simultaneously alter both the wind and fuels within and below the canopy, this study evaluated the wind velocity profiles to aid in interpretation of the fire behavior; specifically, the spatiotemporal mean *U(z)* over the 200 m x 200 m site for a 15 min period.

For each simulation three fire behavior metrics were estimated: rate of spread, fireline intensity, and percent canopy consumed. Rate of spread, the velocity of the fire front (e.g. m s^{-1}), and fireline intensity, a measure of energy output per unit time and unit length (kW m^{-1}) were estimated using 10 belt transects within each simulated site. Transects were oriented in the streamwise direction, spanned the length of each site, were 1 m in width, and spaced 20 m apart each in the cross stream direction. In particular, rate of spread was estimated as the slope parameter derived by regression of the transect-averaged fire front location, specifically at 1 m in height, against time. Fireline intensity was estimated by summing the total heat release on each transect for each second that the fire front was wholly contained within the site, and averaging over time and transects. The third metric of fire behavior, percent of canopy consumed, was calculated as a ratio of the post-fire canopy mass over the initial canopy mass.

To investigate the relative roles of altered fuel loading on potential fire behavior, a series of fixed-effects ANOVA were used of the following form were used:

$$\beta_0 + \beta_1 wind + \beta_2 surfaceload + \beta_3 cbd + \beta_4 cbh$$

, where wind was the inflow, open wind speed, surface load was the average load used in each simulation, and cbd and cbh were the canopy bulk density and canopy base height of each site before and after treatment. Due to the complexities of structural complexity and wind velocity we did not include any direct metrics of structural complexity in this analysis. Rather we used wind velocity which accounts for both the bulk change in wind flow due to reduced fuel loads and the changes in wind flow due to local channeling effects created by various spatial arrangements of the canopy fuels. The ω^2 statistic (Hays, 1994) was used to examine the relative importance of each factor on fire behavior measures in each ANOVA model. Residual diagnostic plots showed the models met assumptions of homoscedasticity and normality of residuals with raw, untransformed data.

1.3 RESULTS

1.3.1 Initial results

Before treatment, stocking among the seven sites varied, ranging from a threefold and twofold difference in tree density and basal area, respectively, from the least to most stocked sites (Table 1.1). Mechanical treatments selectively removed non-ponderosa pine conifers, and varied in intensity, reducing trees per hectare across sites from 29% to 81%, and basal area from 23% to 62%. In addition, canopy base height increased in 5 sites, decreased in one site, MG, and was unchanged in HB (Table1.3).

Although exact objectives in silvicultural prescriptions varied slightly among sites, all sampled areas explicitly stated objectives that sought to reduce potential fire behavior and qualitatively described the desired structure in such a way that might increase structural complexity (e.g. creation of a mosaic of tree patches and openings, increase tree aggregation, promote vertical diversity within patches). All sites were mechanically thinned with targets of 7 to 14 m2 ha-1 of residual basal area.

All treatments increased 1-hr dead and down woody fuel loads among all sites, though total surface fuel load (the cumulative mass of 1-hr dead and down woody fuel, litter, and surface vegetation) increased only in MG (Table 1.3). Among in sites HB, LC, DL, and UM, total surface load decreased, and was unchanged in PC and BW, following treatment. This occurred because litter composed over 90% of surface fuel load by mass, while 1-hr woody fuels and vegetative surface fuels composed less than 10% of mass. The herbaceous and shrubby fuel loads changed inconsistently following treatment though constituted a negligible portion of total surface fuel load.

Table 1.3. Summary of stand-averaged fuels properties, across 7 sites, before and after mechanical treatment. Surface fuel load is the contribution of load from litter, 1-hr time lag fuel, shrubby and herbaceous surface fuels.

Site	Treatment status	Surface fuel load ($kg\ m^{-2}$)	Canopy bulk density ($kg\ m^{-3}$)	Canopy base height (m)
HB	Pre	1.04	0.14	3.6
	Post	0.76	0.10	3.6
LC	Pre	0.36	0.13	3.8
	Post	0.29	0.10	6.6
MG	Pre	0.41	0.14	3.2
	Post	0.62	0.06	2.6
PC	Pre	1.33	0.15	2.2
	Post	1.33	0.06	2.6
DL	Pre	1.16	0.08	2.2
	Post	0.61	0.05	3.0
UM	Pre	1.20	0.15	4.7
	Post	1.06	0.08	5.0
BW	Pre	0.24	0.09	4.6
	Post	0.24	0.03	4.7

Initial inspections of stem maps reveal diverse forest structure and tree arrangement before and after mechanical treatments (Figure 1.3). Most notable is the large elliptical gap in DL before treatment and the large cluster of stumps in the northwestern corner, as well as the linear pattern of removal in HB. Higher resolution stem-maps can be found in Figures 3.1—3.7 (Section 3.2).

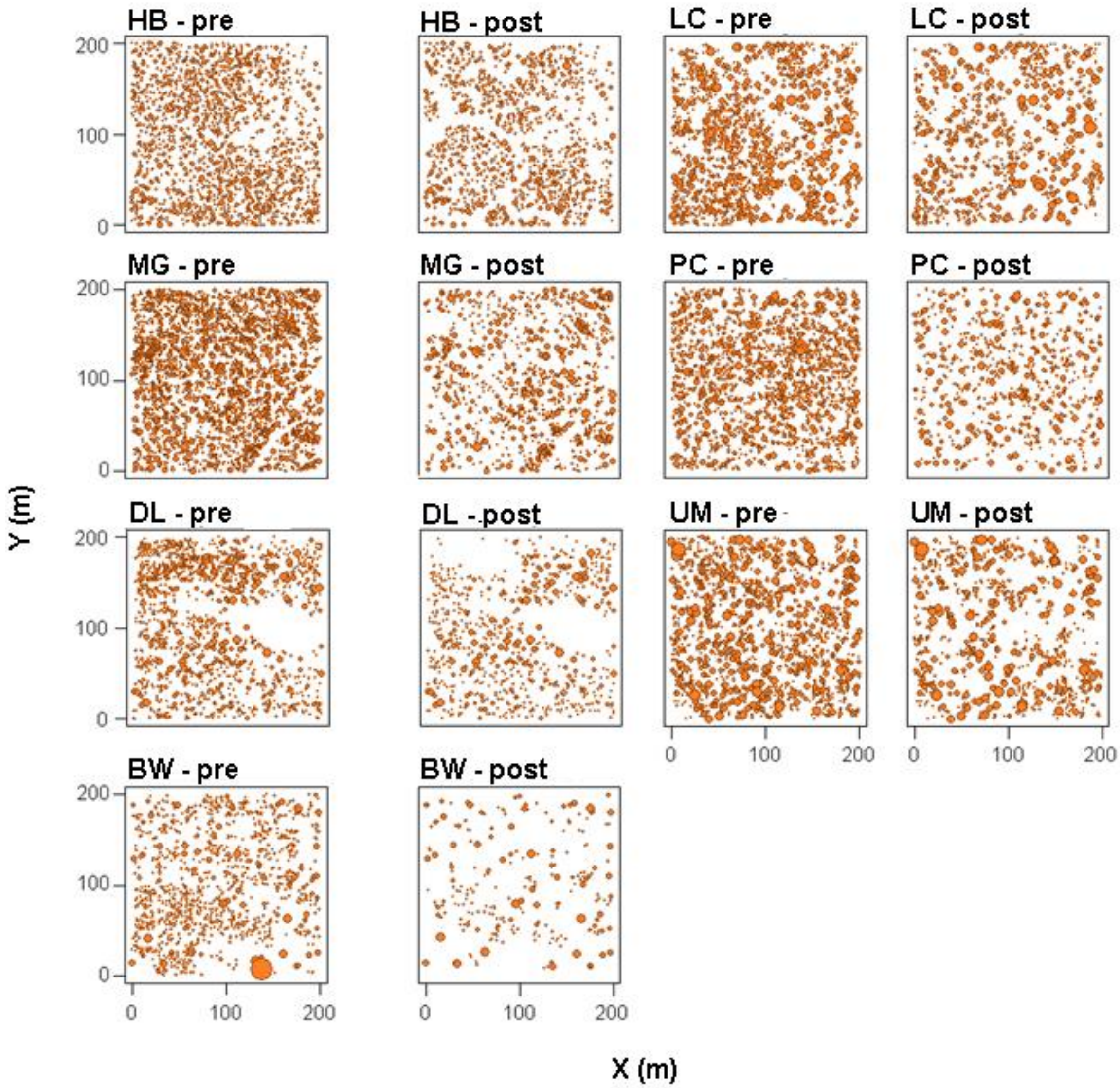

Figure 1.3. Stem-maps from sampled sites, pre- and post-treatment, with points scaled to crown area.

The regressions used to predict DBH, crown width, tree height and crown base height from DSH for estimation of pre-treatment structure generally fit well. In addition, inspection of residuals confirmed linear regressions fit data well. Coefficients of determination were above 0.70 for over half of the 38 linear regression analyses (Table 3.1; Section 3.2). Fits were generally better for predictions of DBH and tree height more so than crown width and crown base height. In addition, measurements on species with larger sample sizes, such as ponderosa pine, quaking aspen (*Populus tremuloides* Michx.), Douglas-fir, and spruce had an improved fit over those with smaller sample sizes, such as Rocky Mountain juniper (*Juniperus scopulorum* Sarg.), Gambel oak (*Quercus gambelii* Nutt.), and two-needle pinyon (*Pinus edulis* Engelm.).

1.3.2 Effect of treatments on structural complexity

The univariate pair correlation function analysis performed both before and after mechanical treatments suggest that stand scale aggregation was present both before and after treatment (Table 1.4), with the exception of site HB, before treatment. The spatial pattern of trees following treatment did not consistently suggest either an increase or decrease in level of aggregation. Sites HB, MG, PC, and DL showed a significant increase in the level of aggregation following treatment (Table 1.4). Of these sites, the increase in level of aggregation did not alter the already aggregated pattern of MG, PC, and DL, though it was associated with the shift from a uniform pattern in HB to an aggregated pattern following treatment. While the level of aggregation did not significantly change in UM, treatments did significantly decrease the level of aggregation in sites LC and UM following treatment. Because the post-treatment patterns were still aggregated, this signifies that the reduction of aggregation was of relatively low magnitude.

Table 1.4. Univariate pair correlation function results describing the spatial tree patterns pre-treatment, $g_{pre}(r)$, post-treatment, $g_{post}(r)$, and the bivariate pair correlation function, $g_{post}(r)$ - $g_{pre}(r)$, results comparing degree of aggregation from post- to pre-treatment.

Site	$g_{pre}(r)$		$g_{post}(r)$		$g_{post}(r)$- $g_{pre}(r)$	
	Pattern	*P*	Pattern	*P*	Difference	*P*
HB	Unif.	0.002	Agg.	0.001	More agg.	0.001
LC	Agg.	0.001	Agg.	0.001	Less agg.	0.001
MG	Agg.	0.001	Agg.	0.001	More agg.	0.003
PC	Agg.	0.001	Agg.	0.001	More agg.	0.001
DL	Agg.	0.001	Agg.	0.001	More agg.	0.001
UM	Agg.	0.001	Agg.	0.001	Less agg.	0.026
BW	Agg.	0.001	Agg.	0.001	None	0.138

Potential patterns are uniform (unif.), aggregated (agg.) or random

At the patch scale, mechanical treatments reduced the frequency of the continuous cover patch size class (> 20 trees), and the large patch size class (10-20 trees), across all sites, and increased relative frequency of the smallest class (2-4 trees) (Table 1.5). Before treatment, the continuous cover patch class of over 20 trees per patch comprised a sizable portion of the patches (e.g. 43% of patches in MG). The treatments substantially reduced the number of these largest patches, completely eliminating their presence in sites HB, LC, and DL.

Before treatment, the frequency of the smallest patches was sometimes as infrequent as 22% of patches, such as in sites MG and UM. After treatment, their presence was much more common, comprising 44% to 100% of all patches. As larger patches comprise more areal cover, a disproportionate removal of these patches would relate to a reduction in total cover represented by patches. Accordingly, the areal cover of patches fell by 8% to 26% across sites following treatment (Table 1.5). Yet, the areal cover of patches still was greater than the areal cover of individual trees across a majority of sites after treatment; BW was the exception where slightly more area was represented by individual trees than patches.

Table 1.5. Patch scale analysis of stem-mapped sites before (pre) and after (post) mechanical treatments.

Site	HB		LC		MG		PC		DL		UM		BW	
Status	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
Aerial cover (%)														
Indiv. Tree	8	8	5	7	4	8	5	6	6	9	4	4	8	5
Openings	62	70	60	68	55	77	60	83	73	82	57	72	76	91
Patches	30	22	35	25	41	15	29	12	21	9	39	24	16	4
Patch metrics														
Frequency by patch size (# of trees) (%)														
Small (2-4)	51	59	32	59	22	53	35	56	36	80	22	44	67	100
Medium (5-10)	23	26	18	29	18	19	25	23	18	17	13	21	13	0
Large (11-20)	19	15	25	12	16	15	21	12	23	3	31	17	20	0
Continuous cover (>20)	6	0	25	0	43	13	18	9	24	0	34	18	0	0

Across sites, mechanical treatments had varying effects on vertical complexity at both the stand and patch scales. At the stand scale, the median height differentiation index (TH) ranged greatly across sites after treatment, from a low of 0.17 in BW to a max of 0.77 in site PC (Figure 1.4). Median height differentiation values significantly increased in sites PC (*W*=2,426,161; p=0.01), MG (*W*=1,232,770; p=0.01), and HB (*W*=885,645; p=0.01), and significantly decreased in sites BW (*W*=203,870; p<0.001), UM (*W*=1,330,682; p=0.01), DL (*W*=835,247; p<0.001), and LC (*W*=1,940,075; p<0.001), following treatment.

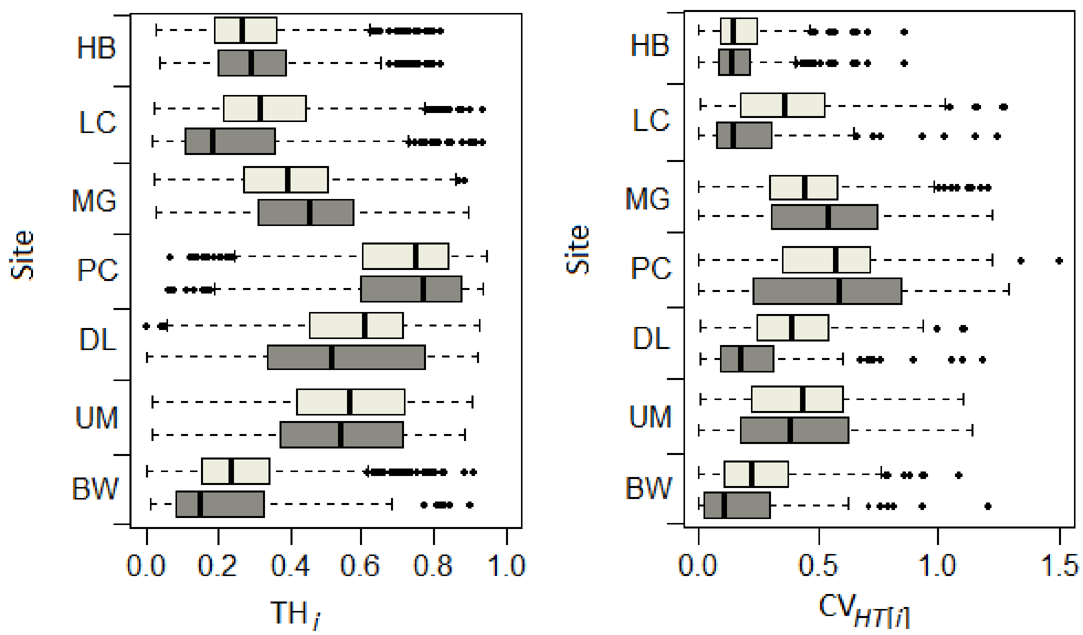

Figure 1.4. Distributions of height differentiation index values (TH$_i$) and of coefficients of variation of tree heights (CV$_{HT[i]}$) among each site before (light grey) and after (dark grey) mechanical treatments. Larger values represent greater degrees of height differentiation.

Patch scale vertical complexity, as measured by CV$_{HT}$, also ranged across sites post-treatment from a minimum median of 0.13 at HB to a maximum median at 0.55 at PC (4). At this measured scale, treatments resulted in significantly greater median values only in MG (*W*=33,889; p=0.001), and significantly lesser median values among sites BW (*W*=7432; p=0.004), DL (*W*=21,836; p<0.001) and LC (*W*=25,088; p<0.001), while treatments did not significantly alter median values among sites UM (*W*=19,113; p=0.330), HB (*W*=38,026; p=0.155), and PC (*W*=40,163; p=0.611).

1.3.3 Simulated treatment effects on fire behavior and wind flow

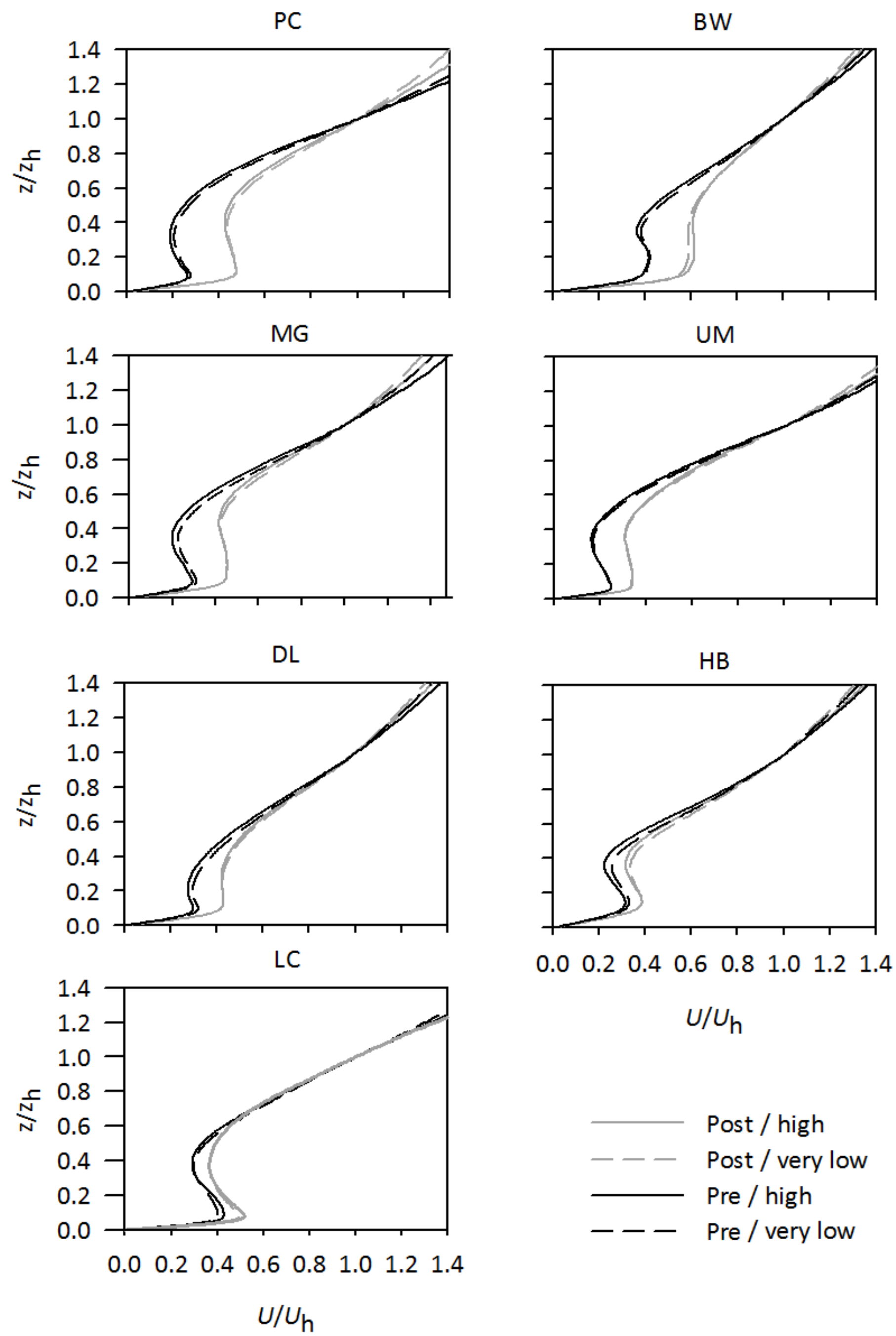

Figure 1.5. Vertical wind profiles of streamwise wind velocity, *U*, by height, z, across 7 different sites before (pre) and after (post) mechanical treatments at two wind scenarios (very low and high) as modeled by WFDS. Normalization is by h, the canopy height, and sites are ranked by basal area reductions from high to low.

Mechanical treatments altered mean within-canopy streamwise wind velocities and wind profile shapes at all sites. Wind profiles normalized by the wind speed at the top of the canopy demonstrate that open wind velocities had little effect on the functional relationship of wind speeds at a given height with respect to wind speeds at the top of the canopy (Figure 1.5). In other words, the wind profile shape below the canopy height was fairly constant across wind speeds. However, the reduction in canopy bulk density following treatments did alter the shape of the wind profile. The reduction of the prominence the inflection point within the canopy following treatment signifies that within-canopy wind speeds were higher after treatment despite similar open wind speeds (Figure 1.5). This finding is consistent among sites, with greater changes in wind associated with heavier thinnings, such as in PC, and lesser changes in wind associated with lighter thinnings, such as in LC.

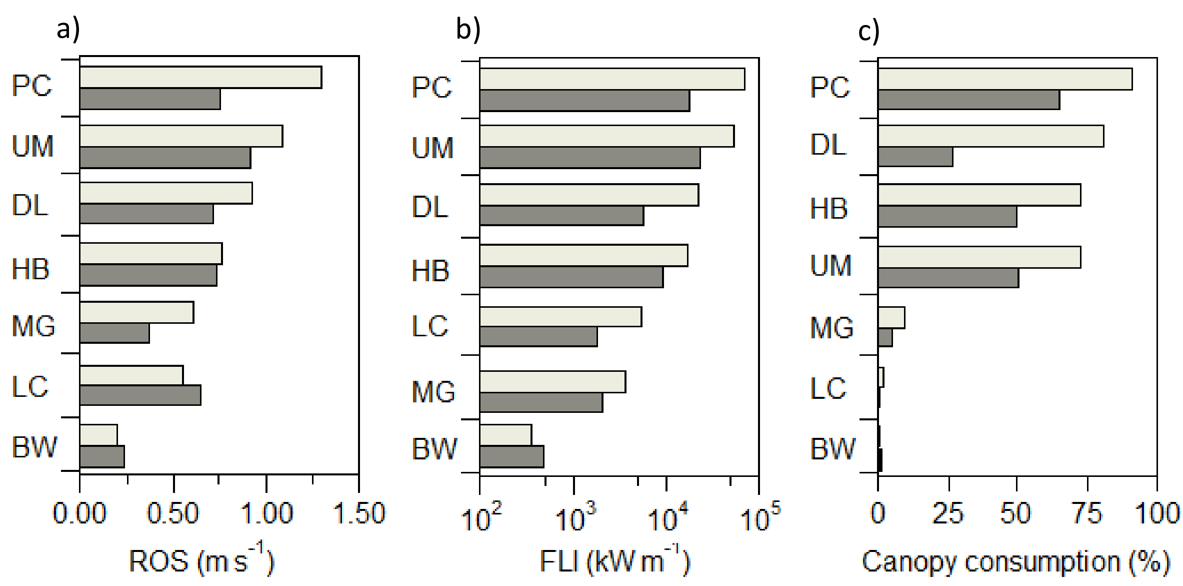

Figure 1.6. a) Rate of spread (ROS), b) log-scaled fireline intensity (FLI), and c) canopy consumption before (light grey) and after (dark grey) mechanical treatments averaged across 4 open wind scenarios, across seven sites, ranked by pre-treatment fire behavior, as predicted by WFDS.

Rate of spread was reduced in five of the seven sites following mechanical treatments. Among these 5 sites, the reduction of rate of spread, averaged over wind scenarios, ranged from 4% in HB to 42% in PC. In LC and BW, sites with relatively moderate rates of spread before treatment, rate of spread had increased by 16% and 20%, respectively (Figure 1.6a). As open wind speed increased, rate of spread increased at a greater rate in pre-treatment simulations than post-treatment (Figure 1.7a). Rates of spread, averaged across the five sites with decreases following treatment, were 8% lower at very low wind scenarios and 32% lower under high wind scenarios, respectively.

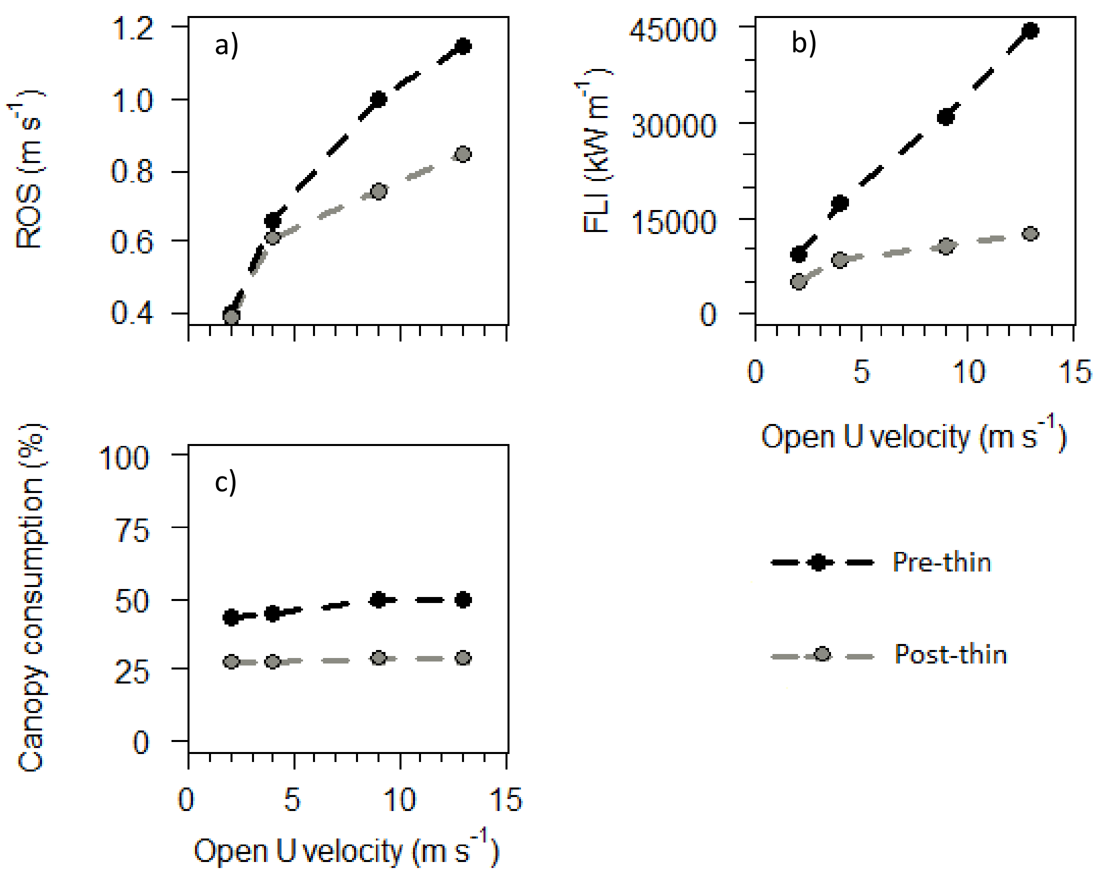

Figure 1.7. Modeled a) rate of spread (ROS), b) fireline intensity (FLI), and c) canopy consumption before and after mechanical treatments, across four open wind scenarios averaged over seven sites, as predicted by WFDS.

Fireline intensity was reduced following mechanical treatments in six of seven sites. In BW, fireline intensity increased over all wind scenarios by an average of 30%, bearing in mind fireline intensity was very low relative to other sites (Figure 1.6b). Across the 6 sites where fireline intensity was reduced, there was a substantial dependency of treatment impact on open wind speed. At the very low wind scenario, treatments resulted in a 47% reduction in fireline intensity, and a 72% reduction in fireline intensity at the high wind scenario, averaged across sites (Figure 1.7b).

WFDS also predicted that treatment in all sites but BW would reduce percent of canopy consumed (Figure 1.6c). Averaged over open wind speeds, the proportion of canopy consumed was reduced by up to 63% in DL and as low as 40%, among 6 sites. Though percent canopy consumed increased in BW following treatment, the effect was less than a 1% increase across all simulations (Figure 1.6c). Unlike rate of spread and fireline intensity, the effect of open wind speed on percent canopy consumed was only weakly correlated to the effect of treatment (Figure 1.7c).

The high site to site variability in fire behavior before treatments and their varying of thinning intensity from treatments is partly accounted by the non-spatial measures of the fuels complex. As illustrated above, the level of open wind does influence rate of spread and fireline intensity, however fuels cumulatively explain a greater portion of variability in measures of fire behavior (Table 1.6). Of the three fuels complex parameters tested, surface fuel load overwhelmingly explained the most variability: 26% of variability in rate of spread and fireline intensity each, and 59% of the variability in percent canopy consumed.

Table 1.6. ANOVA results for fire behavior metrics under four open wind speed scenarios, for seven different stands before and after mechanical treatments, as modeled by WFDS.

Source of variation	Df	Rate of spread			Fireline intensity			% canopy consumed		
		SS	*F*	ω^2	SS	*F*	ω^2	SS	*F*	ω^2
Open wind	1	2.9	54.4*	0.28	3,712,924,635	13.8*	0.10	218.4	2.0	0.00
Surface load	1	2.6	50.2*	0.26	9,388,597,277	35.0*	0.26	37,874.5	338.2*	0.59
CBD	1	0.5	8.8*	0.04	2,808,818,540	10.5.*	0.08	781.4	7.0*	0.01
CBH	1	0.1	2.8*	0.01	263,268,492	1.0	0.00	2.02	0.0	0.00
Error	47	2.7			13,690,593,637			5,258.7		

* Denotes factor was significant ($p < 0.05$)

1.4 DISCUSSION

1.4.1 Changes in structural complexity

Results reveal that the seven different mechanical treatments altered aspects of structural complexity in an inconsistent manner. Various considerations during implementation likely produce mixed impacts. First, implementation of hazard reduction objectives (Agee and Skinner 2005) likely lead to trade-offs with respect to producing greater structural complexity (Naficy et al., 2010; Larson and Churchill, 2012). It is likely that increases in the canopy heights among all sites resulted from targeted removal of ladder fuels. As ladder fuels are chosen due to their close proximity to a larger tree (Menning and Stephens, 2007), their removal would produce less aggregation at the stand scale aggregation and lower vertical complexity at stand and patch scales. Another silvicultural tactic reducing structural complexity is increasing inter-tree distances in order to locally lower canopy bulk density (Agee and Skinner, 2005). This would produce lower levels of aggregation at the stand scale, and, at the patch scale, a shift towards individual tree prevalence over patches. To counteract homogenization of complexity, there are other silvicultural tactics implemented. Stand scale aggregation increases are partly attributable to opening creation. This concentrates tree removals and produces greater variability of inter-tree distances than a regular, dispersed thinning. At the patch scale, Churchill et al. (2013a) demonstrated horizontal complexity is created when specific targets of patch sizes are used. This avoids leaving large areas of continuous cover, or conversely, a stand mainly composed of individual trees.

Though these mechanical treatments often did not increase a specific measure of structural complexity, it does not constitute a general failure to promote some degree of heterogeneity. In these sites, post-treatment structural complexity was still largely defined by an aggregated tree patterns, some degree of vertical complexity, and a matrix of individual trees and smaller patches; these qualities mirror desired attributes of dry forest structure (Larson and Churchill, 2012). Rather, the assumption that contemporary forests are homogeneous (Veblen et al., 2000; Fulé et al., 2009; Naficy et al., 2010) and that treatments ought to increase complexity may be inappropriate. The spatially-dependent processes such as regeneration and mortality still occur in contemporary forests to produce complex structures even in undesirably dense stands (Mast and Veblen, 1999; Boyden et al., 2005). Further, these treatments may result in greater structural complexity through patch dynamics over time. The creation of openings and aggregative tree patterns produce heterogeneous distributions of resources and rates of inter-tree competition (McCarthy, 2001). As a result, trees on the periphery of patches can have greater growth rates than interior trees given greater resource availability. This could lead to higher vertical complexity through time (*sensu* Plotkin et al., 2002).

Division of structural complexity into horizontal and vertical dimensions provides a useful framework of assessing the distribution of trees as they varied in space and size. Through analysis of multiple spatial analyses, this study could individually focus on specific facets of heterogeneity, as recommended by Perry et al. (2006). In a review of reported structure of dry, historic forests, or those with intact disturbance regimes, Larson and Churchill (2012) noted that studies have focused on the horizontal distribution of structure, but have not shed light on the vertical distribution through space. Yet, how structure is vertically distributed through space has ecological implications, such as plant and animal niche diversity (McElhinny et al., 2005). This is emphasized primarily only in moist old-growth forests, despite the many parallels in structural development with dry forests (Franklin, 2002; Zenner, 2004).

1.4.2 Changes in fire behavior

WFDS simulations of the sites suggested all treatments increased wind speeds within and below the canopies. In addition, those treatments with heavier thinnings resulted in greater wind speeds within the canopy. This finding was unsurprising as the causal link between canopy biomass and imposed drag is well-supported in the literature (Landsberg and James, 1971; Raupach and Thom, 1981; Pimont et al., 2011; Linn et al., 2013). Greater open wind speeds did interact with the fire to produce increased surface fireline intensities and increase potential for crown fire initiation. Once in the canopy, higher wind speeds tilt the plume of hot gasses towards horizontal, increasing convective heat flux to unburnt crowns (Linn, 1997), and produce greater rates of spread (Xanthopoulos, 1990, Scott and Reinhardt, 2001).

In the five sites where a notable amount of crown fire activity was observed prior to treatment (>2% canopy consumed), mechanical treatments reduced all three measures of fire behavior despite increases in within and sub-canopy wind flow. Treatments reduced the total amount of available surface and canopy fuel, increased the typical gap from the surface to the canopy. In addition, the sped-up wind speed was not exacerbative as greater winds increase tilt of the plume such that the canopy is less directly contacted (Luke and McArthur, 1978; Cruz et al., 2006) and high wind speeds dilute plumes of hot gasses by introducing ‘fresh’ cool air (Bilbao et al., 2001). Most significantly, I observed that fire behavior, post-treatment, increased with open wind speed at a rate less than pre-treatment. In the more open canopies, convective cooling towards unburnt crowns ahead of the fire front is greater than convective heating (Tachajapong et al., 2014).

Interestingly, mechanical treatments did not lead to a reduction in fire behavior on two of our sampled sites. On these sites, the fire primarily burned through the surface fuels with little surface to crown fire transition occurring. This is likely due to the low surface fuel loads observed in these sites and a sparser canopy which allowed for greater cooling as the fire spread below the canopy. Due to the lack of involvement of the canopy fuels in these sites the simulated fire dynamics parallels wind-driven surface-fires (e.g. Cheney et al., 1993). In these fire dynamics, increases in sub-canopy wind velocities exacerbate the surface fire by tilting the flaming front thus increasing effective radiative heat flux to unburnt surface fuels and increasing rates of spread and fireline intensities. The increased fire rates of spread and intensities in these stands may weaken the ability of fire operations to utilize treated areas during suppression operations (Moghaddas and Craggs, 2007; Reinhardt et al., 2008). Further, though not observed here, treatments in similar stands could lead to increased crown fire activity if conditions are such that the net heat flux exposure of canopy fuels was increased. Such conditions could occur if large accumulations of surface fuels, such as activity fuels or shrub and tree regeneration were present, or if the spatial arrangement of canopy fuels allows high residence time of surface to canopy convective heat flux (Tachajapong et al., 2014). In addition it is unknown how extreme wind scenarios, greater than those tested in this study, will influence fire behavior and the onset of crown fire activity. Further research is needed to better understand under what conditions treatments could lead to greater occurrences of surface to crown fire transitions.

Across simulations of fire behavior both before and after treatment, canopy consumption ranged from negligible, indicative of surface fire, to crown fires combusting up to 93% of all canopy mass. Among simulations of over 20% canopy consumed, the median fireline intensity was ~20,600 kW m^{-1} with an interquartile range from 10,000 kW m^{-1} to 33,000 kW m^{-1}; at the extreme end, two WFDS simulations predicted intensities of ~108,000 kW m^{-1} and 117,000 kW m^{-1}. Accurate comparisons to crown fire fireline intensities are difficult due to the indirect method of fireline intensity estimation (Alexander, 1982), though available data suggest crown fires commonly range from 4,000 kW m^{-1} to 30,000 kW m^{-1}, potentially reaching up to 150,000 kW m^{-1} (Kiil and Grigel, 1969; Trabraud, 1989; Stocks et al., 2004; Alexander and Cruz, 2013). In this study, WFDS predicted rates of spread across simulations varied by an order of magnitude, from 0.18 m s^{-1} to 1.95 m s^{-1}. Crown fire rates of spread in ponderosa pine forests have been observed to range from 0.24 m s^{-1} to 1.80 m under open wind speeds of 4.8 to 8.9 m s^{-1} (Alexander and Cruz, 2006). This range fits well with predictions from WFDS. Performing a rigorous evaluation of results from this study compared against fire behavior observed in field scale experiments is not trivial. WFDS requires a large extent of detailed data for parameterization, while real-world data is limited and requires liberal interpretation (Linn et al., 2012). At face value, WFDS appears to qualitatively fit within observed fire behavior; it should be kept in mind that empirical observations are generally reported over larger spatiotemporal scales (e.g. km and hr), whereas this study's observations occur over only 200 m for 1 to 15 minutes. The variability expected in fire behavior across a landscape produced by changes in topography and fuels, and weather, especially over the course of hours, substantially limit direct comparison.

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2. LITTER BULK DENSITY ESTIMATES FOR USE IN SOUTHERN ROCKY MOUNTAIN FORESTS

2.1 INTRODUCTION

An accurate inventory of the fuels complex is critical for a wide range of current management challenges including assessing fire hazard and effects, the design of fuels treatments, wildlife habitat assessment and the quantification of carbon pools (Agee and Skinner, 2005; Keane and Dickinson, 2007). In addition, quantitative descriptions of wildland fuels are often required as inputs for planning and management models (Black and Opperman, 2005; Hessburg et al., 2007; Keane, 2013). Common fuel assessments often simplify the fuels complex through the use of distinct fuelbed categories and then quantifying the desired characteristics for each category, such as fuel moisture, the mineral content or the heat content. The most common characteristic quantified for any fuelbed category, and the central focus of this paper, is fuel load or the dry weight biomass of fuel per unit area (e.g. kg m^{-2}, tons ac^{-1}).

Multiple methodologies have been developed to quantify fuel load for various fuelbed categories ranging from simple, rapid visual assessments to highly detailed measurements typically along transects or in fixed area (Sikkink and Keane, 2008) . There has been particular interest in accurately evaluating surface fuel loads which encompass dead and down woody material, non-woody vegetation such as grasses, forbs and shrubs, and the forest floor (Keane, 2013). Recent studies have evaluated the accuracy of multiple methods used to quantify dead and down woody material (Sikkink and Keane, 2008), but there has been little assessment or development of more robust methods used to quantify the forest floor or non-woody surface fuel loads.

Despite that litter is often the most abundant source, by mass, of potential surface fuel to carry fire in dry, western forests (Ottmar et al 2007), litter load estimation has received little attention relative to other fuel categories. Litter, also known as the O_i soil horizon, consists of the relatively undecomposed organic matter fallen from vegetation (Reinhardt et al., 1997). Litter is distinguished from woody fuels layers above and the more decomposed duff strata (O_e and O_a soil horizons) below (Ottmar et al., 2003). Litter often burns completely, contributing to fire behavior, as well as influencing fire effects such as nutrient storage, erosion potential, and post-fire vegetation dynamics (Kauffman and Martin, 1989; Bonnet et al., 2005).

The most common approach to characterizing litter load is by directly measuring litter depth and multiplying by a reference bulk density value, hereafter referred to as the depth–to–load method (Sikkink and Keane, 2008). The accuracy associated with this method depends upon a reliable estimate of litter bulk density (e.g. the dry weight of litter biomass per volume, kg m^{-3}). In general, it is assumed that litter bulk density values are specific to forest types due to differences in compactness driven by needle morphology (Van Wagtendonk et al., 1998). However, reference values are limited geographically, by forest type and by recent disturbance history due to difficulty in measuring litter (Schulp et al., 2008). For example, though studies quantifying litter bulk density in ponderosa pine dominated forests (*Pinus ponderosa* Lawson) are well represented, these studies occur primarily only in Arizona, (Ffolliott et al., 1976, 1968; Eakle and Wagle, 1979), California (Van Wagtendonk et al., 1998; Reiner et al., 2009) and the northern Rocky Mountains (e.g. Brown, 1970, 1981), despite its wide range throughout the western United States. Studies in ponderosa pine dominated forests also have either sampled stands without recent management activity (Harrington, 1986), did not differentiate study sites based on disturbance (Ewell, 2006), or did not provide adequate details to determine disturbance history (Brown, 1981). The disparate sampling efforts have precluded analysis into whether recent disturbances, such as management activity may impact litter bulk density, as suggested by Sevgi et al. (2011). These factors add to the uncertainty surrounding the appropriate choice of a litter bulk density value. The need to increase sampling of litter bulk density across forest types and geographical regions, taking into account recent disturbance history cannot be overstated. Without reliable estimates, use of the depth-to-load method is much hindered (Brown et al., 1982).

The primary objective of this study is to provide regional-specific litter bulk density values for two common, fire-frequent forest types of the southern Rocky Mountains, ponderosa pine and dry mixed-conifer, in both recently mechanically thinned and unthinned stands. Values from this investigation are meant to support surface fuel load assessments using the load-to-depth method complimentary to other methods such as dead and down woody fuel transects.

2.2 METHODS

2.2.1 Litter inventory

Litter bulk density was measured across five study sites located throughout the Colorado Front Range and Colorado Plateau. Sampling occurred during summer months of 2012 and 2013. Within each site, a mechanically thinned stand two years after treatment was sampled, as well as an adjacent unthinned stand identified in consultation with local forest managers as having similar structure and composition to that of the thinned stand before thinning. All thinned stands were treated with a cut-to-length removal with the tree tops and branches scattered. Initial and post-treatment stocking varied across thinned stands (Table 2.1). Ponderosa pine was the dominant overstory tree in four sites, comprising over 85 % of total basal area. In the mixed conifer site (Phantom Creek), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), ponderosa pine, and spruce (*Picea engelmannii* Parry ex Engelm.; *Picea pungens* Engelm.) were the most common tree species, comprising 54%, 20% and 19% of pre-thinning basal area, respectively. The thinning favored ponderosa pine, a more fire-tolerant species, as recommended by fuels reduction strategies (Reinhardt 2008). Post-thinning, Douglas-fir, ponderosa pine, and spruce composed 41%, 32% and 10% of basal area, respectively.

In each stand, samples were collected across a 4 hectare plot chosen for having less than 10% average slope and little topographic variability. A starting point was randomly generated within each stand as a corner for establishing a square grid with 64 points spaced regularly every 25 m. At each grid point, an azimuth and distance was randomly selected with 12m from the grid point. At each sampling location, a square 930 cm^2 frame was inserted into the forest floor. Litter depth was measured at nine, uniformly distributed points in the frame, collected and weighed after oven-drying. Dead and down woody material in the litter stratum was removed before weighing as supplementary sampling methods are available for estimating this surface fuel component (e.g. Brown, 1974). Litter bulk density of each sample was calculated by dividing the oven-dry mass over the product of mean litter depth and frame area.

Table 2.1. Composition and stocking of sampled stands, before and after thinning, based on trees at least 1.4 m tall.

Managing agency	Site	Major tree spp.*	Basal area ($m^2 ha^{-1}$)		Density (trees ha^{-1})	
			Before	After	Before	After
Boulder Cty. Parks & Open Spaces	Heil	PIPO	19	15	418	295
Pike NF	Messenger Gulch	PIPO	23	11	685	269
Cibola NF	Bluewater	PIPO	19	8	327	63
Uncompahgre NF	Uncompahgre Mesa	PIPO	30	18	504	314
Pike NF	Phantom Creek	PIPO, PSME, PICEA	25	10	934	352

*Species with at least 15% of total basal area before thinning; nomenclature follows NRCS PLANTS database (USDA NRCS, 2014)

2.2.2 Statistical analyses

Mean litter bulk density of thinned and unthinned ponderosa pine dominated sites was compared with a mixed-effects 2-way analysis of variance with treatment status (thinned or unthinned) as a fixed effect and site as random effect, included to account for stand level variability (PROC MIXED, SAS version 9.3). For the mixed-conifer site, a two-sample *t*-test determined differences among mean litter bulk density by thinning status (PROC TTEST, SAS version 9.3). For all tests, significant differences are reported using $\alpha = 0.05$. Some observations were missing, leading us to perform unbalanced tests. Inspection of residuals revealed tests were balanced and had equal variance, so data were untransformed.

Regression analysis was used to determine if litter bulk density remained constant regardless of total litter depth (PROC GLM, SAS version 9.3). A lack of significance in a regression of observed litter depth and litter bulk density would indicate that a single litter bulk density could be used to estimate litter fuel load.

2.3 RESULTS

Litter bulk densities across all sampled ponderosa pine sites, regardless of treatment status, averaged 20.88 kg m^{-3}, with a standard deviation about 50% of the mean (Table .2). Bulk density was 13 % lower in thinned than unthinned stands ($F = 11.39$, $P < 0.001$). Site explained a significant portion of variance in litter bulk density ($F = 9.33$; $P < 0.001$). A post-hoc site multiple comparison revealed the site at Messenger Gulch significantly differed from the other sites, with a bulk density about 5 kg m^{-3} denser than the others (Table 2.2).

Mean litter bulk density in dry, mixed conifer stands was 58.23 kg m^{-3}. Standard deviation of observations was over twice as great than in ponderosa pine. Litter bulk density was on average 15.61 kg m^{-3} more dense in the thinned versus unthinned stands ($F = 12.35$; $P <$ 0.001) in this forest type.

Average litter depths were similar in ponderosa pine and mixed conifer stands, as well as in thinned and unthinned stands. The total range of litter depths for ponderosa pine and mixed conifer stands were from 0.1 cm to 7.0 cm and from 0.2 cm to 5.6 cm, respectively. Among ponderosa pine stands, there was no significant influence of litter depth on litter bulk density ($F = 1.92$, $P = 0.17$), while among the mixed conifer samples, litter bulk density significantly decreased with litter depth, though the correlation was low (F = 21.481, P < 0.01; adj. r^2 = 0.14).

Table 2.2. Summary statistics of litter bulk densities and depths across sampled stands by forest type, treatment status and site. Superscript letters represent groups from Tukey-Kramer adjusted pairwise comparisons.

	n	Litter bulk density ($kg\ m^{-3}$)		Litter depth (cm)	
		x̄	s	x̄	s
Ponderosa pine					
(all)	506	20.88	10.25	1.9	1.4
by status					
Thinned	254	19.46	10.25	2.1	1.5
Unthinned	251	22.3	10.07	2.3	1.1
by site					
Heil	127	20.18	9.30*a*	2.9	1.6
Messenger Gulch	125	24.69	13.36*b*	1.7	1.0
Bluewater	128	20.28	8.91*a*	1.2	0.6
Uncompahgre Mesa	126	18.43	10.02*a*	2.9	0.9
Dry, mixed conifer – Phantom Creek					
(all)	125	58.23	26.04	1.8	0.9
by status					
Thinned	63	65.91	20.15	1.7	0.6
Unthinned	62	50.30	29.07	2.0	1.1

2.4 DISCUSSION

2.4.1 Litter bulk density by forest type

The results provide litter bulk density values for use with the load-to-depth method for ponderosa pine and dry mixed conifer forests in the Colorado Front Range and Colorado Plateau. Within this study, mean bulk density in dry mixed conifer forest was almost three times denser than in the ponderosa pine forests. Litter bulk density in the mixed conifer site was likely denser due to the prevalence of spruce and Douglas-fir. The morphological characteristics of these species' foliage lead to "stacking" of litter and relatively dense litter layers. Conversely, pine litter beds are more likely to create "fluffy" litter strata as pine needles are bound to fascicles and protrude in various directions. If needle morphology does explain these differences, variability in species diversity could explain the greater variability in litter bulk densities of the mixed conifer site.

Further, results suggest effects of thinning on litter bulk density depended upon forest type. For ponderosa pine forest, results suggest that there is a decrease in the litter bulk density, while in the dry mixed conifer forest site, litter bulk density was higher following thinning. It's possible litter bulk density decreases in treated ponderosa pine stands may be due to a shift in litter inputs from dense material such as strobili and bark slough to a lighter material such as dead grass and forb leaves component of herbaceous material. The contrasting result of the increase in litter bulk density for the thinned mixed conifer stand demonstrates the importance of distinguishing among forest types and management activities. Perhaps the preferential thinning of the Douglas-fir and spruce within the mixed conifer site increased litter bulk density by adding in a large component of small and compact needles. Regardless, the observed effect size of thinning was diminished by site to site variance and replication was limited to only one mixed conifer site. Therefore, managers need not select reference litter bulk density values based on disturbance for these forest types, except for dramatic disturbances of the forest floor such as mastication (Reiner et al., 2009; Battaglia et al., 2010).

Litter bulk density remained constant regardless of total litter depth yielded different results based on forest type. The lack of relationship across the ponderosa pine forest type supports the assumption that the use of a single bulk density value is valid for litter depths up to 7 cm deep in the study region. In contrast, a relationship between sample litter depth and litter bulk density in mixed conifer forests was observed, but it had limited explanatory power and samples were from just one site. Until further research investigating the relationship between litter depth and bulk density, it is suggested a single bulk density value is appropriate for litter samples of varying depths for both forest types.

2.4.2 Comparison to previously published values

Several previously reported values in ponderosa pine forests of other regions compare well with results from this study. In Arizona, Ffolliott et al. (1968, 1976) presented slightly lower mean litter bulk density values (15.89 kg m^{-3} and 17.65 kg m^{-3}) then those reported here (~20 kg m^{-3}). Their relatively low values are believed to be caused by their exclusion of all non-needle material (Harrington, 1986). However, in western Montana, Brown (1970) reported a mean bulk density of 15.8 kg m^{-3} and included all non-woody material. Brown also later reported

(Brown, 1981) median litter bulk density values for two ponderosa pine habitat types sampled across a broader region of the northern Rockies. These median values, 21.9 kg m^{-3} and 22.4 kg m^{-3} are close in agreement to unthinned values here. Again, all vegetative organic matter, with exception of dead and down wood was included.

In contrast, several studies report much higher estimates of litter bulk density in ponderosa pine forests ranging from between 53.50 kg m^{-3} and 88.42 kg m^{-3}(Eakle and Wagle, 1979; Battaglia et al., 2010, 2008; Woodall and Monleon, 2008). However, these studies use alternative methods to identify the litter layer. These studies identified litter similar to protocols of Soil Survey Staff (1975) where the forest floor is divided into litter (O_1) and duff (O_2) strata based on whether or not constituent origin is macroscopically identifiable. In contrast, this study, and those that reported similar values use the stratification consistent with (Soil Classification Working Group, 1998) where the forest floor is divided into: a freshly fallen litter horizon with macroscopically identifiable origins (O_i); a partially decomposed fermentation horizon with macroscopically identifiable origins (O_e); and a much decomposed humus horizon without macroscopically identifiable origins (O_i) (Figure 2.1). Presumably, because forest floor bulk density increases with depth (van Wagtendonk et al., 1998, Stephens et al., 2004), studies based on the method by Soil Survey Staff (1975) are more likely to result in greater estimates of litter bulk density.

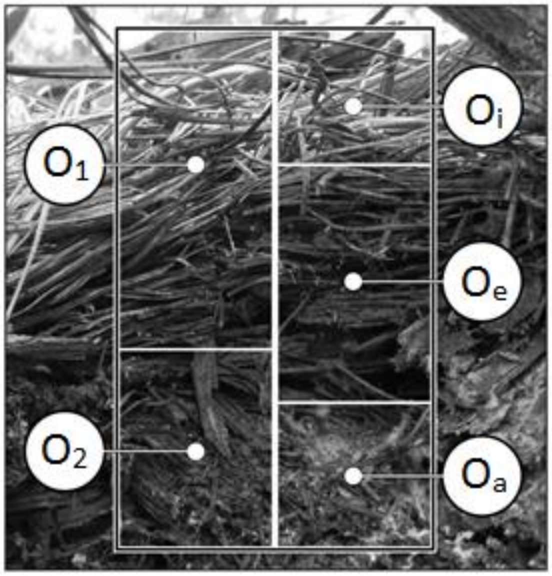

Figure 2.1. Example application of two methods to identify forest floor strata. On the left, Soil Survey Staff (1975) and, on the right, Soil Classification Working Group (1998).

Comparison to other dry mixed conifer forests values is difficult because no other study has quantified these values for a similar species composition mix as sampled in this study. Battaglia et al. (2010) sampled in the Colorado Front Range and reported a median litter bulk density of 46.6 kg m^{-3}. However, the species mixture was of lodgepole pine (*Pinus contorta* Douglas ex Loudon), ponderosa pine, limber pine (*Pinus flexilis* James) and Douglas-fir. In Arizona, Ffolliott et al. (1977) reported a much lower litter bulk density (ca. 18 kg m^{-3}) in stands of Douglas-fir, white fir (*Abies concolor* [Gord. & Glend.] Lindl. Ex Hildebr.), southwestern white pine (*Pinus strobiformis* Englm.), corkbark fir (*Abies lasiocarpa* [Hook] nutt. var. *arizonica* [Merriam] Lemmon), ponderosa pine, blue spruce, Engelmann spruce and quaking aspen (*Populus tremuloides* Michx.). The relative composition of forests in these studies is unclear, but further suggest that mean litter bulk density may be affected by composition. This highlights the challenge and complexity that is faced when attempting to estimate litter fuel loads in these forest types. Further research is needed to examine the changes in litter bulk density in relation to various mixtures of species composition for better estimates of litter fuel loads in these forest types.

2.4.3 Management implications

The difference in choice of a reference litter bulk density carries implications for decision making in forest management. For example, observed mean value for litter bulk density and depth measured here results in an estimated 0.40 kg m^{-2} litter load, versus 1.02 kg m^{-2} using Woodall and Monleon (2008). This impacts fire management decisions as smoke emissions and heat per unit area from litter are predicted using constant factors based on litter load. (Reinhardt et al., 1997). The difference in estimated litter loads may influence decisions on whether or not to employ a prescribed fire for a given set of environmental conditions and operational burning tactics used (Hardy et al., 2001). Accurate litter load estimation is also pertinent to other forest management objectives, such as carbon emission offsets. As carbon content of litter is also estimated as a constant factor of litter load (Law et al., 2001), the predicted amount of stored carbon predicted will greatly affect investment-return expectations of carbon offset projects (Malmsheimer et al., 2011).

2.4.4 Future work

While these results provide a baseline reference value for ponderosa pine and dry mixed conifer forests in Colorado there are several additional factors that may influence litter bulk density. Additional variability in litter deposition and decomposition during a year in response to seasonal changes in litterfall rates, across years during events such as droughts, and in abrupt episodes such as intense storm or wind events is likely. Sampling occurred only during summer months as forest and fire managers are especially interested in the fuels complex at this temporal snap-shot. Further, additional investigation into the spatial distribution of litter properties and how forest type, structure and disturbance history shapes this distribution is warranted. This information can be used in the future to better link litter properties to fire behavior and effects.

2.4 CONCLUSION

Application of the values reported here is best utilized when estimating the litter fuel load using the load-to-depth method. Although fuels monitoring databases, and fuels calculation systems often rely on this method for estimating litter fuel load not all systems allow the user to customize the litter bulk density value. For example, FuelCalc (Reinhardt et al., 2006) and FFI (Lutes et al., 2006) allow for customization of litter bulk density, while FCCS (Prichard et al., 2011) does not allow for user to customize the litter bulk density values. Instead, for FCCS, observed litter depth may need to be altered in order to match a litter load calculated from this study's reported bulk densities. Users should also note definitions of litter constituents may vary by application. FCCS, for example, defines litter as needles, dead herbaceous material, bark slough and fine woody material less than 6.4 mm in diameter. As dead and down woody material was excluded from this study's derived estimates, FCCS users would need to account for my omission.

In conclusion, regional values of litter bulk density are provided here for managers to employ with the depth-to-load method for litter load estimation. These results reinforce the notion that forest type is an important consideration when choosing a litter bulk density value and that this method should work over a range of litter depths. At the same time, site to site variability was larger than that found following thinning. As such differentiating bulk density values according to thinning status may not result in greater accuracy.

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3. APPENDICES

3.1 TYPICAL WFDS INPUT FILE

The following text contains an example of information input into WFDS to perform simulations. The values that were changed according to individual simulations were: CHID, TITLE, surface fuel VEGETATION_LOAD and surface fuel VEGETATION_HEIGHT, the tree list, and inflow VEL.

```
&HEAD CHID='HB_Post_2ms'
TITLE='Numerical simulation of HB after treatment with inflow, open wind speed at 2 ms' /
```

- Mesh set-up

```
&MESH ID='MESH1', IJK=310,25,100, XB=0,620,0,50,0,100 /
&MESH ID='MESH2', IJK=310,25,100, XB=0,620,50,100,0,100 /
&MESH ID='MESH3', IJK=310,25,100, XB=0,620,100,150,0,100 /
&MESH ID='MESH4', IJK=310,25,100, XB=0,620,150,200,0,100 /
&MESH ID='MESH5', IJK=310,25,100, XB=0,620,200,250,0,100 /
&MESH ID='MESH6', IJK=310,25,100, XB=0,620,250,300,0,100 /
&MESH ID='MESH7', IJK=310,25,100, XB=0,620,300,350,0,100 /
&MESH ID='MESH8', IJK=310,25,100, XB=0,620,350,400,0,100 /
&MESH ID='MESH9', IJK=300,20,100, XB=620,920,0,20,0,100 /
&MESH ID='MESH10', IJK=300,20,100, XB=620,920,20,40,0,100 /
&MESH ID='MESH11', IJK=300,20,100, XB=620,920,40,60,0,100 /
&MESH ID='MESH12', IJK=300,20,100, XB=620,920,60,80,0,100 /
```

```
&MESH ID='MESH13', IJK=300,20,100, XB=620,920,80,100,0,100 /

&MESH ID='MESH14', IJK=300,20,100, XB=620,920,100,120,0,100 /

&MESH ID='MESH15', IJK=300,20,100, XB=620,920,120,140,0,100 /

&MESH ID='MESH16', IJK=300,20,100, XB=620,920,140,160,0,100 /

&MESH ID='MESH17', IJK=300,20,100, XB=620,920,160,180,0,100 /

&MESH ID='MESH18', IJK=300,20,100, XB=620,920,180,200,0,100 /

&MESH ID='MESH19', IJK=300,20,100, XB=620,920,200,220,0,100 /

&MESH ID='MESH20', IJK=300,20,100, XB=620,920,220,240,0,100 /

&MESH ID='MESH21', IJK=300,20,100, XB=620,920,240,260,0,100 /

&MESH ID='MESH22', IJK=300,20,100, XB=620,920,260,280,0,100 /

&MESH ID='MESH23', IJK=300,20,100, XB=620,920,280,300,0,100 /

&MESH ID='MESH24', IJK=300,20,100, XB=620,920,300,320,0,100 /

&MESH ID='MESH25', IJK=300,20,100, XB=620,920,320,340,0,100 /

&MESH ID='MESH26', IJK=300,20,100, XB=620,920,340,360,0,100 /

&MESH ID='MESH27', IJK=300,20,100, XB=620,920,360,380,0,100 /

&MESH ID='MESH28', IJK=300,20,100, XB=620,920,380,400,0,100 /

&MESH ID='MESH29', IJK=40,200,100, XB=920,1000,0,400,0,100 /


&TRNZ IDERIV=1,CC=0.,PC=0.5,MESH_NUMBER=1 /

&TRNZ IDERIV=1,CC=0.,PC=0.5,MESH_NUMBER=2 /

&TRNZ IDERIV=1,CC=0.,PC=0.5,MESH_NUMBER=3 /

        ...CONTINUED...

&TRNZ IDERIV=1,CC=0.,PC=0.5,MESH_NUMBER=28 /

&TRNZ IDERIV=1,CC=0.,PC=0.5,MESH_NUMBER=29 /


&TRNZ IDERIV=2,CC=0.,PC=0.0,MESH_NUMBER=1 /

&TRNZ IDERIV=2,CC=0.,PC=0.0,MESH_NUMBER=2 /
```

```
&TRNZ IDERIV=2,CC=0.,PC=0.0,MESH_NUMBER=3 /

        ...CONTINUED...

&TRNZ IDERIV=2,CC=0.,PC=0.0,MESH_NUMBER=28 /

&TRNZ IDERIV=2,CC=0.,PC=0.0,MESH_NUMBER=29 /
```

- Simulation duration

```
&TIME TWFIN=2500. /
```

- Miscellaneous parameters

```
&SPEC ID='WATER VAPOR' /

&MISC RADIATION=.TRUE.,BAROCLINIC=.TRUE.,TERRAIN_CASE=.FALSE., WIND_ONLY=.FALSE. /
```

- Parameters for gaseous combustion following solid fuel pyrolysis

```
&REAC ID='WOOD'

   FYI='Ritchie, et al., 5th IAFSS, C_3.4 H_6.2 O_2.5'

   SOOT_YIELD = 0.02

   O       = 2.5

   C       = 3.4

   H       = 6.2

   HEAT_OF_COMBUSTION = 17700 /
```

- Surface boundary fuels

```
&SURF ID = 'GROUND VEG1'

   VEGETATION = .TRUE.

   VEGETATION_INITIAL_TEMP = 20

   VEGETATION_CDRAG    = 0.05

   VEGETATION_MOISTURE = 0.05
```

```
    VEGETATION_SVRATIO  = 5710
    VEGETATION_CHAR_FRACTION  = 0.265
    VEGETATION_LOAD     = 0.58
    VEGETATION_HEIGHT   = 0.08
    VEGETATION_ELEMENT_DENSITY= 510
    EMISSIVITY = 0.99
    VEGETATION_ARRHENIUS_DEGRAD=.FALSE.
    FIRELINE_MLR_MAX = 0.35
    RGB       = 122,117,48 /
&VENT XB=1,680,0,400,0,0,SURF_ID='GROUND VEG1' /
&VENT XB=683,996,0,400,0,0,SURF_ID='GROUND VEG1' /
```

- Defining crown fuel

```
&PART ID='TREE',TREE=.TRUE.,QUANTITIES='VEG_TEMPERATURE',
    VEG_INITIAL_TEMPERATURE=20.,
    VEG_SV=4000.,VEG_MOISTURE=1.0,VEG_CHAR_FRACTION=0.25,
    VEG_DRAG_COEFFICIENT=0.125,VEG_DENSITY=520.,VEG_BULK_DENSITY=1.2,
    VEG_BURNING_RATE_MAX=0.4,VEG_DEHYDRATION_RATE_MAX=0.4,
    VEG_REMOVE_CHARRED=.TRUE. /
```

- Defining tree stems

```
&PART ID='TRUNK',TREE=.TRUE.,QUANTITIES='VEG_TEMPERATURE',
    VEG_SV=3.,VEG_MOISTURE=1.0,
    VEG_DRAG_COEFFICIENT=0.125,VEG_DENSITY=520.,
    VEG_BULK_DENSITY=520 /
```

- List of trees within study plot

```
&TREE XYZ=779.18,136.87,0,PART_ID="TREE",FUEL_GEOM="CONE",CROWN_WIDTH=1.08
,CROWN_BASE_HEIGHT=0.63,TREE_HEIGHT=1.38,OUTPUT_TREE=.TRUE.,LABEL="TREE1"/
&TREE XYZ=760.66,188.02,0,PART_ID="TREE",FUEL_GEOM="CONE",CROWN_WIDTH=0.9
,CROWN_BASE_HEIGHT=0.82,TREE_HEIGHT=1.42,OUTPUT_TREE=.TRUE.,LABEL="TREE2"/
&TREE XYZ=881.24,246.21,0,PART_ID="TREE",FUEL_GEOM="CONE",CROWN_WIDTH=0.66
,CROWN_BASE_HEIGHT=0.25,TREE_HEIGHT=1.43,OUTPUT_TREE=.TRUE.,LABEL="TREE3"/
          ...CONTINUED...
&TREE XYZ=858.74,256.68,0,PART_ID="TREE",FUEL_GEOM="CONE",CROWN_WIDTH=8.64
,CROWN_BASE_HEIGHT=6.95,TREE_HEIGHT=18.86,OUTPUT_TREE=.TRUE.,LABEL="TREE724"/
&TREE XYZ=827.35,147.72,0,PART_ID="TREE",FUEL_GEOM="CONE",CROWN_WIDTH=2.16
,CROWN_BASE_HEIGHT=0.48,TREE_HEIGHT=20.04,OUTPUT_TREE=.TRUE.,LABEL="TREE725"/

&TREE XYZ=779.18,136.87,0,PART_ID="TRUNK",FUEL_GEOM="CYLINDER", CROWN_WIDTH=0.0178
,CROWN_BASE_HEIGHT=0,TREE_HEIGHT=0.63 /
&TREE XYZ=760.66,188.02,0,PART_ID="TRUNK",FUEL_GEOM="CYLINDER", CROWN_WIDTH=0.0203
,CROWN_BASE_HEIGHT=0,TREE_HEIGHT=0.82 /
&TREE XYZ=881.24,246.21,0,PART_ID="TRUNK",FUEL_GEOM="CYLINDER", CROWN_WIDTH=0.0102
,CROWN_BASE_HEIGHT=0,TREE_HEIGHT=0.25 /
          ...CONTINUED...
&TREE XYZ=858.74,256.68,0,PART_ID="TRUNK",FUEL_GEOM="CYLINDER", CROWN_WIDTH=0.4953
,CROWN_BASE_HEIGHT=0,TREE_HEIGHT=6.95 /
&TREE XYZ=827.35,147.72,0,PART_ID="TRUNK",FUEL_GEOM="CYLINDER", CROWN_WIDTH=0.1041
,CROWN_BASE_HEIGHT=0,TREE_HEIGHT=0.48 /
```

- List of trees outside of study plot

```
&TREE XYZ=79.18,363.13,0,PART_ID="TREE",FUEL_GEOM="CONE",CROWN_WIDTH=1.08
,CROWN_BASE_HEIGHT=0.63,TREE_HEIGHT=1.38,OUTPUT_TREE=.FALSE. /
```

```
&TREE XYZ=60.66,311.98,0,PART_ID="TREE",FUEL_GEOM="CONE",CROWN_WIDTH=0.9
,CROWN_BASE_HEIGHT=0.82,TREE_HEIGHT=1.42,OUTPUT_TREE=.FALSE. /
&TREE XYZ=181.24,253.79,0,PART_ID="TREE",FUEL_GEOM="CONE",CROWN_WIDTH=0.66
,CROWN_BASE_HEIGHT=0.25,TREE_HEIGHT=1.43,OUTPUT_TREE=.FALSE. /
        ...CONTINUED...
&TREE XYZ=958.74,156.68,0,PART_ID="TREE",FUEL_GEOM="CONE",CROWN_WIDTH=8.64
,CROWN_BASE_HEIGHT=6.95,TREE_HEIGHT=18.86,OUTPUT_TREE=.FALSE. /
&TREE XYZ=927.35,47.72,0,PART_ID="TREE",FUEL_GEOM="CONE",CROWN_WIDTH=2.16
,CROWN_BASE_HEIGHT=0.48,TREE_HEIGHT=20.04,OUTPUT_TREE=.FALSE. /


&TREE XYZ=79.18,363.13,0,PART_ID="TRUNK",FUEL_GEOM="CYLINDER", CROWN_WIDTH=0.0178
,CROWN_BASE_HEIGHT=0,TREE_HEIGHT=0.63 /
&TREE XYZ=60.66,311.98,0,PART_ID="TRUNK",FUEL_GEOM="CYLINDER", CROWN_WIDTH=0.0203
,CROWN_BASE_HEIGHT=0,TREE_HEIGHT=0.82 /
&TREE XYZ=181.24,253.79,0,PART_ID="TRUNK",FUEL_GEOM="CYLINDER", CROWN_WIDTH=0.0102
,CROWN_BASE_HEIGHT=0,TREE_HEIGHT=0.25 /
        ...CONTINUED...
&TREE XYZ=958.74,156.68,0,PART_ID="TRUNK",FUEL_GEOM="CYLINDER", CROWN_WIDTH=0.4953
,CROWN_BASE_HEIGHT=0,TREE_HEIGHT=6.95 /
&TREE XYZ=927.35,47.72,0,PART_ID="TRUNK",FUEL_GEOM="CYLINDER", CROWN_WIDTH=0.1041
,CROWN_BASE_HEIGHT=0,TREE_HEIGHT=0.48 /
```

- Ignitor fire

```
&SURF ID='IGN FIRE', HRRPUA=275.,RAMP_Q='RAMPFIRE' /
&RAMP ID='RAMPFIRE',T=0.0,F=0.0 /
&RAMP ID='RAMPFIRE',T=1310.0,F=0.0 /
&RAMP ID='RAMPFIRE',T=1326.0,F=0.5 /
```

```
&RAMP ID='RAMPFIRE',T=1332.0,F=1.0 /

&RAMP ID='RAMPFIRE',T=1338.0,F=1.0 /

&RAMP ID='RAMPFIRE',T=1339.0,F=0.5 /

&RAMP ID='RAMPFIRE',T=1340.0,F=0.0 /

&VENT XB=680,683,50,350,0,0, SURF_ID='IGN FIRE'/
```

- Inflow

```
&SURF ID='INFLOW',VEL=-2.24, RAMP_V='RAMPVEL', PROFILE='ATMOSPHERIC', Z0=20.,PLE=0.143 /

&RAMP ID='RAMPVEL',T=0.0,F=0.0 /

&RAMP ID='RAMPVEL',T=1.0,F=0.1 /

&RAMP ID='RAMPVEL',T=2.0,F=0.2 /

&RAMP ID='RAMPVEL',T=3.0,F=0.3 /

&RAMP ID='RAMPVEL',T=4.0,F=0.4 /

&RAMP ID='RAMPVEL',T=5.0,F=0.5 /

&RAMP ID='RAMPVEL',T=6.0,F=0.6 /

&RAMP ID='RAMPVEL',T=7.0,F=0.7 /

&RAMP ID='RAMPVEL',T=8.0,F=0.8 /

&RAMP ID='RAMPVEL',T=9.0,F=0.9 /

&RAMP ID='RAMPVEL',T=10.0,F=1.0 /
```

- Boundary conditions

```
&VENT XB=0,0,0,400,0,100, SURF_ID='INFLOW'/

&VENT XB=1000,1000,0,400,0,100, SURF_ID='OPEN'/

&VENT XB=0,1000,0,0,0,100, SURF_ID='MIRROR'/

&VENT XB=0,1000,400,400,0,100, SURF_ID='MIRROR'/

&VENT XB=0,1000,0,400,100,100, SURF_ID='MIRROR'/
```

- Output data

```
&DUMP DT_SLCF=1, DT_PART=1, DT_BNDF=1. /
```

- Slice files for wind profiles

```
&SLCF XB=700,900,100,300,1,1, QUANTITY='U-VELOCITY' /
&SLCF XB=700,900,100,300,2,2, QUANTITY='U-VELOCITY' /
&SLCF XB=700,900,100,300,3,3, QUANTITY='U-VELOCITY' /
        ...CONTINUED...
&SLCF XB=700,900,100,300,30,30, QUANTITY='U-VELOCITY' /
&SLCF XB=700,900,100,300,40,40, QUANTITY='U-VELOCITY' /
```

- Slice files for rate of spread and fireline intensity

```
&SLCF XB=700,900,110,110,0,60, QUANTITY='HRRPUV' /
&SLCF XB=700,900,130,130,0,60, QUANTITY='HRRPUV' /
&SLCF XB=700,900,150,150,0,60, QUANTITY='HRRPUV' /
        ...CONTINUED...
&SLCF XB=700,900,270,270,0,60, QUANTITY='HRRPUV' /
&SLCF XB=700,900,290,290,0,60, QUANTITY='HRRPUV' /
```

- Declare end of input file

```
&TAIL /
```

3.2 SUPPLEMENTAL FIGURES AND TABLES

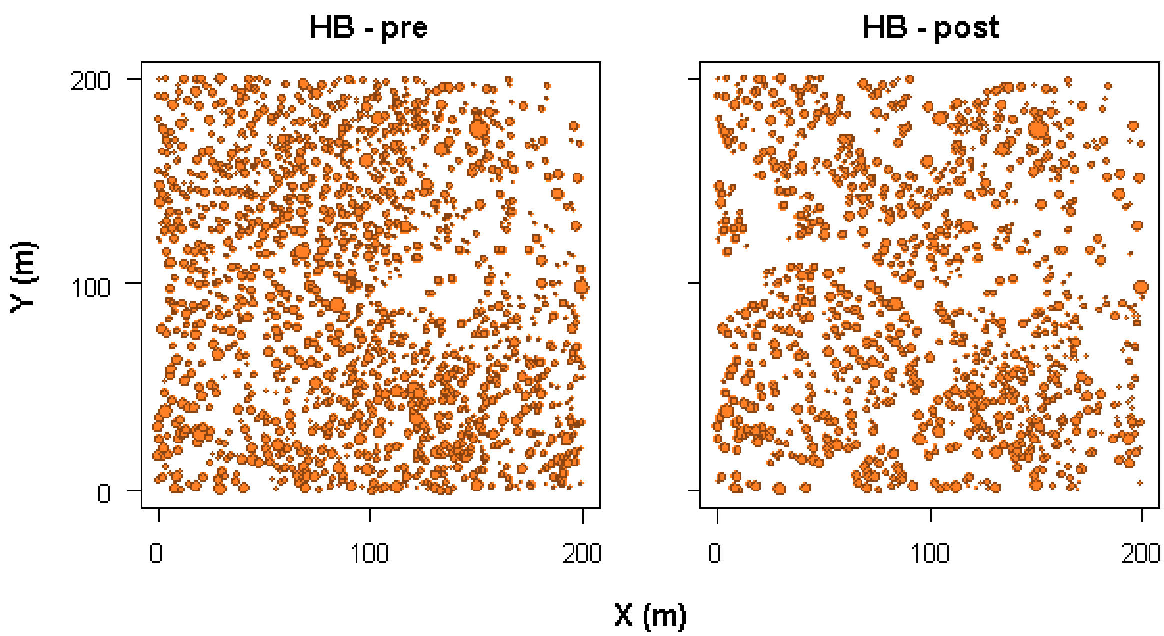

Figure 3.1. Four hectare stem-map of site HB, located within Boulder County Open Space land, before (left) and after (right) thinning. Points scaled to tree crown area.

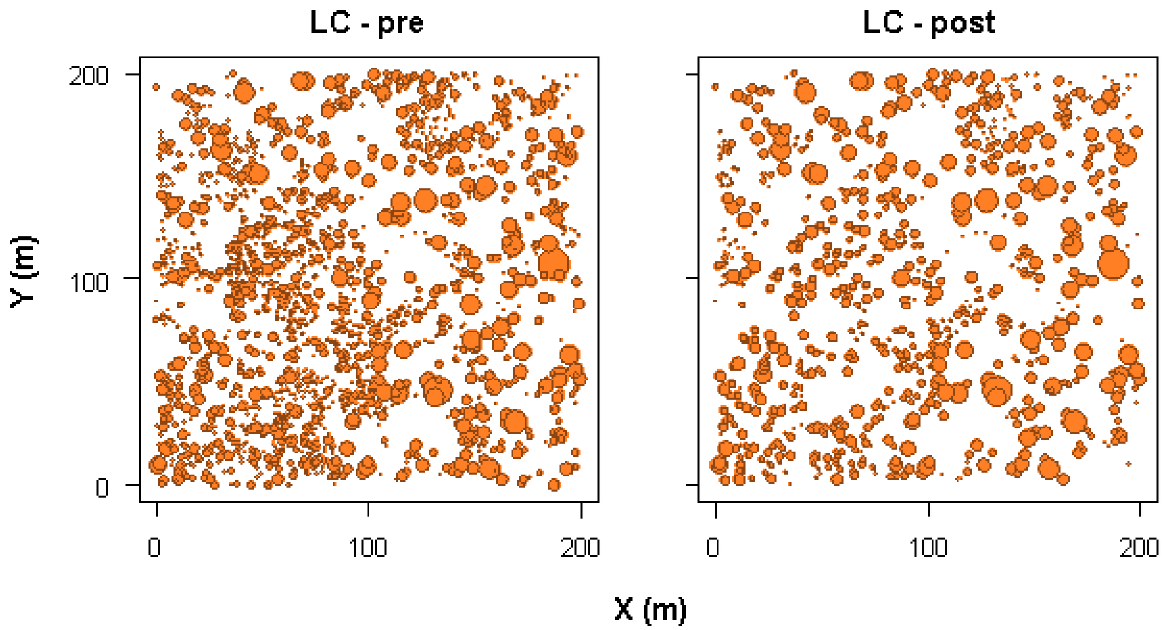

Figure 3.2. Four hectare stem-map of site LC, located within the Kaibab National Forest, before (left) and after (right) thinning. Points scaled to tree crown area.

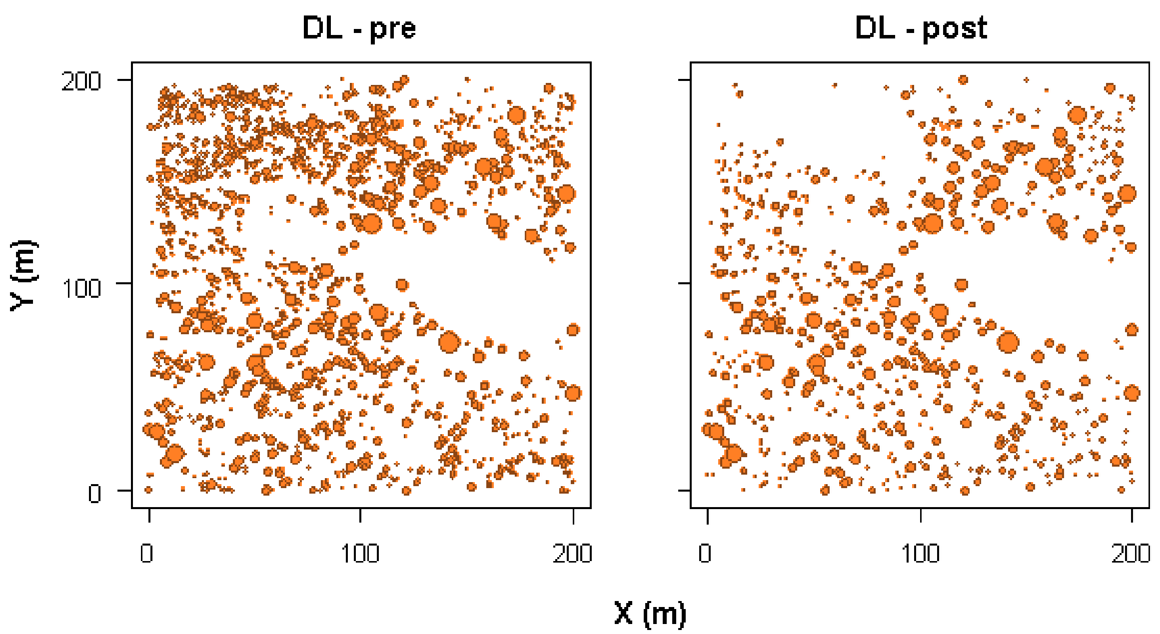

Figure 3.3. Four hectare stem-map of site DL, located within the Roosevelt National Forest, before (left) and after (right) thinning. Points scaled to tree crown area.

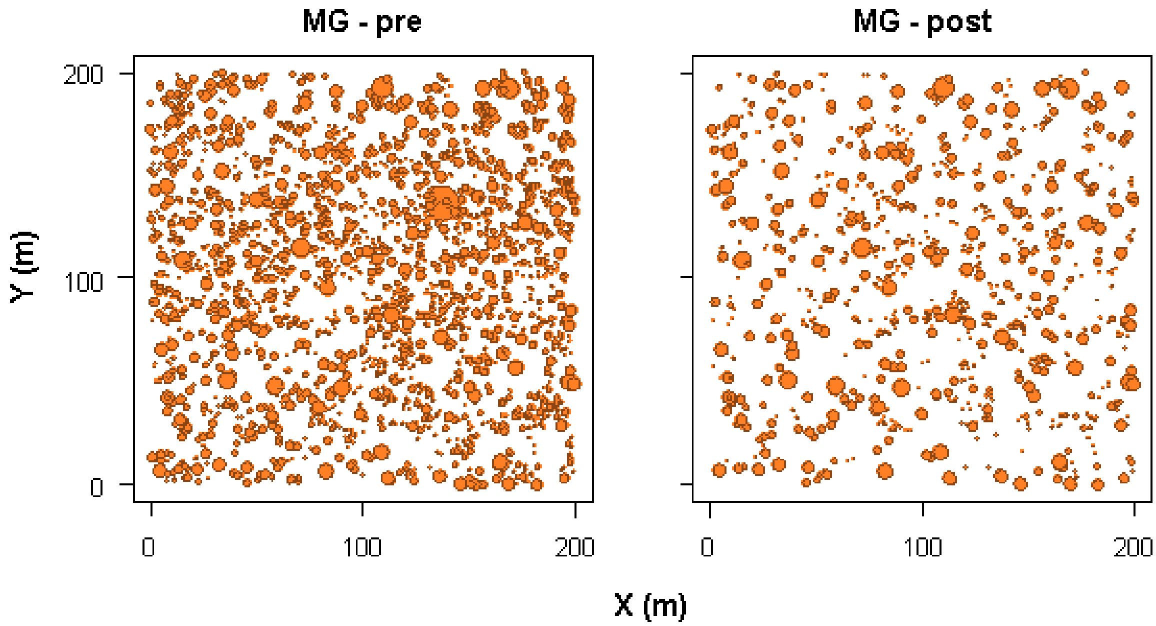

Figure 3.4. Four hectare stem-map of site MG, located within the Pike National Forest, before (left) and after (right) thinning. Points scaled to tree crown area.

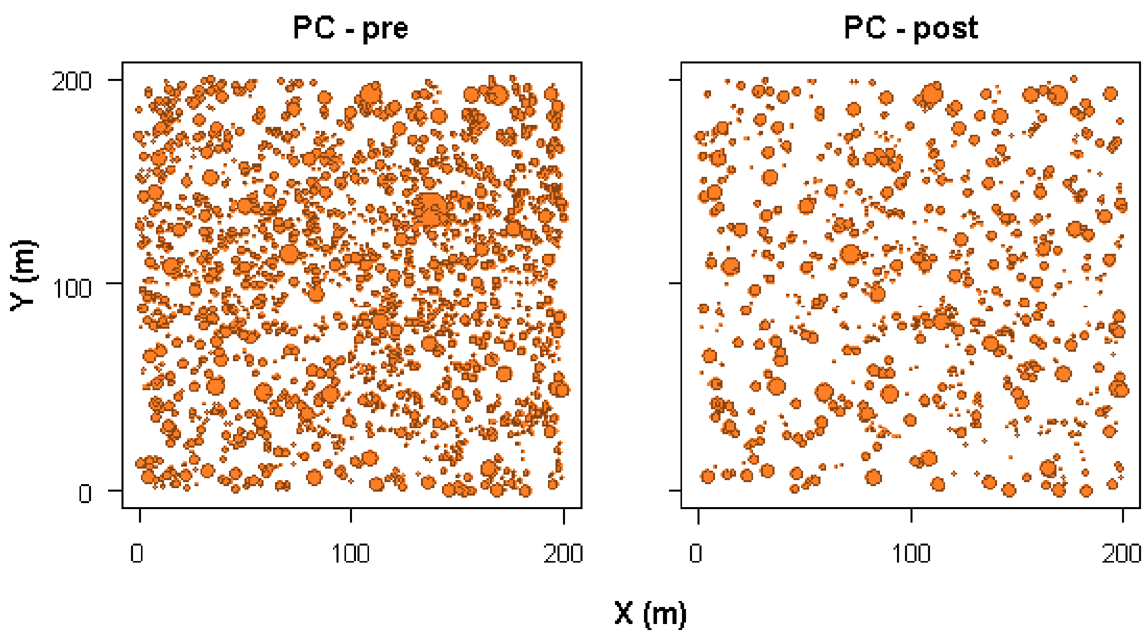

Figure 3.5. Four hectare stem-map of site PC, located within the Pike National Forest, before (left) and after (right) thinning Points scaled to tree crown area.

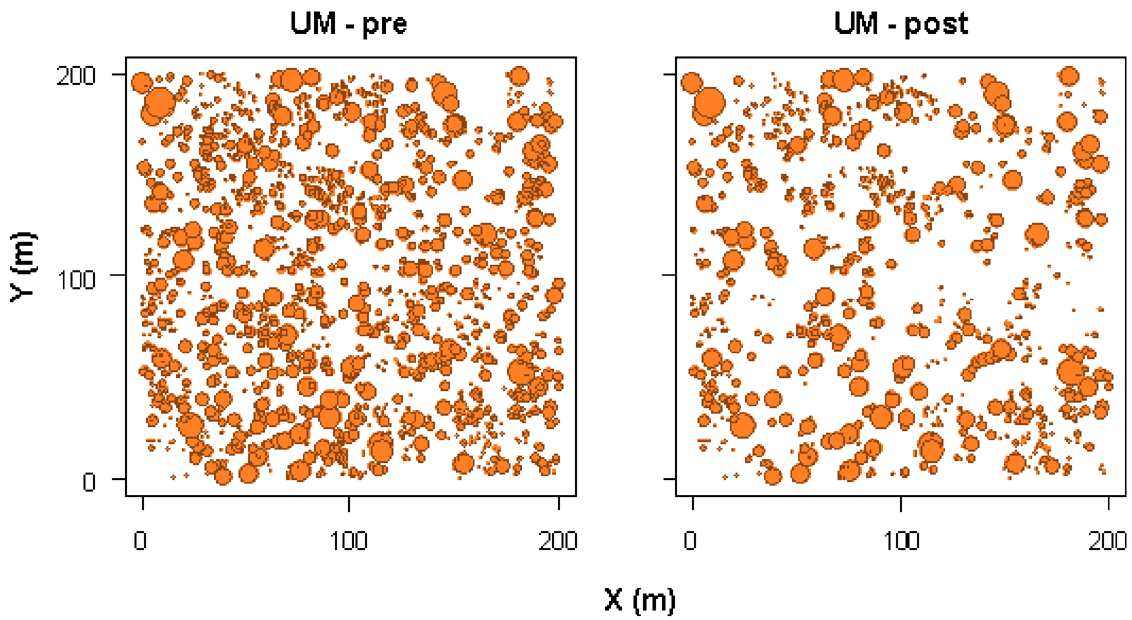

Figure 3.6. Four hectare stem-map of site UM, located within the Kaibab National Forest, before (left) and after (right) thinning. Points scaled to tree crown area.

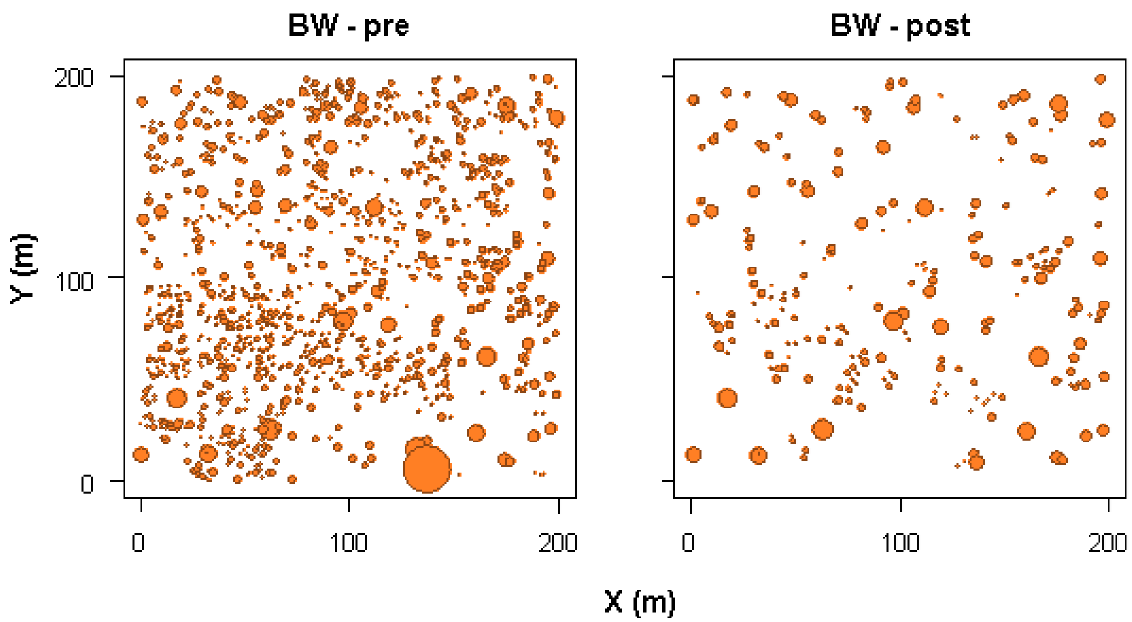

Figure 3.7. Four hectare stem-map of site BW, located within the Cibola National Forest, before (left) and after (right) thinning. Points scaled to tree crown area.

Table 3.1. Summary of linear regressions to estimate tree geometry from diameter at stump height, localized for sites. Full model is y = $\beta_0 + \beta_1 site + \beta_2 dsh + \beta_3$ (*site* x *dsh*). Regressions were reduced following backward selection, removing first *site* x *dsh*, then *site* if the term was not significant ($p < 0.05$).

Spp.	*Y*	β_0	β_2	Site HB		Site LC		Site MG		Site PC		Site DL		Site UM		Site BW		*n*	*F*	r^2	*p*
				β_1	β_3	β_1	β_3	β_1	β_3	β_1	β_3	β_1	β_3	β_1	β_3	β_1	β_3				
PIPO	HT	4.0	0.3	-0.4	-0.1	-2.2	0.1	-2.7	0.1	-0.3	0.0	-2.9	0.0	-2.0	0.1	0.0	0.0	4531	1906	0.85	< 0.001
	CBH	3.9	0.0	-2.0	0.0	-3.4	0.1	-3.0	0.1	-1.9	0.1	-3.7	0.1	-2.8	0.1	0.0	0.0	4528	426	0.55	< 0.001
	DBH	-1.0	0.8	-0.9	0.0	-2.9	0.0	-1.3	0.0	0.2	0.0	-1.3	0.0	-0.7	0.1	0.0	0.0	4552	12131	0.97	< 0.001
	CW	0.1	0.1	0.4	0.0	0.5	0.0	0.3	0.0	0.4	0.0	0.5	0.0	0.4	0.0	0.0	0.0	4501	1042	0.75	< 0.001
POTR5	HT*	1.6	0.5	-	-	-	-	-	-	0.0		-0.7		-0.2		-	-	1255	2131	0.84	< 0.001
	CBH*	0.9	0.3	-	-	-	-	-	-	0.0		-1.5		-0.3		-	-	1255	1176	0.74	< 0.001
	DBH*	-0.6	0.9	-	-	-	-	-	-	0.0	0.0	0.0	-0.1	0.7	0.0	-	-	1255	9732	0.97	< 0.001
	CW	0.7	0.1	-	-	-	-	-	-	0.0	0.0	-0.1	0.1	0.4	0.0	-	-	1251	505	0.67	< 0.001
PSME	HT	0.1	0.4	0.8	0.0	-	-	1.7	-0.2	1.5	-0.1	-2.1	0.0	0.0	0.0	-	-	381	273	0.85	< 0.001
	CBH	1.2	0.0	0.8	0.0	-	-	-1.1	0.1	-1.0	0.1	-0.3	0.0	0.0	0.0	-	-	379	96	0.67	< 0.001
	DBH	0.2	0.9	-2.0	-0.1	-	-	-1.4	-0.2	-1.1	-0.1	-4.1	0.0	0.0	0.0	-	-	381	1565	0.97	< 0.001
	CW†	0.8	0.1			-	-									-	-	381	1451	0.79	< 0.001

Species codes according to NRCS PLANTS Database; HT, CBH, DBH, DSH and CW refer to height (m), crown base height (m), diameter at breast height (cm), diameter at stump height and crown width (m), respectively; * and † indicate reduced models without an interaction term, and without both site and interactions terms, respectively; "-" indicates species not present in site

Table 3.1 (cont.).

Spp.	*Y*	β_0	β_2	Site HB		Site LC		Site MG		Site PC		Site DL		Site UM		Site BW		*n*	*F*	r^2	*p*
				β_1	β_3	β_1	β_3	β_1	β_3	β_1	β_3	β_1	β_3	β_1	β_3	β_1	β_3				
PICEA	HT	0.3	0.4	-	-	-	-	-	-	0.0	0.0	-	-	1.0	0.0	-	-	230	496	0.87	< 0.001
	CBH†	0.4	0.1	-	-	-	-	-	-			-	-			-	-	232	188	0.45	< 0.001
	DBH*	-1.5	0.9	-	-	-	-	-	-	0.0		-	-	0.4		-	-	232	12120	0.99	< 0.001
	CW*	0.8	0.1	-	-	-	-	-	-	0.0		-	-	0.5		-	-	232	260	0.69	< 0.001
QUGA	HT†	2.0	0.2	-	-	-	-	-	-	-	-	-	-			-	-	190	114	0.38	< 0.001
	CBH†	1.3	0.1	-	-	-	-	-	-	-	-	-	-			-	-	190	41	0.18	< 0.001
	DBH†	-0.3	0.9	-	-	-	-	-	-	-	-	-	-			-	-	190	3018	0.94	< 0.001
	CW†	0.8	0.1	-	-	-	-	-	-	-	-	-	-			-	-	190	91	0.33	< 0.001
PIED	HT†	3.0	0.2	-	-	-	-	-	-	-	-	-	-	-	-			33	19	0.38	< 0.001
	CBH†	0.6	0.0	-	-	-	-	-	-	-	-	-	-	-	-			33	5	0.13	0.041
	DBH†	4.5	0.8	-	-	-	-	-	-	-	-	-	-	-	-			33	49	0.61	< 0.001
	CW†	1.8	0.1	-	-	-	-	-	-	-	-	-	-	-	-			33	20	0.39	< 0.001
JUSC2	HT†	1.4	0.2			-	-	-	-	-	-			-	-			21	58	0.20	0.046
	CBH†	-0.3	0.1			-	-	-	-	-	-			-	-			21	4	0.60	< 0.001
	DBH†	-3.8	0.9			-	-	-	-	-	-			-	-			14	122	0.91	< 0.001
	CW†	0.5	0.1			-	-	-	-	-	-			-	-			20	42	0.70	< 0.001